UNITED NATIONS ENVIRONMENT PROGRAMME

REGIONAL SEAS

# Technical annexes to the report on the state of the marine environment

*UNEP Regional Seas Reports and Studies No. 114/2*

*Prepared in co-operation with*

United Nations FAO UNESCO WHO WMO IMO IAEA

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## PART 1



## PART 2



*International Maritime Organization, office for the London Dumping Convention.

## PREFACE

This publication contains 16 technical annexes which were used in preparing the report on the state of the marine environment by the IMO/FAO/Unesco/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) which was published by UNEP as a GESAMP report.[1]

The annexes were written by individual experts commissioned by Professor Alasdair McIntyre who, as the Chairman of a GESAMP Working Group on the state of the marine environment, co-ordinated the preparation of the GESAMP report. The annexes were endorsed by the Working Group but are the responsibility of their individual authors.

The organizations sponsoring GESAMP would like to acknowledge with appreciation the contribution made by the authors of the technical annexes to the GESAMP report on the state of the marine environment.

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[1] Rep. Stud. GESAMP No. 39 and UNEP Regional Seas Reports and Studies No. 115.

# ANNEX VII

# INTERNATIONAL CONVENTIONS ON THE PREVENTION OF MARINE POLLUTION: CONTROL STRATEGIES

LONDON DUMPING CONVENTION UNIT

INTERNATIONAL MARITIME ORGANIZATION
LONDON
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# INTRODUCTION

## I. INTERNATIONAL GLOBAL CONVENTIONS ON THE CONTROL AND PREVENTION OF MARINE POLLUTION

### A. THE LEGAL FRAMEWORK 1950 - 1973

1. Between 1950 and the 1970s it became generally recognized that marine pollution had developed into a serious environmental problem. Possibly the greatest public exposure during these times was given to marine pollution arising from accidental spillages and operational discharges of oil at sea. As a result of such concerns, a number of multilateral conventions were adopted at that time specifically addressing these problems.[1]

2. Besides the concerns related to the pollution of the sea by oil, anxieties were also expressed regarding the impact of radioactive substances on human health and marine life. Article 25 of the High Seas Convention (adopted in 1958) calls upon States to "prevent pollution of the seas from the dumping of radioactive waste" and to co-operate in taking measures "for the prevention of pollution...resulting from any activities with radioactive materials". Accordingly, a number of conventions were adopted between 1950 and 1971 concerning nuclear substances ad-

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- International Convention for the Prevention of Pollution of the Sea by Oil, London 1954; amended 1962, 1969 and 1971;
- Convention on the High Seas, Geneva, 1958 (Article 24);
- International Convention Relating to Intervention on the High Sea in Cases of Oil Pollution Casualties, Brussels 1969;
- Agreement for Co-operation in Dealing with Pollution of the North Sea by Oil, Bonn 1969;
- International Convention on Civil Liability for Oil Pollution Damage, Brussels 1969;
- International Convention on the Establishment of an International Fund for Oil Pollution Damage, Brussels, 1971;
- Agreement between Denmark, Finland, Norway and Sweden concerning Co-operation in Measures to Deal with Pollution of the Sea by Oil, Copenhagen, 1971.

dressing, however, mainly the liability in cases of damage caused by radioactive substances.[2]

3. In the early 1970s public concerns expanded to new substances, types and sources of marine pollution. Thus, it was recognized that the capacity of the marine environment to assimilate wastes dumped at sea was limited. Accordingly, the text of an international convention on the prevention of marine pollution by dumping wastes at sea was prepared by a series of international conferences and subsequently adopted in 1972. This is the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, London, 1972 (London Dumping Convention).

4. Another effort made in the early 1970s with a view to covering in a comprehensive way the various sources of marine pollution, resulted in the adoption of the International Convention for the Prevention of Pollution from Ships, London 1973, as modified by the Protocol of 1978 (MARPOL 73/78). This Convention provides for a comprehensive control system for operational discharges from ships of oil (Annex I), noxious liquid substances carried in bulk (Annex II), harmful substances carried in packaged forms, containers, portable tanks (Annex III), sewage (Annex IV) and garbage (Annex V). The 1973 Conference which adopted this Convention also prepared a Protocol relating to the Intervention on the High Seas in Cases of Marine Pollution by Substances other than Oil. This instrument complements the 1969 Convention Relating to Intervention on the High Seas in Cases of Oil Pollution Casualties.

## B. SOURCES OF MARINE POLLUTION

5. Besides the control of the above sources of marine pollution, *viz.* maritime transport and waste disposal at sea, other sources have not yet been covered by globally applicable legal instruments, e.g. pollution from land-based sources, through the atmosphere and the exploration and exploitation of the sea-bed.

6. A comparison of the amounts of substances that enter the sea from the various sources clearly indicates that the legal control mechanisms currently in force for the prevention of marine pollution by shipping and by waste disposal at sea cover only a relatively small portion of pollu-

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- Convention on the High Seas, Geneva, 1958. See also:
- Convention on Third Party Liability in the Field of Nuclear Energy, Paris, 1960;
- Convention on the Liability of Operators of Nuclear Ships, Brussels, 1962;
- Convention on Civil Liability for Nuclear Damage, Vienna, 1963;
- Treaty Banning Nuclear Weapon Tests in the Atmosphere, in Outer Space and Under Water, Moscow, 1963;
- Convention Relating to Civil Liability in the Field of Maritime Carriage of Nuclear Material, Brussels, 1971.

tion. Indeed, it is assumed that the input to the sea of substances through and from the atmosphere and from land-based sources (i.e. rivers, pipelines, direct outfalls) together is approximately 70 per cent of the total input. A very rough estimate based on figures available for petroleum hydrocarbons through IMO publications and material submitted to the Second International Conference on the Protection of the North Sea (1987) is as follows:

INPUT OF SUBSTANCES TO THE MARINE ENVIRONMENT
(per cent)

| | All Contaminants | Petroleum Hydrocarbons (global) | Petroleum Hydrocarbons (North Sea, 1987) |
|---|---|---|---|
| Natural sources | 8 | 8 | 2 |
| Offshore production | 0.5 | 2 | 28 |
| Maritime transportation | 11 | 45 | 3 |
| Atmosphere | 30 | 10 | 12 |
| Run-off and land-based discharges | 40 | 34 | 43 |
| Dumping | 10 | 1 | 12 |

7. The above figures highlight the main sources of marine pollution, namely the atmosphere and land-based sources. The situations in the various sea areas differ in that major sources depend on such factors as the degree of industrialization of coastal States, population in coastal areas and offshore activities. For example, the estimate of inputs of petroleum hydrocarbons to the North Sea, when compared with global figures, clearly demonstrates the special character of the North Sea as an area with intensive offshore activities.

8. With regard to other contaminants entering the North Sea, the input of nitrogen through river inputs and direct discharges from land was estimated to be 72 per cent of the total nitrogen input (with 26 per cent introduced through the atmosphere), and the input of phosphorus from land-based sources 98 per cent, with two per cent from dumping sewage sludge at sea. For metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn) the atmosphere is the main source in that it contributes 73 per cent, whereas land-based sources provide approximately 12 per cent and dumping at sea 15 per cent. However, this appreciable amount derived from the dumping source originates mainly from the disposal of dredged materials at sea in which the metals are contained in minerals or are firmly bound to particulates. The metals derived from this source are therefore to a large extent not bioavailable and so do not contribute to the pollution burden of the North Sea.

9. It had certainly already been realized in the early 1970s that both shipping and dumping of wastes at sea were minor sources of marine pollution when compared with the input from the atmosphere and from land-based sources (rivers, run-off from land, direct outfalls). However, regulations for the control of disposal of wastes at sea were relatively easy to develop, especially as a number of countries bordering the North Sea and the North East Atlantic, plus Finland were negotiating the Oslo (Dumping Convention) at that time. The Oslo Convention was then used as a model for the globally applicable London Dumping Convention. The global character of shipping indicated that this source should be covered by a global convention. The existence at that time of global conventions on the protection of the marine environment by oil from ships also made it relatively easy to develop additional globally applicable requirements on the prevention of pollution from ships. Subsequent concerns on other pollution sources, in particular from land-based sources, has been met so far in some regions only through regional agreements and regulations (see section E below).

## C. THE STOCKHOLM CONFERENCE (1972)

10. Developments since mid-1972 have certainly been influenced by the outcome of the UN Conference on the Human Environment which took place in Stockholm from 6 to 16 June 1972 (Stockholm Conference). The object of the Conference was to create "a basis for comprehensive considerations" of the problem of the human environment, and to "focus the attention of governments and public opinion in various countries on the importance of the problem". The final documents of the Conference are the Declaration, which contains 26 principles, and an Action Plan which contains 109 recommendations. In this context, principles 7 and 21, which include a general obligation of States to preserve the marine environment are of particular importance.

### *1. Principle 7*

11. "States shall take all possible steps to prevent pollution of the seas by substances that are liable to create hazards to human health, to harm living resources and marine life, to damage amenities or to interfere with other legitimate uses of the sea".

12. Principle 7 reflects the duty of States to protect the marine environment and also introduces a definition of marine pollution based on a definition prepared for the Stockholm Conference by the then IMCO/FAO/UNESCO/WMO/WHO/IAEA/UN Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP) which is cited as follows:[3]

13. Pollution is ... "the introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to

impairment of quality for use of sea water, and reduction of amenities."

14. Conventions on marine pollution adopted prior to 1972 (the date of the Stockholm Conference) do not include any definition of marine pollution. In legal texts adopted or prepared between 1972 and 1974, the definition of marine pollution is not used as such but it is implicitly contained in the basic obligations for the protection of the marine environment or in other relevant concepts such as the concept of harmful substances. For example, the London Dumping Convention in Article I requires that Contracting Parties pledge themselves to:

> take all practicable steps to prevent the pollution of the sea by the dumping of waste and other matter that is liable to create hazards to human health, to harm l ving resources and marine life, to damage amenities or to interfere with other legitimate uses of the sea.

15. The United Nations Convention on the Law of the Sea (UNCLOS) which was adopted in 1982, defines marine pollution in its Article 1, paragraph 1(4) as follows:

> "pollution of the marine environment" means the introduction by man, directly or indirectly, of substances or energy into the marine environment, including estuaries, which results or is likely to result in such deleterious effects as harm to living resources and marine life, hazards to human health, hindrance to marine activities, including fishing and other legitimate uses of the sea, impairment of quality for use of sea water and reduction of amenities.

16. Since 1974 the basic elements of the GESAMP definition, namely human interference with the marine environment and a list of undesirable effects have been included in all internationally agreed legal instruments for the prevention and control of marine pollution.

17. It should also be noted that whereas in accordance with the definition of GESAMP, human interference with the environment causing undesirable effects is called "pollution", the introduction by man of substances and energy into the marine environment which do *not* lead to harmful effects does not fall under the definition of "pollution". While such activities are sometimes referred to as "contamination", this does not, however, imply that they should be exempt from control measures.

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3 The United Nations Environmental Programme (UNEP), which was established by the UN General Assembly pursuant to a recommendation of the Stockholm Conference, joined the group of Sponsoring Organizations of GESAMP at a later stage. IMCO (Inter-Governmental Maritime Consultative Organization) in 1982 changed its name to IMO (International Maritime Organization).

### 2. *Principle 21*

18. "States have, in accordance with the Charter of the United Nations and the principles of international law, the sovereign right to exploit their own resources pursuant to their own environmental policies, and the responsibility to ensure that activities within their jurisdiction or control do not cause damage to the environment of other States or of areas beyond the limits of national jurisdiction."

19. Principle 21, in its first element, recognizes the sovereign right of a State to exploit its own resources. The second element, concerning the responsibility to avoid damage to the environment of other States or of areas beyond the limits of national jurisdiction, is of fundamental importance in terms of controlling transfrontier pollution and in ensuring the protection of coastal interests as well as the shared resources of the high seas.

## D. GLOBAL AGREEMENTS 1973 - 1982

20. In addition to the global rules, standards and procedures regarding the protection of the marine environment from waste disposal at sea (London Dumping Convention, 1972) and from operational vessel-source pollution (MARPOL 73/78), the prevention of accidental pollution is covered by the internationally agreed provisions for the safety of navigation. In this respect the International Convention on the Safety of Life at Sea, 1974 (SOLAS 74) has a significant role concerning the protection of the marine environment.

21. The need to take measures for pollution prevention from land-based sources, from the atmosphere or from sea-bed activities was increasingly recognized. Accordingly, the United Nations Convention on the Law of the Sea (1982) provides a broad outline for action concerning the prevention of marine pollution from *all* sources at the global level, i.e. various articles of the Convention request States to act especially through competent international organizations or diplomatic conferences for the establishment of global rules and standards to prevent, reduce and control marine pollution from various sources, including land-based sources, the atmosphere and sea-bed activities.

22. A further initiative aimed at the control of marine pollution from land-based sources was taken soon after the conclusion of the 1982 UN Convention on the Law of the Sea by UNEP in convening the first session of the Ad Hoc Working Group of Experts on the Protection of the Marine Environment against Pollution from Land-based Sources. This effort resulted in the adoption, in 1985, of international guidelines on the Protection of the Marine Environment Against Pollution from Land-based Sources (Montreal Guidelines).

## E. REGIONAL AGREEMENTS

23. Several of the early global conventions (e.g. the London Dumping Convention) express the necessity that States with common interests to protect the marine environment in a given geographical area enter into regional agreements consistent with global conventions, but taking also into particular account characteristic regional features. Preparatory work in the early seventies aiming at the establishment of regional conventions was supported by decisions made by the Stockholm Conference in 1972. Increased activities in the protection of regional sea areas since 1972 were also due to the establishment in that year of the United Nations Environment Programme (UNEP). UNEP developed regional action plans for the environmental protection of relevant parts of the oceans, its first major regional activities being focussed on the protection of the Mediterranean Sea. The Regional Seas Programme of UNEP has since been expanded. In 1988 it covered eleven different regions.

24. The following Conventions have been adopted within UNEP's Regional Seas Programme:

*Mediterranean Sea*

Convention for the Protection of the Mediterranean Sea against Pollution, 1976 (Barcelona Convention), with the following Protocols:

- Protocol concerning Co-operation in Combating Pollution of the Mediterranean Sea by Oil and Other Harmful Substances in Cases of Emergency, 1976.

- Protocol for the Prevention of Pollution of the Mediterranean Sea by Dumping from Ships and Aircraft, 1976.

- Protocol for the Protection of the Mediterranean Sea against Pollution from Land-based Sources, Athens, 1980.

- Protocol concerning Mediterranean Specially Protected Areas, Geneva, 1982.

*Gulf Area*

Kuwait Regional Convention for Co-operation on the Protection of the Marine Environment from Pollution, 1978 (Kuwait Convention), with the following Protocol:

- Protocol concerning Regional Co-operation in Combating Pollution by Oil and Other Harmful Substances in Cases of Emergency, 1978.

*West and Central Africa*

Convention for Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region, 1981 (Abidjan Convention), with the following Protocol:

- Protocol concerning Co-operation in Combating Pollution in Cases of Emergency, 1981.

*East Africa*

Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region, 1985 (Nairobi Convention), with the following Protocols:

- Protocol concerning Protected Areas and Wild Fauna and Flora in the Eastern African Regions, 1985.

- Protocol concerning Co-operation in Combating Marine Pollution in Cases of Emergency in the Eastern African Region, 1985.

*South-East Pacific*

Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific, 1981 (Lima Convention), with the following Protocols:

- Agreement on Regional Co-operation in Combating Pollution of the South-East Pacific by Oil and Other Harmful Substances in Cases of Emergency, Lima, 1981.

- Protocol for the Protection of the South-East Pacific Against Pollution from Land-based Sources, Quito, 1981.

*Red Sea and Gulf of Aden*

Regional Convention for the Conservation of the Red Sea and Gulf of Aden Environment, 1982 (Jeddah Convention), with the following Protocol:

- Protocol Concerning Regional Co-operation in Combating Pollution by Oil and other Harmful Substances in Cases of Emergency, 1982.

*Caribbean Region*

Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region, 1983 (Cartagena Convention), with the following Protocol:

- Protocol concerning Co-operation in Combating Oil Spills in the Wider Caribbean Region, 1983.

*South Pacific*

Convention for the Protection of the Natural Resources and Environment of the South Pacific Region, 1986 (Noumea Convention), with the following Protocols:

- Protocol for the Prevention of Pollution of the South Pacific Region by Dumping, 1986.

- Protocol concerning Co-operation in Combating Pollution Emergencies in the South Pacific Region, 1986.

25. Regional legal instruments which have been established since 1972 in Europe independently from UNEP's Regional Seas Programme are:

*North East Atlantic and North Sea*

- Convention for the Prevention of Marine Pollution by Dumping from Ships and Aircraft, 1972 (Oslo Convention).

- Convention for the Prevention of Marine Pollution from Land-based Sources, 1974 (Paris Convention).

*Baltic Sea*

- Convention on the Protection of the Marine Environment of the Baltic Sea Area, 1974 (Helsinki Convention).

*Scandinavian waters*

- Convention on the Protection of the Environment between Denmark, Finland, Norway and Sweden (with Protocol), 1974 (Stockholm Convention). Agreement between Denmark, Finland, Norway and Sweden concerning Co-operation in Measures to Deal with Pollution of the Sea by Oil, Copenhagen, 1971.

*North Sea*

- Agreement for Co-operation in Dealing with Pollution of the North Sea by Oil, Bonn 1969.

- Agreement for Co-operation in Dealing with Pollution of the North Sea by Oil and Other Harmful Substances, 1983.

26. It is remarkable that, beside the Paris Convention and the Helsinki Convention, which contain provisions for the protection of the marine environment from land-based sources, only two of the eight UNEP's Regional Seas Conventions have been supplemented by protocols on the prevention and control of marine pollution by land-based sources, despite the considerable amounts of substances entering the other regional seas through these sources. Likewise, no international agreement regulates pollution from land-based sources in the West Atlantic and the whole of the North Pacific.

27. It is less surprising that beside the North Sea and the Baltic, only two regional seas are protected by dumping protocols: the Mediterranean and the South Pacific. This is certainly due to the existence of the London Dumping Convention being considered as the appropriate instrument in controlling and regulating waste disposal in the various regions. However, it should be recalled that the London Dumping Convention itself expresses the need that States with common interests to protect the marine environment in a given geographical area enter into regional agreements.

28. The reluctance to develop and adopt legally binding protocols concerning the prevention and control of marine pollution from land-based sources should be understood in the light of the far-reaching implications they would have on the industries, municipalities, and agriculture of the prospective parties since tight control measures, standards, surveillance and monitoring systems would have to be established. This is also the reason why the establishment of a globally applicable and all-embracing convention on the protection of the marine environment from land-based sources seems at this stage not to be possible, taking into account the many different stages of development (industrialization, population, science and technology) in the various regions of the world. There is also an underlying feeling that land-based sources of pollution are principally a matter to be dealt with at the national level.

29. With regard to the other major source of marine pollution, the atmosphere, control measures covering the Baltic Sea are contained in the Helsinki Convention. Within the framework of the Paris Convention, relevant measures have also been taken for the protection of the North East Atlantic including the North Sea area. The lack of internationally adopted protection measures within other regions again reflects the implications these would have for industry, particularly in less developed countries.

30. A regional convention covering the North Sea is the Convention on Civil Liability for Oil Pollution Damage Resulting from Exploration for and Exploitation of Sea-bed Mineral Resources, London 1977. This Convention has not yet entered into force.

## II. INTERNATIONAL INSTITUTIONS

31. There is a large number of international agreements, global and regional, aiming at the protection of the sea. For the administration of these conventions there is likewise a number of institutions, embodied either in the United Nations system, or regional economic organization or regional secretariats that have been established for this particular purpose.

32. The two global conventions for the prevention and control of marine pollution, the London Dumping Convention and MARPOL 73/78, are both administered by the International Maritime Organization (IMO), which is the only specialized agency of the United Nations totally dedicated to maritime affairs. This organization has developed over the years a strategy for the protection of the marine environment from pollution from ships and from the disposal of wastes at sea. The fundamental components of that strategy, implicit in the charter of the Organization, are:

1. to provide an effective machinery for technical, legal and scientific co-operation among Governments in the field of the protection of the marine environment from pollution from ships and related activities and the mitigation of the environmental effects of such pollution and compensation;

2. to adopt the highest practicable international standards in matters concerning maritime safety and prevention and control of marine pollution from ships and related activities;

3. to encourage the widest possible acceptance and effective implementation of these standards at the global level;

4. to strengthen the capacity for national and regional action to prevent control, combat and mitigate marine pollution and to promote technical co-operation to this end; and

5. to co-operate fully with other organizations within the United Nations family and relevant international, regional and non-governmental organizations to ensure a co-ordinated approach to the problem and avoid wasteful duplication of efforts.

33. The above strategy should be seen as an example for other global institutions involved in the protection of the environment.

34. The strategies of UNEP for each of its regions are similar. They basically include:

1. the preparation of an Action Plan setting out activities for scientific research and co-operation, including environmental assessment and management;

2. the preparation and adoption of a legally binding framework convention (umbrella convention) embodying general obligations;

3. the preparation of protocols dealing with specific problems and sources of marine pollution, such as dumping, land-based sources, co-operation in pollution emergencies, conservation, etc; and

4. the adoption of financial and institutional arrangements that provide back-up for the activities mentioned in sub-paragraph 1 above.

# III. REVIEW OF INTERNATIONAL LEGAL INSTRUMENTS

35. A review of existing international legal instruments shows that the approaches used for controlling the input of substances to the sea and the evaluation of the hazards of substances entering the sea differ to some extent. The basic approaches are outlined in the following paragraphs by using the following examples:

1. the requirements of MARPOL 73/78, for the control of operational discharges of harmful substances from ships;

2. the requirements of the London Dumping Convention, 1972, for the prevention of marine pollution by wastes and other matter dumped at sea;

3. the requirements of the Paris Convention, 1974, for the prevention of marine pollution from land-based sources, as well as the recommendations contained in the Montreal Guidelines on the Prevention of Marine Pollution from Land-Based Sources, 1985.

## A. THE INTERNATIONAL CONVENTION ON THE PREVENTION OF POLLUTION FROM SHIPS, 1973, AS MODIFIED BY THE PROTOCOL OF 1978 RELATING THERETO (MARPOL 73/78)

36. MARPOL 73/78 in its Annexes contains regulations for:

1. the prevention of pollution by oil (Annex I);

2. the control of pollution by noxious liquid substances in bulk (Annex II);

3. the prevention of pollution by harmful substances carried by sea in packaged form, or in freight containers, portable tanks for road and and rail tank wagons (Annex III);

4. the prevention of pollution by sewage from ships (Annex IV); and

5. the prevention of pollution by garbage from ships (Annex V).

### *1. Prevention of pollution by oil*

37. The control or prevention of pollution of the sea by oil from ships is achieved through Annex I of the International Convention for the Prevention of Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78). Approximately 81 per cent by deadweight of the world's merchant fleet is governed by MARPOL 73/78 and some 10.5 per cent follow the International Convention for the Prevention of Pollution of the Sea by Oil, 1954, as amended in 1969 (OILPOL 54/69), which contains provisions almost identical to those of Annex I of MARPOL 73/78.

38. For the purpose of the Annex, "oil" is defined as "petroleum in any form including crude oil, fuel oil, sludge, oil refuse and refined products (other than petrochemicals which are subject to the provisions of Annex II of the present Convention) and, without limiting the generality of the foregoing, includes the substances listed in Appendix I to this Annex."

39. The lists of oils set out in Appendix I to MARPOL 73/78 includes the following:

Asphalt solutions

Blending Stocks
Roofers Flux
Straight Run Residue

Oils

Clarified
Crude Oil
Mixtures containing crude oil
Diesel Oil
Fuel Oil No.4
Fuel Oil No.5
Fuel Oil No.6
Residual Fuel Oil
Road Oil
Transformer Oil
Aromatic Oil (excluding vegetable oil)
Lubricating Oils and Blending Stocks
Mineral Oil
Motor Oil

Gasoline Blending Stocks

Alkylates - fuel
Reformates
Polymer - fuel

Gasolines

Casinghead (natural)
Automotive
Aviation
Straight Run
Fuel Oil No.1 (Kerosene)
Fuel Oil No.1–D
Fuel Oil No.2
Fuel Oil No.2–D

Jet Fuels

JP–1 (Kerosene)
JP–3
JP–4
JP–5 (Kerosene, Heavy)

Turbine Oil

Distillates

Straight Run
Flashed Feed Stocks

Gas Oil

Cracked

Mineral Spirit

Naphta

Solvent
Heartcut Distillate Oil

40. All ships in their machinery spaces use oil as fuel and lubricant for main and auxiliary engines. MARPOL 73/78 prohibits discharges of machinery space bilges, which contain leaked oil etc., unless bilge water is first passed through oily-water separating equipment and the oil content in the effluent is less than 15 ppm or, when the ship is at least 12 miles from shore and proceeding en route, less than 100 ppm.

41. Oil tankers are considered to provide the main source of oil to the sea by discharging ballast waters and oil residues from their cargo tanks. Oil tankers below a certain dead weight while engaged on ballast voyages have to carry ballast water in cargo tanks, producing oil-contaminated water. MARPOL 73/78 requires oil tankers above a certain size to have completely segregated ballast tanks of sufficient capacity, or to apply a crude-oil washing system for the tanks which will be ballasted, or to dedicate certain tanks to the carriage of ballast water only. On tankers under 20,000 dwt, in which cargo tanks are regularly ballasted on the voyage to the loading port, the discharge of ballast must be monitored through approved oil content meters so that the amount of oil discharged to the sea is limited to 60 litres per mile and the maximum amount of oil discharged is less than 1/15,000 of the cargo carried on the loaded voyage. (For new tankers the figure is 1/30,000).

42. In "special areas" listed in MARPOL 73/78, Annex I, no discharges of 15 ppm and above are allowed. Special areas in this context are the Mediterranean Sea, the Baltic Sea, the Black Sea, the Red Sea, the Gulf Area and the Gulf of Aden. These special area requirements will only become operational for the latter three areas upon decisions taken by IMO, which in turn will require notification to IMO that the necessary provisions have been made in their ports for the reception of oily wastes.

43. Governments of parties to the Convention undertake to provide reception facilities for oily residues and oily mixtures.

44. Annex I also contains important regulations to control the accidental outflow of oil from a tanker involved in a collision or grounding, largely through provisions of structural requirements

of the ship in the event of assumed damage levels.

45. The measures adopted within MARPOL 73/78, Annex I, have had tremendous implications for the shipping industry: loss of up to 30 per cent of cargo space due to the introduction of segregated ballast tanks and dedicated clean ballast tanks; investment in crude oil washing as well as in oil separation equipment and oil monitoring meters and the training of crew. The provision of reception facilities for oil wastes on land, the treatment of these wastes and their ultimate disposal also involve additional costs for the Parties to the Convention.

46. It should also be mentioned that the definition of "oil" as contained in the Convention does not take into account any physical, chemical or biological properties of the various types of oils nor of their constituents. The above list of oils covers in fact a wide variety of "oils" (e.g. asphalt solutions, crude oil, fuel and lubricating oils, gasolines, kerosene and naphtha solvents), totally neglecting the effects which the different types of "oil" may have on the marine environment and human health under varying conditions and in different areas.

### 2. *Control of pollution by chemicals*

#### (a) *Operational discharges from ships*

47. The regulations for the control of pollution by noxious liquid substances in bulk (MARPOL 73/78) are based on different sets of standards and procedures for the discharge of tank washings and residues from chemical tankers to the sea (with specific conditions for "special areas"), depending upon the hazards the chemicals may pose to the marine environment and human health. Accordingly, noxious liquid substances transported in bulk by ships have been divided into four pollution categories as follows:

Pollution Category A: Substances which would present a major hazard to either marine resources or human health or cause serious harm to amenities or other legitimate uses of the sea.

Pollution Category B: Substances which would present hazards to either marine resources or human health or cause harm to amenities or other legitimate uses of the sea.

Pollution Category C: Substances which would present a minor hazard to either marine resources or human health or cause minor harm to amenities or other legitimate uses of the sea.

Pollution Category D: Substances which would present a recognizable hazard to either ma-

48. The allocation of substances to pollution categories is based on hazard evaluation procedures established for that purpose by GESAMP in 1972. These procedures take into account the following potential effects and targets:[4]

1. damage to living resources;
2. hazards to human health;
3. reduction of amenities; and
4. interference with other uses of the sea.

49. The hazard evaluation procedures consist of a step-by-step evaluation summarized as follows:

1. Is the substance carried by ships or proposed for carriage by ships? (hazard profiles should not be produced unnecessarily)

2. Is the substance an oil (MARPOL 73/78, Annex I) or is it a noxious liquid product (MARPOL 73/78, Annex II), or is it carried as a so-called dangerous good in packages, containers, etc. (MARPOL 73/78, Annex III)?

3. Is the substance, or its reaction/degradation product(s) liable to be bioaccumulated or to taint? (Column A of GESAMP Hazard Profile)

4. How great is the risk posed to living aquatic organisms by operational or accidental discharges? (Column B of GESAMP Hazard Profile: Assessment of acute toxicity to marine organisms)

5. How great is a hazard posed to human health? (Column C of Hazard Profile: acute oral toxicity; Column D: dermal and inhalation toxicity; Remarks Column: Carcinogenic and other long-term health effects)

6. What impact would a spill have on the recreational use of a beach, amenity interests and aesthetics? (Column E: Highly/moderately/slightly objectionable).

50. The GESAMP hazard evaluation procedure and the criteria used for establishing hazard profiles in abbreviated form are shown in Table 1.

51. The GESAMP Hazard Profiles prepared on the basis of the above evaluation procedure were first used in 1972 for the categorization of substances in preparation for the MARPOL 73/78 Convention; that Convention in its Guidelines for the Categorization of Noxious Liquid

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4 These effects of substances when released to the sea were chosen in light of the definition of marine pollution developed by GESAMP (see paragraph 13 above).

Substances (MARPOL 73/78, Annex II, Appendix I) reflects the procedures used by GESAMP and by the International Conference on Marine Pollution in 1973. An explanatory table to the Guidelines was developed by the IMO Sub-Committee on Bulk Chemicals and approved by the Marine Environment Protection Committee (MEPC) of IMO as shown in Table 2.

52. Since 1973 the hazard evaluation procedure has basically not been changed. However, certain definitions were modified in the light of difficulties encountered throughout the years.

53. Every year IMO distributes a "Composite List of Hazard Profiles" established by GESAMP (e.g. BCH/Circ.25 of 31 October 1988) and this forms a basis for IMO to review the Pollution Categories (A, B, C and D) and the carriage requirements for noxious liquid substances.

*TABLE 1*

GESAMP HAZARD PROFILES (abbreviated legend):

*Column A - Bioaccumulation and Tainting*

| | |
|---|---|
| + | Bioaccumulated to significant extent and known to produce a hazard to aquatic life or human health |
| Z | Bioaccumulated with attendant risk to aquatic organisms or human health, however with short retention of the order of one week or less |
| T | Liable to produce tainting of seafood |
| O | No evidence to support one of the above ratings (+, Z, T) |

*Column B - Damage to living resources*

| *Ratings* | | *96 hr. $LC_{50}$* |
|---|---|---|
| 4 | Highly toxic | less than 1 mg $l^{-1}$ |
| 3 | Moderately toxic | 1–10 mg $l^{-1}$ |
| 2 | Slightly toxic | 10–100 mg $l^{-1}$ |
| 1 | Practically non-toxic | 100–1000 mg $l^{-1}$ |
| 0 | Non-hazardous | greater than 1000 mg $l^{-1}$ |
| D | Substance likely to blanket the sea-bed | |
| BOD | Substance with oxygen demand | |

*Column C - Hazard to human health, oral intake*

| *Ratings* | | $LD_{50}$ (laboratory mammal) |
|---|---|---|
| 4 | Highly hazardous | less than 5 mg $kg^{-1}$ |
| 3 | Moderately hazardous | 5–50 mg $kg^{-1}$ |
| 2 | Slightly hazardous | 50–500 mg $kg^{-1}$ |
| 1 | Practically non-hazardous | 500–5000 mg $kg^{-1}$ |
| 0 | Non-hazardous | greater than 5000 mg $kg^{-1}$ |

*Column D - Hazard to human health, skin contact and inhalation*

| | |
|---|---|
| II | Hazardous |
| I | Slightly hazardous |
| 0 | Non-hazardous |

*Column E - Reduction of amenities*

*Ratings*

| | |
|---|---|
| XXX | Highly objectionable because of persistency, smell or poisonous or irritant characteristics; as a result beaches liable to be closed; also used when there is clear evidence that the substance is a human carcinogen or that the substance has the potential to produce other serious specific long-term adverse health effects in humans. |
| XX | Moderately objectionable because of the above characteristics, but short-term effects leading only to temporary interference with use of beaches; also used when there is credible scientific evidence that the substance is an animal carcinogen but where there is no clear evidence to indicate that the material has caused cancer in humans or when there is evidence from laboratory studies that the substance could have the potential to produce other serious specific long-term adverse health effects |
| X | Slightly objectionable, non-interference with use of beaches |
| 0 | No problem |

*Other Symbols*

Ratings in brackets, ( ), indicate insufficient data available to the GESAMP experts on specific substances, hence extrapolation was required.

| | |
|---|---|
| N/A | Not applicable (e.g. if gases) |
| – | Indicates data was not available to the GESAMP Working Group |

*TABLE 2*

## EXPLANATORY TABLE FOR ALLOCATION OF POLLUTION CATEGORIES

| GESAMP Hazard Profile Columns (see Table 1) | | | | MARPOL 73/78, Annex II Pollution Categories |
|---|---|---|---|---|
| A | B | C | E | |
| + | - | - | - | Category A |
| - | 4 | - | - | |
| T | 3 | - | - | |
| Z | 3 | - | XXX | |
| T | - | - | - | Category B |
| Z | - | - | - | |
| - | 3 | - | - | |
| - | 2 | - | XXX* | |
| - | 2 | - | - | Category C |
| - | 1 | 4 | XX | |
| - | 1 | 3 | XX | |
| - | 1 | - | - | Category D |
| - | - | 4 | - | |
| - | - | 3 | X | |
| - | - | - | XXX | |
| - | - | - | XX | |
| - | D/BOD | - | - | |

* If the substance is non-volatile and insoluble (vapour presure 1mm Hg at 20°C and solubility 2g/100 ml at 20°) otherwise it may be rated as Category C.

*Note by the Secretariat*

It may be noted that the GESAMP Column D rating has been dispensed with because that rating is related to hazard to human health (skin contact and inhalation) which in the view of IMO has no direct bearing on aquatic pollution. GESAMP however continues to evaluate these effects due to their importance for assigning ratings under Column E (reduction of amenities).

### *(b) Prevention of accidental spillages*

54. MARPOL 73/78, Annex II, in addition to establishing provisions for the discharge to the sea of residues and tank washings from chemical tankers in accordance with the pollution categories mentioned above, also emphasizes the need that requirements should be established for the prevention of marine pollution by accidental spillages, and that these be included in the Code for the Construction and Equipment of Ships Carrying Dangerous Chemicals in Bulk (Bulk Chemical Code). At the time of the adoption of MARPOL 73/78, the Bulk Chemical Code contained only provisions for the safety of life at sea (loading and handling of chemicals on board) but did not include requirements for the prevention of marine pollution.

55. In 1985 the Marine Environment Protection Committee (MEPC) of IMO adopted criteria for the allocation of ship types for the transport of chemicals from the marine pollution point of view, which are based on GESAMP ratings for Columns A, B, C and E of the hazard profiles. In this respect MEPC did not take account of all "T" (tainting) ratings assigned by GESAMP under column A of its hazard profiles, but only of those substances with strong tainting properties as identified by the IMO Sub-Committee on Bulk Chemicals. An interpretation of the various requirements is shown in Table 3.

56. The basic philosophy is one of ship types related to the hazards of the products carried, i.e. the more hazardous the products carried, the stricter are the containment standards. To achieve this, ships subject to the Bulk Chemical Codes must be of one of three types, as follows:

*Type 1* ships are designed to transport products with very severe environmental and safety hazards which require maximum preventive measures to preclude an escape of cargo.

*Type 2* ships are designed to carry products with appreciably severe environmental and safety hazards which require significant preventive measures to prevent cargo escaping.

*Type 3* ships are designed to carry products with sufficiently severe environmental and safety hazards which require a moderate degree of containment to increase survival capability in a damaged condition.

*TABLE 3*

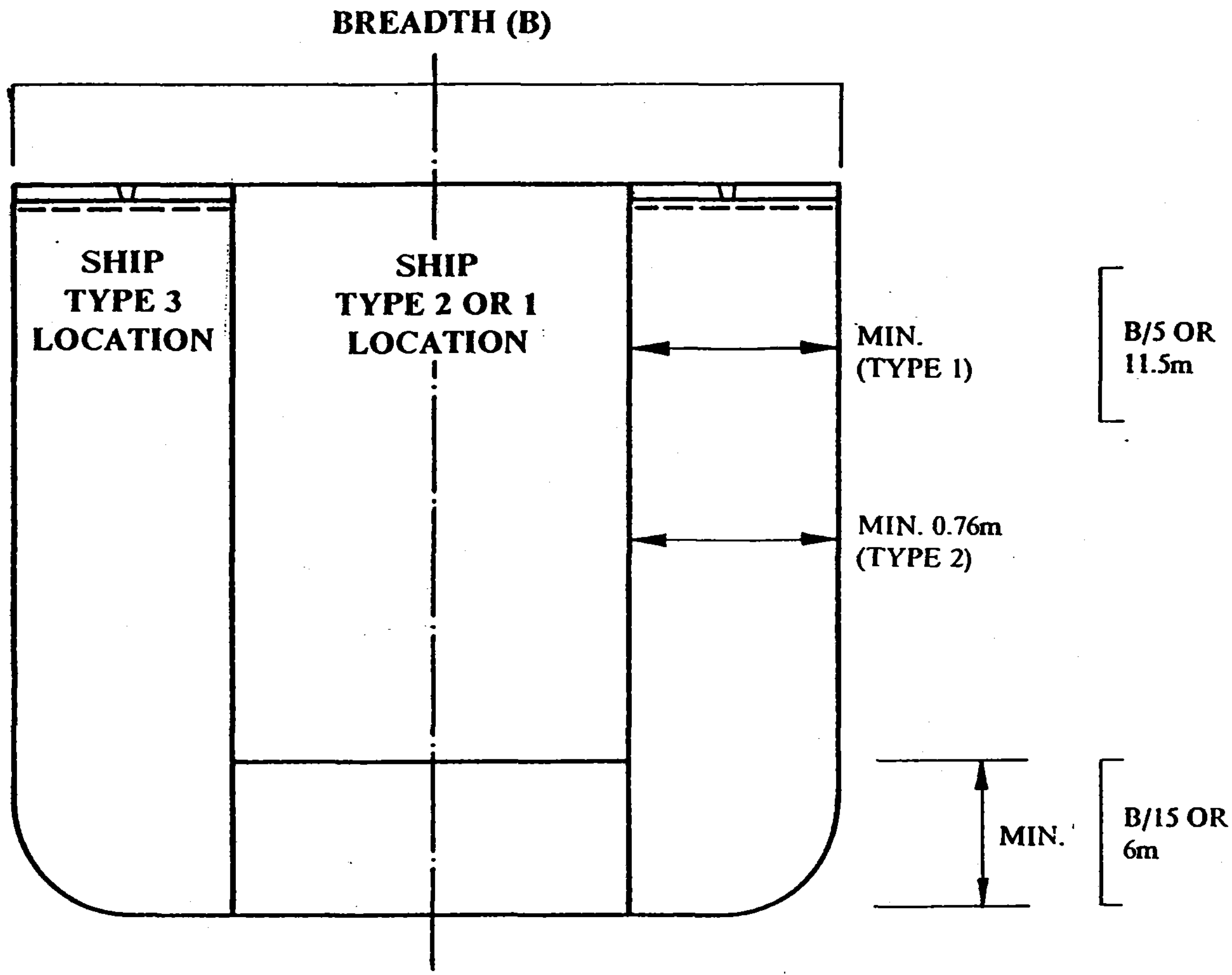
## RATIONALE FOR THE ALLOCATION OF SHIP TYPE FROM THE MARINE POLLUTION POINT OF VIEW

| Ship type | GESAMP Hazard Profiles | | |
|---|---|---|---|
| | Column A Bioaccumulation and tainting | Column B Damage to living resources | Column E Reduction of amenities |
| 1 | + | 4 | |
| | + | | XXX |
| | T* | 4 | |
| 2 | + | | |
| | Z | 4 | |
| | Z | 3 | |
| | Z | | XXX |
| | T* | | |
| | 0 | 4 | |
| | 0 | 3 | XXX |
| 3 | All other substances falling under pollution categories A, B and C. | | |

T*: Substances with strong tainting properties as identified by the Sub-Committee on Bulk Chemicals at its thirteenth session. These are as follows:

- Camphor oil
- Creosote (wood tar)
- Creosols (mixed isomers)
- Carbolic oil
- Dichloroethyl ether
- Dichlorophenols
- Ethyl acrylate
- Naphthalene
- Alpha-methyl naphthalene
- Naphthenic acids

Containment standards that have been designed in order to provide protection from damage in the case of collision or stranding, by locating the tanks at specified minimum distances in board from the ship's shell plating, are summarized in the following figure:




In addition to the tank location requirements, damage survival requirements have been specified for the three chemical tanker types. Ships should be capable of surviving damages as follows:

1. *Type I ships* - this ship should be designed in such a way that it would sustain damage anywhere in its length;

2. *Type 2 ships* - a ship of *more than 150 m* in length should be assumed to sustain damage anywhere in its lenght; a ship of *150 m in length or less* should be assumed to sustain damage anywhere in its lenght except involving either of the bulkheads bounding a machinery space;

3. *Type 3 ships* - a ship of *more than 224 m* in length should be assumed to sustain damage anywhere in its length; a ship of *125 m in length or more but not exceeding 225 m* in length should be assumed to sustain damage anywhere in its length except involving either of the bulkheads bounding a machinery space located aft; and a ship *below 125 m* in length should be assumed to sustain damage anywhere in its length except involving damage to the machinery space when located aft. However, the ability to survive the flooding of the machinery space should be considered by the Administration.

### *(c) Principles of Annexes I and II to MARPOL 73/78*

57. As mentioned above, the definition of "oil" as used in MARPOL 73/78 is very general and does not include any criteria or properties normally used for describing the hazardous characteristics of a substance (e.g. toxic, persistent, bioaccumulative, carcinogenic, corrosive, flammable, explosive, etc.). The list of oils included in MARPOL 73/78 merely provides examples of the wide range of materials considered as "oils".

58. The principle of Annex I may be briefly stated as: the retention of oil on board by means of separating the water-insoluble substance of a density of less than that of sea water by means of gravity separation; and residues of oil and oil products to be discharged into shore reception facilities.

59. The principle of Annex II may be briefly stated as: the discharge of residues to shore reception facilities, and the dilution of remaining tank residues of noxious liquid substances to harmless concentrations prior to be discharged to the sea.

60. In comparing the substances listed in Annex I with those listed in Annex II of MARPOL 73/78, it becomes evident that all substances covered by Annex I consist of mixtures of individual chemical compounds and that these chemical compounds are also listed as Annex II substances. The distinction between Annexes I and II is to an extent arbitrary, but was intended to be pragmatic.

61. In spite of the above different principles., the Convention provides, for both oils and chemicals, a strict control of emissions of tank residues and tank washings, taking into account the vulnerability of coastal areas (no discharges of "dirty" ballast within 12 miles from the coast) and of certain enclosed or semi-enclosed sea areas (so-called "special areas").

62. The introduction of the above control measures and the continuing review of hazard profiles carried out by GESAMP as well as the evaluation by GESAMP of new products proposed for carriage by ships has again resulted in far-reaching implications for the shipping industry; from

the introduction of "efficient stripping devices" for reducing residues in tanks and pipes to changes of contracts and trade patterns to take account of the availability of the respective ship types required for the carriage of the more hazardous substances. In many ports, as for oil residues, reception facilities for discharges of residues from chemicals have had to be built.

### 3. *Prevention of pollution by harmful substances carried in packaged form*

63. The regulations of MARPOL 73/78, Annex III, include provisions on packaging, marking and labelling, stowage, quantity limitations, and notification for preventing or minimizing pollution of the marine environment by harmful substances carried in packaged form.

64. For the purpose of future implementation of these requirements criteria have been developed for the identification of goods shipped in packages as "marine pollutants". These criteria are again based on the hazard profiles established by GESAMP. A rationale listing the various criteria is shown in the following table (Table 4).

*TABLE 4*

PACKAGED AND CONTAINERIZED SUBSTANCES
IDENTIFIED AS "MARINE POLLUTANTS"

| ANNEX III - MARPOL 73/78 | | | |
|---|---|---|---|
| GESAMP Hazard Profiles columns (see table 1) | | | |
| A | B | C | E |
| +<br><br>T<br>Z | 4 | | |

### *4. Prevention of pollution by sewage from ships*

65. MARPOL 73/78, Annex IV, requires that sewage that has not been comminuted and/or disinfected must be discharged more than 12 miles from the nearest land and that comminuted and/or disinfected sewage may only be discharged at a distance of more than four miles from the nearest land. Within four miles from land the discharge of sewage from ships is prohibited.

66. The Government of each Party to the Convention undertakes to ensure the provision of facilities at ports and terminals for the reception of sewage, without causing undue delay to ships.

67. In order to assist Parties in implementing MARPOL 73/78, Annex IV, Recommendation on International Effluent Standards and Guidelines on Performance Tests for Sewage Treatment Plants have been developed by IMO, as well as Guidelines on the Provision of Adequate Reception Facilities in Ports.

### *5. Prevention of pollution by garbage from ships*

68. MARPOL 73/78, Annex V, deals with the prevention of pollution by the discharge of garbage from ships to the sea, which includes all kinds of victual, domestic and operational waste generated during the normal operation of ships. It applies to all kinds of ships.

69. One of the most important provisions in the Annex is the ban on the discharge into the sea of all plastics, such as synthetic ropes, synthetic fishing nets and plastic garbage bags.

70. Some other forms of garbage may be dumped under strictly controlled conditions. Dunnage, lining and packing materials can only be disposed of at sea more than 25 miles from land. Food wastes and all other garbage (including paper products, rags, glass, metal, bottles and crockery) cannot be discharged within 12 miles of land, unless it has first been passed through a comminuter or grinder. Even so, the minimum distance from land when discharge is permitted is set at three miles.

71. Even stricter controls apply in what are termed "Special Areas". These are mainly semi-enclosed sea areas, such as the Mediterranean, Baltic and Black Seas, together with some areas in the Middle East which are particularly at risk from pollution. Here the discharge of all forms of garbage except food wastes is completely banned, and even food wastes cannot be disposed of into the sea within 12 miles of land.

72. To enable ships to get rid of wastes, Contracting Parties to the Convention are obliged to provide facilities in ports for the reception of garbage.

73. Recognizing that direct enforcement of Annex V regulations, particularly at sea, is dif-

ures, including the reporting of inadequate reception facilities for garbage in ports, and the keeping of outboard record books which should document *inter alia* the date, time, location, type and amount of garbage discharged to the sea or in a port reception facility. Other recommended measures include compliance incentive systems as well as voluntary measures addressing seafarers' unions, shipowners' associations and fishermens' organizations.

## B. THE CONVENTION ON THE PREVENTION OF MARINE POLLUTION BY DUMPING OF WASTE AND OTHER MATTER, 1972 (LONDON DUMPING CONVENTION, 1972)

### *1. Definition of dumping*

74. "Dumping" within the framework of the London Dumping Convention, the Oslo Convention and other dumping conventions and protocols, means any deliberate disposal at sea of wastes and other matter loaded on board vessels, aircraft, platforms or other man-made structures at sea, but does not include the placement at sea of matter for a purpose other than mere disposal. Nor does it include the disposal of wastes or other matter arising from offshore processing of sea-bed mineral resources.

75. Since the adoption of the London Dumping Convention in 1972 new disposal methods have been developed. These are:

- the incineration at sea of wastes and other matter; and
- the disposal of hazardous wastes into the sea-bed.

76. With regard to the incineration of wastes at sea, the Consultative Meeting of Contracting Parties to the London Dumping Convention agreed in 1977 that incineration at sea should be considered as a method of sea disposal. Accordingly, amendments to the Annexes to the London Dumping Convention had been adopted in 1978, including regulations for the control of incineration at sea. The status and future use of this technology is currently under review by the Consultative Meeting of Contracting Parties to the Convention.

77. With regard to the disposal of hazardous wastes into the sea-bed, there have been lengthy discussions within the London Dumping Convention as to whether this method (e.g. the shooting of torpedo-shaped canisters into sediments or the storage of containers filled with high-level radioactive wastes or other hazardous substances in holes drilled into the hard rock base underlying the sediments) would fall under the provisions of the Convention. In this respect no agreement could be reached by the Consultative Meeting; however, it was decided that, if future

mentally acceptable, the Consultative Meeting of the London Dumping Convention would be the appropriate forum to prepare the necessary regulations and guidelines for the implementation of such a disposal method.

## 2. *Basic provisions*

78. The London Dumping Convention (as well as other regional conventions established for the prevention of marine pollution by wastes disposed of at sea) is based on the approach that discharges have to be controlled on the basis of lists of substances, classified according to their potential to cause harm to the marine environment and to man.

79. The substances listed in Annex I to the Convention are prohibited to be dumped at sea, unless it can be demonstrated that they are present in wastes only as trace contaminants or are rapidly made harmless in the sea. Annex I (the so-called black list), therefore, calls for control procedures that are, in practice, very restrictive; this is because the substances (e.g. organohalogens, mercury and its compounds, cadmium and its compounds, persistent plastics, oils, high-level radioactive wastes) were considered to be harmful to the marine environment and to human health under the majority of circumstances. The possibility of an exemption to Annex I (i.e. if the substances were contained in wastes as "trace contaminants" or would be "rapidly rendered harmless in the sea by chemical, physical or biological processes") may be explored by a suite of biological tests designed to demonstrate that the dumping at sea of a particular waste would not cause any harm to the marine environment.

80. In 1979 the Contracting Parties to the London Dumping Convention agreed that the prohibition requirements for Annex I substances and wastes do not apply to their disposal by means of incineration at sea, except for high-level radioactive wastes and for materials produced for biological and chemical warfare. Amendments to the Annexes concerning incineration at sea, including a set of regulations for the control of incineration of wastes and other matter at sea, entered into force in 1979.

81. Annex II of the Convention contains a list of substances which require special care when dumped at sea, and accordingly need special authorization by national administrations before a dumping permit is issued. The "special care" provisions of Annex II are somewhat harder to describe because they rely on the application of special techniques for mitigating the effects of dumping, such as the use of specialized disposal methods and site selection criteria. To date, only a few special care techniques and special site criteria have been described, such as capping deposits of contaminated dredged material with clean sand, and site selection criteria for dumping at sea of low-level radioactive wastes. Some substances require special care only when they occur in "significant amounts"; this term has been given an interim definition based on the proportion of the substances in the whole waste. Once again, this corresponds to a form of control procedure.

arsenic, copper, lead, zinc, cyanides, fluorides and pesticides not covered by Annex I, as well as bulky wastes (e.g. shipwrecks, scrap metal) and low-level radioactive wastes.

82. The lists under these Annexes were prepared in the early 1970s and it was then felt that the Annexes should be continuously reviewed and updated in accordance with scientific and technical progress. However, since the entry into force of the Convention, there has been some reluctance to update the lists of Annexes as recommended by the scientific advisory groups established under the Convention. The discussions on amending the position of substances in the Annexes to the Convention automatically raised the question of what classification system for reviewing the list of substances and for allocating new substances to the Annexes should be used.

83. In the light of the difficulties encountered in updating the Annexes, the Consultative Meeting of Contracting Parties to the London Dumping Convention developed criteria for assigning substances to the Annexes, as given below.

### *(a) Allocation guidelines adopted in 1980*

84. As a general guideline it was agreed that substances and materials listed in Annex I should be those which are:

1. simultaneously toxic, persistent and bioaccumulative and therefore have a wide range of action; and

2. while essentially non-toxic, are persistent and float or remain suspended in the sea where they may interfere with legitimate uses of the sea either because of the quantities dumped at a single time or because of their accumulation over a long period of time.

85. The substances listed in Annex II should be those which exhibit one or more of the properties of toxicity, persistence or bioaccumulation, but which may be safely disposed of in the marine environment if special care is used in their disposal.

### *(b) Annexation guidelines adopted in 1982*

86. In 1982 this question was brought up again with a view to developing more precise classification standards. The provisions adopted by the Consultative Meeting in this respect are contained in "annexation guidelines" as follows:

A decision to include a substance in the Annexes should be preceded by a risk evaluation process having three components: these are the evaluation of hazard potential, the evaluation of environmental

With regard to the evaluation of hazard potential it is important that toxicity to man, domestic animals and marine mammals, carcinogenicity and mutagenicity, and ability to interfere with other legitimate uses of the sea, should be added to the original three factors of persistence, toxicity (to marine life) and bioaccumulation potential. Substances should be allocated to the Annexes if they possess any combination of these properties, if they may result in significant environmental exposure, and if they are, or are proposed to be, dumped at sea.

*Annex I*

87. Annex I substances are those which, as a result of being dumped, will or may contribute significantly to environmental exposure on a wide scale, extending far beyond the original location and time of disposal. They will also result in significant adverse environmental effects. Such substances will have in common a high degree of persistence coupled with at least *one* of the following properties:

- the ability to accumulate to levels significant in terms of toxicity to marine organisms, to domestic animals or to man;

- carcinogenic or mutagenic properties to domestic animals or man;

- the ability to cause a high degree of interference with fisheries, amenities or other legitimate uses of the sea.

*Annex II*

88. Annex II substances are those considered suitable for annexation which are not allocated to Annex I.

### 3. *Factors to be taken into account when considering the issue of dumping permits (Annex III)*

89. The implementation of Annexes I and II as well as the disposal at sea of wastes not listed in Annexes I or II is greatly facilitated by Annex III to the London Dumping Convention which includes all the factors to be considered in issuing permits for disposal of waste at sea. By requiring a balanced assessment of factors, including the waste characteristics, the conditions at the dumpsite and the availability of land-based alternatives, Annex III incorporates many of the key elements of a waste management plan and also requires some estimation of the assimilative capacity of a water body to receive waste without unacceptable impact.

90. Annex III to the Convention (and more clearly the guidelines adopted thereto for the uniform interpretation of the requirements set out in Annex III) emphasizes the need for good waste management practices to be taken into account, as well as environmental impact assessments examining the actual and potential effects of disposal at sea compared with disposal in other components of the environment.

91. Waste management, as it is currently understood, incorporates a hierarchical approach. Its highest tier gives preference to those practices and technologies which produce no waste or which can initiate progressive waste reduction programmes. The second tier encourages recycling or re-use of wastes so that requirements for treatment and disposal are minimal. The third tier places emphasis on the treatment of wastes to reduce their potential for environmental impact. At the lower end of the hierarchy are waste destruction (e.g. incineration) and last of all, disposal, either on land or at sea.

92. Even when the above mentioned hierarchy of waste management is rigorously applied, it is inevitable that there will still be some wastes which cannot be avoided or used in a productive manner. Unless there is a reasonable prospect of a use being developed in the near future, wastes should not usually be stored. Under these circumstances, it is necessary to study the alternative options and to select the preferred method of diposal.

### *4. Review of the operational procedures of the Convention*

93. The Scientific Group on Dumping identified a number of reasons why a review of the Annexes might be of benefit for the effective implementation of the London Dumping Convention. These are as follows:

1. to clarify the control requirements set out in the London Dumping Convention for the benefit of regulatory authorities;

2. to remove, as far as possible, non-definitive terminology;

3. to provide a clearer distinction between substances and wastes;

4. to clarify the relationships between the requirements of Annexes I and II on the one hand, and Annex III on the other;

5. to more clearly reflect good waste management principles;

6. to recognize, and justify, the application of a holistic approach to comparative assessments of waste disposal alternatives; and

7. to increase confidence in the control and enforcement procedures of the Convention.

94. A fundamental review of the operational procedures of the Convention is currently being carried out with the goal of eliminating inconsistencies and ambiguities from the existing procedures, overcoming difficulties caused by terminologies, and generally improving the regulation of dumping at sea within a holistic waste management context.

### 5. *Environmental impact assessments*

95. The environmental impact assessment (EIA) required by Annex III to the Convention should examine the actual and potential effects of a disposal method on each component of the environment directly or indirectly influenced by the diposal operation. It should include physical, chemical and biological impacts whether these occur immediately or in the long term. In particular, the assessment should consider any transformations of the waste, such as degradation or complexation, and the bioavailability, toxicity and mobility of the original material and its transformation products.

96. A comprehensive EIA will examine the economic and social effects of the sea disposal option, including the cost of monitoring or the loss of public amenities. In some cases, it may be necessary to take into account the permanency of the impact, for example the permanent closure of a land area and the effect of this on other potential uses.

### 6. *The capacity of the oceans to receive wastes*

97. The concept that the sea may receive waste materials without resulting in effects detrimental to marine life and to human health is in fact embodied in the requirements of the MARPOL 73/78 Convention, as well as in the London Dumping Convention.

According to GESAMP, environmental (i.e. assimilative) capacity is:

> "a property of the marine environment, ... defined as its ability to accomodate a particular activity or rate of activity (e.g. volume of waste discharge per unit time ...) without unacceptable impact."

GESAMP goes on to characterize the environmental or assimilative-capacity-based waste management strategies as:

> "(aimed) ... at selecting the disposal option which involves the least collective impact in terms of human health detriment, disturbance and/or damage to the natural environment and associated social and economic penalties."

98. However, in practice, the above concept does not solve the difficulties related to defining "harmfulness" and "acceptability", because these terms cannot be defined only on the basis of

technical and scientific factors. While knowledge concerning the responses of marine organisms to individual contaminants and wastes is essential, it is not always possible to decide whether a particular response is harmful or significant. For example, many biologists believe that transitory exposures which affect individual organisms, but which do not affect populations or communities, will not be significant in the long term. But although a population of seals, for example, may not be threatened if one seal becomes entangled in a fishing net, society may still be concerned about this situation and take steps to prevent its recurrence. Thus the preservation of marine environmental quality suitable for the long-term preservation of a healthy flora and fauna must also address societal standards on the level of acceptable risk and quality.

99. Different views have been expressed by Contracting Parties to the London Dumping Convention concerning the use of the sea as a receptacle of waste materials. Some Contracting Parties believe that the limitations of knowledge on oceanic processes, and of contaminant assessment procedures, are such that it is not possible to predict the consequences of waste disposal in the marine environment accurately. In this regard, particular concern is expressed about the long-term effects of persistent substances contained in wastes. For this reason, the countries concerned advocate a very cautious approach to disposal at sea and promote the suspension of all such practices until it can be demonstrated that they are harmless to the marine environment.

100. Other Contracting Parties feel that the sea has a capacity to receive certain wastes without harm to the marine environment or to human health, and that existing procedures can be used to estimate the portion of this capacity which can be utilized within reasonable margins of safety. Such use would only be justified where the waste and its source had been subject to the provisions of waste management, and where sea disposal was clearly the option of least detriment to man and the environment.

### 7. *General agreement on waste disposal principles*

101. In spite of the different views of Contracting Parties to the London Dumping Convention expressed above, there is general agreement that:

1. international action is necessary to reduce the deleterious effects of waste on the environment; in this respect national waste management activities should be harmonized;

2. the current approach to waste management emphasizes the importance of waste reduction, recycling and re-use, so that quantities requiring disposal are minimized;

3. where disposal proves necessary, all available options should be assessed and compared to determine the disposal method which is the least detrimental to human health and the environment. It is implicit in this process that all compartments of the environment may contribute to solving waste disposal problems: this is known as a holistic

approach to waste management;

4. in order to determine the acceptable level of waste input to any sector of the environment, it is first necessary to estimate its assimilative capacity for the waste in question. This capacity cannot always be determined exclusively on the basis of scientific and technical factors. A process of optimization, which calls for exposures to be kept as low as reasonably achievable, taking into account social and economic factors, as well as technical factors, is an important step in defining assimilative capacity and, thus, the disposal option of least detriment; and

5. procedures for controlling waste disposal operations include the imposition of pre-operational controls such as environmental impact assessments and screening tests, and post-operational controls such as monitoring.

## C. CONVENTION FOR THE PREVENTION OF MARINE POLLUTION FROM LAND-BASED SOURCES, 1974 (PARIS CONVENTION)

### *1. Basic requirements*

102. The Paris Convention (and other instruments on land-based sources) uses basically the same approach as the London Dumping Conventions (and other dumping agreements, as well as, to a certain extent, MARPOL 73/78, Annex II, concerning operational discharges of noxious liquid substances from ships): emissions are controlled on the basis of lists, with different regimes for each list, one more stringent than the other. However, the conventions for the prevention of marine pollution from land-based sources (covering pollution of the sea through watercourses, pipelines and fixed structures) deal mainly with *indirect* emissions to the sea and accordingly it was recognized to be impossible to request the strict banning of discharges of all the substances compiled in a "black list". Instead, Contracting Parties undertake to eliminate the pollution of the respective areas by these substances. With regard to the substances listed in a second list, the "grey list", strict limitation of pollution of the sea from land-based sources by these substances is required.

103. The Paris Convention, as mentioned above, imposes an obligation on its Parties to adopt and implement, individually or jointly, programmes and measures necessary to eliminate pollution by substances on the black list. The programmes and measures in accordance with that Convention should cover:

- regulations and standards governing the quality of the environment;

maritime area; and

- the composition and use of substances and products.

104. The requested programmes and measures set out in the Convention give the individual Contracting parties the opportunity to fix standards governing the quality of the environment and standards for discharges. The majority of Parties to the Paris Convention favour a policy for elimination of pollution by imposing strict limits on discharges through the adoption of the so-called "uniform emission standards" (UES) approach, while others feel that the best way of preserving and improving the quality of the marine environment is by fixing "environmental quality objectives" (EQO). Supporters of the UES policy stress the preventive aspect of their approach, the fact that it can be based on "the best existing technical methods", and that it authorizes the establishment of standards on an international scale. Upholders of the EQO approach stress that the final objective of the Convention is the protection of the marine environment and of man as the consumer of seafood. They refer to provisions of the Convention which provide that Contracting Parties, with a view to preserving and enhancing the quality of the marine environment, shall endeavour to reduce existing pollution from land-based sources and to forestall any new pollution from such sources, taking into account provisions of the Convention referring to:

- the nature and quantities of the pollutants under consideration;

- the level of existing pollution;

- the quality and absorptive capacity of the receiving waters of the maritime area; and

- the need for an interpreted planning policy consistent with the requirements of environmental protection.

105. The relationship between the UES and EQO approaches has been the object of much debate within the framework of the Paris Convention. Significantly, the most recent programme of work of the Paris Convention includes the following statement: "The Convention allows for use of both the EQO and the UES systems, and there is growing recognition that each, or both, may be a valid method of eliminating or preventing pollution in particular circumstances."

106. In this connection, it should be pointed out that the UES approach cannot easily take into account pathways other than direct discharges to water. As the significance of other sources of contaminants entering the sea, in particular atmospheric deposition, becomes more evident, it is necessary to find some measure to assess the success or failure of the overall effort.

### 2. *Criteria for the selection of substances listed in the Paris Convention*

107. The Paris Convention contains a general statement on the criteria to be used for selecting the "black list" of substances. These criteria are toxicity, persistence and bioaccumulation. The Convention further states that these criteria are not necessarily of equal importance for a particular substance or group of substances, and that other factors, such as the location and quantities of the discharge, may be considered.

108. With regard to the substances listed in the "grey list" of the Paris Convention, it is stated that these were selected because they exhibit similar characteristics to the substances of the "black list", but seem to be less noxious or are more readily rendered harmless by natural processes.

## D. MONTREAL GUIDELINES ON THE PREVENTION OF MARINE POLLUTION FROM LAND-BASED SOURCES

109. On the basis of existing agreements on the prevention of marine pollution from land-based sources (the Paris Convention, the Helsinki Convention, the Athens Protocol, the Quito Protocol), a set of guidelines has been prepared by UNEP (Montreal Guidelines, 1985) with a view to assisting governments in the process of developing appropriate bilateral, regional and multilateral agreements and national legislation for the protection of the marine environment against pollution from land-based sources. The Montreal Guidelines are of a recommendatory nature. They are presented as a checklist of basic provisions, rather than as a model agreement, from which governments may select, adapt or elaborate, as appropriate, to meet the needs of specific regions.

### 1. *Comprehensive environmental management approach*

110. The Guidelines request States to undertake the development, as far as practicable, of a comprehensive environmental management approach to the prevention, reduction and control of pollution from land-based sources, taking into account relevant existing programmes and provisions (strategies for protecting, preserving and enhancing the quality of the marine environment). Such a comprehensive approach should include the identification of desired and attainable water-use objectives for the specific marine environment.

### 2. *Development of control strategies*

111. The Montreal Guidelines recommend that States should develop, adopt and implement

taking into account relevant international or national experiences. The Guidelines further recommend that national standards be adopted, as outlined below.

112. The Guidelines further point out that once the desired present, interim and long-term use and associated objectives for a water body have been determined, a number of control strategies may be employed to achieve those objectives, taking into account the different environmental capacities and other properties of these water bodies as well as differences in regional socio-economic conditions. The proposed principal strategies are based on:

- marine environmental quality standards;
- emission or source standards; and
- environmental planning standards.

113. All these strategies may employ criteria for water, air or sediment quality, as well as criteria related to specific marine life. Receiving environment quality standards are most prevalent for uses (e.g. swimming, direct harvesting of fish for human consumption) where sound scientific criteria exist to determine levels of harm. Usually relationships are drawn between the quality of a receiving environment and emissions of potential pollutants. If the quality needs to be upgraded because of expressed changes in the desired uses, additional controls may be placed on allowable emissions.

114. The various standards and objectives recommended by the Montreal Guidelines are described in the following paragraphs.

### *(a) Marine environmental quality standards*

115. Standards are derived from the quality of water, biota or sediments that must be maintained for a desirable level of quality and intended use. Several applications of these exist, as set out in the following paragraphs.

i. *Environmental quality objectives*

Technical assessments are conducted to determine the maximum allowable inputs that will ensure that the desired levels of environmental quality are met. The assessments consider the fates and effects of various contaminants, amounts of input, and the existing natural characteristics of the relevant marine ecosystem. Numerical standards are then established with which concentrations measured in the receiving environment may be compared. They are usually more restrictive than numbers derived from the technical assessment to allow for monitoring and enforcement capabilities and safety requirements. They may apply to water, sediment, fish or the tissues, health or community

Monitoring is required to detect changes and compliance with the standards. Changes in the items monitored, after adjustment for natural fluctuation, may signal a need further to reduce inputs and vary existing standards and controls.

ii. *No change above ambient*

Standards are set based on existing contaminant levels which must not be exceeded. This strategy is employed in situations where there is a desire to prevent any increase of present specific contaminant levels. It is an interim strategy to allow time to develop a solid scientific base on which more precise quality criteria for a specific use may be employed. It does not imply an existing state of the environment that is satisfactory, nor does it eliminate the need for its improvement.

iii. *Dilution*

Some contaminants discharged at the source are assumed to attenuate as they spread from the source. Dynamic characteristics of the receiving environment are employed to determine rate and level of dilution. Standards are derived from measured parameters taken at given distances from the discharging source. This strategy may accept short-term or local excess of a potential pollutant at the source of discharge. Application is generally associated with effluent that is considered biodegradable, and avoided where scientific evidence suggests that the effluent may accumulate in a given receiving environment.

iv. *Loading allocations*

This alternative imposes priority of control on the larger sources in consideration of the most cost-effective solution. Allowable discharges are measured in terms of the total allowable for an entire receiving environment regardless of specific site quality.

Application is suited to relatively self-contained receiving environments such as lagoons and semi-enclosed bodies of water. It allows flexibility of contaminant output, in that certain sources may emit more than adjacent ones, as long as loading limits applied to the receiving environment are not exceeded.

### *(b) Emission or source standards*

116. Standards that define levels of emission control are generally based on achievable technology and restrictions of contaminant emission or effluent by that technology. For reasons of economy of enforceability, source standards may reflect extrapolation of the criteria defining a desired use of the receiving environment. They differ from marine environmental quality standards

i. *Uniform regional emission standards*

These are considered to be a set of requirements that have universal application for emission sources. They are relatively easily regulated and enforced, and are usually applied in situations where there are existing pollution problems of a similar nature and there is urgent need to reduce such pollution. They do not give primary consideration to the nature of the sources, their economic base or the receiving environment.

ii. *Technology-based standards*

Standards of this nature would be reflected in requirements for discharges on a sectoral basis, thus providing a means of imposing similar costs across that sector and also allowing for relatively simple administration. Various methods have been used and include:

(a) *Best practicable technology*: Normally applied to an industrial sector, it reflects the application of demonstrable and sound treatment technology or spectrum of technologies, which is affordable by the sector.

(b) *Best available technology*: A uniform standard which reflects state-of-the-art technology in contaminant control. In general, the standards would reflect a more stringent level of controls as compared to best practicable technology. Application is generally for the control of emissions of the most noxious substances or to protect a sensitive environmental use.

(c) *As low as reasonable achievable (ALARA)*: Emission standards for a pollutant or range of pollutants are based on the reduction of releases to levels as low as reasonably achievable (ALARA). This concept implies optimization of protection measures taking into account technical, economical and social considerations. The application is mainly for pollutants without a threshold effect, such as radionuclides.

(d) *Zero discharge*: In a situation where a stringent protection of a sensitive marine environment is deemed appropriate, consideration may be given to the denial of any release of a contaminant to the environment. In general, zero discharge does not take account of the reality that at least some of the waste will be released to the air, soil or water after recycling and recovery have occurred.

## *(c) Environmental planning strategies*

117. This set of strategies allows an approach to the management and protection of particular environments which may involve restrictions on, or modifications of, activities and sites, as

zoning and other land-use restrictions. These strategies may include:

i. *Activity management*

Certain activities are deemed inappropriate or inconsistent with the value of uses of an environment. Activities themselves are regulated. From a preventive point of view, perhaps the most critical question that must be asked is whether there is a need for that activity and, if so, whether the need can be accommodated elsewhere or in a different manner:

(a) *Use designation*: Use of the receiving environment is the determining factor for pollution control standards, as well as the basis for regulations or guidelines affecting other activities. For example, if the desire is to maintain or develop a shellfish harvest (a socio-economic decision) then quality standards and uses are developed with this in mind. That application may result from a perceived threat to an established economic base or cultural value, or a conscious effort to change the existing use of a receiving environment.

(b) *Siting requirements*: Siting of any activity affecting the marine environment is subject to an analysis of the ecological characteristics of the receiving environment and the social and economic considerations associated with a particular pollution source and discharge area.

ii. *Regional planning*

Activities are assessed in a regional context where the region may be defined in socio-economic or ecological terms as, for example coastal zones or river basins:

(a) *Coastal zone planning*: This strategy employs planning capabilities to make best use of the coastal zone. It is not use- or source-specific but area-specific. Potential activities are assessed as components of a coastal zone. Planning is based on regional socio-economic and ecological considerations. Zoning and other land-use restrictions or modifications are major regulatory tools. Many States give the task to manage overall resource planning within a particular coastal area to regional planning authorities or councils.

(b) *Watershed or drainage basin planning*: This strategy acknowledges that many contaminants in the marine environment enter via watercourses. It does not necessarily account for inputs via the atmosphere, though air management areas have also been employed for control purposes. Through consideration of socio-economic and environmental factors utilizing a drainage system as the boundary limit, the desired uses determined. Pollution via watercourses is controlled through regulation of point and diffuse sources of such pollution within the given watershed.

(c) *Specially protected areas*: This strategy involves the identification of unique or pristine areas, rare or fragile ecosystems, critical habitats and the habitat of depleted, threatened or endangered species and other forms of marine life. Those areas to be protected or preserved from pollution, including that from land-based sources, are selected on the basis of a comprehensive evaluation of factors, including conservational, ecological, recreational, aesthetic and scientific values. States should notify an appropriate international organization of the establishment of, and any modification to, such areas, with a view to those data being included in an inventory of specially protected areas.

### *3. Factors influencing the choice of strategies and control instruments*

118. Whether strategies to control marine pollution from land-based sources fall under one or more of the above control strategies depends, to a large extent, on the socio-economic status of the State and of the infrastructure of that State to regulate and enforce pollutant abatement decisions. Table 5 outlines major factors influencing the choice of strategies and control instruments that a State must consider before embarking on a pollution control programme. Under an ideal socio-economic and ecological framework, marine environment quality standards acceptable for a desired level of use would be developed and enforced. However, if background scientific data are lacking, a monitoring capability is not available, or State infrastructure is such that effective regulation and enforcement is not feasible, alternative strategies to the use of marine environmental quality standards are required. For any viable strategy, flexibility is critical to the effective management of land-based sources of marine pollution.

## *TABLE 5*

## FACTORS INFLUENCING CHOICE OF STRATEGIES AND CONTROL INSTRUMENTS

*Economic*

General economic conditions and trends (deficit, balance of trade, inflation, etc.)

Availability of public financing

Availability of external funding

Unemployment

Economic viability of various sectors

"Polluter pays" principle

Availability of institutions and infrastructure

*Scientific/technical*

Availability/accessibility of scientific data, such as physical characteristics affecting flushing and mixing, natural nutrient cycles and geochemical cycles; biological processes and nature of communities

Availability/accessibility of technology, including: basic information on industry types; total effluent releases; specific data on waste stream constituents

Availability of expertise

Capability for monitoring

Existing engineering infrastructure

Experience with implementation of strategies or instruments elsewhere

Sensitivity of ecosystems to be affected

Climatic considerations

Current level of pollution of the receiving environment and identified trends, in municipal agricultural and industrial waste releases

*Social/cultural/political*

Infrastructure

Existing and proposed uses of the marine environment

Political realities

Social/cultural awareness of the population

Perception of social and cultural values

### *4. Substances recommended for inclusion in black and grey lists*

119. The Montreal Guidelines recommend that States should undertake to establish priorities for action, based on lists of substances from which pollution should be eliminated, and of substances from which substances should be strictly limited on the basis of their hazard potential. This in fact reflects the situation of the existing legal instruments on the prevention and control of marine pollution from land-based sources.

*Black List*

Substances may be included in this list:

1. because they are not readily degradable or rendered harmless by natural processes; and

2. because they may either:

   - give rise to dangerous accumulation of harmful material in the food chain, or

   - endanger the welfare of living organisms causing undesirable changes in the marine ecosystems, or

   - interfere seriously with the harvesting of seafood or with other legitimate uses of the sea; and

3. because it is considered that pollution by these substances necessitates urgent action.

*Grey List*

Substances may be included in this list because, although exhibiting similar characteristics to the substances in the black list and requiring strict control, they seem less noxious or are more readily rendered harmless by natural processes.

### *(a) Monitoring and data management*

120. In accordance with the Guidelines, States should endeavour to establish directly or, whenever necessary, through competent international organizations, complementary or joint programmes for monitoring. This includes storage and exchange of data, based, when possible, on compatible procedures and methods, taking into account relevant existing programmes at the bilateral, regional or global level in order to:

1. collect data on natural conditions in the region concerned as regards its physical, biological and chemical characteristics;

2. collect data on input of substances or energy that causes or potentially causes pollution emanating from land-based sources, including information on the distribution of sources and the quantities introduced to the region concerned;

3. assess systematically the levels of pollution along their coasts emanating from land-based sources and the fate and effects of pollution in the region concerned; and

4. evaluate the effectiveness of measures in meeting the environmental objectives for specific marine environments.

# IV. DISCUSSION OF APPLIED CONTROL STRATEGIES USED WITHIN THE FRAMEWORK OF INTERNATIONAL CONVENTIONS

121. Each of the pollution control approaches mentioned in the preceding section has its merits, but none of its own provides a fully workable guide for action. Conflicts between approaches will sometimes arise, and these will not be resolved by an insistence on the "correctness" of any particular one. Any move towards an integrated international approach that does justice to each national pattern of policy is dependent upon a clear understanding of the strengths and weaknesses of each particular approach, and the possible relationship to policy in different countries.

122. Should consideration of the assimilative capacity of the receiving waters, and of the quality objectives implied by present or intended future use, guide regulations (the environmental quality objectives approach), or should discharges be regulated on the basis of the technical capabilities of water pollution control technologies (the emission standards approach)? What other approaches have been developed and used?

## A. ENVIRONMENTAL QUALITY OBJECTIVES (EQO)

123. The application of environmental quality objectives requires extensive research on dose-effect relationships and comprehensive programmes for monitoring the conditions of the sea. Whereas such objectives take into account the input to the sea from all sources, including non-point sources, e.g. from the atmosphere, it would be difficult, if applied to large water bodies, to control and to regulate discharges from individual sources. When environmental levels approach quality standards, it becomes difficult to decide on how the remaining "receiving capacity" could be apportioned among proposed additional discharges to the respective marine area.

## B. EMISSION STANDARDS

124. Emission standards or source standards and the limit values associated with them may effectively control point sources, but they cannot deal with non-point source inputs such as agricultural run-off or atmospheric deposition. As these standards are based on "available technology" or "best available technology", there is the problem of determining these and of adjusting the re-

## C. THE "BEST PRACTICABLE ENVIRONMENTAL OPTION" APPROACH

125. The "best practicable environmental option" approach is based on the view that once wastes have been produced, the environmental costs of all disposal options must be assessed before one medium, such as the sea, rather than another, such as land, is chosen. Air, water and land should not be considered in separate compartments but must be considered together when solutions have to be found for the ultimate disposal of contaminants. The aim should be to choose the least vulnerable compartment, taking into account the pathway of the respective contaminants to man and their potential impact on human health.

126. The application of this approach requires the carrying out of a comprehensive comparative assessment of the potential impact on the environmental compartments, risks to human health, hazards (including accidents) associated with treatment, transport and disposal, the financial costs involved, and the possible exclusion of other future uses of discharge or disposal sites. The principles of this approach are embodied in Annex III to the London Dumping Convention.

## D. THE PRINCIPLE OF PREVENTION AND ANTICIPATORY ENVIRONMENTAL PROTECTION

127. The idea that prevention is better than cure and that emissions should be prevented, even without evidence of damage, has been propounded for many years and was stated in 1973 as a principle of an action programme on the environment adopted by the European Community as follows:

> The best environment policy consists in preventing the creation of pollution or nuisances at source, rather than subsequently trying to counteract their effects. To this end, technical progress must be conceived and devised so as to take into account the concern for protection of the environment and for the improvement of the quality of life at the lowest cost to the community. This environment policy can and must be compatible with economic and social development. This also applies to technical progress.

128. It will be noticed that this statement is qualified by a reference to costs to society and "technical progress". The need to balance limitations on emissions of substances with the costs of doing so, is also found in the words which guide the setting of emission standards, and certainly is also incorporated in the best practicable

129. Thus far, the prevention principle has rarely been used in practice to justify specific measures which go beyond what may be implied by the Uniform Emission Standards or the Environmental Quality Objectives approach. It appears rather as a general imperative without operational application.

130. A more stringent version of the prevention principle developed in the Federal Republic of Germany is called the anticipatory protection principle or precautionary principle ("Vorsorgeprinzip"). This principle was introduced at an international level at the first International Conference on the Protection of the North Sea in 1984, where it was recognized by other Parties as an important approach to a comprehensive protection of the North Sea ecosystem.

131. The precautionary principle advocates that in addition to taking steps for the control of environmental pollution , precautionary action is necessary to relieve the burdens on the environment resulting from the input of foreign substances. On the basis of the precautionary principle, rather stringent standards for releases of contaminants have been applied by the Federal Republic of Germany in certain areas despite the absence of incontrovertible evidence linking observed environmental deterioration to releases rather than to other factors, such as natural variation.

132. The precautionary principle is part of a policy of avoidance or at least reduction of the risks of damage to man and the environment from a given source or practice, even when the magnitude and/or frequency of such damage cannot yet be assessed.

133. Risk prevention, i.e. avoidance or reduction of risks to the environment, is considered to be a main element of a concept to progressively reduce the emission levels of all substances introduced by man into the atmosphere, water and soil.

134. The precautionary principle implies a change of emphasis towards indicators of potential effects (hazard and risk), irrespective of their origin, and away from strong reliance on scientific considerations of cause-effect relationships and on technological solutions. Anticipatory environmental protection, as it is evolving, also raises an essential issue: are the actions for the protection of the environment, taken on the basis of our present level of knowledge, sufficient, or do we have to assume that the future holds risks which are beyond our knowledge and therefore need to be catered for in our current pollution prevention strategies?

# V. THE EFFECTS OF SUBSTANCES ON THE ENVIRONMENT AND HUMAN HEALTH

135. Sooner or later all chemicals will find their way back to the natural environment either in their original composition or as conversion products. The following properties are of significance to the behaviour of substances in the environment:

- chemical properties, in particular reaction to air, water and other substances;

- physical properties, such as density and solubility in the various media and vapour pressure;

- biological properties, such as bioavailability, biodegradability, bioaccumulation, resorbability, persistence and depositability as well as effects of the substances on the inhabited environment; and

- properties endangering human health, such as acute and chronic toxic effects, carcinogenic, mutagenic, neurotoxic and embryotoxic effects.

136. The above properties play a more or less important role in the evaluation of the hazards of substances, as incorporated in the various international conventions on the prevention and control of marine pollution.

137. Any hazard evaluation or hazard assessment should not only take into account the intrinsic properties of substances (chemical, physical and biological properties) but also extrinsic conditions which govern the distribution and fate of a substance once released to the environment. On the basis of results of laboratory tests alone (e.g. aquatic toxicity tests, degradability tests, bioaccumulation tests, tainting tests, etc.) it is not possible to predict biological and toxicological effects and the harm the substance may pose to the marine environment once released to the sea. Other factors such as background levels, other fluxes to the sea, oceanographic and meteorological processes, as well as operational discharge and dumping techniques and other uses of the sea, have to be taken into account for assessing the hazards of substances to the marine environment.

138. Of the conventions discussed above, the MARPOL 73/78 Convention in three of its five Annexes regulates distinctive substances or wastes (i.e. oil (Annex I), sewage (Annex IV) and garbage (Annex V)) without providing any guidance for evaluating substances or wastes. With regard to chemicals transported by ships (in bulk or as packaged goods, Annexes II and III). MARPOL 73/78 refers to assessment procedures which include only intrinsic properties of the chemicals concerned: aquatic toxicity, bioaccumulative properties, tainting properties, and human

toxicity. Certain physical properties (solubility, density, volatility) are, besides human health factors, also taken into account for assessing the potential "reduction of amenities" (i.e. closure of beaches).

139. Likewise, the areas designated in MARPOL 73/78 as "special areas", i.e. areas with "zero discharge", have not been selected on the basis of scientific criteria, and it is only now, more than fifteen years after adoption of the Convention, that such criteria for the identification of particularly sensitive sea areas and special areas are being developed.

140. The current rather rigid and restrictive system used within MARPOL 73/78 for evaluating substances carried by ships makes it possible to establish hazard profiles within rather short periods for the many chemicals proposed every year for maritime transport, once the necessary data are available. It also provides from the administrative viewpoint a rather simple rationale by which the hazard profiles can be transferred into regulatory discharge and carriage requirements.

141. The London Dumping Convention's lists of substances have also been established on the chemical, physical and biological properties (i.e. the intrinsic properties of substances) summarized in the first part of this section. However, through the application of exemption rules and specific conditions ("trace contaminants", "rapidly rendered harmless", "significant amounts", "special care") and in particular by Annex III of the London Dumping Convention, Contracting Parties, when considering the disposal at sea of substances and wastes, have to take into account the results of an exposure assessment, in addition to the effects' assessment (based on the properties of substances) which is implicitly built into the list of substances. The London Dumping Convention further requests comparative studies between land and sea disposal options (including their risks) as well as the carrying out of monitoring on the condition of disposal sites.

142. Although the combination of baseline studies, comprehensive hazard assessments, comparative disposal alternatives studies, and monitoring programmes which have been established within the London Dumping Convention would provide an ideal combination of steps towards an integrated marine environment protection and management approach, there still remains a number of inconsistencies and terminologies which have to be clarified. In light of the increasing acceptance by Contracting Parties of the precautionary principle, additional issues related to social and political aspects of marine pollution prevention have to be addressed.

# VI. EFFECTIVENESS OF MEASURES TAKEN FOR THE CONTROL AND PREVENTION OF MARINE POLLUTION

## A. MARPOL 73/78

### *1. Oil*

143. It is difficult to assess precisely how effective the requirements of MARPOL 73/78 and of the 1954 Oil Pollution Conventions have been in reducing marine pollution by oil. However, there is evidence indicating that oil pollution is now far less serious than it had been a decade ago.

144. The United States National Academy of Sciences (NAS) in 1973 made its first estimate of the amount of oil entering the sea, using data relating to 1971. It showed that about 2 million tonnes of oil entered the sea that year as a result of maritime transportation. For 1980 the amount estimated by NAS was 1.5 million tonnes per year, of which 400,000 tonnes were the result of maritime accidents. This indicates that there has been a considerable improvement, especially since the amount of oil carried by sea has increased in the period between the two estimates.

145. The improvement is even more marked if one considers what might have happened if no action had been taken to prevent oil pollution from ships. It has been estimated that without the application of the regulatory measures as much as 8 to 10 million tonnes of oil would enter the sea each year as a result of pumping oil-contaminated tank-cleaning and ballast water straight into the sea. In addition there would be the large amounts of oil spilled at sea due to incidents.

146. There are many reasons for this improvement. One is the greater use of load-on-top procedures and another is the application of crude oil washing on tankers. These techniques, coupled with other measures adopted by IMO, have resulted in the great reduction of the amounts of oil entering the sea due to routine tanker operations, such as tank cleaning.

147. Similarly, the increased availability of reception facilities in ports, and improved cleaning techniques, have led to a substantial reduction in the amount of oil emptied into the sea by ships going into dry dock (in the earlier NAS study it was estimated that in 1971 one-half of tankers arrived for drydocking without tank washing residues).

148. The amounts of oil entering the sea due to maritime accidents have also fallen sharply in recent years, due to the development by IMO of improved safety standards, navigational aids, training and watchkeeping, traffic separation schemes, etc..

149. For the last few years, the IMO Steering Group on Casualty Statistics has been preparing and analysing statistics on casualties involving tankers, chemical carriers and combination carriers (ships which are designed to carry oil and/or bulk cargoes such as ores). The statistics go back to 1971 and show that since 1971 the serious casualty rate per hundred tankers at risk has averaged 2.29 per year. In very recent years the rate has been well below average and since 1986 has been down to 1.83.

150. Statistics from other sources support the view that the situation is improving. The International Tanker Owners' Pollution Federation (ITOPF) reported in 1986 that the average incidence of oil spills in the 1980s was 70 per cent lower than in the previous decade. There were 26 spills a year in the 1970s - and only 8 in the 1980s.

151. There is reason to hope that this improvement can be maintained in the years to come as more stringent controls become internationally mandatory and their implementation becomes more effective.

## *2. Chemicals*

152. The requirements of the MARPOL Convention related to the control of marine pollution by noxious liquid substances carried in bulk (MARPOL 73/78, Annex II) entered into force in 1987. Since that time discharges of chemical residues and tank washings to the marine environment have considerably decreased. In light of the requirements of MARPOL 73/78 to limit discharges of chemical residues to the sea (as well as the aim of the industry to recover as much as possible of the products due to economic reasons) efficient tank stripping systems (and line cleaning systems) have been developed. It might also be pointed out that there have been no accidents involving chemicals carried in bulk by ships which have led to major impacts on the marine environment.

153. The introduction of these measures and the continuing review of hazard profiles carried out by GESAMP as well as the evaluation by GESAMP of new products proposed for carriage by ships has again resulted in far-reaching implications for the shipping industry: from the introduction of "efficient stripping devices" for reducing residues in tanks and pipes to change of contracts and trade patterns, to take account of the availability of the respective ship types required for the carriage of the more hazardous substances. In many ports, as for oil residues, reception facilities for discharges of residues from chemicals have had to be built.

## B. DISPOSAL OF WASTES AT SEA (SEA DUMPING)

154. Since the entry into force of the London Dumping Convention in 1975, a notification procedure has been established by which Contracting Parties submit information on their dumping and related activities (e.g. compliance monitoring, environmental impact assessments) to the Secretariat. On the basis of the material received by the Secretariat, attempts have been made to prepare trend analyses, with a view to finding out whether disposal at sea activities have decreased or increased since the London Dumping Convention entered into force.

155. With regard to the disposal at sea of industrial wastes, the largest amount dumped at sea was about 17 million tonnes in 1979, and the smallest amount was 6 million tonnes in 1984. The amounts of sewage sludge dumped at sea were between 1977 and 1982, 15 millions tonnes per year; a small gradual decrease has been observed after 1982.

156. Dredged material is dumped annually in amounts between 160 to 260 million tonnes; however, there are some fluctuations due to variations in the maintenance of shipping lanes and harbours.

157. With regard to the incineration of wastes at sea, the annual amounts burned at sea have increased after the introduction of this disposal method in 1969 from 10,000 tonnes to approximately 100,000 tonnes in 1973. Since that time the amounts have been rather constant, between 85,000 and 100,000 tonnes per year.

158. Histograms reflecting the information made available to the IMO Secretariat are shown annexed hereto. During the period of 1976 to1980 amounts of wastes annually dumped at sea are in general increasing. This probably does not reflect increasing dumping activities but indicates the growing number of countries which have joined the London Dumping Convention as Contracting Parties and made the respective information available.

159. In light of public concerns expressed on the input of industrial wastes to the North Sea, governments of countries bordering the North Sea have in 1987 agreed to phase out the dumping of hazardous wastes in that area. This decision has been based on the precautionary principle (anticipatory principle, Vorsorgeprinzip) adopted by some North Sea States and is strictly speaking not based on scientific evidence and cause/effect relationships, but the result of concerns that wastes dumped or incinerated in the North Sea may have contributed to the deterioration of its quality. It also reflects the advanced technological development of North Sea countries in having increasingly available possibilities and methods for reducing and avoiding the introduction of substances into the environment by adopting low-waste production and recycling processes.

## C. PREVENTION OF POLLUTION FROM LAND-BASED SOURCES

160. Measures have been adopted within regional conventions to limit or eliminate pollution caused by discharges to the sea from rivers, pipelines and man-made structures at sea. However, only very few areas are as yet covered by such regional instruments.

161. The contribution of contaminants to the sea through land-based discharges by far exceeds the amounts discharged from ships and dumped at sea. Direct discharges to coastal waters from the reception facilities required by MARPOL 73/78 for oily and chemical residues from tankers may have a more serious impact on the marine environment than the discharge from these tankers on the high seas. Likewise, the many uncontrolled discharges of hazardous effluents to rivers and coastal areas would, from a holistic environmental viewpoint, result in less harm to the marine environment if dumped or incinerated at sea under the control of the London Dumping Convention.

162. It is therefore unfortunate that no globally applicable legally binding instrument has yet been developed. The recommendations of the Montreal Guidelines could provide a good basis for establishing a global convention for the prevention of marine pollution from land-based sources. On the other side, the state and conditions of industrial development in the many regions of the world might be too different to cover these in one global instrument. Nevertheless, more efforts could be made to establish more regional conventions in this field.

# VII. CONCLUSIONS

163. The behaviour of each human being contributes to the nature and extent of environmental stress. In implementing the objectives of an environmental strategy, a high level of environmental awareness of the population concerned is crucial. It can be achieved by proper advice and information services and by establishing educational programmes. Research results and scientific reports as well as data bases should be made publicly available by the national agencies concerned. In many cases it is advisable to present scientific material in such a way that complicated scientific issues can be better understood by the layman.

164. In developing and implementing environmental strategies close co-operation with industry (manufacturers, commerce, agriculture, transport, etc.) is necessary. Representatives from industry know better than anyone else the possibilities for reducing and avoiding wastes and the introduction of substances into the environment. Industry should commit itself to act in a manner consistent with environmental prevention and control measures. This self-commitment could be supported by governments through economic incentives, such as tax incentives and investment aid.

165. Substances introduced into the environment spread across national borders and, if persistent, are able to spread worldwide. Therefore national measures alone are not sufficient to protect man and the environment. International co-operation is necessary in order to achieve, through international conventions, uniform measures for the prevention and control of pollution. In this connection it is the responsibility of the highly developed countries to assist the less developed ones in achieving common goals. Such co-operation is being developed within the international conventions introduced in this document.

166. It is not possible at this stage to introduce a single overall principle which would guide all actions to control and prevent the pollution of the marine environment. Many approaches have been introduced and all of them have their advantages and disadvantages. It does not seem reasonable to apply in the various regions only one concept, but to seek a comprehensive approach which includes proper hazard assessment, including the distribution and fate of substances in the marine environment, covers all sources of marine pollution (point sources and non-point sources), takes into account the best available technology, applies a tiered waste management strategy and aims to protect all compartments of the environment (air, land and sea) at the same level. In this connection, it is important to recall that the foremost objective of an environmental policy should be the protection of human life and health both now and in the future.

167. Before considering and deciding upon control measures and regulations, science has to answer a number of

introduced into the environment. Decisions would, however, eventually be made at a political level rather than by scientists. The effectiveness of such political decisions should be continuously controlled and examined, and, if necessary, improved on the basis of scientific findings and the development of technology. In this connection it is crucial that monitoring programmes be developed in order to observe the effectiveness of the measures taken and to verify how far these have been complied with.

168. In 1987 Report of the World Commission on Environment and Development as a central message expresses the need for sustainable development based on a new era of economic growth and the integration of environment and development, and rightly lays stress on the importance of the oceans to our survival: "sustainable development, if not survival itself, depends on significant advances in the management of the oceans. Considerable changes will be required in our institutions and policies and more resources will have to be committed to oceans management".

# VIII. LITERATURE

The text of this study does not include references to publications used in preparation of the text. The background material consists mainly of IMO documents, in particular with regard to matters related to the London Dumping Convention and MARPOL 73/78.

Other literature used in preparing this document is listed as follows:

1. IMO documents providing background material on matters related to MARPOL 73/78 and the London Dumping Convention.

2. The Oslo and Paris Commissions: The First Decade, 1984.

3. Mary Quing-nan: *Land-based marine pollution: international law development*, Graham and Trotman Ltd., London 1987.

4. Protection of the Marine Environment Against Pollution from Land-based Sources, Montreal Guidelines, 1985.

5. Hairvonen, H. and R. P. Coté: Control strategies for the protection of the marine environment, Marine Policy 10 (1), 1986.

6. UNEP: Final Report of the Ad Hoc Working Group of Experts on the Protection of the Marine Environment Against Pollution from land-based Sources, Third Meeting, Montreal, 11-19 April 1985, UNEP/W.G. 120/3, 1985.

7. E. von Weizsäcker: Towards an Integrated Approach; paper submitted to the International Conference on Environmental Protection of the North Sea, London, 1987.

8. The Federal Minister for the Environment, Nature Conservation and Nuclear Safety, Federal Republic of Germany: Guidelines on Anticipatory Environmental Protection, 1986.

9. GESAMP: The Evaluation of the Hazards of Harmful Substances Carried by Ships, GESAMP Rep. Stud. 17, 1983.

10. GESAMP: Environmental capacity. An approach to marine pollution prevention, GESAMP Rep. Stud. 30, 1986.

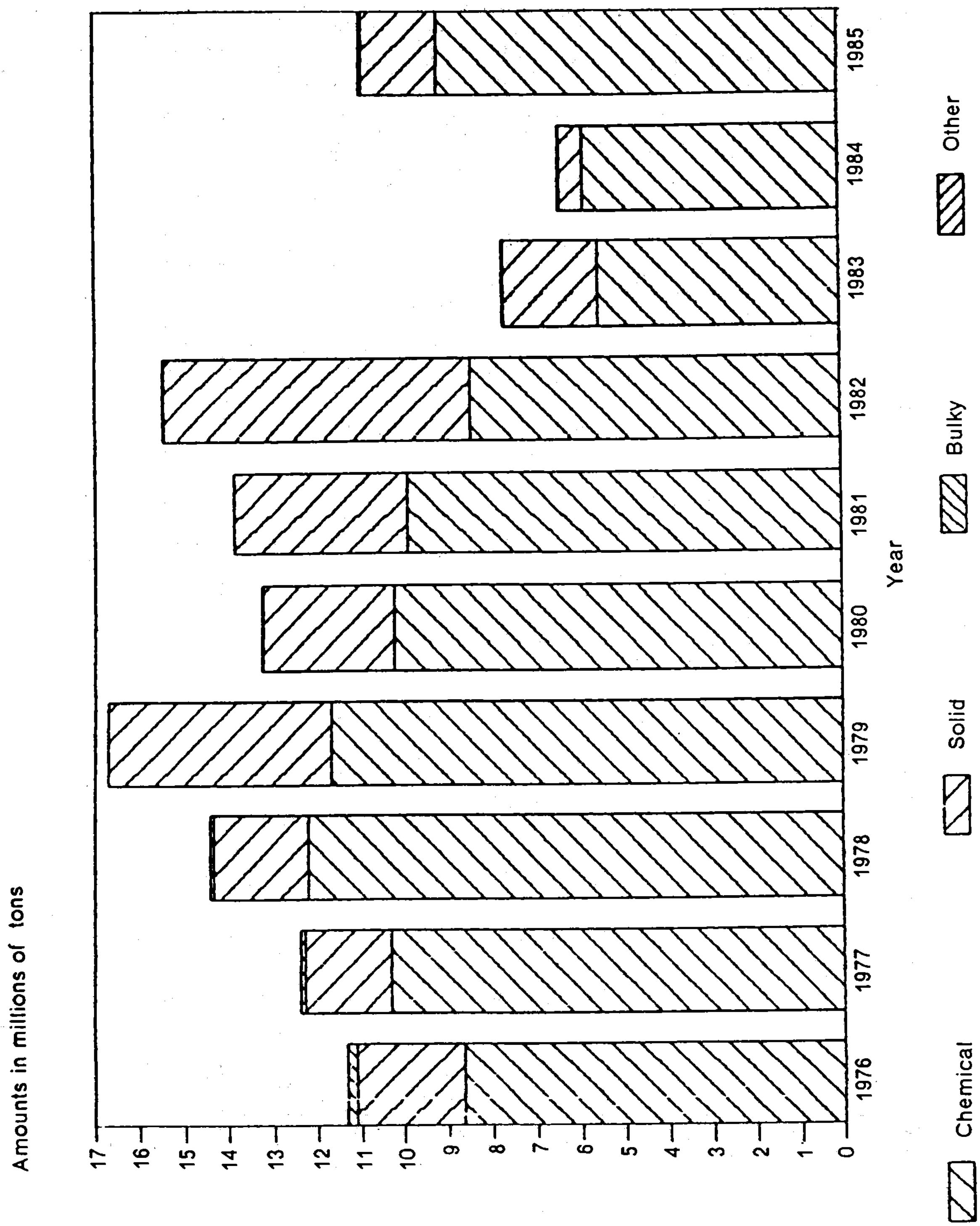
***

# APPENDIX

## AMOUNTS OF WASTES DUMPED AND INCINERATED AT SEA
## 1976-1985

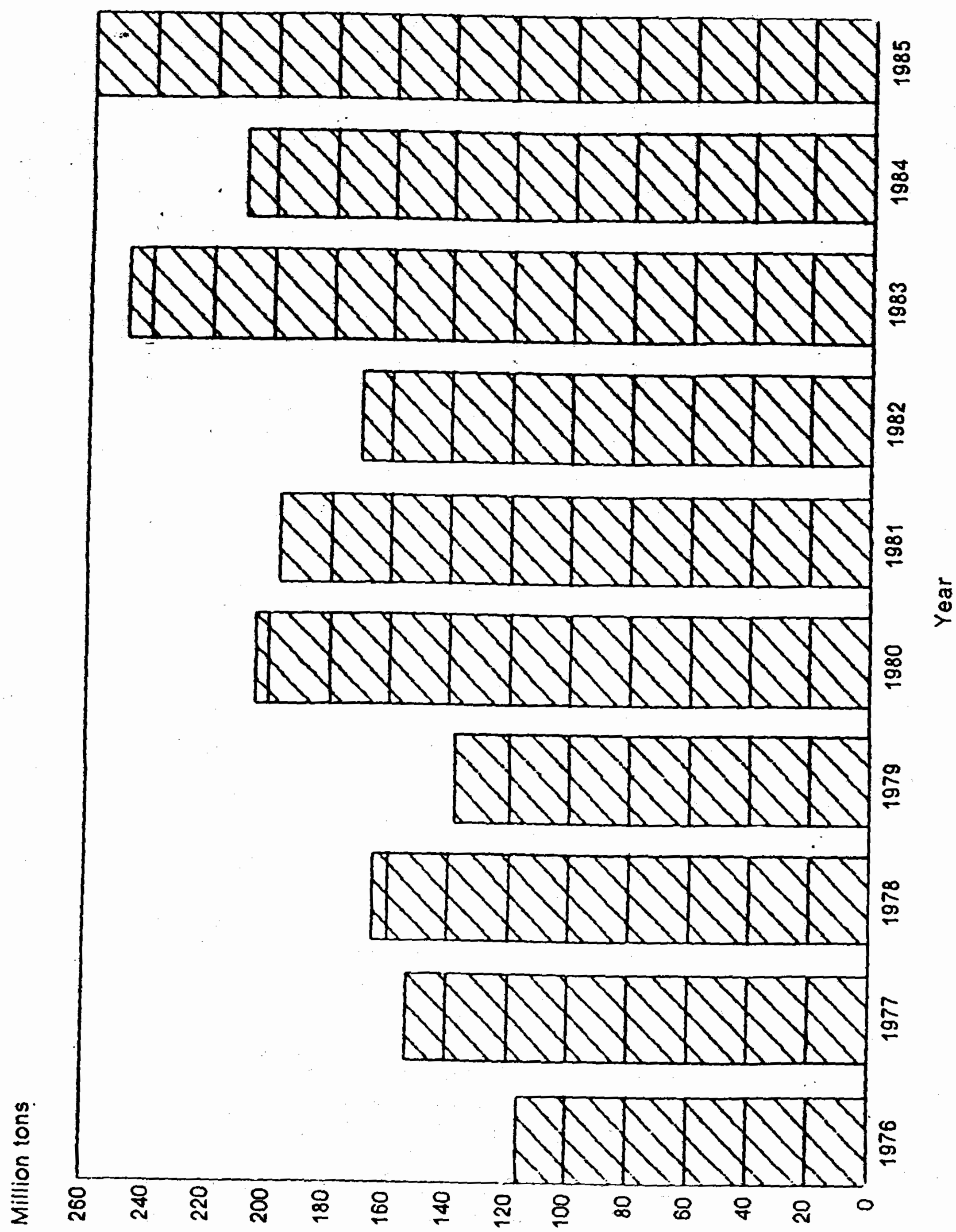
# SEA DISPOSAL OF INDUSTRIAL WASTE

During period from 1976 to 1985



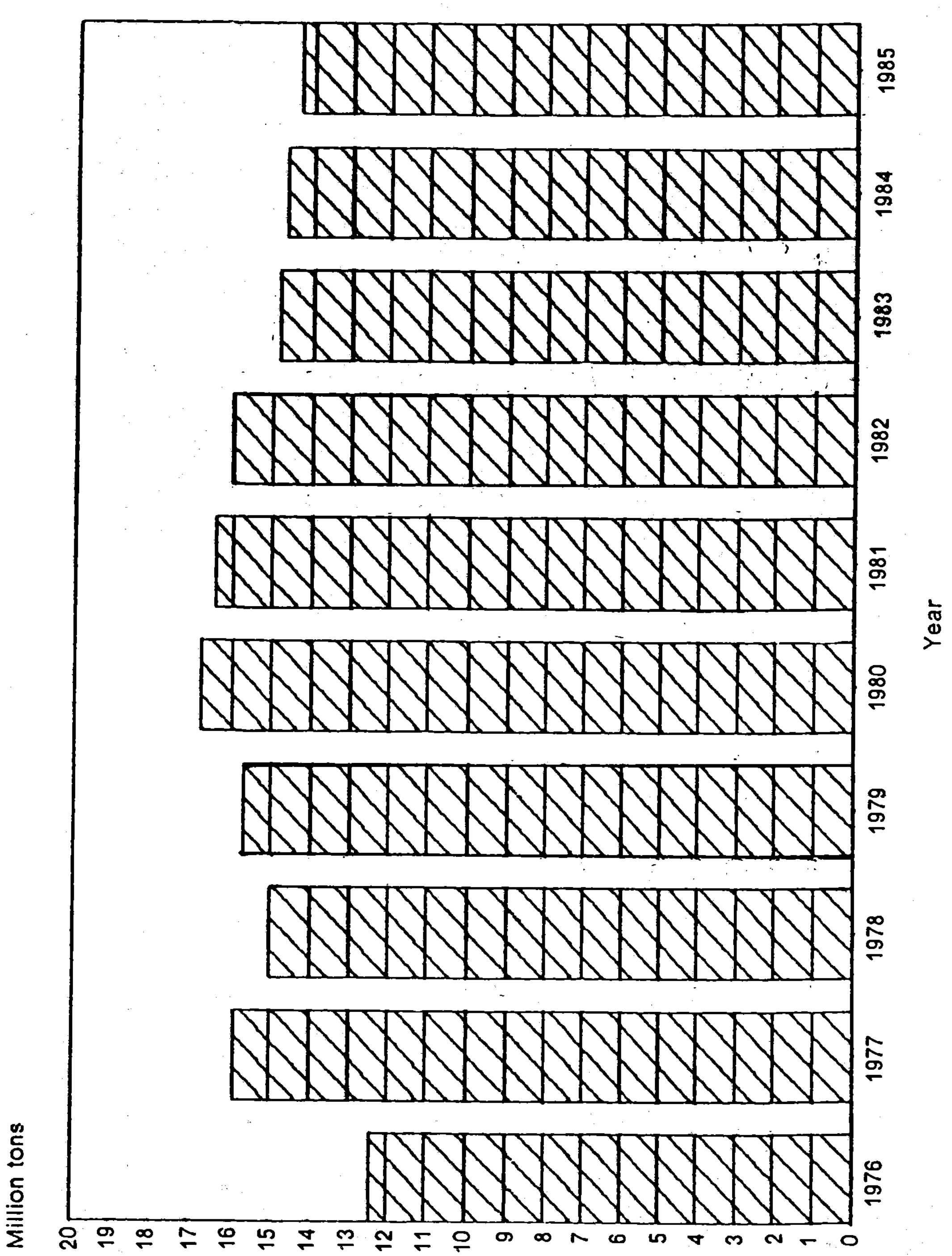
# DUMPING OF DREDGED MATERIAL

Total of received reports, 1976 - 1985



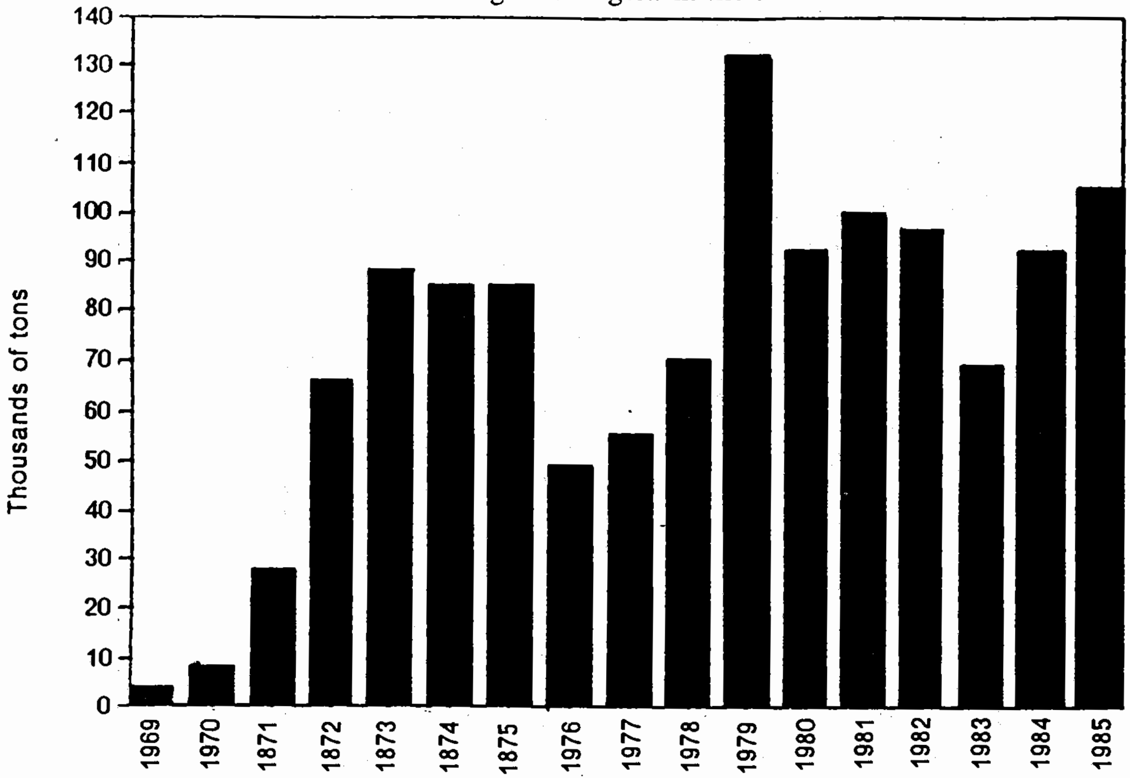
# DISPOSAL OF SEWAGE SLUDGE

Total of received reports, 1976 - 1985



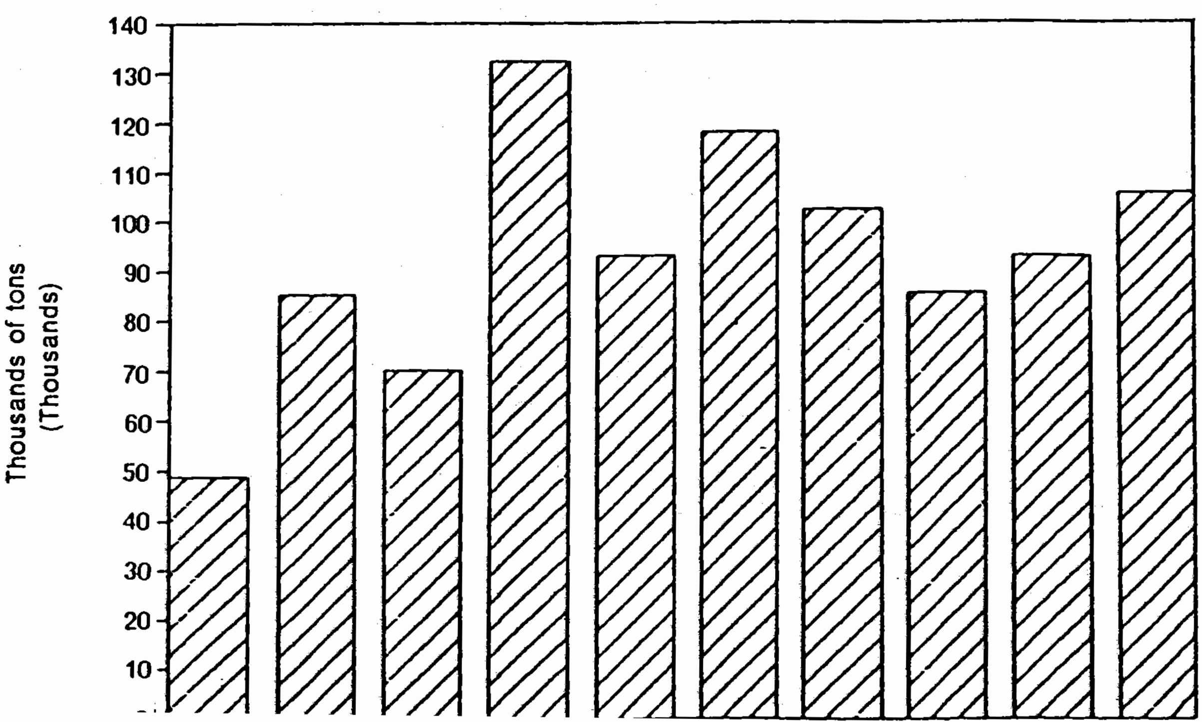
## INCINERATION OF INDUSTRIAL WASTES

Organohalogens in the North Sea




## INCINERATION OF INDUSTRIAL WASTES

Organohalogens



# ANNEX VIII

# RECOVERY OF DAMAGED ECOSYSTEMS

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INSTITUTET FÖR VATTEN-OCH LUFTVÅRDSFORSKNING
STOCKHOLM
SWEDEN

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# INTRODUCTION

1. An ecosystem that is affected by pollution may differ from an unaffected system in a number of ways:

- Species diversity is lower

- Food chains are shorter

- Energy transfer between trophic levels is less efficient.

2. The most simplistic definition of ecosystem recovery is the re-establishment of the unaffected ecosystem. There are, however, some obvious differences between the changes that occur as effects of pollution and those occuring during recovery, beside the direction of the change.

3. One is the time element. Damage frequently occurs much faster than recovery. Further, an ecosystem is mostly composed of a number of subsystems, e.g. pelagic, littoral and profundal, which react in that order both as pollution affects develop and as recovery progresses. This means that at the initial stage of pollution, the pelagic community may be severely damaged, the littoral moderately so, while the profundal may remain largely intact. Once the latter has been affected, however, the recovery process may take much longer to complete.

4. Another aspect is that, after the pollution episode, the ecosystem may not necessarily return to the same state as before. The system may have several stable positions and the pollution event may have displaced it from one to another. This could mean that, although species diversity increases, the food chain again grows longer and the energy transfer between trophic levels becomes more efficient, the dominating species being different ones.

5. In short, a recovery process may be quite different from the reversal of the pollution process, even if the end result is expected to be an ecosystem very similar to that at the onset of pollution.

# I. RECOVERY PROCESSES IN MARINE SYSTEMS FOLLOWING REDUCTION IN MUNICIPAL WASTEWATER DISCHARGES

## A. THE STOCKHOLM ARCHIPELAGO

6. Since the beginning of this century there has been a heavy increase of municipal wastewater discharge by the city of Stockholm to the inner archipelago. As a consequence, the phosphorus load in the water surrounding Stockholm has increased. The maximum content of phosphorus carried in by wastewater, 980 tons, was reached in 1968 (Cronholm *et al.*, 1978). During the period 1968-1973 all purification plants in Stockholm introduced biological and chemical purification (Brattberg, 1986). This caused a dramatic decrease in phosphorus releases to the waters. In 1986 the discharge from the plants contained 81 tons of phosphorus (Brattberg, 1987) and the phosphorus concentration in the surface water not far from Stockholm decreased from 170 to 40 $\mu g\ l^{-1}$(Brattberg, 1987).

7. The total nitrogen and phosphorus discharged from the plants in 1970 were in a ratio of 4.4:1 (Waern *et al.*, 1973), while the relative demand of nitrogen and phosphorus in most phytoplankton is 7:1 (Brattberg, 1987). Melin *et al.* (1973) showed that nitrogen was the limiting factor for primary production during 1970.

8. The unlimited supply of inorganic phosphorus lead to heavy blooming of blue-green algae, which caused decreased transparency of the water (Cronholm *et al.*, 1978). Periods of oxygen deficiency and subsequent hydrogen sulphide production occurred now and then in the centre of Stockholm before 1970 (Brattberg, 1987).

9. The reduced discharge of phosphorus in combination with a lesser increase in nitrogen discharges caused the nitrogen-phosphorus ratio to rise (Brattberg, 1986), so much so that in 1986 the average ratio in the water discharged by the plants reached 51:1 (Brattberg, 1987). Enrichment experiments with the natural phytoplankton population in 1973 and 1974 showed a transition from nitrogen to phosphorus as a limiting factor (Norin, 1977). Diminished nitrogen fixation and diminished occurrence of nitrogen fixing blue-green algae was reported by Brattberg (1977), probably due to the reduced phosphorus discharge and the subsequent increase in the nitrogen-phosphorus discharge ratio. Another effect of the reduced phosphorus discharge is that the chlorophyll (a) content of the water body has been halved and the transparency of the water has increased (Brattberg, 1987). The oxygen content of the water in the inner archipelago (close to Stockholm) is now slowly increasing and occurrence of hydrogen sulphide is rare (Brattberg, 1987).

10. The bottom fauna in the archipelago has recently been investigated (Brattberg *et al.*, 1986). It is difficult to draw conclusions from the results of the measurements since sampling started several years after the sewage treatment had begun. However, it seems likely that the bottom fauna community close to Stockholm contains fewer species than the community in the outer archipelago, indicating a more stable bottom fauna far from Stockholm. Salmon (*Salmo salar*) and salmo trout (*Salmo trutta*) were introduced in the centre of Stockholm in 1973, (Stockholms Fritidsförvaltning, 1987) and have thrived. Since salmon requires good water quality, its presence is an indication of the water condition in the Stockholm surroundings.

## B. THE RIVER THAMES ESTUARY

11. In the 18th century the Thames contained great amounts of salmon (*Salmo salar*). With the introduction and spreading of the water closet the water condition deteriorated due to the increasing outflow of untreated wastewater.

12. By the early 19th century major outbreaks of cholera caused by contaminated drinking water from the Thames occurred now and then (Elkington, 1977), and the smell of hydrogen sulphide was often annoying in the vicinity of the river. The sewerage system was built during the second half of the 19th century, and around 1890 experimental sewage treatment started (Elkington, 1977). In spite of that, the water quality did not improve significantly, mainly because of the expanding industry and the bomb damage to the sewage works during the Second World War.

13. By 1949 the condition of the Thames was again very bad, with around 20 to 30 miles of the waterway anaerobic and the smell of hydrogen sulphide felt during hot, dry weather spells (Water Pollution Research Laboratory, 1965). A special committee established to survey the river (Elkington, 1977) decided to enlarge and modernize the Beckton and Crossness outfalls. An investigation of the Thames in 1957 did not produce evidence of any fish at all from Richmond to Gravesend in the London area (Wheeler, 1970).

14. In 1964 and 1974, respectively, the two purification plants were fitted with the latest treatment equipment. Reduction of wastewater discharges to the Thames was followed by an increase in the oxygen content of the water (Andrews *et al.*, 1980). Subsequently, large numbers of whiting (*Merlangius merlangus*) occurred. Later, in the second half of the 1970's, the amount of smelt (*Osmerus eperlanus*) increased dramatically. More than 1,000 individuals were taken in a one hour sample at the West Thurrock power station in November 1979 (Andrews *et al.*, 1980) compared to less than 10 individuals each year between 1968 and 1973 (Wheeler, 1979 and Andrews *et al.*, 1980).

15. Smelt is a close relative to salmon and shares its requirements of good water quality. The recolonization of smelt is therefore an indication of the improvement of the Thames (Andrews *et al.*, 1980). Since 1964 the cumulative number of fish species has increased every year and by 1979 almost 100 species had been recorded (Andrews *et al.*, 1980). In 1983 the first salmon taken on rod and line since 1833 was caught in the Thames (Thames Water Authority, 1984). The recorded catches of salmon since then indicate a positive stock development (Thames Water Auth., 1985, 1986).

16. Studies of macro-invertebrates since the beginning of the 1970s show a shift from dominating tubificid worms to a multi-species community which indicates more stable conditions (Andrews *et al.*, 1980). Huddart *et al.*, (1971) examined more than 40,000 shrimps from the inner Thames in 1969. Among them only one berried female was discovered. This is in sharp contrast with the observation of thousands of gravid shrimps and prawns reported by Andrews *et al.*, (1980). Recolonization of algae was reported by Tittley (1972), whereas investigations in the early 1960s showed no or little evidence of macroscopic algal flora. The marine and brackish water algal flora now seen in the Thames is generally similar to that found in an unpolluted estuary elsewhere (Tittley *et al.*, 1977). Andrews et al (1980) also reported the return of waterfowl to the Thames. From the late 1960s to the late 1970s the species composition of the bird population has become more varied, probably due to diminished pollution and the development of a more diversified macroinvertebrate fauna.

## C. THE SOUTHERN CALIFORNIA BIGHT

17. Studies of ecological recovery due to diminished sewage discharge to the marine environment has been carried out in some places in the Southern California Bight and, to a lesser degree, in San Francisco Bay.

18. Great amounts of municipal sewage are discharged to nearshore southern California waters every day. The wastewater (including chemicals) results from the household, industrial and agricultural activities of the eleven million people living in the southern California coastal basin (Smith, 1974) in the late 60s and early 70s.

19. In March 1971, the Orange County Sanitation district in the Southern California Bight diverted their wastewater from an inshore outfall, discharging at 16 - 18 m depth, to a deeper system discharging at 53 - 60 m depth. Sampling at both outfalls during discharge and non-discharge periods made possible the identification of sewage-caused effects.

20. The invertebrate and fish fauna was studied and the quality of sediments and waters were determined prior to the switch at both discharge areas and at two control stations (Smith, 1974).

by a few species of small, opportunistic polychaetes, particularly *Capitata ambiseta* and *Capitella capitata*. Small size might confer advantages of passive food uptake capabilities, short generation times and reduced predation, characters that might be of importance in a polluted environment containing large amounts of organic detrital particles. Among the fish, high measures of total abundance, numerical dominance of white croaker (*Genyonemus lineatus*), probably due to their generalized feeding behaviour (Skogsberg, 1939 quoted in Smith, 1974), and low diversity were recorded.

21. When the discharge was drastically reduced, a rapid change in invertebrate and fish fauna occurred, probably as a result of diminished detrital (food) supply. Previously abundant invertebrate species became rare, the total number of individuals decreased, species diversity increased and individuals increased in average size. White croaker disappeared from the area. The water and sediment quality improved quickly: within three months after the discharge termination, organic carbon in sediment and sulphide concentrations decreased to essentially background values.

22. The ecological effects of the new, deeper sewage outfall were apparently responses to increased detrital supply. Species biomass and abundances increased, especially small, opportunistic polychaetes like *Capitata ambiseta*. Shifts in the size composition of the fauna toward smaller individuals reflected increased abundance of juveniles and small-sized species.

23. Another study was carried out close to the Los Angeles County Sanitation District's sewage outfalls (Swartz *et al.*, 1986). During 1980 and 1983 sediment and macrobenthos samples were collected along a pollution gradient from the outfalls. At the same time the sewage discharge was reduced by 18.6 per cent.

24. The investigation showed that species richness, biomass and density of the benthos 5 to 11 km from the outfalls were significantly reduced in 1983 compared to 1980. Close to the outfalls - 1 to 3 km - the situation was different. In 1980 the fauna was poor, with few species and low biomass. The strongly dominating species was the opportunistic polychaete *Capitella spp.*. Three years later species richness and biomass significantly increased and *Capitella spp.* was much less abundant. The overall dominating invertebrates in 1983 were ostracods of the genus *Euphilomedes* that are characteristic of background conditions or of areas only slightly affected by wastewater discharges.

25. Sediment contamination by most measured chemicals, and organic enrichment indicators, decreased between 1980 and 1983. In 1980 the sediment was acutely toxic to the amphipod *Rhepoxynius abronius* but showed no toxicity in 1983.

26. At the beginning of the 1960s sewage treatment started to reduce some of the wastewater effects in the San Francisco Bay (Nichols *et al.*, 1986). The result is that summer oxygen depletion ... Summer concentrations of enteric bacteria in the South Bay have declined by a

permitted during summer. All these improvements are positive, but bay sediments and organisms are still contaminated with trace organic and inorganic materials. Toxic organic materials sometimes occur at levels comparable to those in highly contaminated estuaries.

27. In conclusion, it looks as if diminished sewage discharge, implying less detrital supply close to the outfall, results in a shift from a community with few, small species, often dominated by a small opportunistic polychaete in large amounts, to a more multi-specied community with increasing average individual size and fewer individuals. The situation farther away from the outfall is different; species richness, biomass and density of the benthos diminish following reduced discharges, probably due to decreasing food supply.

## II. RECOVERY FROM MERCURY POLLUTION

28. Phenylmercury acetate was used as a slimicide in the paper and pulp industry in Sweden till it was banned in 1968. Consequently part of the fibre banks that resulted from the industrial processes before wastewater treatment practices were installed to remove suspended material contained mercury-contaminated fibre.

29. Phenylmercury in the fibre banks is gradually converted to inorganic mercury (divalent or elementary) and then biologically methylated to form volatile dimethylmercury or monomethylmercury which leaks to the surrounding water and is accumulated in water-living organisms.

30. The rate of formation of methylmercury is low if counted as a fraction of the total mercury annually converted to methylated forms, typically 0.1 per cent a year (Beijer and Jernelöv, 1979). A depot of mercury in a fibre bank may thus remain a source of methylmercury for the environment for a very long time. But the rate of formation and release of methylmercury from a fibre bank can be high - if seen in relation to the resulting level of methylmercury in local predatory fish such as pike.

31. Along the Swedish coast of the Baltic there are some 10 areas with mercury-contaminated fibre banks (Lundberg and von Post, 1979) which, decades after the ban on the use of mercury as a slimicide, contribute to contamination of fish with methylmercury. In two cases restoration measures in the form of dredging of the fibre banks have been undertaken. One of those is the bay of Ovserum close to the town of Västervik.

32. In 1978-1979 a total of 184,000 $m^3$ fibre-containing sediment, corresponding to 15,000 tons of dry matter, was removed by suction-dredging, followed by a wastewater treatment. Of the total mercury 99 per cent was removed from the bay (Eriksson *et al.*, 1981).

33. Subsequently methylmercury levels in pike have dropped significantly, the ban on fishing in the Bay of Ovserum has been lifted (Jernelöv, 1977; Aronsson, 1980) and considerable improvement in the water quality and the structure of this bottom substrate has also been achieved (Lindeström *et al.*, 1978).

# III. RECOVERY FROM POLLUTION FROM PULP AND PAPER INDUSTRY

34. Untreated wastewater from paper and pulp industry contains dissolved organic matter with high biological oxygen demand (BOD), lignosulphonic acids and related compounds with brownish colour and fibres. Local physical and chemical effects consequently include decreased oxygen concentration, decreased light penetration and changes in bottom sediment structure.

35. In the Baltic, which has a brackish water ranging in salinity from just a few per thousand in the Botnian Bay to about 20 at the outlets to the North Sea, chironomids are frequently used as indicators of organic pollution.

36. In order of tolerance the following list has been proposed (Siederholm, 1974; Landner *et al.*, 1977):

Highest tolerance

*Chironomus plumosus*
*C. halophilus*
*C. halophilus*
*C. antracinus*
*Proctadius spp.*
*Camptochironomus tentans*

Moderate tolerance

*Chryptochironomus str. sp.*
*Parachironomus sp.*
*Glyptotendipos sp.*
*Polypedilum convinctum gr. sp.*
*Harmischia viridula*

Slight tolerance

*Monodiamosa bathyphila*
*Limnochironomus sp.*
*Pagastiella orophila*

Low tolerance (clean-water species)

*Microspecta spp.*
*Tanytansus s. str. sp.*
*Heteretrissocladius subpilosus*

37. Among oligochaetes the typical species stimulated by organic pollution are *Potamotrix hammoniensis* and *Limnodrilus hoffmeisteri.* These two species generally occur in the very transition zone between anaerobic fibre-covered sediments and less affected bottom areas. Next in tolerance to the oligochaetes come several species belonging to the genus *Tubifex*. Typical clean-water oligochaetes, i.e. species susceptible to organic pollution, are *Peloscolex ferox* and *Psammosycbides barbatus.*

38. Among bivalves *Pisidium sp.* dominates in the Bothnian Bay and *Macoma baltica* in all regions south thereof. Both are relatively tolerant to organic pollution, and are found close to fibre carpets or anaerobic sediments but never within them.

39. *Desidotea automon* is found in low numbers in all areas. It is relatively insensitive to pollution. However, the mobility of this crustacean makes quantitative sampling difficult.

40. In a summary of studies made around 15 paper and pulp industries in the Baltic in 1978, Pearson and Rosenberg (1977) found anaerobic zones with no macrofauna covering up to a few $km^2$ in most but not all investigated areas. Since that time water treatment has reduced discharges of BOD and fibre by more than 20 per cent and the anaerobic zones around the paper and pulp industries do not exist any more. While fibre banks persist, they have partly been colonized with species like *Chironomus halophilus* and *Potamotrix hammoniensis.*

# IV. RECOVERY FOLLOWING BAN OR RESTRICTION IN USE OF DDT/PCB

41. In the 1960s it was found the many chlorinated insecticides such as DDT, dieldrin and aldrin were found well outside areas of application, and that biomagnification in food chains resulted in high levels in predators. Similarly organo-chlorine compounds with industrial applications such as PCBs were found in unexpected substrates such as the fat tissue of eagles.

42. Environmental effects such as egg-shell thinning in birds and reproductive failure in seals were correlated to the presence of DDT/DDE and PCBs, respectively, and resulted in ban or restriction in the use of DDT and PCBs among other organo-chlorine compounds in more OECD countries in the early 1970s.

43. Subsequently environmental levels dropped along the Californian coast in the Japanese Inland Sea and in the Baltic (Jernelöv, 1978).

44. In the late 1970s levels of PCBs in seals in the Baltic dropped by 50 per cent following a lag period after the ban. Surprisingly, however, little further decrease has occurred since then. The levels of total DDT (mainly consisting of DDE) dropped continuously from the early 1970s following the ban and had by 1984 reached a level of about 10 per cent of those in the early 1970s.

45. However, after 1984 the level of total DDT, and specifically of DDT itself, rose again, indicating a fresh recontamination. While eggshells returned to pre-DDT thickness in the late 1970s, they have now become thinner again. However, the correlation with total DDT is not perfect, suggesting other contributing causes.

46. Seals remain badly damaged and the population remains on the verge of extinction. Here also the correlation between total PCBs and effects is far from perfect and suggests that other factors may be contributing to the damage (Olsson *et al.*, 1979 and 1986).

47. In summary, one can conclude that the ban and restriction on use of DDT and PCB resulted in decreased levels in the marine environment, but that the promising trend has been halted.

# V. RECOVERY FOLLOWING ACCIDENTAL OIL SPILLS

48. A large number of studies regarding the effects of oil pollution are reported in the literature. However, the major part of these investigations deal with laboratory studies and probably less than 25 per cent are direct field studies of oil spills and their impact on the surrounding environment. It is also obvious that only a small minority of the field investigations have actually attempted to show the full spectrum of effects of the oil spills on all the various parts of the ecosystem. Most of the studies have only involved some aspects of the possible impact on selected sub-systems of the ecosystem. However, some studies have aimed at illustrating the total effect on all the various parts of the ecosystem including the littoral, supra-littoral, benthic, and pelagic zones. The best examples of such studies are probably the ones carried out after the Torrey Canyon, West Falmouth, Arrow, Amoco Cadiz, Argo Merchant and Tsesis spills. A summary of the results of these studies together with some results of other pertinent investigations, and the conclusions and implications as far as the long-term impact and the recovery of the ecosystem are concerned is given in this report.

## A. FACTORS CONTRIBUTING TO THE EXTENT OF BIOLOGICAL DAMAGE

49. The results of the study of the episodes listed above indicate that the recovery of ecosystems affected by oil pollution varies considerably. In some cases large or extremely large spills have caused a minor impact, while in other cases only very small quantities have caused severe and indeed long-term effects on large parts of the marine ecosystem. Both abiotic and biotic factors govern the extent of the biological consequences of each oil spill, and it is the interaction and relative contribution of each of these factors that are important.

50. A physical factor of importance for the extent of biological damage and the time required for complete recovery is the capacity of the polluted water to be diluted to concentrations too low to cause any lethal or important sublethal effects. It is clear that the quantity of oil and the morphology and hydrography of the affected area are important here. Therefore it seems quite obvious that single oil spills in offshore areas with considerable water depth cause less biological damage than oil spilled close to the coast or in shallow and confined water bodies. The impact

the ecosystems in the area, while the Florida spill in West Falmouth, Mass. is an example of a spill close to the shore where the oil concentration in the confined water body rapidly reched toxic concentrations. However, the type of oil involved is also important in these two cases (see below).

51. In addition, the spills close to open coasts in areas with large tidal water amplitudes and good water exchange appear to cause considerably less damage than spills in atidal bays and archipelagos, where winds and currents cannot dilute the oil. Although locally extensive, the damage caused by the large quantities of oil from the Torrey Canyon did not seem to cause very long-term damage. This does, however, not apply to the areas where dispersants were used extensively in the clean-up of the oil. Considering the large amount of spilled oil, the limited results obtained so far from the Amoco Cadiz may also indicate a fairly rapid recovery, at least in exposed locations. The Tsesis oil spill on the contrary, although only involving a very small quantity of oil, caused comparatively long-lasting damage in the enclosed low-turbulence archipelago.

52. Most of the spill studies that have shown particularly long-term damage to the marine communities have been spills in which the oil is accumulated in fine-particle sediments in the intertidal or subtidal region, where the degradation and evaporation of the oil is slow or almost non-existent. The long-term action of the higher molecular weight aromatic hydrocarbons on living organisms will become important under such circumstances. Examples of spills where the oil has been accumulated in sediments, thus prolonging the impact, are the Florida , the Arrow and the Tsesis oil spills.

53. Another factor of high significance for the extent of the damage is the composition of the spilled oil. The light refined products such as no.2 fuel oil or similar oils containing high proportions of light, readily soluble, aromatic hydrocarbons are considerably more toxic than normal crude or heavy refined oils. Furthermore, light refined oils are usually more easily emulsified into the water body by wave action. The Florida oil spill involved such a light diesel oil with considerable long-term effects on the near-shore communities. The Tsesis oil spill, which also occurred close to the coast and involved approximately the same quantity of a heavy distillate (No. 5 fuel oil) caused less impact on the coastal ecosystem.

54. Meteorological conditions at the time of the spill affect the extent of the biological damage. Generally, less impact is caused if wind and wave action are low so that mixing of the oil into the water body is limited, and evaporation is facilitated. High temperatures also speed up the evaporation. An example of unfavourable meteorological conditions were those that occurred during the Florida oil spill, when strong wind rapidly emulsified the oil and low temperatures prolonged the leaching of toxic aromatic compounds from the oil.

55. The geographical distances in the affected area constitute a factor that is important for the recolonization of an area affected by an oil spill. Basically, recolonization can occur either through the direct migration of adult or juvenile mobile stages (not planktonic), or by the settlement of planktonic stages distributed by current. For some species, attaching to drifting substrates

of a coastline is affected by a spill, recolonization of all major types of organisms may occur rather rapidly. On the other hand, if longer stretches of a coast are affected, then only the organisms distributed by the currents may be able to recolonize within a few seasons. Organisms which have to migrate along the sea floor may require considerably longer time to re-enter the area. This was, for example, observed after the Torrey Canyon oil spill.

56. The biological factor of significance for the extent and duration of the damage relates to the season of the year when the spill occurs. In certain months, particularly in the spring and summer, biological activity is higher, with large numbers of particularly sensitive juvenile stages of several species present. During late autumn on the other hand, biological activity may be very low with small stocks of most species. This was the case when the Tsesis spill occurred.

## B. THE EXTENT OF DAMAGE IN DIFFERENT COMMUNITIES

57. Few studies on the effects of single oil spills on planktonic communities exist. Some effects on phytoplankton were observed following the Torrey Canyon, Santa Barbara (Straughan, 1971) and Tsesis oil spills. These effects were, however, minor. Some impact on zooplankton was observed after the Torrey Canyon spill, although the effect was probably related to the toxicity of dispersants rather than of oil. The Amoco Cadiz oil spill apparently resulted in effects on zooplankton up to a few months after the spill in offshore areas. Following the Argo Merchant spill some effects were observed on zooplankton in the oil-contaminated area. These effects did, however, not appear to be very drastic. After the Tsesis oil spill, zooplankton was severely affected only immediately after the spill and in the close vicinity of the wreck.

58. Based on these observations it would appear that the impact of the oil spills on the planktonic community is not of a long-term nature. The water exchange and turbulence in offshore areas rapidly dilute the oil and replace the affected communities. It seems likely that the period necessary for recovery of the plankton community from single spills is usually a question of weeks rather than months.

59. Studies in the littoral zone are more frequent. Extensive and lasting damage was caused to the littoral communities after the Tampico Maru, Florida, Arrow and Tsesis oil spills. These accidents occurred in bays and estuaries where the spilled oil was not sufficiently diluted. In several cases the oil was also accumulated and retained in sediments. The spills from Tampico Maru and Florida also involved highly toxic products. That from Amoco Cadiz caused severe acute effects along the coast of Brittany. Except in the estuaries, however, the spill does not appear to have caused very long-term impacts in the littoral zone. Not even the oil from the Torrey Canyon

60. It appears from these studies that the impact on littoral communities may be drastic and lasting depending on a number of factors. The recovery of affected littoral communities is usually a question of several years. In the worst cases the time necessary for a complete recovery of the ecosystems may take one or more decades.

61. Only few studies are available on the impact of acute oil spills in benthic subtidal and sublittoral communities. The studies following the Florida and Tsesis oil spills indicate, however, that the impact in this zone may be severe, and may perhaps last longer than in any other part of the ecosystem. The Tsesis oil spill caused effects in soft bottom sublittoral communities that lasted longer than those in the littoral zone. The oil was incorporated into sediments and organisms and as the water exchange was limited and the temperature and oxygen content were low, the oil was preserved for a longer period than in the littoral zone.

62. The figure shows schematically the impact of the Tsesis oil spill on various parts of the ecosystem.

63. It is not known if the picture applies to the general recovery process that follows all spills in which the oil affects both pelagic, littoral and sublittoral zones, or if it reflects a unique situation. The reason why this is not known is mainly because only in very few cases have investigations been carried out concurrently in all of these three habitats. General knowledge of the stress tolerance of littoral and tidal organisms compared with that of deeper living organisms and of the rate of water exchange and consequent dilution capacity in surface nearshore areas compared to bottom areas does, however, suggest that the figure may show a general recovery process applicable to other oil spill cases. In addition, the conditions for oil degradation with regard to temperature and oxygen support the generalizations about the extent of the damage and the recovery pattern observed in the three major marine habitats after the Tsesis oil spill.

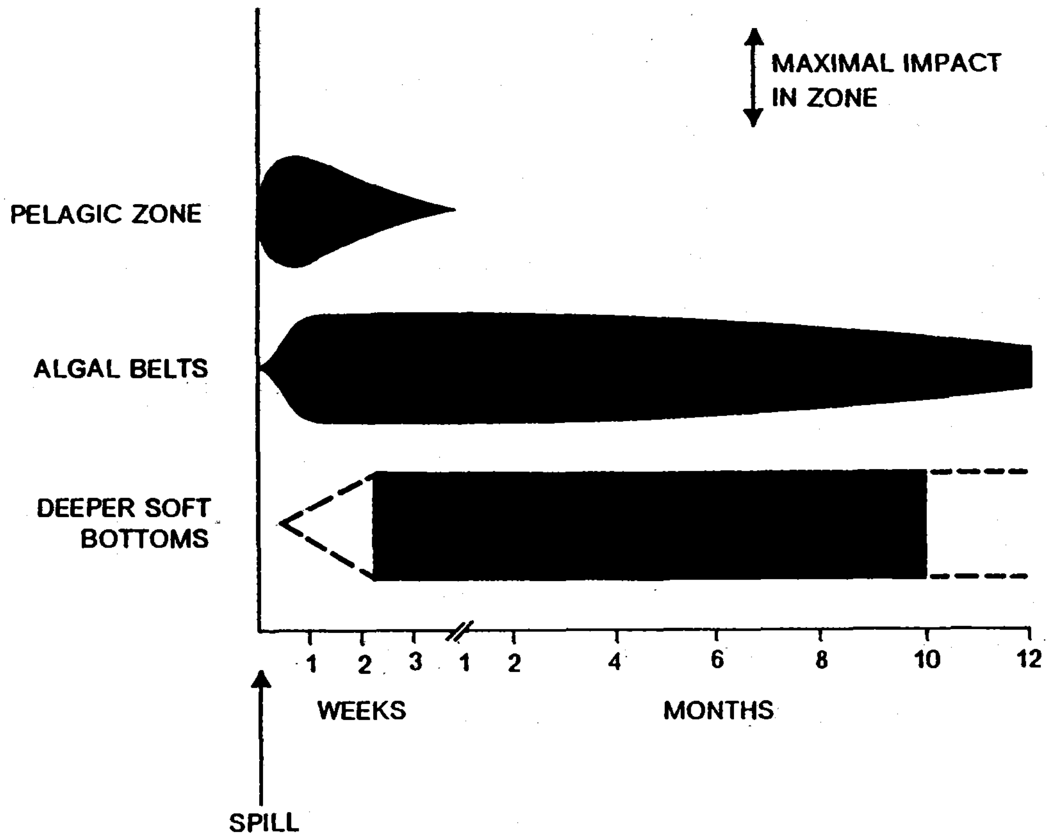


FIGURE. Duration and intensity of ecological disturbance following the Tsesis oil spill in the three major biotopes studied. Intensity is given as a subjective estimate relative to maximum recorded impact in each biotope, since comparisons between biotopes are of questionable validity.

## C. CONCLUSIONS

64. The extent of damage to the marine ecosystem and the time required for recolonization and recovery of affected ecosystems is the result of interactions with, and relative contributions of, a number of factors:

- The dilution capacity of the polluted water which depends on turbulence, currents, winds and water exchange, closeness to the shore, depth and quantity of oil.

- The presence of low-energy environments such as muddy estuaries and bays, soft bottom sediments in sublittoral and subtidal areas.

- The type of oil is important both with regard to the quantity of toxic compounds in the oil and the tendency to form emulsions.

- The meteorological conditions such as winds and wave action, and temperature.

- The geographical distances within the area affected by the spill.

- The season of the year when the spill occurs.

65. The on-scene spill studies carried out so far indicate that:

- The effects on the pelagic communities are rarely very drastic and the recovery period is usually a matter of weeks or one or two months.

- The effects on the littoral and tidal communities can be extremely severe and recovery may take years or decades.

- The effects in sublittoral and subtidal communities may also be extremely drastic, and the effects may be present even longer than in the shoreline communities.

- The effects on bird populations are often severe and some species are threatened with extinction.

# VI. RESTORATION OF MARINE ECOSYSTEMS

## A. RECOVERY OF CORAL REEFS FROM PHYSICAL DAMAGE

66. Destruction of coral reefs can be induced by human activities and natural catastrophes. Alcala *et al.* (1986) reported that the major cause of catastrophic coral mortality is mechanical destruction by tropical storms. They studied the recovery of a coral reef in Pescador island, Philippines, after typhoon damage. A year after the damage, little recovery in terms of recolonization by living corals on the bare rocky substrate had occurred.

67. Fishing by dynamite-blasting causes destruction of coral reefs. Studies in the Central Visayas, Philippines, indicated a recovery time (to 50 per cent cover) of about 38 years (Alcala *et al.*, 1979). In another study in Cangaluyan Island, Philippines, all living corals in a previously dynamite-blasted reef community were removed to simulate severe damage (Alino *et al.*, 1985). After a year only 1 - 3 per cent of the area was covered by living corals.

## B. MANGROVES

68. Large areas of the world's tropical and subtropical shorelines are dominated by mangroves. They are usually regarded as land builders and coastline stabilizers (Rehm, 1976; Teas, 1977) although the land-building effects of the trees are perhaps overestimated (Carlton, 1974).

69. Mangroves in Florida often grow in waters with wide salinity fluctuations (Rehm, 1976; Teas, 1977). Because of the salt tolerance, they are ordinarily successful competitively where other plants are inhibited by salinity (Teas, 1977). Semidiurnally the trees are inundated by the tide. To be able to withstand the environmental stress, mangroves have evolved specific modifications like aerial roots and prop roots (Carlton, 1974). The aerial roots support the trees with oxygen during inundation and the prop roots acts a stabilizers. The roots also function as sediment and debris traps which prevent erosion (Rehm, 1976; Teas, 1977).

70. The role of mangroves as habitat providers for many organisms, e.g. algae, invertebrates, fishes and birds is described by many authors (e.g. Teas, 1977; Thayer *et al.*, 1978; Thayer *et al.*, 1987).

- impact of man, e.g. dredge-and-fill operations in the construction of ports and harbours (Teas, 1977)

- hurricanes (Teas, 1977)

- isopod parasites (Rehm, 1976).

72. In order to re-establish mangrove ecosystems, several transplanting experiments have been carried out (e.g. Carlton, 1974; Teas, 1977; Hoffman *et al.*, 1981). In Florida three kinds of mangroves are found: Red Mangrove (*Rhizophora mangle*), White Mangrove (*Laguncularia racemosa*), and Black Mangrove (*Avicennia germinans*). The species usually transplanted is the Red Mangrove (Teas, 1977) although Savage (1972b quoted in Carlton, 1974) concluded that the Black Mangrove would serve equally well or better for this purpose, among other things depending on its ability to tolerate disturbed substrates. Different types of material have been used for planting, namely propagules - deriving from mature fruits of Red Mangroves - seedlings and trees (Goforth *et al.*, 1979; Teas, 1977).

73. The results of transplantation are highly variable from no to 100 per cent survival, depending on the degree of shoreline exposure, plant material, tidal depth of planting, substrate type, water salinity, root-parasite prevalence and vandalism (Teas, 1977).

74. Rehm (1976) questions the extent to which mangroves can be used to stabilize eroding shorelines in Florida due to the destruction of mangroves (especially Red Mangroves) caused by the wood-boring isopod *Sphaeroma terebrans*. Hannan (1975 quoted in Teas, 1977) points out the importance of transplanting mangroves where they are not easily killed by *Sphaeroma*. By transplanting root-protected plants to sites at or above the mid-tide zone on the east coast of Florida, he obtained 80 per cent to 100 per cent survival after 13 months, compared to no survival (due to *Sphaeroma*) when planting mangroves without root protection below the mid-tide zone.

75. Already during the beginning of this century, attempts to transplant mangroves were reported, e.g. Bowman in 1917 (quoted in Carlton, 1974) planted mangroves to prevent erosive action by waves on railroad ballast. The transplantation was reported successful.

76. In 1940 a total of 4,100 Red Mangroves were transplanted in an area south-west of Florida (Dalton, 1940 quoted in Teas, 1977). One year later 3,300, around 80 per cent of the plants had survived. However, 32 years after planting, none at all could be found. This shows the importance of following-up the plantings for several years to be sure of the possible success.

77. In a Red Mangrove transplanting project at Key West, Florida propagules, seedlings and small trees were each planted at three sites with different exposure rates (Goforth *et al.*, 1979). After 23 months the mean value of survival for propagules at the three sites was 45 per cent. The survival rate was inversely related to the degree of exposure. The mean value for seedlings was about the same: 48 per cent. Transplants of 2-3 year old trees (0.4 - 0.8 m high) were highly

78. Hoffmann *et al.*, (1981) report 73 per cent survival of Black and White Mangroves (initial height: 0.3 - 1.9 m) after 13 months from a project in an area of low wave energy in Tampa Bay, Florida. In a review of Red Mangroves transplantations, Teas (1977) describes several examples of transplantation of seedlings (propagules), and of small and large trees. His conclusion is that the survival rates of older plants are lower than for younger plants, probably due to lesser root damage when transplanting smaller mangroves. He reports that of the largest transplants attempted (4.6 - 6.1 m high) none survived. Contradictory data are supplied by Carlton (1974). Up to 6.5 m high Red Mangroves were transplanted in Miami (Gill, 1971 quoted in Carlton, 1974). Revisiting about two years later indicated survival of all trees but one. An example of planting mangroves in Hawaii proved successful; they developed into forests and spread to other areas (Teas, 1975 quoted in Teas, 1977).

79. In conclusion mangrove restoration seems to be a technique with very varying results depending on planting type and environmental factors.

## C. SEAGRASS BEDS

80. Seagrasses provide rich habitat and food resources which contribute to the large productivity in these areas (Thayer *et al.*, 1978, 1982). The canopy of the plants provides a shelter for preys and at the same time supplies predators with food. In addition, herbivores consume the living plant, detritus from the dead plant is ingested by detrivores, and micro-organisms exploite dissolved organic compounds derived from the seagrasses (Thayer *et al.*, 1982). Seagrasses also function as sediment stabilizers through their roots and rhizomes (Fonseca *et al.*, 1985).

81. Increasing human activities like fishing, coastal development and pollution causes damage to seagrass communities. Research into the function of these ecosystems did not start until the late 1950s and seriously only in the 1970s. As a result, the technology for seagrass transplantation is quite new (NOAA, 1985). However, of later years seagrass transplanting has increased.

82. The technique is used to enhance and accelerate the establishment of seagrasses and gradually fishery production in damaged areas. Recommendations for future coastal revegetation are supplied by NOAA (1985).

83. It is believed that the fishery production in a transplanted area is equal to that in a natural meadow, but no or few data exist to support this assumption (Thayer *et al.*, 1986). According to Thayer *et al.*, (1986), both successes and failures have been reported following seagrass transplantation.

84. The importance of site evaluation prior to the execution of a transplanting project is

Zedler *et al.*, 1982). Critical environmental factors are temperature, salinity, light, depth, hydrodynamics, sediment characteristics and fluctuation. Planting on dredged material may involve problems with chemical pollution (Fonseca, 1987).

85. It is also important to use plants that normally recolonize damaged seagrass beds and to get the plants from non-destructive sources (Mangrove Systems, Inc., 1985).

86. One example of seagrass restoration is a project in the Florida Keys. The purpose was to restore seagrass meadows destroyed as a result of bridge construction (Mangrove Systems, Inc., 1985). A little more than half of the damaged area was considered restorable. In April and August 1983 the greater part of the area was planted with shoots from shoalgrass (*Halodule wrightii*) anchored with staples or nails to the substrate. After a year 73 per cent of the restored area met the requirements of 80 per cent cover.

87. This example does not mention anything about faunal abundance in the transplanted area. However, Fonseca *et al.*, (1979) replanted eelgrass (*Zostera marina*) into a meadow in North Carolina which had been denuded by scallop dredging. Most of the planting sites developed a significant cover of eelgrass. A little more than four months later, there were significantly higher scallop densities in the vegetated treatments as compared to the controls. Even the total faunal abundance, in terms of densities and species numbers was higher in the vegetated areas (Homziak *et al.*, 1982 quoted in Thayer *et al.*, 1985).

88. In a study adjacent to the one just mentioned, it was found that the epifaunal communities in disturbed eelgrass meadows recovered quite quickly (Stuart, 1982 quoted in Thayer *et al.*, 1985). In another study *Zostera marina* was planted on dredged material in North Carolina (Fonseca 1987). Even though the transplanted meadows 10 months after planting showed higher faunal abundance than the unvegetated areas, distinct differences appeared between the transplanted areas and adjacent, natural, meadows, the latter having a much higher faunal abundance.

89. In summary, it seems that seagrass transplantation enhances and hastens the rebuilding of the ecosystem, in spite of the rather limited experience of the technique. Even though both successes and failures are reported, many projects seem to end up in a relative success, especially when the planting sites are carefully evaluated before transplanting.

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# ANNEX IX

# LAND-TO-OCEAN TRANSPORT OF CONTAMINANTS:

# COMPARISON OF RIVER AND ATMOSPHERIC FLUXES

**This annex, prepared by P. S. LISS (University of East Anglia, Norwich, U.K.) is the result of workshops of GESAMP Working Groups 14 (Interchange of Pollutants between the Atmosphere and Oceans) and 22 (Land-Sea Boundary Flux of Pollutants). The participants were: J. M. Bewers (Canada), R. A. Duce (U.S.A.), T. D. Jickells (U.K.), P. S. Liss (U.K.), J. M. Miller (U.S.A.), H. L. Windom (U.S.A.) and R. Wollast (Belgium).**

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# INTRODUCTION

1. The state of the marine environment with respect to anthropogenic contaminant inputs is probably best assessed by evaluating changes in fluxes through major input pathways rather than by monitoring oceanic reservoirs. The two dominant natural pathways through which most contaminants are transported from the continents to ocean basins are rivers and the atmosphere. This section presents a brief review of the present understanding of river and atmospheric transport of contaminants to the marine environment. Where possible, fluxes through these pathways for specific regional seas and oceanic basins are provided and compared. Much of this review is based on results of deliberations of GESAMP Working Groups 14 and 22, and more detailed discussions are provided in GESAMP Reports and Studies Nos. 26 (WMO, 1985) and 32 (UNESCO, 1987).

# I. CURRENT STATE OF UNDERSTANDING AND RESEARCH NEEDS

## A. RIVER INPUT

2. The following discussions of the fluxes of contaminants to the ocean through rivers is based on the report of GESAMP Working Group No. 22 (UNESCO, 1987). This document defines two categories of land-sea fluxes from rivers to the oceans. The first is the gross flux, which corresponds to the rate of contaminant transport to the marine environment within the river itself. The second category is the net flux, which corresponds to the flux of river-derived material that escapes from the nearshore and estuarine region and is transported to the open ocean beyond the continental shelf.

### *1. Gross river fluxes*

3. Estimates of the gross global run-off of trace constituents from rivers have generally been derived from scaling of the fluxes of chemicals from individual major rivers (e.g. cadmium and nickel derived from scaling of Amazon discharge by Boyle *et al.* (1976) and Sclater *et al.* (1976), respectively). While such projections of global run-off fluxes from individual rivers are useful and undoubtedly warranted, it is likely that they provide relatively inaccurate estimates of global river

chemicals would be by obtaining data from a number of rivers representing a range of climatic, geological, biological, hydrologic and demographic regimes (UNESCO, 1987). The emphasis in these studies should be on elucidating the natural river-water composition in terms of the drainage basin characteristics. Such studies could later be extended to comparisons among similar basins subject to differing anthropogenic influences, as a result of either population density or industrial activity, and to investigations of the nature and extent of anthropogenically-mediated chemical mobilization.

4. The approach suggested above to obtain estimates of gross river fluxes requires the acquisition of data of comparable analytical quality from a number of river systems. At the present time, there are reliable data for only a relatively few river systems for the contaminants of interest (i.e. nutrients, trace metals and high molecular weight organics). The following discussions, however, provide a review of the present level of understanding regarding gross river fluxes and, to the degree possible, global gross fluxes are estimated.

### *(a) Nutrients*

5. Estimates of natural gross river fluxes of nutrients have been obtained by several authors from the study of uncontaminated river systems by scaling the results to world run-off. Such estimates indicate that the natural global dissolved fluxes of nitrogen, phosphorus and silicon are 15, 1 and 100 Mt $y^{-1}$, respectively. The global fluxes of particulate nitrogen and phosphorus are each 20 Mt $y^{-1}$.

6. The nutrient fluxes in rivers resulting from human activities have been estimated by a variety of approaches, including comparison of concentrations of nitrogen and phosphorus species in pristine and polluted streams, historical evolution of these concentrations in some large rivers, and direct estimations of fluxes due to domestic, agricultural or industrial activities. Examples of anthropogenic nutrient flux estimates from rivers are given in Table 1. Although there are large discrepancies among the estimates of the various authors, it is obvious from a comparison of these estimates with natural flux estimates that the anthropogenic flux of dissolved nutrients is at least comparable to, and probably significantly larger than, the natural fluxes. Except for the estimates of increased fluxes of particulate nitrogen and phosphorus due to soil erosion made by Van Bennekom and Salomons (1981), there are no other data available on a global basis.

### *(b) Trace metals*

7. As pointed out above, the use of individual river compositions to estimate global run-off fluxes for trace chemical constituents is fraught with uncertainties. However, these uncertainties are usually not so great as to preclude these flux values being order-of-magnitude (or better) estimates of gross global river discharge fluxes of trace metals.

of a number of rivers representing different drainage basin characteristics (see UNESCO, 1987), the mean dissolved and particulate composition of global run-off can be estimated, and used, in turn, to estimate the global gross river fluxes of trace metals given in Table 2.

### *(c) High molecular weight synthetic organic compounds*

8. Unlike the two categories of contaminants considered above, there is no natural flux component of high molecular weight organic compounds. The complex chemistry of these substances, along with the number of individual compounds involved and their heterogeneous environmental distribution, make global gross river flux estimates irrelevant. Any consideration of gross river inputs should therefore be region- and compound-specific.

## *2. Net river fluxes*

9. Many of the chemical constituents transported by rivers to the sea are transformed or removed from the water during its transport through the estuarine and continental-shelf environments. Thus, to understand the transport of chemicals to various regions of the ocean, particularly the deep ocean, it is necessary to determine what the net fluxes of river-transported chemicals are at some offshore boundary. When combined with a gross flux estimate, net flux determinations provide estimates of what proportions of river-transported materials are retained within estuarine, nearshore or shelf environments. Such estimates make possible the assessment of the effects of such retention on amenities and communities in these areas.

10. It has been pointed out that net fluxes apply to the transport of material across the boundary between the continental shelf environment and the pelagic ocean. Thus, the processes that are responsible for the removal of material in the coastal zone are those occurring both in estuaries and on continental shelves. The extent to which these processes modify river discharge fluxes is likely to vary from extreme, for minor river run-off to large slowly-flushed shelf regions, to small, for rivers having external estuaries on narrow continental shelves, such as the Amazon. This argument is based upon the premise that the longer the residence time of water on the shelf, the greater the time for removal to occur. There are also varieties of continental shelf environments that will have a pronounced effect on the nature and extent of coastal zone removal. Rivers discharging to constrained marginal seas having limited exchange with offshore shelf waters, such as the Baltic Sea and the Gulf of St. Lawrence, are likely to have smaller net discharges to the deep ocean than similar rivers discharging directly to narrow, well-flushed shelf areas. It is this very heterogeneity of conditions that makes the estimation of net river influxes so difficult. However, there exist several different approaches to the estimation of net river influxes to the ocean that can be applied to various chemical substances. These approaches fall into three main categories:

1. Estimates based upon process studies;

2. Estimates based upon chemical-salinity relationships;

3. Estimates based upon mass balances.

11. The first of these approaches attempts to understand and model the biogeochemical behaviour of substances in the estuarine/shelf region. The models are then used to determine the relationship between gross and net fluxes of specific chemicals, and to estimate the net flux on the basis of the best estimate of aggregate gross influx to the area concerned.

12. The second approach uses relationships between chemicals and a conservative property (salinity) to calculate the effective composition of the freshwater end-member for mixing in the coastal zone. A good example of this approach is the calculation of effective zero-salinity values from the relationships between metals and salinity in offshore areas.

13. The third method is the construction of mass balances for areas in which all but one of the fluxes can be determined by direct measurement and physical oceanographic understanding. The use of this technique is either confined to coastal areas and marginal seas having constrained boundaries, thereby facilitating the development of hydrodynamic models, or to estimations of net fluxes based upon crude estimates of removal (sedimentation) rates. Systems such as the Baltic Sea and the Gulf of St. Lawrence have been used for the first method, and calculations of net nutrient fluxes have been based on sedimentation rates and sediment composition.

14. None of the above approaches is very distinct, and often a specific approach to net flux estimation will embody aspects of all three. Estimates of the global net river fluxes of various substances have been made and are described below.

### *(a) Nutrients*

15. From process studies it may be concluded that only small proportions of the nutrients survive nearshore and continental-shelf processes to be transported offshore. Estimates of the net nitrogen and phosphorus fluxes can be obtained from the rate of nutrient incorporation into pelagic sediments based upon C/N and C/P ratios and the rate of pelagic carbon accumulation. These estimates are 0.44 and 0.3 Mt $y^{-1}$ for nitrogen and phosphorus, respectively. A similar calculation for silica is not possible due to the lack of data on the concentrations and fluxes of opal in the ocean. The nitrogen and phosphorus fluxes so calculated should represent upper limits to the net river fluxes, because they include the net phosphorus and nitrogen contributions from the atmosphere to the pelagic ocean. However, the values obtained are relatively small proportions of the aggregate dissolved and particulate nitrogen and phosphorus discharged by rivers (the gross

### *(b) Trace metals*

16. Estimates of the global net river fluxes of trace metals have been made largely on the basis of either mass-balance calculations for marginal sea areas or metal-salinity relationships in offshore waters. Table 3 presents net flux estimates derived from these two separate approaches. In comparing these it should be remembered that the values obtained from mass-balance calculations may be biased by the fact that the characteristics of the river discharges to specific marginal sea areas may not be representative of global river composition. There is, however, some logic to the salinity regression estimates being consistently lower than those obtained from mass-balance calculations. These differences will be partly due to the inclusion in the net-flux estimates of metals associated with particulate material that remain bound and not subject to exchange with the dissolved phase in transport through the coastal zone. Such particulate fluxes will remain unquantified in the salinity regression approach.

17. As a check on the representativeness of total net river flux (Table 3), the results of mass-balance calculations on transport in the Gulf of St. Lawrence (Yeats and Bewers, 1983) have been used to calculate an independent set of dissolved and particulate metal fluxes for the North Atlantic by scaling. The results of these latter calculations confirm the relative proportions of dissolved and particulate transports for each metal, and also yield flux values that in all cases but one are within a factor of four (and in most instances within a factor of two) of those given here.

## B. ATMOSPHERIC INPUT

### *1. Sources and speciation*

18. There are many sources, both anthropogenic (man-made) and natural, of trace species (trace metals, organics and nutrients) in the global atmosphere. Sources of atmospheric trace metals are diverse and complex and include natural inputs (e.g., windblown dust, volcanoes, vegetation, sea-spray) and inputs arising from man's activities (e.g. emissions from metal production and processing industries, energy utilization and production, and waste disposal). Global and regional estimates of these emissions are available (Pacyna, 1986). Organic substances of primary environmental concern are high molecular weight organochlorine compounds (pesticides and PCBs) and polyaromatic hydrocarbons (PAHs). The organochlorine compounds are entirely synthetic and sources include emissions during production and disposal, as well as during application, in the case of pesticides. PAHs are produced during combustion processes, which may be from man-made or natural material. Estimates of global emissions of these compounds are complicated by the enormous numbers of different species involved, the diverse sources and the rapidly chang-

sources. In the case of nitrogen, the natural sources are emissions of nitrogen oxides from soils and the oceans, and ammonia from decaying organic matter, while the primary anthropogenic sources are high-temperature combustion processes. Global atmospheric emission estimates for nitrogen (Logan, 1983) and for phosphorus (Graham and Duce, 1982) are available.

19. Contaminants can travel in the atmosphere as gases and aerosol particles and the distinction is of considerable importance because of its effect on deposition processes. Phosphorus and all trace metals, except mercury, travel primarily in association with aerosol particles. The size class of aerosol particles with which a metal is associated is largely a function of the emission process. Mercury and boron, by contrast, are present in the atmosphere primarily as gases. Fixed inorganic nitrogen can travel as gases (nitrogen oxides or ammonia) or aerosol particles, and complex interconversions between the two forms occur. The distribution of organic compounds between the gas and aerosol phase depends on the vapour pressure of the compound and the ambient temperature.

## 2. *Atmospheric transport*

20. As for the rivers, the atmosphere is an important pathway by which pollutants are transported from their sources to the ocean. Generally, the substances of interest are injected into the atmosphere at or near the surface of the earth, where they are mixed vertically and can be transported hundreds to thousands of kilometres. Because sources are primarily in the mid-latitudes in the Northern Hemisphere, these materials move principally from west to east. Thus, North America contributes significantly to the North Atlantic Ocean, and the Asian continent influences the North Pacific Ocean. On the other hand, in the trade wind zones, movement is from east to west, such as flow from southern North America across the Pacific toward the Hawaiian Islands. However, even in the westerlies, disruptions in the meteorological patterns, particularly by such events as the El Niño, can cause significant north-south transport of pollutants (Merrill, 1986).

21. To describe these atmospheric movements, meteorologists have developed several types of models. These models at their most sophisticated can incorporate emissions, transport, chemical transformation and removal processes. There have been three types of models that are relevant to atmospheric transport to the oceans:

- Models based on multiple trajectory calculations (Lagrangian) or data specified at fixed grid points (Eulerian) can be used for transport calculations, especially near coastal areas. A disadvantage of these models is their need for a considerable amount of meteorological data not available over the oceans (Rodhe, 1985).

- General calculation models have been applied to describe transport on a global scale for trace substances. However, their suitability for describing the transport of trace metals, organics and nutrients, with the exception of nitrogen compounds, has not been attempted (Levy *et al.*, 1985).

- Single back trajectory models have been the most successful to date in describing transport paths to the ocean. This approach has the advantage that a measurement can be assigned a trajectory that describes the movement of the sampled parcel of air back to its possible source. Numerous examples of the use of trajectories can be cited from deposition to long-transport across both the Atlantic and Pacific Oceans (Miller, 1987).

### 3. *Atmospheric input processes to the ocean*

22. The total atmospheric input of chemicals to the surface oceans is the sum of the amount entering in gas and solid phase ('dry' deposition) and of that falling in rain and snow ('wet' deposition). The latter will comprise water and its dissolved gases and solutes, together with any insoluble material contained therein.

23. A significant amount of the published data for air-sea fluxes of chemicals has been obtained by direct measurement in precipitation, since it is conceptually and practically easy to collect samples. However, there are very real problems with this simple approach for precipitation chemistry, and even greater difficulties arise in attempting to measure dry deposition directly.

24. Contamination-free collection of precipitation in remote marine regions is difficult, with relatively few reliable and extensive data bases available for pollutants in rain and snow. Even when large amounts of high-quality precipitation chemistry data are available, as is becoming the case for certain areas over the North Atlantic and for some coastal regions, complications in the interpretation arise because marine rains are associated with a variety of types of meteorological systems. These range from simple trade wind showers with a typical horizontal cross section of 500 m and vertical extent of 2,000 m to mesoscale storm systems with horizontal dimensions of over 1,000 km and vertical extent through the tropopause. The chemical composition may be significantly different as a result of differences in the vertical distribution, duration, intensity, and droplet size of the precipitation. There has been little effort to distinguish among these different types of systems when evaluating wet input of trace substances to the ocean over large geographical and temporal scales.

25. Direct measurement of aerosol particle dry deposition to the ocean is currently not feasible, except for substances present on giant ($r > 5$ µm) particles, whose deposition is controlled largely by gravitational settling. The use of surrogate surfaces for deposition of particles with $r < 5$ µm has been seriously criticised because these surfaces do not accurately mimic the microstructure, roughness, etc., of a natural water surface (Hicks *et al.*, 1980).

26. Satisfactory techniques are also presently not available to measure directly pollutant gas

fluxes of interest across the air-sea interface. In theory the measurement of vertical gradients of gaseous pollutants above the air-sea interface or simultaneous rapid measurements of the pollutant concentrations and the vertical wind component near the ocean surface (eddy correlation) could provide accurate fluxes. However, current analytical capabilities for the pollutants of interest do not allow sufficiently accurate or rapid measurements to be made.

27. Because of these various problems, indirect methods for the estimation of pollutant fluxes are often used. In indirect approaches the input flux is generally calculated from the product of a concentration term, which drives the flux, and a kinetic parameter, which quantifies the rate of mass transfer.

28. For gases the flux driving term (F) is the concentration difference across the air-sea interface ($\Delta C$), and the rate expression is often called a transfer velocity (K):

$$F = K * \Delta C \qquad (1)$$

29. The $\Delta C$ term must be obtained by direct field measurements of air and water concentrations. The transfer velocity is controlled by near-surface processes in either the air or the water phase, or in both. Which phase(s) control(s) K depend(s) on the Henry's law constant of the gas and its chemical reactivity in the water. The magnitude of K is dependent on the degree of mixing in the rate-controlling phase, but not necessarily in a simple manner (Liss, 1983; Liss and Merlivat, 1986). Application of (1) yields the net flux across the interface. However, this model cannot represent accurately the detailed, complex and variable processes occurring at the air-sea interface. Other difficulties arise from inaccurate values of Henry's law constants and difficulties in obtaining accurate values of the appropriate transfer velocities for the gases of interest. Nevertheless, models of this type make it possible to predict the direction of the air-sea flux and may yield the flux at a particular location, and in some cases (e.g. dimethyl sulphide) globally, within a factor of 2 to 3, provided accurate measurements of the pollutant of interest can be made in the surface air and sea water.

30. In the case of aerosol-particle transfer, the flux is obtained from the product of the measured concentration of particles in the air ($C_a$) and the deposition velocity ($V_d$):

$$F = V_d * C_a \qquad (2)$$

31. The term $V_d$ is derived from theoretical considerations, taking into account the processes involved, e.g. gravitational settling, impaction and diffusion. The theory is very complex because all of these processes act simultaneously and because each is dependent on a number of variables (i.e., wind speed, particle size, relative humidity, air viscosity, sea-surface roughness,etc.). Because $V_d$ varies with the size of the particles, the term $C_a$ must be obtained by accurate measurements

of the size-fractionated aerosol, especially in the largest size fraction (Slinn and Slinn, 1980; Williams, 1982; Arimoto and Duce, 1986). However, these calculated fluxes still have very large uncertainties, and development of methods for the accurate measurement of dry deposition is urgently needed.

32. The flux of material to the sea surface in rain can be formally expressed as the product of the precipitation rate (P) and the concentration of the substance in the rain ($C_r$):

$$F = P * C_r \qquad (3)$$

33. Equation (3) is often expressed not in terms of $C_r$, which effectively implies making a direct flux measurement, but through a wash-out factor (W), which is the ratio $C_r / C_a$, where,

$$F = P * W * C_a \qquad (4)$$

34. By analogy between (2) and (4), P * W is equivalent to a 'wet' deposition velocity. The advantage of (4) over (3) is that, if W is known, measurement of $C_a$, provides the concentration driving term for both equations and hence makes possible the estimation of deposition of both 'dry' particles and material in rain.

35. It should be noted that application of (2), (3) and (4) gives only gross fluxes. A number of chemical substances can be preferentially injected into the atmosphere from the ocean surface, either on sea-salt particles produced by bursting bubbles or directly via gas exchange. When this material re-enters the ocean, via either wet or dry deposition, it does not contribute to a net input of that substance to the ocean. For example, the accurate measurement of lead in rain may not be representative of the net input of that metal to the ocean because a fraction of the lead may be recycled from the ocean surface. Similarly, the measurement of selenium in rain will not give an accurate indication of the net input of selenium to the ocean because much of the selenium present in rain and on aerosol particles is apparently derived from gas-phase selenium released by the sea. The importance of the atmosphere as a transport path for material from the continents can only be assessed accurately if the relative contributions of the net and recycled components can be distinguished (Weisel *et al.*, 1984; Arimoto *et al.*, 1985; Settle and Patterson, 1982). Unless the recycled component is accounted for, the calculated net deposition to the ocean will be anomalously high.

## 4. *Data quality and availability*

36. The existing data bases which can be used to make estimates of fluxes of contaminants from the atmosphere to the oceans are of variable size and quality, depending on the substance.

while ammonia and nitrogen oxide concentrations in air are less well known (Knap *et al.*, 1986; Duce, 1986). Relatively few data are available on phosphorus concentrations in the atmosphere (Duce, 1986). Few recent high-quality data sets are available for trace metal concentrations in aerosols and precipitation in oceanic regions (Church *et al.*, 1984; Arimoto and Duce, 1987). Recent advances have provided a preliminary geochemical budget for mercury (Fitzgerald, 1986). Older data may be inaccurate because of contamination during sampling and analysis. For trace organic contaminants, very few data exist on concentrations in the atmosphere or precipitation because of the formidable analytical problems associated with the collection and analysis of these substances (Atlas and Giam, 1986). The data that are available for these contaminants are from widely dispersed sites, and the extrapolation of measurements from these sites to the whole of the surrounding ocean basin is inevitably uncertain. A further complication is that atmospheric concentrations and deposition fluxes show considerable temporal variability. Therefore, in order to be useful for estimating fluxes, measurements must be obtained over extended periods of time. The few areas for which measurements that satisfy these criteria are available will be considered subsequently.

## 5. *Specific examples of atmospheric fluxes to the ocean*

37. There are limited atmospheric flux data available for trace metals, high molecular weight organic compounds and nutrients in the literature for enclosed seas and open ocean regions. The available fluxes have been obtained by a combination of direct measurements in precipitation and indirect calculations using models applied to measured ambient concentrations, as described earlier. For the details concerning any particular flux values given, the original reference should be consulted. Note also that when ranges of flux values are reported in the original citation, the mid-point is presented in this report. Thus all flux values have uncertainties of at least a factor of 2-3 and in some cases much larger. The geographical areas compared include the North Sea, the western Mediterranean Sea, the Sargasso Sea (near Bermuda in the North Atlantic), Enewetak Atoll at 10°N in the tropical North Pacific, and American Samoa at 14°S in the tropical South Pacific. These sites give a representative span of marine locations, from enclosed seas surrounded by highly populous and industrialized regions to one of the cleanest regions of the atmosphere, the tropical South Pacific Ocean.

38. Atmospheric fluxes of several trace metals to the ocean are presented in Table 4. With the exception of mercury, it is apparent that the metal fluxes to the North Sea and the Mediterranean are generally a factor of 2 to 5 times higher than those to the Sargasso Sea in the mid-latitude North Atlantic. However, an even greater difference of a factor of 4 to 20 is found between the North Atlantic and the tropical North Pacific. The tropical South Pacific is even lower, generally by a factor of 2 to 10 compared with the North Pacific. These flux differences are consistent with the increasing distances of these locations from continental sources, both natural and anthropogenic, for these metals, and with the relatively short atmospheric residence time

(days) of the aerosol particles on which most of the metals are found. Note the relatively small difference in the mercury fluxes at all sites. This is a result of the relatively long (several months at least) atmospheric residence time for mercury, which is primarily found in the gas phase, leading to a rather homogenous geographical concentration and flux distribution for this metal.

39. Atmospheric fluxes of several high molecular weight organic compounds are presented in Table 5. It must be emphasized that all the atmospheric fluxes in Table 5 have very large uncertainties. The determination of organic fluxes is very complex because of the importance of gas exchange for many of the compounds. While concentrations have been reported from several regions, flux estimates are only available for the North Sea (van Aalst, 1982) and North Pacific (Atlas and Giam, 1986). The latter authors report a range of estimates, which in this report have been given as the average of their minimum and maximum ranges. Their flux estimates have been combined with the atmospheric concentrations reported from the same site by these authors to yield an effective deposition velocity. This deposition velocity has then been applied to the concentrations reported for Samoa and the Sargasso Sea (Atlas and Giam, 1986 and references therein) to derive the flux estimates reported in table 5.

40. Even with these large uncertainties, a trend of decreasing atmospheric fluxes can be seen as one moves from the polluted enclosed seas to the clean open ocean regions. However, the overall decrease in flux for any particular organic compound is less than that for the trace metals in Table 4, with the exception of mercury. Again, the smaller decrease of these organic compound fluxes is likely to be related to the longer atmospheric residence time for these substances compared with the metals present on aerosol particles, since most of these organic species are found predominantly in the gas phase.

41. Atmospheric fluxes of the nutrients nitrogen and phosphorus are presented in Table 6. A gradient from the North Sea to the South Pacific is again evident for fixed nitrogen species. This gradient is much less marked for fixed nitrogen than for trace metals, despite the fact that the atmospheric lifetime of these nitrogen species are short (days), as for trace metals. This may reflect important sources for fixed nitrogen in remote marine areas.

## 6. *Research needs*

42. It is obvious from the above discussion that our ability to describe the atmospheric input of metals, organics and nutrients to the oceans is still not complete. In order to improve our estimates, the following are recommended:

- Further work is needed to develop methods to measure directly the flux of both gases and particles to the ocean surface.

areas of the global oceans on isolated islands and by ships and aircraft.

- Atmospheric models have an important role to play in calculating deposition, particularly for inland seas. The first pilot set of deposition values is being developed for the nitrogen compounds. After these results are evaluated for this initial case, the transport of other substances can be evaluated using transport and deposition model techniques.

# II. COMPARISON OF RIVER AND ATMOSPHERIC FLUXES

43. As pointed out in the introduction, one of the aims of this report is to make comparative estimates of the atmospheric and river fluxes of contaminants to specific oceanic basins or regional sea areas. Such information can then be used in mass-balance calculations and is also useful for determining the main principal transport pathways of chemical contaminants. In this section comparisons are made for the North Sea, the North Atlantic and the North Pacific. While it is possible to estimate fluxes by particular pathways in many coastal and pelagic areas, it is in most cases not possible to obtain concomitant estimates of both atmospheric and river inputs. Furthermore, even in the cases of the North Sea and the North Atlantic, the data restrict the estimation of inputs by both routes to a limited number of chemicals. None the less, North Sea data provide a case where atmospheric and river fluxes of several contaminants to a coastal/marginal sea area can be compared, and the North Atlantic data provide a similar case for trace metal transport to a large marine basin which receives considerable inflow of fresh water from rivers, compared with other ocean basins. Finally, comparisons of atmospheric and river fluxes of nutrients are provided for the North Atlantic and North Pacific Basins.

## A. COMPARISON OF SOME TRACE METAL AND SYNTHETIC ORGANIC COMPOUND FLUXES TO THE NORTH SEA

44. Table 7 compares the gross river and atmospheric inputs of selected contaminants to the North Sea. Results of two separate studies are presented. The results of both studies suggest that the atmospheric fluxes of lead, perhaps cadmium, and certain organic species are more important than river fluxes. The relatively large disagreement in the remaining estimated fluxes between the two studies, however, indicates that there is still uncertainty about the most important pathway for the entry of these contaminants to the North Sea. It appears, however, that rivers do not dominate the fluxes to the North Sea.

## B. ESTIMATES AND COMPARISON OF NET RIVER AND ATMOSPHERIC FLUXES OF TRACE METALS TO THE NORTH ATLANTIC

45. The data on metal-salinity relationships (UNESCO, 1987) and on global hydrologic transports (Baumgartner and Reichel, 1975) permit estimation of the net dissolved river influxes of several trace metals into the North Atlantic. These estimates are given in Table 8 and apply to the aggregate dissolved and exchangeable phase transport. It should be noted that the effective zero-salinity projections given in UNESCO (1987) are not available for all Atlantic boundary areas for a uniform set of metals. Therefore, in several cases arbitrary assignments of the effective zero-salinity concentrations of metals have been made for the net flux calculations. Flux estimates are reported in Table 8.

46. The river flux of particulate matter to the open ocean is more difficult to evaluate due to lack of detailed compositional data on river-borne suspended matter. Furthermore, a large fraction of this suspended river load is trapped in the estuaries or in the adjacent coastal zone, and small errors in the estimation of the flux deposited in these areas influences drastically the estimation of the flux of material transferred to the open ocean. Nevertheless rough estimates of these fluxes are calculated by considering:

- the total particulate river flux to the North Atlantic and North Pacific estimated from the data of Milliman and Meade (1983);

- that 95 per cent of that flux is trapped in the estuarine zone and the coastal area and that thus only 5 per cent reaches the open ocean;

- that the elemental composition of the suspended river load corresponds to the mean composition given by Martine and Meybeck (1979).

47. In Table 8 estimates of trace metal fluxes through rivers are compared to estimates of atmospheric trace metal fluxes for the North Atlantic. The atmospheric estimates are based on scaling the estimates for the Sargasso Sea to the whole North Atlantic, and must therefore be interpreted with caution. The ratio of atmospheric flux to net river flux for each trace metal for the North Atlantic Ocean (Table 8), indicates that manganese, iron and nickel have similar and low ratios, consistent with their geochemistry and their relatively small anthropogenic versus natural contributions. Lead, however, shows the highest atmosphere to river flux ratio, indicating that

previously (Jickells *et al.*, 1987). The atmospheric fluxes of zinc and cadmium also substantially exceed those due to rivers, which is consistent with their major input to the atmosphere from industrial sources.

## C. COMPARISON OF ATMOSPHERIC AND RIVER FLUXES OF NUTRIENTS TO THE NORTH ATLANTIC AND THE NORTH PACIFIC

48. The gross river input of nitrogen and phosphorus to the North Atlantic and North Pacific was estimated from the river discharge to the basins and mean concentrations of the dissolved and particulate nutrients in river water (UNESCO, 1987). It was estimated in this report that the transfer of nutrients to the open ocean represents only 1 to 2 percent of the gross river input due to the intensive utilization of these elements by the estuarine and coastal plankton. The atmospheric fluxes are mean values calculated from the range of fluxes estimated by Duce (1986) for these areas. In both regions the atmospheric input of nitrogen species is considerably larger than that estimated for the contribution of river input to the open ocean (Table 9). The atmospheric input is, in fact, comparable to the gross river input of nitrogen. Conversely, the phosphorus contribution of the atmosphere is negligible compared to the river input.

49. An estimate of the new production for the two oceanic areas was also made on the basis of the mean annual primary productivity of Platt and Subba-Rao (1975), and the fractions of new production given by Eppley and Peterson (1979). The new production was expressed in terms of nitrogen and phosphorus fluxes by using the Redfield ratio. These fluxes should represent the total inputs of nutrients to the photic zone. It is interesting to note that the atmospheric input represents a significant fraction of the nitrogen required by the phytoplankton. A change in the atmospheric flux may thus affect significantly the productivity of the open ocean if nitrogen is the limiting nutrient.

## D. COMPARISON OF ATMOSPHERIC AND RIVER FLUXES OF NUTRIENTS TO THE NORTH SEA

50. Estimates of river fluxes, direct discharges and atmospheric fluxes of nutrients in the North Sea leads to the same conclusions as for the North Atlantic and North Pacific (Table 10). The atmospheric input of nitrogen represents a significant fraction of the total input, although the river input has been considerably increased by human activities. The potential importance of atmospheric nitrogen input to the ocean as a source of nutrients has been discussed by Duce (1986).

# III. CONCLUSIONS AND RECOMMENDATIONS

51. From published sources, the workshop compiled the river and atmospheric inputs of metals, organics, and nutrients to the global oceans. These results were combined to produce a preliminary comparison of the importance of river and atmospheric inputs to certain ocean areas. The results suggest that in coastal areas river inputs are generally more important than those from the atmosphere, although in certain areas and for certain substances (e.g. lead in the North Sea) atmospheric inputs dominate. In contrast, in open ocean basins the atmospheric rate for man-made contaminants is generally more important than the inflow via rivers. The group noted that:

- Lack of data is the major obstacle to estimating inputs to the oceans. The atmosphere data base is particularly deficient. Further, there is an almost complete lack of river and atmospheric input data collected simultaneously in the same marine area. Generation of such spatially and temporally coherent data sets should be a high priority.

- A compilation of existing information on atmospheric input to the oceans should be made. This exercise would follow the examples of the report already completed by GESAMP Working Group 22 on river input to the oceans.

TABLE 1. ESTIMATES OF THE GLOBAL ANTHROPOGENIC FLUXES OF DISSOLVED NUTRIENTS IN RIVERS (in $10^9$ kg $y^{-1}$)

| AUTHORS | N | P |
|---|---|---|
| Stumm (1973) | - | 0.6 |
| Lerman *et al.* (1975) | - | 1.8 |
| Soderlund and Svensson (1976) | 18 | - |
| Delwiche and Likens (1977) | 35 | - |
| Van Bennekom and Salomons (1981) | | |
| domestic | 11 | 1.7 |
| agricultural (*) | 8 | 0.35 |
| industrial | 13 | 1.7 |
| Total | 32 | 3.75 |
| Meybeck (1982) | 7 | 1.0 |
| Wollast (1983) | 21 | 1.7 |

(*) Particulate N and P fluxes due to land erosion estimated as 28 $10^9$ kg N $y^{-1}$ and 10 $10^9$ kg P $y^{-1}$ by these authors.

TABLE 2. ESTIMATES OF THE GLOBAL RIVER FLUX OF TRACE METALS IN RIVERS *

| | Gross River Flux (kg $y^{-1}$) | |
|---|---|---|
| | Dissolved | Particulate |
| As | 1 $10^7$ | 7.8 $10^7$ |
| Cd | 3.4 $10^5$ | 1.5 $10^7$ |
| Co | 1.7 $10^6$ | 3.1 $10^8$ |
| Cu | 1 $10^7$ | 1.5 $10^9$ |
| Fe | 1.4 $10^9$ | 7.4 $10^{11}$ |
| Hg | 3.4 $10^4$ | 1.6 $10^6$ |
| Mn | 2.8 $10^8$ | 1.6 $10^{10}$ |
| Pb | 2 $10^6$ | 1.6 $10^9$ |
| Ni | 1.1 $10^7$ | 1.4 $10^9$ |
| Zn | 5.8 $10^6$ | 3.9 $10^9$ |

* (See UNESCO, 1987 for sources of data)

TABLE 3. NET RIVER FLUXES OF TRACE METALS TO THE PELAGIC OCEAN

| | Mn | Ni | Cu | Zn | Cd | Pb | Co |
|---|---|---|---|---|---|---|---|
| Mean zero-salinity intercept ($g\ l^{-1}$) | 5.1 ± 2.6 | 2.0 ± 2.8 | 2.5 ± 1.6 | 1.6 ± 1.2 | 0.18 ± 0.23 | 0.03 ± 0.01 | 0.12 ± 0.15 |
| Mean net flux to the ocean ($kg\ y^{-1}$) | 1.9 $10^8$ | 7.5 $10^7$ | 9 $10^7$ | 6 $10^7$ | 6.7 $10^6$ | 1.1 $10^6$ | 4.5 $10^6$ |
| Mean net flux to the ocean based on Gulf of St. Lawrence mass balance ($kg\ y^{-1}$) | 2.7 $10^9$ | 8.4 $10^7$ | 1.3 $10^8$ | 1.8 $10^8$ | 7.6 $10^6$ | | 2.5 $10^7$ |
| Accumulation rate in pelagic clays ($kg\ y^{-1}$) | 2 $10^9$ | 8.0 $10^7$ | 8.0 $10^7$ | 7.8 $10^7$ | 6.3 $10^5$ | | 3.9 $10^7$ |

TABLE 4. ATMOSPHERIC FLUXES OF TRACE METALS TO THE OCEAN in mg $m^{-2}$ $y^{-1}$

| OCEANS | Pb | Cd | Hg | Zn | Cu | Fe | Mn | Ni | V | REFERENCES |
|---|---|---|---|---|---|---|---|---|---|---|
| North Sea | 8.6 | 0.24 | 0.03 | 14 | 1.7 | - | - | 1.1 | - | Anonymous (1987) |
| | 12 | 0.47 | <0.06 | 52 | 9.5 | - | - | 3.4 | - | van Aalst (1982) |
| Western Mediterranean Sea | 7.3 | 0.3 | 0.024 | 6.0 | - | - | 2.2[a] | - | - | WMO (1985) |
| Sargasso Sea | 1.6 | 0.13 | 0.018[b] | 3.4 | 0.58 | 32 | 1.1 | 0.43 | 0.17 | Jickells *et al.* (1987) |
| Enewetak, North Pacific | 0.07 | 0.0035 | 0.004[b] | 0.67 | 0.09 | 5.6 | 0.09 | - | 0.08 | Arimoto *et al.* (1985) |
| American Samoa, South Pacific | 0.03 | - | 0.007[b] | 0.15 | 0.06 | 0.05 | 0.04 | - | - | Arimoto *et al.* (1987) |

[a] Fitzgerald (1986)
[b] Bergametti (1987)

TABLE 5. ATMOSPHERIC FLUXES OF HIGH MOLECULAR WEIGHT ORGANIC COMPOUNDS TO THE OCEAN in $\mu g\ m^{-2}\ y^{-1}$

| | HCB | HCH | DDT | Chlordane | Dieldrin | PCB | References |
|---|---|---|---|---|---|---|---|
| North Sea | - | 90 | 20 | - | - | 150 | Anonymous (1987) |
| Sargasso Sea | 2.2 | 24 | 0.14 | 0.27 | 2.0 | 21 | Atlas and Giam (1986) |
| Enewetak, North Pacific | 1.45 | 16 | 0.14 | 0.12 | 1.1 | 1.5 | Atlas and Giam (1986) |
| American Samoa, South Pacific | 0.80 | 2.0 | 0.03 | 0.009 | 0.22 | 0.37 | Atlas and Giam (1986) |

TABLE 6. ATMOSPHERIC FLUXES OF FIXED NITROGEN AND PHOSPHORUS TO THE OCEAN in $mg\ m^{-2}\ y^{-1}$

| | Fixed Nitrogen | Phosphorus | References |
|---|---|---|---|
| North Sea | 688 | | Anonymous (1987) |
| Western Mediterranean Sea | | 32 | Bergametti (1987) |
| Sargasso Sea | 133 – 276 | 0.38 – 0.93 | Duce (1986) |
| North Pacific Gyre | 41 – 133 | 0.06 – 0.14 | Duce (1986) |

TABLE 7. COMPARISON OF CONTAMINANT FLUXES TO THE NORTH SEA

| | $10^6$ kg $y^{-1}$ | | | | | | | $10^3$ kg $y^{-1}$ | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Cd | Cu | Cr | Hg | Ni | Pb | Zn | PAH | PCB | HCH, HCB |
| DATA FROM (1) | | | | | | | | | | |
| Gross River Input | 0.23 | 1.6 | 1.9 | 0.11 | 1.4 | 2.4 | 17 | 50 | - | 7 |
| Atmospheric Input | 0.27 | 5.7 | 0.75 | 0.04 | 2.0 | 8.3 | 33 | 800 | 95 | 45 |
| Atmospheric Flux/River Flux | 1.2 | 3.5 | 0.39 | 0.36 | 0.70 | 3.4 | 1.9 | 16 | - | 6.4 |
| DATA FROM (2) | | | | | | | | | | |
| Gross River Input | 0.049 | 1.3 | 0.6 | 0.02 | 0.25 | 0.9 | 7.6 | - | 3.3 | 3.3 |
| Atmospheric Input | 0.14 | 1.0 | 0.6 | 0.02 | 0.62 | 5.0 | 8.0 | - | - | - |
| Atmospheric Flux/River Flux | 2.9 | 0.8 | 1.0 | 1.0 | 2.5 | 5.5 | 1.1 | - | - | - |

(1) Van Aalst *et al.* (1982)

(2) Report of the Scientific and Technical Working Group on the North Sea. Submitted to the Second Ministerial Conference on the North Sea 1987. Anonymous (1987).

TABLE 8. COMPARISON OF TRACE METAL FLUXES TO THE NORTH ATLANTIC OCEAN

| | NET RIVER DISCHARGE ($10^5$ $y^{-1}$) | NET METAL INPUTS ($10^6$ kg $y^{-1}$) | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Mn | Ni | Cu | Zn | Cd | Pb | Co | Fe |
| RIVERINE INPUT | | | | | | | | | |
| From N. America | 3.6 | 14 | 4.7 | 6.5 | 6.5 | 0.7 | 0.14 | 0.18 | |
| From S. America | 3.8 | 15* | 0.87 | 6.1 | 6.8* | 0.04 | 0.15* | 0.19* | |
| From Europe | 4.0 | 31 | 38 | 23 | 7.2* | 2.6 | 0.16* | 0.20* | |
| NET DISSOLVED INPUT | 11.4 | 60 | 44 | 36 | 21 | 3.7 | 0.45 | 0.57 | - |
| NET PARTICULATE INPUT | | 130 | 11 | 13 | 31 | 0.13 | 13 | 2.5 | 6,000 |
| TOTAL RIVERINE INPUT | | 190 | 55 | 49 | 52 | 3.8 | 13 | 3.1 | 6,000 |
| DISSOLVED/PARTICULATE | | 0.46 | 4.0 | 2.8 | 0.68 | 28 | 0.03 | 0.22 | - |
| TOTAL ATMOSPHERIC INPUT | | 65 | 21 | 29 | 170 | 6.5 | 80 | - | 1.600 |
| TOTAL INPUT TO N. ATLANTIC** | | 260 | 76 | 78 | 220 | 10.3 | 93 | - | 7,600 |
| Atmospheric Flux/River Flux | | 0.34 | 0.38 | 0.59 | 3.3 | 1.7 | 6.1 | - | 0.27 |

* Values estimated using North American net river concentrations.

** Based on values given in this Table.

TABLE 9. FLUXES OF NUTRIENTS IN THE NORTH ATLANTIC AND NORTH PACIFIC BASINS in mg $m^{-2}$ $y^{-1}$

| | North Atlantic | | North Pacific | |
|---|---|---|---|---|
| | N | P | N | P |
| RIVER INPUT | | | | |
| Gross Input | 340 | 200 | 140 | 80 |
| Net Input (1.5%) | 5 | 3 | 2 | 1 |
| ATMOSPHERIC INPUT | 200 | 0.7 | 85 | 0.1 |
| NEW PRODUCTION | 1,300 | 170 | 365 | 50 |

TABLE 10. FLUXES OF NUTRIENTS TO THE NORTH SEA IN $10^9$ kg $y^{-1}$ (*)

| | N | P |
|---|---|---|
| River Input | 1,010 | 76 |
| Direct Discharge | 90 | 22 |
| Total | 1,100 | 98 |
| Atmosphere | 400 | - |

(*) Source: Report of the Scientific and Technical Working Group on the North Sea. Submitted to the Second Ministerial Conference on the North Sea 1987 (Anon.,1987).

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# ANNEX X

# MARINE HEALTH HAZARDS OF ANTHROPOGENIC AND NATURAL ORIGIN

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# INTRODUCTION

1. Contaminants are transported from land to sea by rivers, direct run-off or point discharges. They are partly retained in the water and sediments of estuaries, bays, beaches or open coastal waters, and partly transported through these boundary areas to the open ocean. The disposal of domestic sewage and industrial effluents from population and industrial centres located along the coastlines extends coastal pollution beyond the estuaries. Moreover, coastal waters are not only more polluted than the open ocean, they also offer significantly more opportunities for exposure through contamination of the world's most important fisheries and recreation areas.

2. However, anthropogenic emissions are not the only reason for the aggregation of human health hazards in coastal waters. There are toxic substances that are ubiquitous in the oceans (e.g. methylmercury) and microbiological agents that are part of the bacterial flora of estuarine waters (e.g. *Vibrio cholerae*). Anthropogenic emissions can only increase their concentrations to hazardous levels. Other biological and toxic agents have no anthropogenic input at all. Within the genus *Vibrio*, the halophilic (adapted to salt) vibrios, many of them pathogenic for humans, belong to the normal flora of coastal and offshore waters. A toxicologically important natural coastal phenomenon is the algal bloom. When the bloom is toxic, the biotoxin is mainly accumulated by bivalve molluscs or reef and semi-pelagic fishes. The scenes of attacks by venomous marine animals are also mostly coastal and reef areas. However, human activities may have some influence even on the production of marine biotoxins.

3. This review will deal with thalassogenic (i.e. associated with the sea) health hazards, irrespective of origin (natural or anthropogenic). It covers both biological and chemical agents and concentrates on exposures associated with the use of recreational areas and with the consumption of seafood. The first part deals with microbiological agents, extensively discussed in UNEP Regional Seas Report and Studies No. 79 (Shuval, 1986). The second part deals with toxic agents. The section on toxic metals is based on two GESAMP (1985 and 1986) documents, whilst the WHO (1984) health criteria document No. 37, was consulted for the relevant part on marine biotoxins.

# I. MICROBIOLOGICAL HEALTH HAZARDS

## A. SOURCE, PRESENCE AND SURVIVAL OF MICRO-ORGANISMS

4. The sea can contain a number of viral, bacterial and fungal agents, many of them pathogenic for man. The source of most of these pathogens is faecal and the vehicle of transport to rivers, estuaries and coastal water is domestic sewage. Even a clinically healthy individual excretes in one g of faeces more than one million infectious particles consisting of more than 100 different viruses and bacteria, and in raw sewage the concentration can be as high as 500,000 particles per litre (Wallis *et al.*, 1979). However, there are several micro-organisms that are pathogenic for humans and indigenous in the marine environment (e.g. *Clostridium botulinum* type E) and, therefore, exist in the sea in the absence of sewage disposal.

### *1. Sewage treatment*

5. A major problem on a world-wide scale is the existence of pathogenic organisms discharged with the domestic sewage from rapidly expanding coastal communities to coastal waters or estuaries. According to Calvert (1975) the discharge of untreated sewage to an adequate depth of water is the most cost-effective disposal method when (1) the configuration of the coast is such that the required dilution can be achieved and (b) currents prevent the contamination of bathing beaches. In the absence of such favourable conditions there is a case for treatment in sewage plants by mechanical (including filtration), biological (to speed up degradation) and chemical (to precipitate and flocculate) methods. The efficiencies of different sewage treatments for different microbial agents are different. According to Bonde (1975) while filtration reduced *Clostridium perfringens* and *Escherichia coli* nearly to the same extent (by 83 and 95 per cent), sedimentation decreased the counts of *Cl. perfringens* by 63 per cent and those of *E. coli* only by 10 per cent. Conventional sewage treatment has less effect on viruses than on bacteria and even chlorination may not remove live viruses completely. Removal of viruses by adsorption to solids results in high concentration in sludge which may be disposed of to the sea (Goyal *et al.*, 1984).

### *2. Survival of micro-organisms in sea water*

6. In sea water there is a significant difference in the survival of faecal microbes. A decrease

requires 2.5, 7.5 and 12.5 days, respectively (Dufour, 1984). It has been shown that the concentration of enteric bacteria decreases with increasing distance from the sewage outfall faster than the concentration of enteric viruses (Loh *et al.*, 1979), and solid-associated enteric viruses can be found in substantial amounts in shellfish waters approved on the basis of low coliform levels (Rao, 1986). The main factors responsible for the disappearance of enteric bacteria from sea water are: salinity, temperature, antibiosis (the anti-bacterial effect of indigenous micro-organisms), heavy metals, solar radiation, and predation by protozoa, grazers or filter feeders. Unlike bacilli (rod-shaped bacteria) or cocci (spherical bacteria), *Vibrio cholerae* can survive for a long period of time and some vibrios are indigenous to sea water (Blake *et al.*, 1980b).

7. Viruses are protein-coated genetic material. They can multiply only within the cells of a host, but can survive outside cells for long periods even under unfavourable conditions. Adsorption of viruses to solid particles helps viral survival. Thus, viruses of faecal origin were isolated from sediments even 17 months after the cessation of sludge dumping (Goyal *et al.*, 1984). The survival of solid-associated enteroviruses in recreational and shellfish-growing areas has public health implications which are frequently unrelated to the bacterial indicators of pollution (Wallis *et al.*, 1979; Steinman, 1987). Experiments indicated the nearly complete (over 99 per cent) adsorption of enteric viruses to sediment. Viruses in the sedimental layer are not immobile. An important factor in their transport is the resuspension of sediments at current speeds of 6 cm $s^{-1}$ or more (LaBelle and Gerba, 1979).

## B. THE CHARACTERIZATION OF HUMAN PATHOGENS IN SEA WATER OR SEAFOOD

### 1. *Pathogens of faecal origin*

#### (a) *Salmonella*

8. All these rod-like micro-organisms are able to cause acute gastro-enteritis and focal manifestations. Generalized infection with fever is caused by *S. typhi* (typhoid fever). Infection by *S. paratyphi* can mimic typhoid fever. It has been suggested that the minimal infective dose of *S. typhi* is 3-5 organisms. Besides these two salmonellae there are over 1,000 serotypes that can cause typhoid-like infections in their natural animal host and food poisoning in man. The infective dose in food poisoning may be 100 million bacilli or over. *S. typhimurium*, *S. enteritidis*, *S. heidelberg* and *S. newport* are important members of this group. In sediments the frequency of occurrence of salmonellae is higher than in surface water (Sayler *et al.*, 1976). Pollution of both recreational waters and seafood has caused illness (Shuval, 1986).

### (b) *Vibrio cholerae*

9. In spite of identical biochemical characteristics, these short curved aerobic cells, propelled by one or more flagellum, are divided into *V. cholerae* 0-group 1 and non-0-group 1. Vibrios in the latter group do not agglutinate in *V. cholerae* O-group 1 anti-serum. *V. cholerae* have been isolated with increasing frequency from estuarine and sea water where a few can survive $10^{\circ}C$ temperature. They can survive 2.5 per cent salinity for up to 42 days and have a preference for moderate salinity between 1.5 and 2.5 per cent (Singleton *et al.*, 1982).

#### (i) *Vibrio cholerae* 0-group 1

10. These vibrios are the causative agents of Asiatic or epidemic cholera. There are two biotypes - classical and El Tor - both of which may contain toxigenic and non-toxigenic strains. The infective dose for *V. cholerae* is $10^9$ cells and only 1 in 25-100 infections by the El Tor biotype and in 5-10 infections by the classical biotype results in severe diseases (cholera gravis). Though *V. cholerae* O-group 1 is widely distributed and is probably part of the indigenous bacterial flora in estuarine waters (see Hackney and Dicharry, 1988), outbreaks require the faecal input of *V. cholerae* in sewage and the accumulation of vibrios in seafood (Blake *et al.*, 1980b; Shuval, 1986).

#### (ii) *Vibrio cholerae* non-0-group 1

11. It is a free-living orgânisms and part of the autochtonous flora of bays and estuaries with salinity in the area of 0.4 to 1.7 per cent. Strains are regularly isolated from the water of Chesapeake Bay in late fall and early spring but not in winter (Colwell *et al.*, 1978), and from the Baltic Sea (Blake *et al.*, 1980b). They are not recognized as a serious epidemic threat, although they have caused cholera-like diarrhoea sporadically. In the absence of faecal input from patients, the majority (more than 95.5 per cent) of the strains isolated from seafood is non-toxigenic. The source of human infection is almost exclusively associated with eating raw oysters (Hackney and Dicharry, 1988).

### (c) *Enteric viruses*

12. Hepatitis A virus, non-A-non-B hepatitis virus, Norwalk virus and rotavirus are important members of this group. Others are Snow Mountain agent, calcivirus and astrovirus. Uptake by shellfish prolongs their survival without multiplication, and has a significant role in transmission (Gerba, 1988). Rotavirus, a causative agent of infantile diarrhoea, was implicated in swimming-associated gastroenteritis (Shuval, 1986). Unlike these viruses, those belonging to the enterovirus group (different types of polio-, echo-, and Cocksackie A and B viruses) are commonly found in human faeces and therefore their presence in seafood indicates faecal contamination

## 2. *Pathogens indigenous to the marine environment*

### (a) *Clostridium botulinum* type E

13. Type E is frequently isolated from marine sediments in temperate zones and occurs fairly regularly in small numbers in fish. Boat-docking facilities and fish-processing plants have been found highly contaminated with spores. Type E spores germinate to form vegetative cells and toxin at minimum temperature (3-4 $^{o}C$), lower than the minimum growth temperature of 10$^{o}C$ for other types. Like other types of *Cl. botulinum*, type E grows and produces the lethal neurotoxin only when foods are neither acid nor alkaline, and when most other bacteria (the oxygen producers) are eliminated and oxygen (air) is excluded. The antigenic nature of the type E toxin is different from other botulinum toxins. (Lewis and Cassel, 1964; Schantz, 1973; Liston, 1980).

### (b) *Halophilic vibrios*

14. The group includes vibrios, indigenous to coastal waters and estuaries, which cause human infections through the consumption of seafood or by direct contact with wounds. The best known human pathogens are *V. alginolyticus*, *V. parahaemolyticus* and *V. vulnificus*. Others are *V. damsela*, *V. fluvialis*, *V. furnissii*, *V. hollisae* and *V. mimicus*. Only *V. parahaemolyticus* and *V. vulnificus* ferment lactose. *V. alginolyticus* can survive in 10 per cent NaCl solution while *V. parahaemolyticus* and *V. mimicus* were found not only in sea water, but also in freshwater (Blake *et al.*, 1980b; Hackney and Dicharry, 1988). *V. mimicus* was previously identified as an atypical non-0-group 1 *V. cholerae*. In fish caught in the coastal waters of Senegal the most frequently isolated halophilic vibrio was *V. parahaemolyticus*, followed by *V. alginolyticus*, *V. vulnificus*, and *V. fluvialis*. Counts reached $10^5$ $g^{-1}$ in the flesh (Schanderyl *et al.*, 1984). Based on biochemical characteristics and virulence in mice, it has been suggested that there are many more potentially human pathogens among unidentified vibrio species (Oliver *et al.*, 1983).

#### (i) *Vibrio alginolyticus*

15. Its normal habitat is sea water and it has been isolated from sea water and seafood in many parts of the world. It is abundant in summer and disappears in winter. Disease caused by this vibrio is usually not severe (Blake *et al.*, 1980b).

#### (ii) *Vibrio vulnificus*

16. Its normal habitat is sea water, where its numbers are generally low. The chance of isolation showed a negative correlation with salinity and a positive correlation with pH, turbidity, presence of vibrios in the sediment and oysters, and total bacterial counts in oysters. The majority

mostly clams (Oliver *et al.*, 1983). Of the 29 isolates from geographically different marine sources, 25 were pathogenic to mice and all had the same cytolysin and cytotoxin level (Tison and Kelly, 1986). *V. vulnificus* produces septicaemia after ingestion, or wound infection after surface exposure. Because it is one of the most invasive species and the outcome of infection is often lethal, *V. vulnificus* is called the new "terror of the deep" (Hackney and Dicharry, 1988; Blake *et al.*, 1980b).

(iii) *Vibrio parahaemolyticus*

17. The salinity of full-strength sea water is inimical to this species which is widely distributed in coastal waters and a common contaminant of seafood. Vibrios surviving winter in sediments are released to water when temperature rises to 14 to 19°C. They are associated with plankton, mainly zooplankton, when the temperature is higher (Kaneko and Colwell, 1975). Only a small per cent is virulent enough to produce heat-stable haemolysin in a special agar medium (Kanagawa positivity) and is therefore pathogenic (Blake *et al.*, 1980b).

(iv) *Other halophilic vibrios*

18. *V. mimicus* is widely distributed in nature and can be found in fresh as well as in brackish waters. *V. damsela* is an important pathogen for damsel fish. The ecology of *V. fluvialis, V. furnissii and V. hollisae* are not well understood. All these pathogens are less virulent than *V. vulnificus* (Hackney and Dicharry, 1988; Morris *et al.*, 1982).

### *(c) Genus Beneckea*

19. These straight or curved bacteria are, like vibrios, motile by means of flagella. They can be readily isolated from ocean water and from the surfaces and intestinal content of fish. The most extensively studied marine species is *B. parahaemolytica* (Baumann and Baumann, 1977).

### *(d) Parasitic worms*

20. *Anisakis simplex* and *Pseudoterranova decipiens* are nematodes belonging to the *Ascaroidea* family. These anisakid worms occur in the intestines of aquatic birds or fish eating vertebrates (e.g. seals), and the larvae in the connective tissue of muscles of fish. The most frequently implicated fish are: herring, Pacific salmon, Pacific rockfish (marketed as Pacific red snapper). The prevalence of larval anisakid nematodes in various fish caught along the U.S. Pacific Coast may be higher than 80 per cent (McKerrow *et al.*, 1988). In Germany, the average number of larval nematodes in 1 kg fish was given as 1 in rock fish, 2 in cod, 6 in saithe (coalfish) and 40

# C. THE MICROBIOLOGICAL HAZARDS OF MARINE RECREATIONAL WATERS

## *1. Exposure characteristics*

21. The major source of marine contamination is domestic sewage, though bathers can also contaminate the water. Exposure to contaminated water depends on the form of recreation. There is an important difference within beach-going populations between the exposures of swimmers and non-swimmers. Only swimming results in the prolonged exposure of upper body orifices to water. Swimmers also ingest water and, according to Shuval (1974), 10 ml per bathing day is a conservative estimate. Swimmers in cold Scandinavian beaches may ingest less, but children on a Mediterranean beach may spend as many as 2-4 full hours in water and ingest 100 ml sea water during the day. Moreover, young children paddle in shallow water where sedimented viruses are most likely to be resuspended during bathing time. Though the ingestion of polluted sea water is the most important source of microbial intake, contact of polluted water with ear, eye, nose, urogenital orifices, and the inspiration of water droplets can result in non-gastric infections. Wounds can be infected by microbes in water or from the surface of fish.

## *2. Case reports*

22. Shuval (1986) reviewed several case reports of infectious diseases associated with bathing in contaminated waters. The pathogens were *Salmonella typhy*, *Shigella sonnei* and viruses like coxsackie A and B, hepatitis A, human rotavirus, Norwalk agent or parvolike virus. With the exception of six reports, all the others quoted by Shuval (1986) were of non-marine origin (swimming pools, rivers or lakes). Many of these studies suffered from cardinal weaknesses in the identification of the aetiological factors and sources, or in the elimination of other possibilities. The six publications which dealt with marine pollution described typhoid fever outbreaks following bathing in sewage-polluted sea water near New Haven (in 1921, 1929) and New York Harbour, U.S.A. (1932), Australia (1959), Tel Aviv, Israel (1972) and Alexandria, Egypt (1983).

23. Infection of the external ear canal is more common during the summer swimming season and it is often called swimmer's ear. Irritation is a known predisposing factor and the causative agent may be *Pseudomonas aeruginosa*, *E. coli* or *Proteus vulgaris* (Berkow, 1977). According to Cabelli *et al.* (1975), *Pseudomonas aeruginosa* is the major aetiological agent. *V. alginolyticus* was isolated from the ears of eight patients (Blake *et al.*, 1980) and *V. mimicus* from the ears of two

*damsela*, *V. parahaemolyticus*, and *V. vulnificus* infected pre-existing wounds or lacerated skin, or the infection was linked to wounding by clam shell or puncturing of the leg while cleaning fish (Blake *et al.*, 1980b; McMeeking *et al.*, 1986, Morris *et al.*, 1982; Oliver *et al.*, 1986; Tyring and Lee, 1986). Three of 18 patients with *V. vulnificus* infection died. The onset of the illness started 12 hours after bathing in contaminated sea water in most cases (Blake *et al.*, 1979). Wound infections were also caused by *B. parahaemolyticus* (Baumann and Baumann, 1977). Brisou (1975) reported increased incidence of vaginal *Candida* infections in users of marine beaches.

### 3. *Epidemiological studies*

24. As far as the range of pathogens are concerned, epidemiological studies were more restrictive than case reports. Either a disease specific to a pathogen (e.g. paratyphoid fever, poliomyelitis), or a non-specific clinical syndrome (e.g. gastrointestinal disorder) was selected. The aim of such studies was not only to estimate incidence rates, but to relate incidence rate to sewage pollution through the microbiological contamination of recreational waters. Owing to methodological difficulties, the responsible microbiological agents were not isolated from the patients and/or quantitated in recreational waters, and illness was related to the concentration of indicator micro-organisms.

25. Shuval (1986) reviewed the first two systematic epidemiological studies, the U.K. Public Health Laboratory Study by Moore and the U.S. Public Health Service Study by Stevenson. The U.K. Public Health Laboratory Study (carried out between 1953 and 1959) compared the frequency of bathing in polluted sea water among 3,000 cases of paratyphoid fever or 150 poliomyelitis patients on the one side, and control groups matched for age, sex, and social class on the other. The conclusion was that bathing in sewage-polluted sea water carried a negligible risk to health. This study was criticized on many points, but mainly because it was too restrictive so that its conclusion might not be valid for other diseases. This critical comment has been proved correct by new epidemiological studies.

26. The U.S. Public Health Service Study was published in 1953 and concluded that disease incidence significantly increased at total coliform concentrations in excess of 2,400 $10^{-2}$ ml water. Another conclusion was that the increase in ear, nose and throat infections in swimmers was independent of the microbiological quality of the water. This study has been seriously criticized because opposite findings and day-to-day variations in pollution levels were ignored, no distinction was made between skin irritation and diarrhoea, and swimming was not defined rigorously. Inspite of these faults in design, the basic conclusions of the study were later confirmed.

27. A U.S. Environmental Protection Agency Study, summarized by Cabelli *et al.*, (1983) and by Dufour (1984), aimed to establish the relationship between the concentrations of microbiological agents at three marine beaches during the time of maximum swimming activities

(between 11:00 a.m. and 5:00 p.m.) and the incidence of gastro-intestinal disorders in the following week. Swimming was defined as the exposure of upper-body orifices. The first part of the study was carried out at two New York beaches with the aim to find the best bacterial indicators. Participants were recruited as family groups at beaches on weekends. Individuals who swam in mid-week or spent more than two days on the beach were eliminated. From New York beach-goers 15,882 usable responses were obtained during the three-year study. 'Highly credible gastro-intestinal symptoms' were defined as including at least one of the following groups of symptoms: (a) vomiting; (b) diarrhoea accompanied by fever or with symptoms disabling enough for the individual to remain home, stay in bed, or seek medical advice; or (c) stomachache or nausea accompanied by fever. Highly credible gastro-intestinal symptoms were equated with gastro-enteritis. The overall incidence of swimming-associated gastro-enteritis was 1.52 per cent. Gastro-enteritis rates correlated best with *Enterococcus* concentrations, *E. coli* was a poor second, and faecal coliforms showed no correlation. The freshwater study confirmed this finding. The regression of gastro-enteritis rate per 1,000 persons (Y) with *Enterococcus* counts per 100 ml water (X) was defined as:

$$Y = 0.2 + 12.17 \log X$$

and for total gastro-intestinal symptoms as:

$$Y = 5.09 + 24.19 X$$

28. The incidence of eye, ear, nose and throat ailments was higher in swimmers than in non-swimmers, but the increase was independent of the quality of water. Thus, the EPA study confirmed the conclusions of the Stevenson - U.S. Public Health Study.

29. The design of the epidemiological study of Fattal *et al.* (1987) was basically the same. Families on three beaches near Tel Aviv were recruited during weekends, when water samples were collected for the quantitative estimation of six bacteria. Bacterial concentrations for each day were categorized as 'low' or 'high', depending on whether they were below or above the overall median. The responses of 609 families covering 2,231 persons were evaluated. A swimmer was defined as a person whose head had been immersed in water or who had swallowed sea water. Swimming increased the incidence of respiratory, ear and skin infections regardless of the microbial quality of water.

30. Results indicated that in the 0-4 year age group significantly more swimmers exposed to 'high' density of *Enterococcus* (24-410 $10^{-2}$ $ml^{-1}$) had symptoms of enteric morbidity than non-swimmers or swimmers exposed to 'low' density. These significant differences were not observed in the total population because in the older age groups the effect of swimming was not consistent and the effect of exposure was not significant. However, when predicted incidence rates were calculated from the regression line of the EPA study (Cabelli *et al.*, 1983), observed incidence

and predicted incidence rates showed reasonable agreement. The average *Enterococcus* densities on the three beaches studied by Fattal *et al.* (1987) were 10, 15 and 41 per 100 ml water. Based on these counts the predicted incidence of swimming related to gastro-intestinal symptoms were 2.9 - 4.4 per cent. The observed difference between the incidence rates of swimmers (9.2 per cent) and non-swimmers (6.8 per cent) was 2.4 per cent in the total population and 6.5 per cent (17.8 - 11.3 per cent) in the 0-4 year age group.

31. These epidemiological studies demonstrated unequivocally that swimming in polluted sea water had increased the incidence of gastric disorders and that the increase correlated with *Enterococcus* counts. The studies also indicated that the incidence of non-gastric disorders in swimmers increased without any correlation with *Enterococcus* counts or other indicators of sewage pollution. Though *P. aeruginosa* is the most commonly isolated organism from cases of external otitis, the bacterial indicators of water quality, including *P. aeruginosa*, do not correlate with the incidence of this illness in swimmers (Shuval, 1986). An explanation for the lack of correlation between swimming-associated non-gastric illnesses and the microbiological quality of sea water is not yet at hand. It may be that pathogens responsible for non-gastric disorders have a longer survival time and consequently show less fluctuation in their concentration in sea water than either enteric pathogens or indicator bacteria. For example, the concentrations of halophilic vibrios are not indicated by the enterobacterial indicators and they are able to produce ear and wound infections (Blake *et al.*, 1980b). However, it is possible that exposure of the upper body orifices to sea water activates the bacterial flora of the swimmer or decreases resistance to infection.

## *4. The microbiological control of recreational waters*

32. The recognition of the role of the faecal route in the transmission of enteric pathogens led to the application of the indicator concept for developing standards for recreational waters. The first three indicators suggested were *E. coli*, *Streptococcus faecalis* and *Cl. perfringens*. Methodological considerations led to the use of total coliforms and faecal coliforms instead of *E. coli*, faecal streptococci instead of *S. faecalis* and sulfite-reducing anaerobes instead of *Cl. perfringens*. Though classifications based on total coliforms counts agreed well with sanitary survey classifications, the total coliform count was replaced by faecal coliforms because some genera within the total coliforms (*Klebsiella*, *Citrobacter* and *Enterobacter*) are not faecal specific (Cabelli *et al.*, 1983). Thus, both the WHO/UNEP and the EPA guidelines used faecal coliforms. The WHO/UNEP (1977) guidelines recommended that bathing water should not contain more than 100 faecal coliforms per 100 ml of water in 90 per cent of the samples or not more than 1,000 per 100 ml in 10 per cent of the samples. USEPA (1976) recommended that the geometric mean counts of faecal coliforms should not exceed 200 per 100 ml water in at least 5 samples taken within 30 days, or 400 faecal coliforms per 100 ml water in 10 per cent of the samples.

33. The case against microbial standards was presented by Moore (1975). He argued that (1) bathing beaches cannot be satisfactorily graded in terms of coliform or faecal coli counts, (2) no good evidence of a quantitative relationship between coli counts and health risk is available, and (3) the allocation of resources needed to check conformity to an imposed standard cannot be justified on public health grounds. The EPA study (Cabelli *et al.*, 1983) and the report of Fattal *et al.* (1986) supported the first two contentions of Moore. Swimming in waters of acceptable quality (based on faecal coliforms) increased the incidence rate of enteric disorders and therefore faecal coliforms do not satisfy the characteristics which justify their use as indicators of risk to health.

34 The ideal characteristics of bacterial indicators, which would allow the analysis of risk associated with the use of recreational waters are: (1) to be present where pathogens, (2) to be unable to grow in the aquatic environment, (3) to be more resistant to disinfection than pathogens, (4) to be easy to isolate and enumerate, (5) to be absent from sources other than sewage or be exclusively associated with sewage, (6) and that the indicator density should correlate with health hazard from a given type of pollution (Dufour, 1984). The finding of the EPA study (Cabelli *et al.*, 1983) proved that faecal coliforms are not good indicators of health risk, but the study presented good evidence of a quantitative relationship between *Enterococcus* and the risk of gastro-enteritis. Thus, *Enterococcus* counts, unlike faecal coli counts, allow the grading of beaches. Though *Enterococcus* seems to be the best presently available bacterial indicator, enteric bacteria do not indicate the risk of non-gastro-intestinal disorders. Further research may find a more human-specific and even more resistant faecal indicator or an indicator of non-enteric illnesses.

35. The correlation between *Enterococcus* counts and enteric disorders does not answer the third argument of Moore (1975) against microbial standards for bathing beaches. Increases in the incidence of a benign and self-limiting illness may be more acceptable than the allocation of resources to check and assure conformity to a standard. Nevertheless, others point out that the use of *Enterococcus* counts for grading beaches helps risk appreciation and allows considered decision making (Cabelli *et al.*, 1983).

## D. MICROBIAL PATHOGENS OF MARINE ORIGIN IN SEAFOOD

36. The contamination of seafood with pathogenic micro-organisms, either naturally present in the sea or derived from sewage, is often responsible for acute gastro-intestinal and other disorders (Gerba and Goyal, 1978; Cliver, 1984). Though fish and crabs that live in water contaminated by domestic wastes are frequently found to contain enteric bacteria and viruses (Gerba, 1988), an overwhelming majority of the reported food-borne outbreaks of infectious diseases are attributable to contaminated molluscan shellfish (Shuval, 1986).

### *1. Accumulation and elimination*

#### *(a) Crabs*

37. Compared with molluscs, accumulation of micro-organisms by crabs has attracted little attention. Viral concentrations in the blue crab (*Callinectus sapidus*) were always below concentrations in water. The highest concentration was always in the haemolymph and the digestive tract. Meat also contained viruses, but because muscle is not isolated from the digestive tract, contamination by haemolymph and digestive tract could not be ruled out (Heikal and Gerba, 1981). Other studies indicated that the muscle of crabs collected from cold waters have bacterial concentrations three orders of magnitude lower than the gill.

38. It has been shown that crabs have an efficient defense system that limits the bacterial contamination of the muscle tissue. However, muscle becomes susceptible to contamination with bacteria, including human pathogens, when the defense system is impaired by oxygen depletion or the crab is injured and dies (Faghri *et al.*, 1984).

#### *(b) Shellfish*

39. The reason why shellfish is the most important vehicle for the transmission of marine-borne microbial pathogens is that (a) shellfish are frequently cultivated in the vicinity of urban centres with inadequate wastewater treatment and disposal systems; (b) shellfish are usually eaten either raw or after gentle cooking; (c) the feeding activity (filter-feeding) of bivalve organisms, such as molluscs, clams and oysters, involves the circulation of large volumes of sea water through the shell cavity (Metcalf, 1982). Owing to this feeding activity, shellfish concentrations of pathogenic

oyster meat contained 6 to 224 viruses, 70 coliforms and 2 faecal coli, the respective numbers in 100 ml water over oyster beds were 4 to 167, 3 and less than 2 (Goyal *et al.*, 1979). Experimental studies indicated that the capacity of molluscs to concentrate microorganisms is high but finite.

40. Rock oysters (*Crassostrea glomerulata*) accumulated 4 $10^{10}$of 9.2 $10^{10}$reovirus particles in 600 ml static water system. Higher concentrations only increased shell adsorption, but not tissue concentration (Bedford *et al.*, 1978). Soft-shelled clams (*Mya arenaria*) placed in a tank for 6 h accumulated 28-35 per cent of the added faeces-associated enterovirus. After this relatively short exposure the body concentration of viruses was 19-34 times higher than the concentration in water. Tissues which participate in accumulation and elimination (hepatopancreas and siphon) had the highest concentration (Metcalf *et al.*, 1979). Though the source of viral contamination in actively pumping shellfish is particle-associated virus, bioaccumulation from undisturbed sediments is negligible. Bioaccumulation only represents a significant threat when sediment is suspended in the water column either by current or perhaps by the feeding activities (opening and closing the shell) of molluscs (Landry *et al.*, 1983). In model experiments, rock oyster accumulated and eliminated coli faster than reovirus. The clearance of *E. coli* both from water by the oyster (accumulation) and from the oyster to water had a 3 h half-time, while the corresponding rates were 9-12 and 9 h for reovirus (Bedford *et al.*, 1978).

41. The concentration factor for *E. coli* in hard clams (*Mercenaria mercenaria*) varied between 6.5 and 8.5, depending on the bacterial count in water (Cabelli and Heffernan, 1970). Hard clams exposed for 15 minutes at 20$^{o}$C to 5 $10^4$ *E. coli* or *S. typhimurium* per ml water accumulated both bacteria to the same extent. The incorporation of 0.5 to 1.0 $10^4$ bacteria per clam resulted in 1-2 $10^5$ $g^{-1}$ tissue concentrations. Accumulation and clearance depended on temperature, the microorganism and the host. Accumulations and clearance were slower at 6$^{o}$ than at 20$^{o}$C, and *Sh. flexneri* was eliminated faster than *E. coli* or *S. typhimurium* (Hartland and Timoney, 1979). In the first few hours after a 15 minutes exposure, the counts of *E. coli* and *S. typhimurium* in the hard clam declined at the same fast rate, but the initial fast elimination phase was longer for the former than for the latter. By 24 h, bacterial counts had fallen by factors of $10^4$ and $10^3$, respectively. The higher persistence in *S. typhimurium* may have contributed to the difference through the re-ingestion of faecal and pseudo-faecal particulates by pumping clams (Timoney and Abston, 1984). American oysters (*Crassostrea virginica*) absorbed more *S. typhimurium* and eliminated this bacterium more slowly than clams (Hartland and Timoney, 1979). *Sh. flexneri* was removed from oysters within 24 h, while the excretion of *S. typhimurium* remained at detectable levels up to 14 days and below detection level perhaps up to 7 weeks (Janssen, 1974).

## 2. *Epidemiology*

42. As shellfish are often supplied from one polluted source to restaurants and markets over a wide area, many outbreaks remain unrecognised. In 1973, 278 people developed hepatitis A in Houston, Texas and Coulhun, Georgia, U.S.A.. When these patients were investigated carefully, it was found that they had eaten oysters at one of 11 restaurants served by a single supplier who had harvested them from waters which had been closed to harvesting earlier owing to faecal contamination (Coulepis *et al.*, 1987). Another outbreak of clam-associated infectious hepatitis numbering an apparent 459 cases in New Jersey was recognised only after a post-epidemic study of shellfish-associated hepatitis in New Jersey connected disease with the ingestion of raw clams. The actual incidence during the outbreak was estimated to be several times higher higher (Goldfield, 1976).

43. When the outcome is not hepatitis but a short-lasting gastro-enteritis, the patient may not seek medical attention at all. But even when medical attention is required, the most likely diagnosis will be non-bacterial gastro-enteritis without any attempt to identify the pathogenic agent. A further problem is to link the causative agent with its presence in seafood. One type of difficulty stems from the unavailability of routine assays for the suspected pathogen in environmental samples. The examples are hepatitis A and Norwalk viruses, two of the most important shellfish pathogens. Another type of difficulty is presented by *V. parahaemolyticus*. It is a common contaminant of seafood, but only 0.18 per cent of the isolates from water, shellfish, and sediments showed virulence by the Kanagawa reaction (see Hackney and Dicharry, 1988), while nearly all the isolates (96.5 per cent) from patients with diarrhoea were Kanagawa positive (Blake *et al.*, 1980b).

### (a) *Seafood-borne bacterial diseases*

#### (i) *Typhoid fever*

44. Epidemics of oyster-associated typhoid fever was reported from France as early as 1816, but the importance of shellfish in the transmission of this disease was recognised only at the end of the last century (Mosley, 1975). Between 1894 and 1896 large outbreaks were reported from the U.S.A., the U.K. and France. Typhoid bacillus (*S. typhi*) remained the leading cause of shellfish-associated disease until the middle of the present century when non-specific gastro-enteritis became the prevalent seafood-borne malady. Nevertheless, shellfish-associated typhoid fever remains a hazard wherever the disease is endemic and sewage reaches the vicinity of shellfish harvesting grounds. Historically, shellfish-associated typhoid fever prompted the development of shellfish-harvest water criteria based on coliforms (Richards, 1985).

(ii) *Cholera*

45. The classical biotype of *V. cholerae* predominated worldwide until the 1960s, when the El Tor biotype became dominant. This biotype caused large epidemics in 1961 in the Philippines (Joseph *et al.*, 1965), where the transmitters were raw shrimps, and in 1973 in Italy, where mussels were probably the transmitters (Baine *et al.*, 1974).

46. Though the causative agent is part of the indigenous bacterial flora in estuarine waters, seafood-borne outbreaks are secondary to the release of cholera vibrios to sea water. Cholera was recognised in the South Pacific in early September 1977, when the El Tor biotype serotype Inaba was isolated from the stool of patients with severe diarrhoea in Tarawa atoll, Gilbert Islands. Most of the 572 patients (3.3 per cent of the population) admitted to hospital by 10 October 1977 had eaten raw, partially dried, saltfish. At the declining phase of the epidemics the source of infection was clams and fish caught in the lagoon, contaminated by the faeces of the first batch of victims (McIntyre *et al.*, 1979). In an outbreak in Louisiana (U.S.A.) all the 11 cases of cholera had eaten cooked crabs. The El Tor biotype of *V. cholerae*, serotype Inaba, was isolated from the patient's stool, a left-over crab, estuarine waters and shrimps, and also in the sewage of six neighbouring towns, including three without identified cases (Blake *et al.*, 1980a). The presence of the El Tor biotype and Inaba serotype vibrios is more widespread than the outbreaks. For example it was isolated in Morro Bay, California, U.S.A. but the low frequency of toxicogenic strains and the low estuary concentrations may explain why no cases of cholera have been reported from the West Coast (Kaysner *et al.*, 1987).

(iii) *Illnesses caused by non-cholera vibrios and Beneckea*

47. In cases of gastro-enteritis following the consumption of raw shellfish the only pathogenic isolate in the stool of the patients may be one of the following bacteria: *Vibrio cholerae* non-0 1, *V. parahaemolyticus*, *V. mimicus*, *V. furnissii*, *V. hollisae* or *B. parahaemolytica*. Since its discovery, *V. paraheamolyticus* has been implicated in Japan in more than 1,000 outbreaks per year and it accounts for 45-70 per cent of that country's bacterial food poisonings (Hackney and Dicharry, 1988). *V. parahaemolyticus*-associated gastro-enteritis has been reported from North America, Central America, Africa, Europe and Asia. In the U.S.A., from August 1971 to August 1972, approximately 1,200 people became ill in 13 outbreaks. These occured between June and October and the most frequent symptoms were diarrhoea, abdominal cramps and nausea. Attacks started mostly 15-24 h after eating contaminated seafood (raw or steamed crab, boiled shrimp, raw or roasted oyster) and half the exposed population became ill. Some of the infections were attributed to cross-contamination of cooked seafood by raw seafood (Barker, 1974).

48. *B. parahaemolytica*, a bacterium closely related to vibrios, caused gastro-enteritis in Japan and in Maryland, U.S.A. (Baumann and Baumann, 1979) and *V. hollisae* along the coastline

have been implicated as a vehicle for *V. mimicus* (Shandera *et al.*, 1983). Though *V. furnissii* has been isolated from patients with acute gastro-enteritis in at least two outbreaks of food poisoning, its pathogenic role needs further study (Brenner *et al.*, 1983).

49. *V. vulnificus* differs from the other non-0 1 vibrios in both virulence and clinical signs. Its main effect after ingestion is septicaemia. In 24 patients onset followed eating raw oysters 24 h earlier. Eighteen of them had pre-existing liver disease and 11 of them died within 24 h of the onset. Two-thirds of the patients developed skin lesions which can take the form of blistering and necrotic ulcers (Blake *et al.*, 1979). Mortality is as high as 67 per cent (Tyring and Lee, 1986). Reduced gastric acidity predisposes to infection (Johnson *et al.*, 1986). There is a case report which implicates *V. hollisae* in septicaemia after the consumption of catfish (Lowry *et al.*, 1986).

(iv) *Botulism*

50. Seafood-borne botulism is caused by the toxin of *Cl. botulinum* type E. Poorly processed hot-smoked whitefish (Schantz, 1983) and canned tuna or salmon (Liston, 1980) are the source of this frequently lethal disease.

## *(b) Seafood-borne viral diseases*

(i) *Infectious hepatitis*

51. Infectious hepatitis or jaundice is known to be associated with the consumption of oysters (Roos, 1983), clams (Goldfield, 1976), mussels (Bostock *et al.*, 1979) and cockles (O'Mahony *et al.*, 1983). The largest outbreak occured in Sweden in 1955-56, when 629 people developed oyster-associated hepatitis (Roos, 1956). The first two large shellfish-related hepatitis outbreaks in the United States occurred in 1961 - an oyster-related outbreak in Mississippi and Alabama with 84 cases, and a clam-associated outbreak in New Jersey with at least 459 cases (Richards, 1985). According to a U.S. Food and Drug Administration publication (see Guzewich and Morse, 1986), there were 28 shellfish-related outbreaks of hepatitis in the United States, in 1961-1981, involving 1,293 people.

52. The spread of the disease in the U.S.A. is demonstrated by an outbreak in 1973, when 293 people developed hepatitis A in New Mexico, Texas, Oklahoma, Missouri and Georgia after eating Louisiana oysters supplied from a single source (Richards, 1985; Coulepis *et al.*, 1987). Sporadic occurrence may be as important as the epidemic one. A study group in Boston found a relationship between the occurrence of hospitalized cases of infectious hepatitis and the consumption of shellfish even in non-epidemic periods. Ingestion of raw shellfish was reported by 18.4 per cent of the patients and 5.4 per cent of controls (Koff *et al.*, 1967). In the U.K. a survey, which included all cases of infective jaundice notified within six months from districts of the

Thames estuary and Sheffield, found that 42.6 per cent of the 450 patients and only 7.5 per cent of the controls had reported the consumption of steamed cockles (O'Mahony, 1983). At present there are no procedures for detecting hepatitis A virus in environmental samples.

53. In all these cases the disease was caused by hepatitis A virus which is spread primarily by faecal-oral contact, and not by the less infective hepatitis B virus (Barkow, 1977), which is mainly transmitted by other routes (infected syringe, blood transfusion). Though hepatitis B virus was isolated in oysters near a hospital waste outfall (Mahoney *et al.*, 1974), its importance as a marine health hazard is uncertain. Three reports also implicated shellfish as a source of non-A, non-B hepatitis. One study found that 12.5 per cent of the cases of non-A, non-B hepatitis had ingested raw shellfish (Gerba, 1988).

(ii) *Viral gastro-enteritis*

54. While bacterial illness from shellfish is on the decline, shellfish-associated viral enteric illness is on the increase. The clinical features of gastro-enteritis produced by various viral agents is similar. A typical case report describes a party attended by 24 individuals. Raw clams were served, and within 6-24 hours 18 of the 20 persons who had eaten clams developed diarrhoea and abdominal cramps which lasted one to three days. None of the four persons who refrained from eating clams developed any symptoms. Clams from the same lot had 0.027 and 0.040 μm virus-like particles and no pathogenic enteric bacterium (Center for Disease Control, 1982).

55. Before the development of sophisticated methods, the diagnosis of viral gastro-enteritis was based on the absence of bacterial pathogenic agents. Lately, positive diagnosis has become possible, as immune electron microscopy and radio-immune assay allow the demonstration of immune reactions in serum and of the virus in stool. Thus, in an Australia-wide outbreak of gastro-enteritis when 2,000 people became ill, Norwalk virus was found in 39 per cent of the faecal samples and antibody in 75 per cent of the tested sera (Murphy *et al.*, 1979). Depuration of oysters collected from an infected area did not prevent Norwalk virus gastro-enteritis in human volunteers (Graham *et al.*, 1981).

56. In the last decade, Norwalk virus was probably the causative agent in most of the shellfish-associated outbreaks in the U.S.A. In 1980-1984 there were 58 outbreaks and seven of them involved more than 200 people. In the last eight months of 1982 there were 103 well-documented outbreaks in which 1,017 persons became ill after eating clams (813 cases) or oysters (204 cases). In five of seven outbreaks serum tests were positive for Norwalk virus and the virus was identified in clams and oysters from two outbreaks (Morse *et al.*, 1986). Increased awareness of shellfish-borne diseases by health professionals, and better reporting practices, may explain why the majority of documented outbreaks were in the New York - New Jersey area. The high number of outbreaks of enteric illnesses associated with raw or steamed clams caused such a public apprehension that in 1983 dealers imported depurated English clams which caused at least 14 outbreaks of gastroenteritis involving over 2,000 consumers (Richards, 1985). A similar outbreak

was caused in the U.K. by imported and depurated Pacific oysters offered at eight subsequent receptions to 1,300 guests. Oysters were ingested by 39 per cent of the guests and 79 per cent of these became ill, while only 4.9 per cent of the non-eaters had gastric symptoms. The causative role of the Norwalk virus was confirmed by laboratory tests (Gill *et al.*, 1983).

57. Besides Norwalk virus, a group of small round viruses - Snow Mountain agent, astroviruses and calciviruses - have been reported as the cause of shellfish-associated gastro-enteritis (Gerba, 1988).

### *(c) Seafood-borne parasitic diseases*

58. The larvae of *Pseudoterranova decipiens* cause the so-called tingling-throat syndrome which ends when the worm is coughed up within 48 hours of meal. Clinically, a more important problem is infection by *Anisakis simplex*. The 50 cases were reported from the U.S.A. may be only the tip of the iceberg (McKerrow *et al.*, 1988). As the larval *A. simplex* is capable of penetrating the stomach and intestinal walls, infection elicits marked inflammation, obstruction, abdominal pain, fever, diarrhoea. The worms can be identified by endoscopy, and the surgical resection of the inflamed area is the only definitive treatment (Hsiu *et al.*, 1986; McKerrow *et al.*, 1988). Both the excretory product of the dorsal oesophageal glands of the larval parasite and the excretory-secretory products of the worm produce antibodies which can be used for immunological diagnosis (Sakanari *et al.*, 1988). Though the main source of infection is raw fish (e.g. sushi), herring marinated in vinegar caused several infections (Verhamme and Ramboer, 1988).

## *3. Preventive measures*

59. There are four possible ways to prevent seafood associated illnesses: (1) to prevent shellfish contamination by the regulation of coastal sewage disposal; (2) to prevent harvesting from polluted grounds; (3) to remove pathogens from the shellfish by efficient depuration; (4) to educate the public in the handling and preparation of such products.

60. The causal relationship between the pollution of shellfish-harvesting areas with domestic sewage and the outbreaks of shellfish-associated enteric bacterial infections led to the application of bacterial indicators for the assessment of the sanitary quality of shellfish and shellfish-growing waters. The use of faecal coliforms as indicators provided a useful information when the faecal pollution was moderate to excessive. It became less and less reliable when the numbers were below $70\ 10^{-2}\ ml^{-1}$ in water and $130\ 10^{-2}\ g^{-1}$ in shellfish (Metcalf, 1975). The cause of discrepancy between coliform standards and viral concentrations is due to the difference between the resistance of faecal coliforms and viruses to chlorination and natural inactivation factors in both sea water

on bacterial quality, a depurated shellfish may pass the sanitary requirement when viral concentration is still in the pathogenic range (Richards, 1985). Thus, adequate depuration characteristics (time and rate of water flow) for bacteria are not adequate for viruses. In addition to viral contamination, faecal coliforms do not indicate hazards presented by bacteria indigenous in sea water.

61. The proven unreliability of coliform standards for shellfish flesh (Sobsey *et al.*, 1980; Johnson *et al.*, 1981) led to proposals to develop standards which include an index of viral risk (Vaughn *et al.*, 1980), irrespective of the lack of correlation between the concentrations of enteric viruses in shellfish flesh and in overlying waters (Gerba *et al.*, 1980). As direct environmental assays for hepatitis A and Norwalk viruses are unavailable at present, the indicator use of rotavirus or poliovirus has been advocated (Richards, 1985). Other possible measures are improved shellfish-borne disease surveillance helped by shellfish tagging to facilitate the identification of the source of shellfish and ideally the harvesting area (Guzewich and Morse, 1986).

62. An additional measure is the education of consumers about the hazard of eating raw or partially cooked shellfish. It has been shown that the inactivation of enteric viruses in crabs requires 8 minutes boiling (Hejkal and Gerba, 1981), while in cockles the same was achieved by the maintenance of internal temperature of 85-90°C for 1 minute (Millard *et al.*, 1987). These reports confirmed the finding of an early study which had shown that clams usually open during the first minute of steaming, but 4-6 minutes are required for clam tissue to approach 100°C, the temperature required for sterilization and prevention of infective hepatitis (Koff and Sear, 1967). Proper cooking and freezing also kills worms (e.g. *A. simplex*) (Liston, 1980).

# II. TOXIC HEALTH HAZARDS

## A. TOXIC AGENTS IN THE SEA AND SEAFOOD

63. The presence of toxic substances has been detected around the world from busy estuaries to the Arctic Ocean. Owing to volcanic emanations, mobilization of material from ocean beds, erosion of coastal areas, precipitation from the atmosphere and marine biological activities, some of these substances were there before the advent of industrialization. Examples are heavy metals and marine biotoxins. Industrialization increased both the volume and, through the synthesis of organic compounds, the variety of potential pollutants. Even the production of marine biotoxins by dinoflagellates may be influenced by human activities, such as land drainage, construction works in lagoons, and the release of trace elements (WHO, 1984).

64. Synthetic organics are exclusively the products and by-products of industrial synthesis. The most important organic synthetic compounds belong to the group of chlorinated hydrocarbons. This group includes biocides such as DDT, hexachlorobenzene, hexachlorocyclohexane, dieldrin and toxaphene as well as polychlorinated biphenyls (PCB). They move horizontally with air and ocean currents and vertically through the air/water interface by precipitation (wet and dry) and volatilization. Contamination by PCB and DDT decreases from the Great Lakes of North America to the Northwest Atlantic Ocean, North Sea and Baltic Sea to the Northwest Pacific Ocean to the Arctic Ocean and is lowest in the Antarctic Ocean. The other organochlorines, especially hexachlorocyclohexane, appear to be more evenly distributed in the northern hemisphere (Norstrom and Muir, 1988). In spite of their global distribution, they are not generally present in industrial effluents, sewage and run-off. For example, in urban run-off the most frequently identified synthetic organic compound, hexachlorocyclohexane, was present only in 20 per cent of the samples, while seven metals, including arsenic, cadmium and lead, had over 40 per cent incidence rates (Pollman and Danek, 1988).

65. The presence of metals in every form of anthropogenic emission, their environmental behaviour, and their toxic potential in seafood, prompted the Twelfth Session of GESAMP (Geneva, 22-29 October, 1981) to decide that priority should be given to the evaluation of the harmful effects of cadmium, lead and tin released into the marine environment. Subsequently, six metals became the subject of two reports, the first on cadmium, lead and tin (GESAMP, 1985) and the second on arsenic, mercury and selenium (GESAMP, 1986). The same six metals and marine biotoxins are the subjects of this section.

# B. SOURCE AND FATE OF TOXIC METALS IN THE MARINE ENVIRONMENT

## *1. Source and size of metal inputs*

66. The sources of metal releases to the environment are mainly the combustion of coal and oil, pyrometallurgical processes, industrial, agricultural or consumer use, and transport. All these activities contribute to releases to the atmosphere, soil or aquatic ecosystems. For example in 1983, 14.9 million tons of metals were emitted and the share of the aquatic ecosystem was 5 per cent. Automotive exhausts explain partly the substantial contribution of lead to metal emissions. Thus in the same year the estimated anthropogenic emission of lead was 2.6 million tons and nearly 13 per cent of this reached aquatic ecosystems (Nriagu and Pacyna, 1988).

67. Inputs to the marine ecosystem are derived from atmospheric emissions, river transport, and the discharge of sewage and dredged soil. Atmospheric emissions and river transport are shown in Table 1. About 30 per cent of the atmospheric emissions reach the aquatic ecosystems. Owing to the high marine-to-inland water-surface ratio, most of the atmospheric fallout reaches the oceans and is distributed over the whole area. River transport, sewage, and industrial waste outfalls, or the deposition of sludge or dredged spoil can produce local hot spots. In the Humber estuary deposition of dredged spoil was the third major form of cadmium input (18.3 per cent) after river transport (40.4 per cent) and industrial discharges (21.5 per cent), and it was the major form (76.3 per cent) of lead input, followed by atmospheric input (8.6 per cent) and river transport (7.3 per cent) (GESAMP, 1985).

68. A metal can reach the marine environment in both inorganic (free, charged and uncharged) or organic forms. Pollution by the organic form needs special attention when the organometal has kinetic (absorption, distribution, excretion) and toxic characteristics distinctly different from inorganic forms. The organic compounds of arsenic, lead, mercury and tin, belong to this group. Compared with the total input, the contribution of organometals is minute. In 1974 10 per cent of the total lead produced in Europe, the U.S. and Japan was used for the synthesis of organolead. As 95 per cent of the organolead additives are decomposed to inorganic lead during combustion, only 0.5 per cent of the total lead production was released as organolead to the atmosphere and a fraction of this could reach the oceans. The ban on alkylmercurial fungicides in most industrial countries, and the alerting effect of the Minamata and Niigata epidemics on industrial practices, substantially reduced the presence of organomercurials in agricultural and industrial wastes. Restrictions on the concentration of organotin in antifouling paints used for small

and on mariculture equipment in the U.K. (Abel *et al.*, 1987), and the shift from the physically dispersed to the more economic chemically bound tributyltin copolymer (Anderson and Dalley, 1986) are attempts to decrease organotin input.

### 2. *Transport and concentrations*

69. Rivers, sewage, sludge, and spoils carry metals, mostly in particulate form, which has less opportunity to escape to the open sea than dissolved components. The transport and trapping mechanisms are (a) movement with coastal water bodies; (b) sedimentation; (c) leaching from, and sorption on, particles; (d) flocculation (Duinker, 1985). From the discharge towards the open ocean both the particulate-to-dissolved metal concentration ratio and the sediment metal concentration decrease as a result of sedimentation. Recent data collected in the Mediterranean Sea area suggest that mercury and lead levels in sediments near discharge sites are 4 to 10 times higher than background values (i.e. 0.1 $\mu g$ Hg $g^{-1}$ and 20 $\mu g$ Pb $g^{-1}$ dry weight), and that concentrations return to background levels within 10-20 km. Concentrations in water (including suspended solids) declined similarly (Osborn, 1988). Surface-water cadmium concentrations in the North Sea were 0.6-1.6 $\mu g\ l^{-1}$ soluble and 0.1-0.3 $\mu g\ l^{-1}$ particulate, whilst in the open North Atlantic Ocean the mean concentrations were 0.06 and 0.002 $\mu g\ l^{-1}$, respectively (Coombs, 1979). A similar strong vertical concentration gradient follows the deposition of soluble atmospheric lead on open ocean surfaces (Buat-Menard, 1987). In the absence of comprehensive and comparable data, Table 2 gives only a rough outline of metal concentrations in unpolluted and polluted waters and sediments.

### 3. *Speciation and transformation*

70. In sea water a metal can be dissolved or particulate, free or complexed, charged or uncharged, inorganic or organic. The chloride affinity of lead, cadmium, and mercury increases from lead to cadmium to mercury, and affects speciation. No mercury is free in sea water while 3 per cent of cadmium or mercury is. Only 47-75 per cent of lead forms chloride complexes, while all the complexed cadmium and mercury is associated with chloride. The higher affinity of mercury is shown by the formation of higher chloride complexes, resulting in the dominance of negatively charged species (Turner, 1987). The oxidation states of lead and tin vary between II and IV. They are mostly bivalent in water and tetravalent in sediment. Tin (IV) is the substrate for methylation (GESAMP, 1985).

71. Depending on pH and redox conditions in aerobic waters, arsenite (III) is oxidised to arsenate (IV) and selenide (II) to selenite (IV) which are the predominant and bioavailable forms

selenium as selenite, after accumulation they are reconverted to trivalent arsenic and bivalent selenide which are the substrates for biomethylation (GESAMP, 1986).

72. When organic lead and tin reach sea water, total decomposition follows entry within a few weeks (GESAMP, 1985). Tributyltin degrades with a half-time of 7-15 days (Seligman *et al.*, 1986) to dibutyltin, and dibutyltin degrades with a longer half-time (Thain *et al.*, 1987). In winter no degradation could be detected (Olson and Brinckman, 1986). The decomposition of organomercurials, with the exception of methylmercury, is fast. In sea water, change in methylmercury concentration depends on the rate of synthesis (GESAMP, 1986).

73. Though abiotic processes may contribute to biomethylation, arsenic, mercury, and tin are methylated by bacterial and, to a smaller extent, by algal activities. Methylation produces mono- and dimethylated mercury, and mono-, di-, and trimethylated arsenic and tin. Several organisms are probably involved in the synthesis of more elaborate organic arsenic compounds, such as arsenobetaine. It has been suggested (GESAMP, 1986) that the last step in arsenobetaine synthesis requires a microbially mediated step in anaerobic sediments where either the final compound or its precursor enter the food chain through consumption by detrivores (eaters of decomposed remnants of living materials).

## 4. *Bioaccumulation*

74. A marine species takes up a metal from water or sediment either directly or indirectly by ingestion with food. Diffusion, facilitated transport, or absorption on gill and surface mucus are the mechanisms of uptake from water.

75. The significance of accumulation from sediments is small, but may be important for the biological cycling of metals. Detrivores may be responsible for the sudden increase in cadmium concentration at 2,900 m depth in the Sargasso Sea (Coombs, 1979) and for the biosynthesis of arsenobetaine (GESAMP, 1986).

76. Marine algae can accumulate arsenic, lead, mercury and selenium with high (x 1,000) bioconcentration factors. Bioaccumulation of arsenic, cadmium, lead, mercury, selenium and tin from water is the important uptake process for mussels and oysters. When both water and food (algae) contained cadmium, 10 per cent of the cadmium found in the tissues of *Crassostrea virginica* originated from the food and 90 per cent from the water. Uptake increases with increasing concentration and temperature (Coombs, 1979; GESAMP, 1985, 1986).

77. The origin of the accumulated metals affects their distribution. Most of the metal absorbed from water is deposited in the shell of molluscs, the exoskeleton of crustaceans and the gill and skin mucus of fish. Uptake from food results in deposition in the viscera of molluscs and the

78. Two species of flatfish, plaice (*Pleuronectes platessa*) and thornback ray (*Raja clavata*) accumulated 3-14 per cent of the inorganic mercury and 80-100 per cent of the methylmercury from food, and methylmercury was preferentially deposited in the muscles. In the muscle of marine fish, but not in their liver, about 90 per cent of mercury is methylmercury. Ingested arsenic is also deposited in the flesh. Arsenobetaine is the major form of arsenic in lobsters, crabs, shrimps and several teleost fishes. Arsenic in the oil-rich tissues (e.g. liver) of the herring (*Clupea harengus*) may also be a lipid-soluble conjugate of arsenobetaine (GESAMP, 1986).

79. There is no reliable information on the bioaccumulation of inorganic tin. The bioconcentration factor of tributyltin was over 1,000 in the Pacific oyster (*Crassostrea gigas*) and the sheep minnow (*Cyrinodon variegatus*) (GESAMP, 1985). *Mytilus edulis* and the mud crab (*Rhithropanopeus harrissi*) accumulated tributyltin from food very much faster than from water (Laughlin, 1986).

80. Of the six metals, only mercury in the form of methylmercury is biomagnified in the aquatic food chain.

### 5. *The toxic potential of metals in seafood*

81. The anthropogenic emissions of toxic metals and their accumulation in marine organisms raised the possibility of seafood-borne outbreaks of intoxication. Two methylmercury epidemics in Japan demonstrated that metal pollution can reach toxic levels in seafood.

82. The possibility of adverse health effects depends on the toxicity of the metal and on the degree of exposure. Toxicity is an inherent quality, but frequently differs with the various compounds of the same metal. Dose or exposure for seafood consumers depends on the concentration of metals in seafood, the chemical form of the metal, and the quantity of seafood consumed. The following evaluation is based mainly on tolerable daily/weekly intakes which are assumed to include reasonable safety margins. Only those adverse reactions are mentioned which may be associated with exposure through the consumption of seafood. When a metal exists in different chemical forms of widely different toxicities, GESAMP (1985, 1986) based hazard evaluation on the concentration of the most toxic form (e.g. arsenic versus arsenobetaine), but called attention to uncertainties concerning other components. If not stated otherwise, concentrations in molluscs, crustaceans and fish, are expressed on the wet weight basis. Where necessary values have been converted on the basis of a wet/dry weight ratio of six.

83. Worldwide, national, or even regional consumption averages give misleading estimates of risk associated with the consumption of seafood. GESAMP (1986) considered that risk is likely to be confined to the upper percentile of the population which has a consumption rate several times the average. Thus risk assessment must cover daily seafood consumptions from 20 g (about one meal per week) through 150 g (one meal per day) to the extreme of 1,000 g.

### *(a) Arsenic*

84. Arsenic concentration in the brown alga *Hizikia fusiforme* is generally 10-100 µg $g^{-1}$ dry weight, while in the red and green algae it is 1-20 µg $g^{-1}$ dry weight. In crustaceans e.g. prawns, shrimps, crabs, lobsters and crayfish, arsenic concentrations are between 1-50 µg $g^{-1}$ wet weight, in bivalve molluscs 1-15 µg $g^{-1}$and in fish 0.2-10 µg $g^{-1}$. In many commercially important fish 90 per cent - and in crustaceans and molluscs 95 per cent - of the arsenic is arsenobetaine (GESAMP, 1986; WHO, 1981).

85. The health effects of drinking water with high arsenic concentration are well documented. In certain parts of Chile 600 µg $l^{-1}$ arsenic in the drinking water caused severe circulatory disorders in the extremities (white fingers, blueness of hands and feet) in one-third of the population after 10-15 years. Half of the patients who received about 3.0 g arsenic in medicine during 6 to 26 years of treatment developed hyperkeratosis and hyper- or depigmentation (GESAMP, 1986). The WHO (1981) suggested that the lifetime risk for skin cancer due to arsenic in drinking-water is about 5 per cent for a total dose of 10 mg in an assumed life span of 70 years. This total dose corresponds to a daily intake of 400 µg from drinking water and it is at least 40 times higher than the daily intake in countries where average fish consumption is about one meal a week and drinking water is not polluted (GESAMP, 1986).

86. Table 3 indicates that only the extreme consumption of flatfish, crustaceans, and molluscs can supply inorganic arsenic in excess of 400 µg per day. Even the daily inorganic arsenic intake of extreme fish eaters remains below the level which is expected to increase cancer risk by 5 per cent. This prediction assumes that inorganic arsenic in seafood has the same bioavailability and toxicity as in drinking water. However, seafood contains more organic than inorganic arsenic and there are no adequate toxicological data on the organoarsenicals present in seafood. Because animals fed for a long time with seafood containing 14 or 30 µg $g^{-1}$ arsenic as arsenobetaine showed no signs of intoxication, arsenobetaine seems significantly less toxic than inorganic arsenic. However, even if the two forms were equally toxic, it would be possible to classify the weekly consumption of one or two seafood meals made from products containing 10 µg As $g^{-1}$ and the daily consumption of fish with 0.1 µg As $g^{-1}$ as non-hazardous. Nevertheless, the effects of high arsenobetaine concentrations in seafood eaten over a lifetime need to be investigated (GESAMP, 1986).

87. If the daily seaweed consumption is 5 g dry weight, assuming 20 µg $g^{-1}$ arsenic concentration, the daily inorganic arsenic intake increases by 70 and the organic arsenic intake by 30 µg. Organic arsenic in seaweed consists of ribofuranoside derivatives of unknown toxicity (GESAMP, 1986).

### *(b) Cadmium*

88. As expected from differences in bioaccumulation, fish and crustaceans usually contain low cadmium levels. According to a joint European monitoring programme, cadmium levels in fish are mostly below 0.2 μg $g^{-1}$ wet weight. Decapods (shrimp, prawn, lobster) can accumulate moderately less than 0.7 μg $g^{-1}$ and molluscs substantially higher concentrations. The maximum concentrations measured in limpets and ormers approached 60 μg $g^{-1}$ and in whelks it was even higher (GESAMP, 1985). In Bluff oysters (*Tiostrea lutaria*), rock oysters (*Crassostrea glomerulata*), and American oysters (*Crassostrea virginica*), concentrations ranged from 0.12 to 7.9 μg $g^{-1}$. In New Zealand and Australia concentrations of 0.15 to 46.0 μg $g^{-1}$ were found in the Pacific oyster, *Crassostrea gigas* (McKenzie-Parnell *et al.*, 1988).

89. About 30 per cent of cadmium is distributed to kidneys where, owing to the extremely long half-time of 10-30 years, concentration increases with age. Above a critical renal cadmium concentration the possibility of kidney damage resulting in proteinuria is increased. This critical concentration varies from individual to individual, but it is assumed that the daily ingestion of 200 μg cadmium over a 50 year period is the critical exposure for the occurrence of increased amounts of low molecular weight proteins in urine (WHO, 1979; GESAMP, 1985).

90. The average daily cadmium ingestion of 10-20 μg is doubled by one fishmeal or by the consumption of three oysters (assuming an average weight 6 g of each) per week. However, only extreme fish eaters and those who consume regularly about two dozen oysters per day are at risk of developing cadmium-related proteinuria.

### *(c) Lead*

91. Seaweeds commonly have lead contents of 3.0-20 μg $g^{-1}$ dry weight, but occasionally 300 μg $g^{-1}$ was measured. Lead concentrations in zooplankton, crustaceans, mussels, squid, octopuses are usually below 1.0 μg $g^{-1}$ wet weight and in common commercial fish below 0.1 μg $g^{-1}$(GESAMP, 1985). However, in samples of *Mytilus* species from the Ligurian Sea, the Gulf of Trieste and the Saronikos Gulf, peak concentrations were 19.5, 2.5 and 18.3 μg $g^{-1}$, respectively (Fowler, 1985).

92. There are significant quantitative (dose) and qualitative (target) differences between adults and children in their reactions to lead. In adults the most sensitive indicators of toxic exposure are interference with both the biosynthesis of haem (which may results in aneamia) and the conduction velocity of peripheral nerves. In children, learning ability can be impaired or more severe brain disorders develop (GESAMP, 1985). The daily lead intake by ingestion is below 100 μg in the majority of adults (GESAMP, 1985), while the tolerable daily intake recommended provisionally by an Expert Committee on Food Additives (FAO/WHO, 1972) is 430 μg or

6 μg $kg^{-1}$ body weight. As the onset of detectable haematological effects is associated with 20 per cent lower blood lead concentration in children than in adults (WHO, 1977), the tolerable daily intake of lead in children may be also 20 per cent lower (5 instead of 6 μg $kg^{-1}$). However, neuropsychological effects mayoccur even at a lower level.

93. Table 3 shows that one standard seafood meal a day, made from fresh fish or mussel, does not elevate lead ingestion above the tolerable daily intake. Only the extreme consumption of lobsters or mussels, but not the extreme consumption of fish, can supply lead in excess of the tolerable daily intake of adults. A child of 20 kg body weight probably reaches the tolerable intake by eating daily 800-1,000 g fish, or 80-100 g mussels or crabs, for a prolonged period of time. In households where canned food is widely used, allowance must be made for lead released from lead-soldered tin cans (GESAMP, 1985).

## *(d) Mercury*

94. The highest mercury concentrations had been measured in fish and shellfish caught in the Minamata Bay (Japan) before the discharge of methylmercury from the acetaldehyde plant of the Minamata factory was modified. Shellfish contained 1.3-14.0 μg $g^{-1}$ and fish 0.4 - 30.0 μg $g^{-1}$ total mercury. In 1960, the year when methylmercury was positively identified in seafood, no fish caught in this area had less than 10 μg $g^{-1}$ (Irukayama, 1977).

95. In fish (sardine, herring, sprat, catelin, cod, hake, haddock, whiting, plaice, sole) caught in the North Atlantic, the total mercury concentration ranged from 0.006 to 0.5 μg $g^{-1}$. The lowest value was found in sardine and the highest in plaice. In the Mediterranean, the mean concentration exceeded 0.5 μg $g^{-1}$ in the striped mullet (*Mullus barbatus*), the bluefin tuna (*Thunnus thynnus thynnus*), and the Atlantic bonita (*Sarda sarda*). The maximum concentrations were 7.9, 6.3 and 2.3 μg $g^{-1}$, respectively (GESAMP, 1986).

96. In the U.S.A. marketed crustaceans have commonly more mercury than molluscs. The highest concentrations were found in lobsters, 0.42 μg $g^{-1}$ mean and 2,310 μg $g^{-1}$maximum. In oysters, clams, scallops, squids, octopuses, mean concentrations were below 0.06 μg $g^{-1}$. The less than 0.1 μg $g^{-1}$ fish group included (in order of increasing concentration): mullet, herring, anchovy, Atlantic mackerel, salmon, flounder; in the 0.1-0.2 μg $g^{-1}$ range: haddock, cod, skipjack tuna, seabass; in the 0.2-0.5 μg $g^{-1}$ range: marine trout, jack mackerel, yellowfin tuna, bonito, halibut, white tuna, Spanish mackerel. Four species had more than 0.5 μg $g^{-1}$ average mercury concentrations: striped bass, 0.75 μg $g^{-1}$; king mackerel, 0.98 μg $g^{-1}$; swordfish, 1.2 μg $g^{-1}$; sharks, 1.24 μg $g^{-1}$. In Australia, black marlin and 10 shark species had more than 1.0 μg $g^{-1}$ total mercury. Black marlin had the highest average (7.3 μg $g^{-1}$) and the highest maximum (16.5 μg $g^{-1}$) concentrations (GESAMP, 1986).

97. In the muscle tissue of fish, about 90 per cent of the total mercury is in the methylated form, but in black marlin it is only 10 per cent. In the liver of fish and marine mammals mercury is mostly inorganic (GESAMP, 1986). While the gastro-intestinal absorption of methylmercury is nearly complete, less than 15 per cent of the inorganic mercury is absorbed. Only methylmercury in seafood has sufficiently high concentration to be of health concern (GESAMP, 1986).

98. Methylmercury is able to damage the sensory part of the nervous system. The first symptom is paraesthesia (abnormal sensations), which is followed by unco-ordinated movement, constricted visual field, slurred speech, and hearing difficulties in more severe cases. The long-term daily intake of 3-7 μg Hg $kg^{-1}$ body weight (or 200-500 μg $g^{-1}$ for 70 kg weight) is expected to cause symptoms in 5 per cent of an adult population (WHO, 1976). Clinical and epidemiological studies have indicated that the foetus is more sensitive than the adult. As the number of children studied was too small, the confidence limits for pre-natal effects was wide, but did not exclude the possibility that the foetus can be 10 times more sensitive than the adult, and that therefore daily intakes exceeding 20-50 μg by the mother may be harmful to the foetus (GESAMP, 1986).

99. Fish and shellfish are the main source of methylmercury intake. The daily intake of 4-5 μg Hg from all other sources is mostly in the inorganic form.

100. Table 6 shows that limiting the number of seafood meals to one per day keeps methylmercury exposure within tolerable limits. When the number of daily seafood meals is increased to two (300 g seafood daily), the consumer must be selective and avoid any fish with more than 0.5 μg $g^{-1}$ mercury. An extreme consumer of 1.0 kg seafood per day must be satisfied with mullet, herring, anchovy, Atlantic mackerel, salmon and flounder. The prevention of foetal damage requires more stringent rules.

### *(e) Selenium*

101. Selenium concentration in zooplankton, mussels and crustaceans is usually less than 1 μg $g^{-1}$, but in the Adriatic 3.5 μg $g^{-1}$ was found in European oysters (*Ostrea* sp) and, in the Long Island Sound, 2.8-5.5 μg $g^{-1}$ in crabs. Concentrations in the muscle tissue of fish fall mostly in the range of 0.2-1.0 μg $g^{-1}$. The liver concentrations of selenium in sharks and marlin can reach the level of 15.0 μg $g^{-1}$. In marine mammals, like whales and seals, the liver can contain even more selenium, mostly in the form of mercury selenide. The range for whales is 4-80 μg $g^{-1}$ and for seals 4-400 μg $g^{-1}$.

102. Selenium is both essential and toxic. Thus, human health can be adversely affected when exposure is too low or too high. In the general population the dietary intake of selenium is within the range of 20-300 μg $day^{-1}$ and the main source is food. The minimal human nutritional

accurate estimation of the tolerable daily intake than that it is between 0.75 and 5.0 mg. Dry and brittle hair, brittle nails with white spots, skin lesions, tooth decay, and the hypersensitivity of the nervous system were the signs and symptoms in one food-related outbreak of selenium intoxication (WHO, 1987).

103. Seafood is a good source of nutritionally desirable selenium levels in areas where soil and plants are deficient in selenium. Table 7 shows that the risk of overexposure associated with seafood consumption is slight. Even the uninterrupted consumption of 1,000 g seafood at the upper end of the concentration range would result in daily consumption only marginally higher than the tentative lower limit of undesirable exposure. Moreover, some selenium is associated with mercury or cadmium in a less toxic form (GESAMP, 1986).

## *(f) Tin*

104. In molluscs and crustaceans the concentration of total tin may reach 2.0 $\mu g\ g^{-1}$, while for fish 0.5 $\mu g\ g^{-1}$ is a more reasonable estimate (GESAMP, 1985). In tributyltin-polluted harbour sites, organotin contributed less than 0.5 $\mu g\ g^{-1}$ to the total tin concentration of oysters (Grovhoug *et al.*, 1986). Organotin concentrations in molluscs peaks during summer months and declines below detection limit in winter (Alzieu, 1986; Weber *et al.*, 1986).

105. The daily intake of tin ranges from 200 to 17,000 mg. The main source is food stored in cans or PVC containers. Less than 5 per cent of the ingested inorganic tin is absorbed. Acute gastro-intestinal effects are dependent more on the concentration of tin in food or drink (and on its irritative effect) than on the dose. Prolonged daily dietary exposure to 210-250 mg had no adverse effects (WHO, 1980). Alkyltins are more toxic than inorganic tin, but there are large quantitative and qualitative differences in their toxicities. Trimethyltin and triethyltin are the most toxic organotins and the only ones with neurotoxic effects. Decreasing the number of alkyl groups and increasing the length of alkyl radicals decreases toxicity (Snoeij *et al.*, 1987). All the reported cases of organotin intoxications were associated with trimethyl- and triethyltin exposure. There are no established tolerable daily intakes for organotins but 3.2 $\mu g\ kg^{-1}$ body weight was suggested for tributyltinoxide (Schweinfurth and Günzel, 1986). This is equivalent to a tin intake 1.26 $\mu g\ kg^{-1}$ body weight or 88 $\mu g$ for 70 kg body weight.

106. Table 8 shows that it is very unlikely that inorganic tin in seafood presents any hazard to human health. Owing to the sparsity of data on the concentrations of organotins in marine food and to the lack of quantitative data on human toxicities, any prediction on the hazard presented by organotins in seafood must be tentative. Nevertheless, it is known that the concentrations of organotins in water and molluscs show wide seasonal variations. Concentrations are high from May to September, and below detection level from December to March (Alziu, 1986; Weber *et*

June to September, less than 0.1 µg $g^{-1}$ was found from December to March. Secondly, only 80-90 per cent of butyltin (Waldock *et al*, 1987) and 10-20 per cent of methyltin is trialkyltin (GESAMP, 1985). Thus, it seems a reasonable assumption that the yearly average concentration of organotin in molluscs does not exceed 0.5 µg $g^{-1}$ or 25 per cent of the total tin concentration and therefore the daily consumption of one oyster meal does not exceed the tolerable daily intake for tributyltin suggested by Schweinfurth *et al.* (1987).

## C. MARINE BIOTOXINS

107. Most of the marine toxins and all the most toxic ones are neurotoxins. The mechanism of action of several marine biotoxins is high-affinity binding to ion channels which disturbs ion movement through membranes and results in membrane depolarization (Wu and Nakahashi, 1988). Depolarization deprives nerve cells of excitability and conductivity, and muscle cells of contractibility. These neurotoxic effects are important for a venomous animal in both defense and aggression.

108. Owing to the variety of conditions for exposure and the possibility of severe consequences, these biotoxins are important human health hazards. The route of communication is either parenteral or oral. Parenteral exposure is associated with fishing and underwater activities (recreational or occupational) or bathing. However, the oral route is a more common path of exposure to marine biotoxins. As a result of modern freezing and shipping techniques, poisoning by seafood has become a general problem, not restricted to coastal areas or local seafood products.

### *1. Parenteral exposure to marine biotoxins*

109. Parenteral exposure can result from superficial contact or from envenomation of human flesh by the traumatogenic apparatus of a poisonous animal. The outcome may range from itching to death. The reaction depends on the size of the contact, the depth of the trauma, the strenght of the toxin and the delivered dose.

110. Usually mild pain and dermatitis are the consequences of contact with the algae *Lyngbya majuscula* (only regions not covered by the swimming trunk are affected), with sponges (*Porifera*), stinging corals (*Millipora*) and stony corals (*Acropora*). Repeated contact may result in chronic dermatitis. The "red moss dermatitis" of North American oyster fishermen is caused by a scarlet sponge (*Suberitus ficus*) found on the shell of oysters while the "sponge fisherman disease" is caused by actinian parasites (*Sagartia rosea* and *Sarortia elegans*). *Alcyonidium*, a member of the group of coelenterate worms (*Polyzoa*), which forms moss-like colonies, can cause

sea anemone *Anemonia sulcata* is of medical importance in the Adriatic Sea where it inflicts numerous stings on bathers. The venom of sea anemones is a relatively mild neurotoxin (about 10,000 times less toxic than palytoxin) and its main effect is local oedema. Palytoxin, the toxin of the female of zoanthid polyps (*Palythoa*) is so powerful that swimming in tidal pools inhabited by *Palythoa* results in numbness of the lips and tongue. Contact of an open wound with the slime of *Palythoa* can be fatal.

111. The stings of starfishes (*Asteroidea*) and sea urchins (*Echinoidea*) can penetrate the skin and can cause severe, burning pain and oedema. A patient lost consciousness after slipping and landing on the starfish, *Acanthaster planci*, and tearing his hand against it. Other symptoms were nausea, dizziness, weakness, partial paralysis of the lips. Even more severe local and general signs and symptoms can be caused by contact with the Portuguese man-of-war (*Physalia physalis*), the cubomedusae jellyfish (*Chironex fleckeri*), some cone-shells members of (*Conidae*) family, the octopus *Hepalochlaena maculosa*, stingrays (*Urobatis halleri* and others), scorpion-fish (*Scorpaena guttata*), and weever-fish (*Trachinidae*). Haemorrhage and necrosis of and around the injured area is common, pain can spread to the whole affected extremity, and respiratory-circulatory disorders with lethal outcome may develop. The cardiotoxic poison of the jellyfish (*Chironex fleckeri*) may kill within 5-150 min. The punctures inflicted by this jellyfish and the Portuguese man-of-war are microscopic, but they may amount to many thousands envenomations. Marine snails of the genus *Conus* contain one of the most virulent of animal venoms, but only the piscivorous species (e.g. *Conus geographus*) poses a serious threat to humans. There are at least 30 cone-shell-related envenomations reported, of which eight had a fatal outcome. There are reports on envenomations, often fatal, by the blue-ringed octopus, *H. maculosa*. The toxin of this species is tetrodotoxin, the toxic agent in pufferfish poisoning.

112. Wounds from stingray, scorpion- and weever-fish are painful. Redness, oedema and haemorrhage extend beyond the trauma to the surrounding area and the general symptoms are often severe. Wounds inflicted by the stingray can penetrate deep into the flesh or even into internal organs. In the U.S.A. alone, the yearly incidence rates of stings by scorpion-fish is 300 and by stingrays 750. Injuries inflicted by weever-fish are common along the southern North Sea, the English Channel and the Mediterranean and Adriatic Seas. The toxins are not virulent and no death attributable to stings of these fishes have been reported in recent years. (This section is based on reviews by Halstead, 1981; Russel, 1986; and Zannini, 1983).

## 2. *Oral exposure to marine biotoxins*

113. Marine biotoxins responsible for food-borne intoxications can be divided into (1) those produced by dinoflagellates and diatoms, which reach edible marine animals through the food chain; (2) those produced in certain species of live fish or crab; (3) those produced in the dead flesh.

extensively described in Environmental Health Criteria document No. 37 (WHO, 1984).

### *(a) Toxins produced by phytoplankton*

114. A sudden upsurge in the reproduction of dinoflagellates becomes visible by the discolouration of the sea when at least 10 cells $l^{-1}$ are present. When the algae are toxin producers, the toxin concentration will also be high in marine animals feeding on plankton or reached through the food web. Nevertheless, it has been shown that mussels can accumulate toxic concentrations of paralytic shellfish poisons without visible algal bloom (Ayres and Cullum, 1978) and that in Alaska the butterclam contains toxic concentrations of paralytic shellfish poisons in all seasons (Mills and Passmore, 1988).

(i) *Paralytic shellfish poisons*

115. Paralytic shellfish poisons (PSP) are produced by the genus *Gonyaulax* of dinoflagellates (e.g. *G. catanella*). The toxic potential varies between species or strains. The toxins produced by these dinoflagellates are collectivley known as saxitoxins. Saxitoxin is heat-stable at acidic pH, but easily oxidized in alkaline conditions. It can cause paralysis, respiratory depression and circulatory failure after the ingestion of mussels, clams, cockles, or scallops that have fed on PSP-producing dinoflagellates. In the Philippines, the xanthid crab, *Zosimus aenus,* which accumulates high levels of saxitoxin, was frequently implicated in intoxications, with high fatality rate. The source of toxin in xanthid crabs appeared to be the PSP-containing algae *Jania* sp. (WHO, 1984).

116. The signs and symptoms of PSP range from slight tingling and numbness around the lips to complete paralysis and death. The tingling sensation develops 5-30 min after consumption and in fatal cases the victim dies within 2-12 hours. In moderate and severe cases the progression of numbness to arms and legs makes voluntary movement difficult, speech becomes incoherent, pulse rapid and respiration superficial (WHO, 1984).

117. A survey, carried out after 78 people had mussel-related paralytic shellfish poisoning in the north-east coast of England, found that in mussels collected from the implicated beds the yearly maximum concentration of PSP varied between 0.4 and 100 mg $kg^{-1}$ (Ayres and Cullum, 1978). Data tabulated by WHO (1984) indicate that PSP concentrations in shellfish ranged between 0.79 and 430 mg $kg^{-1}$ in 12 outbreaks. According to one estimate, a mild case of poisoning can be caused by the ingestion of 1 mg of toxin, moderate poisoning by 2 mg, serious by 3 mg and lethal by 4 mg of PSP (Russel, 1986). In the U.S.A., quarantine is enacted for shellfish beds when the toxin level in the edible part of the shellfish reaches 0.8 mg $kg^{-1}$ (WHO, 1984).

(ii) *Neurotoxic shellfish poisons*

118. The source of the neurotoxic shellfish poisons (NSP) is the dinoflagellate *Gymnodinium breve* (reclassified as *Ptychodiscus brevis*). Blooms on the west coast of Florida occur every 3-4 years and are associated with massive fish kills. The transmission of the toxin is through filter-feeding molluscs which retain the toxins in their edible tissues. Symptoms are very similar to paralytic shellfish poisoning, but paralysis has not been observed. Abnormal sensation is localised to lips, tongue, throat and the area around the mouth. Other symptoms are gastro-intestinal complaints, dizziness and muscular aches. The onset time ranges from few minutes to few hours after ingestion, and recovery is within a few hours to a couple of days (Taylor, 1988). In the U.S.A. shellfish is considered unsafe for human consumption when the concentration of NSP in 100 g edible tissue exceeds the limit of detection by the mouse assay (WHO, 1984). Monitoring red tides alerts shellfish harvesters.

(iii) *Diarrhoeic shellfish poisons*

119. Diarrhoeic shellfish poisons (DSP) are produced by the species *Dinophysis*, an armoured marine dinoflagellate. In Japan, *D. fortii* was identified as a producer of DSP, in the Netherland the bloom of *D. acuminata* was associated with gastro-intestinal disorders among consumers of raw or cooked *Mytilus edulis* (Kat, 1983). During 1976-1980 more than 1,300 people were diagnosed as DSP cases in Japan. Diarrhoea, nausea, and vomiting were the most frequent complaints, followed by abdominal pain. The poison is a mixture of okadaic acid, pectenotoxin and their derivatives (WHO, 1984).

(iv) *Ciguatera poisons*

120. The principal toxin is ciguatoxin, but structurally related maitotoxin and scaritoxin were also isolated from ciguatoxic fish. Maitotoxin ranks among the most potent marine toxins. Ciguatoxins are produced by the dinoflagellate *Gambierdiscus toxicus* which lives around coral reefs, closely attached to macroalgae. Poisoning can be caused by ingestion of fish, usually bottom feeders and reef fishes that have become toxic by feeding on toxic dinoflagellates, and the larger carnivores which prey on these herbivores. Among the most commonly implicated species are groupers, snappers, barracuda, and sea bass. In Hawaii the herbivorous surgeonfish caused numerous outbreaks (WHO, 1984). In Australia the narrow-barred Spanish mackerel (*Scomberomorus commersoni*) accounted for most of the 527 cases between 1965 and 1984 (Wu and Narahashi, 1988). The internal organs are more toxic than the muscle. The occurrence of ciguatoxic fish is limited to the circumglobal tropical and sub-tropical belt, but interregional transport results in human exposure in other parts of the world. Thus, the sources of outbreaks in Maryland (U.S.A.), France and Canada were fish imported from Florida (U.S.A.), Taiwan and Jamaica. The worldwide incidence of ciguatera poisoning may approach 50,000 cases yearly (WHO, 1984, Taylor, 1988).

121. Ciguatera primarily affects the gastro-intestinal and neurological systems. Onset is usually a few hours after ingestion. Initial symptoms are nausea, numbness and tingling of the lips, tongue and throat. Patients later may develop vomiting, abdominal cramps, diarrhoea, abnormal sensation, itching muscle and joint pain. The patient may die from convulsions or respiratory arrest in the first 24 hours, but lethality is less than 0.5 per cent. The prognosis of the survivors is good. The gastro-intestinal complaints are of short duration, but the neurological symptoms may persist for weeks or even a year in severe cases.

122. No monitoring programme has been established, but a newly developed simple stick enzyme immunoassay (Hokama, 1988) seems promising. The risk may be lowered by avoiding unusually large specimens of reef fish and excluding the internal organs and roe of these fish from the diet (Taylor, 1988).

(v) *Amnesic shellfish poison*

123. Amnesic shellfish poisoning is due to domoic acid. Though domoic acid is known to be synthetized by several macro-algal species, the source of toxic concentrations were the pennate diatom *Nitschia pungens* (Bird *et al.*, 1988; Bates *et al.*, 1988) in the first known outbreak. Domoic acid concentration in cultured blue mussels (*Mytilus edulis* L.) collected just before the outbreak from the affected area of the Prince Edward Islands (Canada) was mostly in the range of 0.1 to 1.0 mg $g^{-1}$ wet weight. The total amount of domoic acid in cultivated mussels was estimated at about 6 kg, probably only 1 per cent of the total domoic acid production (Bates *et al.*, 1988). Vomiting, abdominal cramps or diarrhoea occured within 24 hours of consumption, and confusion, disorientation, loss of memory and/or other objective neurological signs within 48 hours. Twenty-two people were hospitalized, 10 of them required intensive care and three eldery patients died (Bird *et al.*, 1988; ARL Shellfish Toxin Team, 1988). The standard mouse bioassay used for the detection of paralytic shellfish poison can detect domoic acid in mussels (ARL Shellfish Toxin Team, 1988). HPLC (high pressure liquid chromatography) allows the estimation of 0.5 $\mu$ $g^{-1}$ wet weight domoic acid in mussel tissue and 1.0 $\mu g$ $g^{-1}$ dry weight in plankton (Bates *et al.*, 1988).

## *(b) Toxins produced in live edible marine animals*

(i) *Tetrodotoxin*

124. Tetrodotoxin (puffer or fugu poison) is found mainly in various pufferfish (*Tetraodontiformes*), but also in the Japanese ivory shell (*Babylonia japonica*), the trumpet shell (*Charonia sauliae*), some newts and vividly coloured frogs, and in the venom of the blue-ringed octopus *H. maculosa* (Mills and Passmore, 1988). The presence of tetrodotoxin in animals of six different classes in four different phyla argues against the hypothesis that it is an inherent metabolic product. Recently it has been shown that members of the *Vibrionaceae* family (e.g. *V.*

*alginolyticus*), which are indigenous in sea water and are associated with marine animals, are producers of tetrodotoxin, anhydrotetrodoxin, or both (Simidu *et al.*, 1987).

125. In the pufferfish, the poison is located mainly in the gonads, liver, intestines, and skin. Only one species, *Logocephalus lunaris lunaris,* can contain fatal amounts of tetrodotoxin in muscle tissue (WHO, 1984). The presence and concentration is related to the reproductive cycle of the pufferfish. Concentration is greatest just prior to spawing (Russel, 1986). Tetrodotoxin can be estimated by a chemical method which is based on its conversion to fluorescent compounds or by a modification of the mouse bioassay for PSP (WHO, 1984).

126. Tetrodotoxin is a heat-stable neurotoxin. Onset is from 10 to 45 min after ingestion. The illness starts with abnormal sensations (paraesthesia), followed by nausea, vomiting, diarrhoea, shallow respiration, low blood pressure, abnormal heart beat. Death usually takes place within the first six hours. The average number of deaths from eating pufferfish in Japan was 100 per year out of 200 reported cases (Schantz, 1973), but since the licensing of fugu restaurants the number of cases has declined to 60 per year with 20 deaths. Licensees of fugu restaurants are trained for species identification and the removal of poisonous visceral organs without contaminating the white meat. Imported fish caused 10 cases of intoxication in Italy (WHO, 1984).

(ii) *Palytoxin*

127. Palytoxin, first isolated from female zoanthid polyps (*Palythoa*), has been identified in several crabs (*Demania alcalai*, *Laphozozymus pictor* and *Demania reynaudii*) in the Philippines. Onset is within few minutes after ingestion and death within a day. The first symptoms in a man who died after consuming one-fourth of a cooked crab were dizziness, nausea, tiredness and diarrhoea followed by low heartbeat, rapid and shallow breathing, bluish discolouration and renal failure (Alcala *et al.*, 1988). Palytoxin is the most toxic marine toxin to date, having intraperitoneal LD of 50-100 ng $kg^{-1}$ for mice (Wu and Narahashi, 1988).

### *(c) Toxins associated with bacterial activity in dead flesh*

128. There are two bacterial activities associated with toxin production. The first is when the bacterium converts a normal body constituent of an edible species to a potentially toxic one (scombrotoxin), the second when the sea-borne bacterium synthesizes its own specific toxin (botulinum toxin type E). The condition for the formation of biotoxins of bacterial origin is improper storage.

(i) *Scombrotoxin*

129. Scombrotoxin is histamine which acts in association with other substances (probably putrefactive amines) formed during spoilage. Histamine is formed from L-histidine, present in the muscle, by the activity of the enzyme histidine decarboxylase. Scombroid fishes (e.g. mackerel, tuna and bonito), which are implicated in 58 per cent of the scombroid fish poisoning outbreaks, have plenty of histamine-producing bacteria in their gut, skin or gills, and they also have characteristically high levels of free histidine in the muscles (Taylor, 1988).

130. The poisoning is rarely serious and is of short duration. In an outbreak in Taiwan, 41 employees of a department store were hospitalized and 115 became ill after eating fish left at room temperature for three to four hours before frying. Symptoms occurred after an incubation period of 10 min to 24 hours. Symptoms in decreasing order were dizziness, flushing, headache, nausea, peri-oral numbness, palpitation, pruritus, fever and diarrhoea. All cases recovered within 24 hours. The U.S. Food and Drug Administration has established 500 $\mu g\ g^{-1}$ as the hazard action level for histamine in tuna (Kow-Tong and Malison, 1987).

(ii) *Botulinum toxin type E*

131. *Cl. botulinum* type E is a toxin-producing bacterium that may be present in seafood products. The spores are highly heat resistant, but the toxin is readily destroyed by heat. As its growth and toxin production require the exclusion of oxygen, the source of infection consists of improperly prepared canned or vacuum-packed seafood products. Onset is within 18-36 hours after ingestion. The first symptoms may be nausea and vomiting, but the illness is dominated by severe neurotoxic effects, and ends in death in 65 per cent of the cases (Berkow, 1977; Schantz, 1973).

### *(d) Preventive measures*

132. Control measures for poisonings caused by scombrotoxin or botulinum toxin centre on proper storage. Scombrotoxin formation is prevented by refrigeration between catching and consumption or canning. *Cl. botulinum* requires heat treatment (80°C internal temperature), the avoidance of sloppy can-filling practices, and the freezing or at least chilling of hot smoked products (Liston, 1980). Botulinum toxin is heat-labile, thus heating the food before consumption is an effective way of prevention.

133. The heat stability of tetrodotoxin, shellfish and ciguatera poisons at normal pH does not offer such a simple protective measure. In the case of tetrodotoxin the Japanese licensing law required restaurants to employ chefs who (a) can recognise poisonous pufferfish species, (b) are aware of seasonal variations in toxicity and (c) can remove liver, gonads, and roe without con-

exist for intoxications caused by plankton-synthetised biotoxins. Awareness of plankton blooms, the assay of products from suspected areas, and quarantining the affected shellfish beds are the only control measures. Depending on conditions, the quarantine can be temporary (until toxin concentration declines below the acceptable level), seasonally permanent or continuously permanent. Ciguatera poses even more problems than shellfish poisons. About 300-400 fish species have been implicated in ciguatera poisonings and toxicity is almost unpredictable with respect to species, location, and the time of the year (Russel, 1986).

# III. CONCLUSIONS

134. The microbiological and toxic agents present in the marine environment and identified as human health hazards are either anthropogenic or occur naturally. They show differences in both the frequency and the severity of consequences. The probability of becoming ill is higher with anthropogenic microorganisms than with those naturally occurring in sea water, and higher with marine biotoxins than with anthropogenic toxic emissions.

135. Ear and gastric disorders caused by anthropogenic microbes dominate infections associated with both recreational activities and seafood consumption. Seafood-related gastric disorders became less severe as Norwalk or other viruses replaced salmonellae as the main causative pathogens. Norwalk virus gastro-enteritis is usually of short duration and requires no medical attention. Hepatitis A virus and *V. cholerae* are the micro-organisms primarily responsible for severe seafood-borne infections. In addition, some of the micro-organisms that have their normal habitat in estuaries and sea waters are human health hazards. Thus, halophilic vibrios have caused seafood-related and wound infections, some of which were lethal.

136. Since the Minamata and Niigata epidemics, anthropogenic toxic emissions have not been associated with seafood-borne intoxications. Nevertheless, the lack of overt intoxication does not exclude the possibility of adverse reactions. Thus, the very high consumption of seafood with high methylmercury concentrations must be counted as a risk factor. As the effect of cadmium may be only mild proteinuria which becomes apparent after 20-30 years of exposure and after 50 years of age, the odds are against detection and the establishment of association. In the absence of established tolerable daily intakes of lead for children, of methylmercury for pregnant women, and of organo-arsenicals and and organotins for the general population, the limits of safe seafood consumption are uncertain. In contrast with these uncertainties, there are plenty of clinical reports and epidemiological studies on illnesses, often lethal, caused by parenteral or oral exposure to marine biotoxins.

137. Prevention depends on the nature of marine health hazards. While conventional cooking methods protect against microbiological pathogens and botulinum toxin, they are ineffective against most of the seafood toxins of algal origin. The input of sewage or toxic effluents can be regulated, but the synthesis of biotoxins is outside human control. Infections by anthropogenic microbiological agents are preventable, while the prevention of wound infections by halophilic vibrios is not possible.

TABLE 1. ESTIMATES OF GLOBAL ATMOSPHERIC EMISSIONS AND GROSS RIVER FLUXES TO ESTUARIES FOR SIX METALS (tonnes per year).

| | Atmospheric Emissions | | Gross River Fluxes |
|---|---|---|---|
| | Natural | Anthropogenic | |
| Arsenic | 7,800[1] | 18,800[1] | 88,000[2] |
| Cadmium | 1,000[1] | 7,570[1] | 34,000[2] |
| Mercury | 6,000[1] | 3,500[1] | 1,600[2] |
| Lead | 19,000[1] | 332,350[1] | 1,600,000[2] |
| Selenium | 9,500[1] | 6,320[1] | 18,200[3] |
| Tin | 2,000[4] | 6,140[1] | < 83,000[4] |

[1] Nriagu and Pacyna, 1988
[2] GESAMP, 1987
[3] GESAMP, 1986
[4] GESAMP, 1985

TABLE 2. EXAMPLES OF TYPICAL METAL CONCENTRATIONS IN MARINE WATERS AND SEDIMENTS

| | Concentration in Water $\mu g\ l^{-1}$ | | Concentration in Sediment $mg\ kg^{-1}$ dry weight | |
|---|---|---|---|---|
| | Unpolluted | Polluted | Unpolluted | Polluted |
| Arsenic | < <2.0 | 5-42 | 13-18 | 50-300 |
| Cadmium | 0.1 | 2-45 | 0.2-0.6 | 10-1,000 |
| Lead | < <0.015 | 1-9(?) | 13-17 | 60-250 |
| Mercury | < <0.003 | 0.02-0.4 | < <0.08 | 0.4-350 |
| Selenium | < <0.17(?) | ? | 0.17 | ? |
| Tin | <0.002 | 0.005-0.09 | 2.0 | 20 |

The second values of lead in polluted water and sediment were taken from Melhuus *et al.*, (1978) and Osborn (1988). All the other values were taken from GESAMP (1985, 1986).

TABLE 3. THE CONTRIBUTION OF INORGANIC AND ORGANIC ARSENIC FROM SEAFOOD TO THE TOTAL DAILY INTAKE

| Mean concentration in seafood ($\mu g$ As $g^{-1}$) | Daily seafood consumption in g 20 | | 60 | | 150 | | 1,000 | |
|---|---|---|---|---|---|---|---|---|
| | $\mu g$ As $d^{-1}$ | | | | | | | |
| | inorg. | org. | inorg. | org. | inorg. | org. | inorg. | org. |
| 1.0[1] | 2 | 18 | 6 | 54 | 15 | 135 | 100 | 900 |
| 10.0[2] | 10 | 190 | 30 | 570 | 75 | 1425 | 500 | 9500 |

[1] Normal concentration in most commercially important fish species. 10 % is assumed to be inorganic.

[2] Concentration likely to be found in crustaceans, molluscs and bottom feeding fish (e.g. flounder, sole). 5 % is assumed to be inorganic.

TABLE 4. THE CONTRIBUTION OF CADMIUM FROM SEAFOOD TO THE TOTAL DAILY INTAKE

| Mean concentration in seafood ($\mu g$ Cd $g^{-1}$) | Daily seafood consumption in g | | | |
|---|---|---|---|---|
| | 20 | 60 | 150 | 1,000 |
| | | | $\mu g$ Cd $d^{-1}$ | |
| 0.4[1] | 8 | 12 | 60 | 400 |
| 4.0[2] | 80 | 120 | 600 | 4,000 |

[1] Normal concentration in fish and crustaceans.

[2] Representative concentration in molluscs.

TABLE 5. THE CONTRIBUTION OF LEAD FROM FRESH SEAFOOD TO THE TOTAL DAILY INTAKE

| Mean concentration in seafood (mu.g Pb $g^{-1}$) | Daily seafood consumption in g | | | |
|---|---|---|---|---|
| | 20 | 60 | 150 | 1,000 |
| | μg Pb $d^{-1}$ | | | |
| 0.1[1] | 2 | 6 | 15 | 100 |
| 1.0[2] | 20 | 60 | 150 | 1,000 |

[1] Concentrations in common commercial fish are below this level.

[2] Concentrations in crustaceans, mussels, octopus are generally below this level.

TABLE 6. TOTAL DAILY INTAKE OF MERCURY AS METHYLMERCURY BY CONSUMERS OF VARIOUS AMOUNTS AND TYPES OF SEAFOOD

| Mean concentration in seafood[1] ($\mu g\ Hg\ g^{-1}$) | Daily seafood consumption in g | | | | | | |
|---|---|---|---|---|---|---|---|
| | 20 | 40 | 60 | 100 | 150 | 300 | 1,000 |
| | $\mu g\ Hg\ d^{-1}$ | | | | | | |
| 0.1 | 2 | 4 | 6 | 10 | 15 | 30 | 100 |
| 0.25 | 5 | 10 | 15 | 25 | 38 | 75 | 250 |
| 0.5 | 10 | 20 | 30 | 50 | 75 | 150 | 500 |
| 0.75 | 15 | 30 | 45 | 75 | 113 | 225 | 750 |
| 1.0 | 20 | 40 | 60 | 100 | 150 | 300 | 1,000 |
| 1.25 | 25 | 50 | 75 | 125 | 188 | 375 | 1,250 |
| 1.5 | 30 | 60 | 90 | 150 | 225 | 450 | 1,500 |

[1]See preceeding text for concentration in fresh seafood.

TABLE 7. THE CONTRIBUTION OF SELENIUM FROM SEAFOOD TO THE TOTAL DAILY INTAKE

| Mean concentration in seafood ($\mu g\ Se\ g^{-1}$) | Daily seafood consumption in g 20 | 60 | 150 | 1,000 |
|---|---|---|---|---|
| | $\mu g\ Se\ d^{-1}$ | | | |
| 0.5[1] | 2 | 6 | 15 | 100 |
| 1.5[2] | 20 | 60 | 150 | 1,000 |

[1] Normal concentration in common commercial fish.

[2] Concentrations in crustaceans and mussels.

TABLE 8. THE CONTRIBUTION OF TIN FROM FRESH SEAFOOD TO THE TOTAL DAILY INTAKE

| Mean concentration in seafood ($\mu g\ Sn\ g^{-1}$) | Daily seafood consumption in g | | | |
|---|---|---|---|---|
| | 20 | 60 | 150 | 1,000 |
| | $\mu g\ Sn\ d^{-1}$ | | | |
| 0.5[1] | 10 | 30 | 75 | 500 |
| 2.0[2] | 40 | 120 | 300 | 2,000 |

[1] Concentrations in common commercial fish do not exceed this value.

[2] Concentrations in crustaceans and molluscs do not generally exceed this value.

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# ANNEX XI

# EXPLOITATION OF MARINE LIVING RESOURCES

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# INTRODUCTION

1. In the first GESAMP Review of the Health of the Oceans, several pages were devoted to the exploitation of living resources. It was noted that the global yield of fisheries was continuing to increase, although at a significantly lower rate than in the past. It was also noted that the major threat to fish stocks arose from over-exploitation, but the report directed its attention mainly to the impact on living resources of pollution from incidents, dumping, discharges and other inputs generated by human activities. It thus dealt with the effects of the environment on fishing and did not expand on the reverse possibility - that fishing activities themselves may damage the environment. It is now felt that this latter aspect should be explored and would perhaps merit a brief comment in this report. The notes below are a first approach to the topic. They deal in particular with the three most obvious considerations - the physical effects of demersal fishing operations on the sea bed and on the benthic organisms; alterations to the food web through over exploitation of single species; the effects of mariculture, including pollution, and the potential implications for ecological balance and gene pool diversity of intentionally or accidentally introduced species.

# I. PHYSICAL EFFECTS OF FISHING

2. Beyond the immediate objective of catching fish, fishing operations can have physical effects on the environment. Topics such as the impact of using explosives in coral reef fisheries and the damage caused by lost and drifting nets are dealt with in other annexes, and here we consider particularly the effects that towing demersal gear can have on the bottom and on non-target benthic organisms.

3. Concern about these effects has a long history, and there are records from more than 600 years ago indicating complaints about the destruction of benthic fish food and about damage to the bottom itself (Anonymous, 1921). It is difficult to evaluate the early observations, but since in these days the intensity and geographical extent of fishing was very much less than at present, it seems reasonable to suggest that the problem would be localised. However, as boats began to go further afield, and particularly as steam replaced sail and was in turn replaced by more effective power for both towing and net recovery, heavier and more complex gear was increasingly intro-

and a study of the effect of these tickler chains in 1938 indicated that some animals such as crabs and sea urchins were broken up but that other species were scarcely damaged. From the evidence collected at that time, it was concluded that the effects would not be serious (Graham, 1955).

4. However, modern fishing vessels cover much more ground and operate even heavier gear. Dutch trawlers, for example, may use as many as 15 ticklers weighing up to 12 tonnes per trawl and the undersides of the nets are often protected by chain mats (de Groot, 1984), while the passage of other types of trawl, which can now be adapted to work on both hard and soft ground conditions, causes major upheaval (Main and Sangster, 1979).

5. In addition to the exploitation of finfish, shell fisheries can contribute to the impact. The dredges used for clams, oysters and mussels may leave ruts and trenches up to 8 cm deep in sand and 10 cm in mud, and can damage the benthos (Gruffydd 1972; Caddy 1973; Franklin and Pickett 1978). Even more significant may be the impact of hydraulic dredging for molluscs. A commercial hydraulic dredge with blades more than 2 m wide may be supplied with water at rates of over 680 $m^3$ $h^{-1}$, and is known to cut trenches of up to 30 cm deep. Harvesting seaweeds can also cause effects on the ecosystem, altering the balance and structure of both plant and animal populations, increasing sediment mobility and resulting in a loss of organic material from the inshore systems (Boaden and Dring, 1980; Sharp, 1985). Finally, even the apparently innocuous operation of digging for bait can have ecological effects. It has, for example, been shown by Jackson and James (1979) that digging for lugworm and ragworm bait on sandy intertidal areas off the English east coast caused a drastic decline in the populations of cockles which were the subject of a commerical fishery. Such impact on non-target species was confirmed from observations in the Dutch Wadden sea (Van den Heiligenberg, 1982), which showed that most of the major species were severely reduced by bait collection, mechanical digging causing worse effects than manual operations. However, in general, harvested areas showed a better recruitment thereafter than non-disturbed grounds.

6. The impact of the disturbance caused by fishing operations is difficult to assess. As indicated above, there are obvious direct effects on the sea bottom, and these will depend on the nature of the substratum. They range from relatively insignificant ruts which will be smoothed over with each tide, to radical changes in the distribution of sediment and rocks. In areas of coarse sand, material resettles quickly but more muddy sediments put into suspension will be transported by tidal currents and settle elsewhere. Thus, repeated trawling on one ground with a mixed sediment where tidal currents are strong might eventually alter the nature of the bottom from fine to coarse particles. While tickler chains have been shown to disturb the sediment to a depth of 1 cm in sand and to 3 cm on muddy ground, the effects can be much more dramatic in areas where boulders are present. The otter trawls used some 15 years ago could tear large stones out of the bottom and deposit them some distance away, where, in their new elevated position clear of the sea bed, they damaged light fishing gear in the same area (Bridger, 1970). The most modern heavy

to a general uneveness of the bottom, they adversely affect the operations of fishermen using light gear.

7. The action of towed gear also damages the fauna directly. Maximum effects may be expected on hard bottoms which support a rich epifauna. For example, off the south eastern United States, a large part of the habitat of the south Atlantic Bight consists of low-relief hard bottom characterised by a sessile invertebrate fauna consisting predominantly of sponges and corals but with species of ascidians, hydroids, bryozoans, and algae also common. A recent study in such an area (Van Dolah *et al.*, 1987), using a research trawl with a roller-fitted headline suggested that sponges and corals were damaged but showed good recovery so that one year later the effect was not detectable. However, earlier work (Tilmant, 1979) on similar grounds but using commercial shrimp frame-trawls with more rigid rollers indicated that when this type of gear is used repeatedly in the same bottom, the damage was much more severe, with more than 80 per cent of the stony coral, 50 per cent of the sponges and 38 per cent of the soft corals in the trawl path crushed or torn loose.

8. On soft bottoms of sand or mud, where a significant proportion of the benthos consists of burrowing organisms, the effects may be less for roller-fitted trawls, but, as already indicated, damage has been recorded when tickler chains are used, and in general it is clear that demersal gear does dislodge and often break up animals in the sea-bed and that in extreme cases the structure of the benthic animal community can be altered. For example, banks of the polychaete worm *Sabellaria*, which forms reefs, are destroyed and cannot be re-established in areas regularly swept by trawls (Riesen and Riese, 1982), while some major taxonomic groups such as burrowing worms are favoured over others which are more susceptible to crushing. On the other hand, it can be argued that the ploughing action of trawls on the bottom may be beneficial, bringing up buried nutrients and food, and indeed it has been observed that fish feed actively in the fresh tracks of trawls (Arntz and Weber, 1970).

## A. FOOD-WEB CHANGES THROUGH RESOURCE EXPLOITATION

9. The previous section considered environmental and biological changes arising from the physical impact of fishing gears. Another concern is that the removal of large numbers of individuals of commercially important species, could, as well as affecting fishermen and consumers, in itself alter the structure of natural food webs. Perhaps the best studied example of this is in the Antarctic, where commercial sealing and whaling activities in the nineteenth and twentieth centuries brought some of these mammal species near to extinction (Laws, 1985). The key prey species in the Antarctic is the krill, *Euphausia superba*, which is the staple food not only of many whales and seals, but also of birds, fish and squid. When the baleen whales declined, the reduction in their grazing pressure released large quantities of krill for other predators, so that there were significant

increases in numbers of several bird species, as well as of the crabeater and fur seals, all of which are great consumers of krill.

10. Similar, although less well documented, situations exist in relation to fish stocks in many other parts of the world. In the North Sea, for example, there has in recent years been overfishing of pelagic stocks and a big rise in 'industrial' fishing (i.e. for meal and oil) involving small species such as sprats, sand-eels and some gadoids, as well as juveniles of various species normally taken as adults for direct human consumption (Bailey, 1987), and it is being suggested that this could have adverse effects on sea-birds which feed on these species. It is certainly true that the increases in some North Sea bird species that have been going on for more than 50 years do seem to be slowing down, and that some populations are levelling off or even declining locally. It is also true that there are records of adult auks found dead in large numbers off the east coast of Britain (Blake *et al.*, 1984) which might be accounted for by food shortage, and of low chick survival of puffins at the Lofoten Islands, and terns and kittiwakes at the Shetlands, which again has been attributed by some observers to insufficient food. However, even if it could be clearly demonstrated (and so far it has not been) that the effects on birds are due to reduced availability of fish, it must be recognised that not all the recent changes in fish stocks in the North Sea are due to the fisheries, and, at this stage in our understanding, it is unwarranted to accept that the main factor affecting the bird populations is exploitation of fish. Nevertheless, this sort of argument has been put forward in other areas. For example, it has recently been suggested that the deaths of thousands of penguins round the Falkland Islands can be attributed to overfishing of the squid, which is the main diet of the penguin in that area.

11. Another facet of the impact of commercial fishing on the structure of ecosystems concerns the capture of non-target organisms. A fishery directed towards one or more species will inevitably catch other organisms, and, on most commercial hauls, large numbers of juveniles of the target species as well as non-target fish and invertebrates are discarded, usually to die. There are some dramatic examples of this. For example, shrimp fishermen in the south-east of the United States catch large numbers of turtles in their nets and turtle exclusion devices (T.E.D.s) are now commonly used in an attempt at conservation. Again, the international tuna fishery in the eastern Pacific Ocean, using large purse seine nets, catches each year thousands of dolphins (the spotted dolphin, *Stenella plagiodon* and the spinner dolphin, *S. longirostris*) which are associated with the yellowfin tuna concentrations. An added complication in this case is that the fishermen use the surface-swimming dolphins as an aid in locating the tuna. There is much concern about the effects of this on the dolphin population, and dolphin-saving fishing equipment and procedures have been developed.

12. The cycle, from over-exploitation of a resource species to major changes in the food web which had supported it, is unfortunately being repeated in many parts of the world, but the complexity of the interactions and the lack of detailed information make it impossible at present to forecast the pattern of events in any individual case.

## B. MARICULTURE ACTIVITIES

13. Mariculture is expanding dramatically in many parts of the world, and now includes a wide range of operational approaches. At its broadest definition, it ranges from simple arrangements for on-growing or fattening of fish or shellfish, through increasingly complex controlled artificial culture systems to, at the other extreme, the practice of ranching, which is usually seen as the release of juvenile specimens into the sea where they range freely and are later harvested as adults. The construction of artificial habitats for fish is becoming particularly popular and reaches its maximum sophistication in Japan, where large areas of coastal waters are set aside for such developments. Any form of mariculture conducted on a large scale will have impacts on the environment, including pollution, alteration of habitat, effects on natural food webs and on ecological interactions, particularly when new species are introduced, visual amenity deterioration and interference with other users of the sea. With regard to the physical aspect, in areas of intensive culture the structures can influence natural sedimentation by modifying the velocity and direction of the currents. In extreme cases the effects can be significant. For example, hanging and stick shellfish culture in the Philippines has increased sedimentation to such an extent that coastline topography has been altered in some areas (Davies, 1964). Direct effects on the natural food web may also be seen. Where filter feeding bivalves are cultivated at very high densities, the amounts of phytoplankton will be measurably decreased, and it has been demonstrated that the filtering efficiency of mussels is so great that the growth of filter feeders downstream can be affected (Wildish and Kristmanson, 1984). However, any local reduction in phytoplankton biomass might be compensated for by increase of primary production due to stimulation from a higher rate of nutrient cycle arising from the filtering activity (Kasper *et al.*, 1985). Some other aspects are discussed briefly below.

### *1. Pollution effects*

14. Cage culture in the sea brings together large numbers of fish in a small space and generates major inputs of material, the nature and impact of which depends on the applied technique, the site location, the size of the production and the capacity of the receiving water. Relevant information is accumulating, particularly with respect to salmonid farming, which has been rapidly expanding in several parts of the world in recent years. The inputs to the sea from these farms consist largely of uneaten food and faecal and other excretory material. The bulk of the wastes are therefore organic carbon and nitrogen, along with ammonium, urea, bicarbonate, phosphate, and some vitamins, therapeutics and pigments.

15. Of the lesser constituents, the vitamins biotin and $B_{12}$ are constituents of fish feed.

moratorium was imposed pending the introduction of new licencing regulations.

18. Mollusc species are intensively cultured round the world, particularly the mussel and the oyster. However, the impact is different from that around fish cages since these molluscs consume phytoplankton and no fodder is supplied, but they do produce large quantities of faeces and pseudofaeces which accumulate on the bottom with the same general effects as described for salmon farms. In the Netherlands it has been estimated that the rate of sedimentation from mussel culture can be 10 mm per year (Misdorp *et al.*, 1984), while in Hiroshima Bay a typical oyster raft will produce 16 tonnes of faeces and pseudofaeces in less than 1 year, so that with over 1,000 rafts in operation, oyster culture will have a major impact on sedimentation in the bay (Arakawa *et al.*, 1971). Assessment of the long-term effect is difficult, but it is considered that when mussel cultivation ceases there is rapid recolonisation of the bottom and it is estimated that within two years the normal bottom fauna will become re-established (Ackefors and Sodergren, 1985).

### 2. *Introduction and transfer of marine organisms*

19. Not infrequently man introduces, either as a planned exercise or by accident, non-indigenous species into an established habitat. Given the complexity of ecological interactions, it is probably impossible to forecast the effects of such introductions, but it may appear that an alien species would have some difficulty in finding a niche and surviving. In some cases, however, the new species thrives, and its competition with or predation on native species can cause major ecosystem dislocation. Once plants or animals have been introduced, it is extremely difficult to eradicate them, and there is a real danger from associated pests or diseases that may be introduced with them. In the sea the problem is particularly difficult because of the ease of dispersal.

20. Some organisms are introduced completely by accident. The American slipper limpet (*Crepidula fornicata*), the Australasian barnacle (*Elminius modestus*) and the Mediterranean crustacean parasite of mussels (*Mytilicola intestinalis*) are thought to have been introduced to Europe on the hulls of ships and are all potential threats to molluscan fisheries. The Chinese mitten crab (*Eriocheir sinensis*), which damages fishing gear and river constructions, is thought to have been introduced to Germany and Belgium in ballast tanks of ships. Man-made alterations to the environment are another source of introductions, the opening of the Panama Canal (Hildebrand, 1939) and the Suez Canal (Por, 1978) being obvious examples. More recently, the construction of canals linking the U.S. and Canadian Great Lakes with the sea allowed the entry of marine species, and the invasion of the lakes above Lake Ontario by the sea lamprey (*Petromyzon marinus*) brought about the destruction of the lake-trout fishery and dramatic changes in prey-predation relationships.

21. There are also records of accidentally introduced marine plants. The green alga (*Codium fragile tomentosoides*) was first reported from off New York in 1957 and has since spread

affecting the behaviour of commercial species. In Northern Europe, the Japanese seaweed (*Sargassum muticum*) was recorded in southern England and northern France in 1973 and in spite of attempts at control is increasing its range, interfering with navigation and causing problems on amenity beaches and in harbours.

22. While it is clearly difficult to control accidental introductions the situation is different when a species is introduced intentionally. There are several reasons for deliberately introducing animals or plants to a new area, including fishery enhancement, mariculture, and hatchery rearing.

23. The fishery enhancement objective, which includes recreational activities, is usually associated with releases to the field, often to establish new reproducing populations of fish or shellfish such as oysters or lobsters, and the introductions are made as eggs, juveniles, or adult specimens. Salmonids have been very much involved, and attempts have, for example, been made to introduce Pacific salmon (both the chum, *Oncorhynchus keta* and the pink, *O. gorbuscha*) by importing eggs into the North Atlantic area without much success (Walford and Wicklund, 1971), although there are records of the establishment of such species as sturgeon, mullet, striped bass, herring and carp in new areas. However, the greatest success is probably with oysters. The European flat oyster (*Ostrea edulis*) has been transplanted between the United Kingdom, France, the Netherlands, Denmark, Norway and the Adriatic, sometimes to relay for fattening, sometimes to augment impoverished breeding stocks, and it has also been introduced to Canada and the United States. Other species, the Portuguese oyster (*Crassostrea angulata*), the Japanese Pacific oyster (*C. gigas*) and the American Atlantic oyster (*C. Virginica*) have all been transferred, usually successfully.

24. In some cases the imports are not for release in the wild, but are for intensive mariculture as, for example, the introduction of coho salmon from the U.S.A. to France and Spain to maintain a sea-cage culture industry, and in recent years the eggs and smolts of Atlantic salmon are being exchanged between European countries to maintain the rapidly growing cultivation of that species. In other cases the importation is aimed not at replenishing stocks but rather at storing live organisms in good condition prior to marketing. This applies in particular to eels, lobsters, crawfish and to a number of molluscs.

25. As already indicated, introductions and transfers of species may cause concern from several points of view - they may introduce pests and non-indigenous disease organisms, they may harm the ecology and they may alter the genetic constitution of existing species.

26. The introduction of disease organisms and pests along with imported species is of very serious concern. One recent example of this is the transfer to Norway from Sweden of the parasite *Gyrodactilus salaris* in imported salmon. This is now found in 22 Norwegian rivers with dramatic effects on wild fish. In addition, there are records of disease introductions with imported crustaceans, molluscs and finfish, which lead to infections of wild stocks (Rosenthal, 1980), and the establishment of pests such as *Crepidula* and *Sargassum* already referred to is thought to have

27. On the ecological side, the major worry is probably the effect of predation and of competition for food and space, and examples have already been given of the damage that can be done in these respects. The genetic implications of introductions have not been discussed, but since this raises wider issues it is dealth with in the next section.

## II. GENETIC EFFECTS OF LIVING RESOURCE EXPLOITATION

28. Some fishing practices may have effects on wild stocks at the genetic level. Very high exploitation rates will reduce the effective size of a stock with the result that the rates of genetic drift and of inbreeding could be increased. In addition, genetic changes may be produced in a stock over time by fishing methods that are in some way selective, as for example in a gill net fishery where the gear used imposed a size selection. Effects of this kind are more likely in freshwater fisheries in rivers or lakes, where the stocks are small and relatively confined geographically, but the possibility of comparable effects in the sea should not be ignored.

29. Pollution could also result in genetic selection, and there are examples of this in sewage sludge dumping grounds where the diversity of the fauna is greatly reduced and a small number of worm species dominate the biomass, and polychaetes like *Capitella capitata*, which have evolved several sub-species, can opportunistically expand in circumstances which are adverse to the normal community.

30. However, the most probable genetic effect may be found in the context of mariculture, since planned selective breeding could produce individuals better suited to the requirements of the consumer but less fit to survive in the wild. There is a fear that genetically inferior stocks of domesticated fish might escape during early phases of the rearing operation, or as adult fish from sea cages, or might migrate to sea from stocked rivers, and breed with wild fish. The results of this include hybridization, change of gene frequency, and changes in disease or parasite resistance. The effects of these possibilities could be either positive or negative, for example hybridization could result in either 'hybrid vigour' or loss of fitness. The fear, however, is that changes would be for the worse and would degrade the natural stocks.

31. Finally, the possibilities of genetic engineering are only just being investigated in aquaculture. For example, experiments are under way in Ireland to develop 'transgenic' salmon which have been injected with a foreign gene for growth hormone in the hope of enhancing productivity of local fish farms.

# III. SUMMARY AND CONCLUSION

32. From the notes above it will be clear that man's exploitation of the sea's living resources can have adverse effects on several compartments of the environment and on a number of marine processes and activities. The sea bed can be stirred up and its nature changed, damaging flora and fauna and interfering with some fishing operations; the structure of food webs can be altered by fishing and by mariculture; intensive culture programmes can pollute the environment, spread pests and diseases, and reduce general amenity; genetic impoverishment may result from overfishing, from breeding practices and from the introduction of exotic species.

33. All such effects come within the GESAMP definition of pollution, and some attempt should be made to assess their significance. This can be done with some confidence on the small scale of fish farms where the impacts can be clearly detected and in some way quantified, tending to be restricted to inshore waters and even there to be highly localised. Pollution around sea cages may either be accepted as part of the price to pay for intensive mariculture, or various steps may be taken to reduce the effects. Among the more serious effects is the spreading of pests and diseases, which can endanger the livelihood of individual fish farmers or can have more extensive and less controllable impact on commercial fisheries when wild stocks are affected. However, once the problem has been recognised, appropriate countermeasures can usually be devised.

34. It is more difficult to address food-web alterations which take place on a wider scale, such as those associated with open-ocean fisheries or in the general Antarctic region. In these cases it may be that the changes themselves affecting the food web are of less concern than the actions that cause the changes and in this context at least it can be said that attempts are made to regulate fishing at both national and international levels.

35. Similar movements towards control are evident in the field of aquaculture, and this is fortunate since it is forecasted that the yield of around 10 million tonnes in 1983 will increase to 25 million tonnes by the end of the century and most of the increase is likely to come from brackish and marine waters. ICES has for some time been interested in the problem of introduced species, and in 1969 set up a working group with broad terms of reference to advise on the matter. The group reviewed existing knowledge and made recommendations (ICES, 1972), as a result of which the Council of ICES in 1973 approved a Code of Practice for introductions. The topic was reviewed again and brought up to date some years later (ICES, 1982) and a further meeting of the working group produced detailed guidelines for the Code of Practice (ICES, 1984). The problem of introductions with particular reference to salmonids was taken up in 1987 by the North Atlantic Salmon Conservation Organisation (NASCO) and a working group has been formed which will

Law of the Sea, in article 196 paragraph 1, takes up the matter, and rules that 'states shall take all measures necessary to prevent, reduce and control the intentional and accidental introduction of species, alien or new, to a particular part of the marine environment which may cause significant and harmful changes thereto'. The machinery for control does appear to exist.

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# ANNEX XII

# SEWAGE IN THE SEA

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# INTRODUCTION

1. The term 'sewage' is here used to refer to human excreta and domestic wastes, with or without industrial additions. It may be taken that, on the average, human beings produce 100 g of raw sewage per person per day, and that in sewerage systems the flow is 180 litres per person per day. This must be disposed of, and disposal may be on the land, by incineration, or by discharge to rivers or the sea. Before disposal, sewage may be subjected to treatment, which can vary from simple screening or comminution to filtering and finally to biological treatment which removes first the large particles and finally those fractions which exert a biochemical oxygen demand. Secondary treatment produces a liquid rich in nutrients and a sludge high in BOD. Typical analyses of sludge discharged from ships and outfalls are given in the Table.

2. Sewage may be introduced to the sea in several different ways. In countries with poor or no domestic plumbing systems, human faeces from coastal communities are often excreted directly in the vicinity of the beach. Sewage and its treatment products also reach the marine environment via rivers when discharges take place into fresh water. Finally, it may disposed of directly to the sea through coastal outfalls or by dumping from ships. Pipeline discharges may be either raw sewage or some fraction from a treatment process, while dumped sewage is usually sludge from treatment, with or without the solids from coarse screening. The effects of sewage in the sea will depend on the method of disposal and on the characteristics of the site, as well as on the composition of the sewage.

# I. POTENTIAL EFFECTS

3. The general effects of sewage at sea may be discussed in terms of the constituents of the input and considered under four heads - inert solids, nutrients and BOD materials, toxic substances and living organisms.

4. Inert solids, particularly grit and inorganic material, will eventually settle on to the bottom, in a manner which will be determined by the hydrographic regime and by the rate and volume of the input. This can have a direct smothering effect on living benthic organisms and can have indirect effects by altering the physical nature of the sediments.

5. Nutrients such as nitrate and phosphate stimulate plant growth, and, in certain conditions, their presence in excess can result in massive blooms of algae which alter the chemistry of the water by their life processes and can cause oxygen deficiency when they die and decay. The carbon-rich material in sewage can also give rise to an oxygen demand in the water column and on the bottom.

6. Toxic substances such as metals, synthetic organics and oil may be associated with sewage in substantial amounts particularly when industrial wastes are present, and enhanced concentrations of these substances in the water and sediments can lead to accumulation in plants and animals, with potentially toxic effects.

7. Finally, sewage may contain a variety of living organisms, microbes or metazoans, some of which may be pathogens from infected human populations. There can thus be a public health threat, by direct infection of humans while bathing in the sea, or indirectly via the food if pathogens are accumulated in commercial fish and shellfish species.

8. The impact of sewage disposal at sea will depend not only on the composition of the input, but also on the hydrographic conditions at the disposal site. It has been suggested (McIntyre, 1977) that disposal areas can be thought of in terms of two extreme types - 'accumulating' and 'dispersing' grounds.

9. Accumulating grounds would ideally be situated in relatively still water, with little current or tidal movement and at sufficient depth to prevent normal wave action from stirring up the bottom. Material dumped would quickly reach the bottom and stay there. Dispersing grounds, on the other hand, would be located in regions of strong and extensive water movement near the bottom, so that dumped material would be carried away and widely dispersed, although it should be noted that even areas of net erosion may have lengthy periods of deposition when contaminants may be mixed into sediments by biological activity. While most areas will be expected to show

considerable range in accumulating and dispersing characteristics, and no ground will be totally one type or the other, the Scottish dumping ground at Garroch Head in the Firth of Clyde and the recently closed spoil area in the New York Bight may be regarded as accumulating grounds, while those in England used by London and Liverpool could be called dispersing grounds, and the disposal area in Delaware Bay would be classified as intermediate between the two.

10. Having outlined above the potential effects, we may now consider specific sites of sewage discharge and examine the impact on the various compartments of the marine environment.

## A. EFFECTS ON THE PHYSICAL ENVIRONMENT

11. In the short term, the impact of sewage disposed of at sea is on the water column, and in the longer term on the sediments. Changes in the abiotic environment are physical (e.g. by the addition of particulate material) and chemical, by the addition of nutrients and potentially toxic substances.

### *1. The water column*

12. Enhanced winter levels of phosphate and nitrate have been measured at the surface over the Clyde sewage sludge dumping ground at Garroch Head - 40 and 171 $\mu g\ l^{-1}$, respectively - but these cannot be considered striking increases above the normal found in certain industrialized coastal stretches in the area. A similar picture holds for the New York Bight with ortho- and total phosphate levels of 42 and 86 $\mu g\ l^{-1}$ found in winter, although values as high as 165 and 232 $\mu g\ l^{-1}$ have been measured at other times. While higher than unpolluted waters in the area (range 6-28 $\mu g\ l^{-1}$) they are exceeded in nearby coastal zones. In England, at the Liverpool Bay sludge disposal area, nitrogen and phosphorus concentrations tend to be about twice those found in unpolluted offshore water around the U.K., but are lower than in areas directly influenced by the outflow of the nearby river Mersey. Duedall *et al.* (1975) studied ammonium in the water of the New York Bight dump site at the end of July when a thermocline prevented mixing. At most surface and midwater stations, concentrations of $NH^+$ were less than 14 $\mu g\ l^{-1}$, but at bottom stations elevated concentrations were usually found, with a mean of 32 $\mu g\ l^{-1}$. They concluded that a strong local effect from sludge dumping could produce high levels in the water which could enhance phytoplankton growth, or even reach toxic concentrations in places, but that these conditions were transient, quickly reduced by dilution and dispersion.

13. Because of the high BOD of sewage sludge, deoxygenation of the receiving water is a possibility, but on dumping grounds in areas of rapid water exchange, this has not been found to be significant. Little oxygen depletion was observed in the outer Thames estuary (Shelton, 1970),

and in Liverpool Bay it was calculated that even in the most adverse circumstances saturation would not drop below 75 per cent, and that this condition would not extend over an area of more than 5 km (DOE, 1972). It might be expected that the least favourable conditions would occur on grounds where sludge tends to accumulate, and indeed decreased oxygen saturation is sometimes recorded in the New York Bight about 1 m above the bottom in several months of the year (Pearce 1972). In the Clyde, on the other hand, no reduction of oxygen has been detected in the water over the dumping ground.

14. The water column will also be affected by increased turbidity during the discharge and settlement of a sludge load, but this is likely to be of only temporary significance. Coulter counter studies show that on the Garroch Head dumping ground, particle counts in the upper layer of water returned to previous values a few minutes after dumping, and on Southampton grounds (Jenkinson, 1972) particulate organic carbon was high in the top one metre 15 minutes after discharge, but was substantially reduced one hour later. Experimental work confirmed the rapid settlement of this sludge. However, in some areas, particularly in deeper water, the presence of density gradients can affect the dispersal pattern, and delays in sinking or even horizontal transport of particulates has been recorded (Orr *et al.*, 1980).

15. In general, it would appear that changes in the water column resulting from sludge dumping will be local and transient and, except near the bottom, of no major significance to the ecosystem.

## 2. *The sediments*

16. Changes in the bottom as a result of dumping are likely to be both more obvious and more significant. On accumulating grounds, the heavier material quickly settles out and can be detected on the sea-bed. At Garroch Head marked changes in the sediment can be observed over an area of about 10 km (Topping and McIntyre, 1972). The normally loose grey sediment becomes black and fibrous in texture and can be clearly distinguished by TV camera. The increased organic content ranges from 3.0 to 8.2 per cent compared with 0.3 to 2.0 per cent in the clean areas. Also, heavy metals, such as copper, lead and zinc, were found to be substantially enriched in sediments at the centre of the dumping ground (Mackay *et al.*, 1972). In the New York Bight, the picture is similar, and even higher levels of organic carbon have been recorded. In Delaware Bay, in areas of sludge dumping from Philadelphia and Camden, the situation was less clear-cut (Watling *et al.*, 1974). Sludge dumping (about 389,000$m^3$ $y^{-1}$) started in 1961, in water of about 20 m depth. Conditions were such that even accurately dumped sludge would probably not settle directly on the site, and sediment analyses suggested that particulate organic matter may have settled some 12 km to the south-east. However, the highest organic content of this sediment was only 3.6 per cent. By contrast, on dispersing grounds, such as where London sludge is dumped, rapid dispersal makes it difficult to detect significant sediment accumulation.

17. The effects discussed so far on the abiotic environment have been illustrated by reference to sites where sludge is dumped from ships, causing initially a surface slick, a transiently turbid water column and, eventually, sedimentation on the sea-bed. Experimental study of this type of disposal shows that fractionation of the dissolved and particulate sludge occurs, so that in stratified waters the particulate plume rapidly penetrates the thermocline while the dissolved plume remains in the upper layer (Hatcher *et al.*, 1981). When the discharge is from an outfall on the bottom, the sequence of events is clearly different. A useful example is the disposal of sewage sludge in Santa Monica Bay from the city of Los Angeles, done through a 7-mile-long pipe discharging on the bottom at a depth of 100 m at the edge of a steep canyon (Bascom, 1980). The relatively warm, low-salinity waste plume rises in a way which depends on the temperature and salinity of the surrounding water. If the water column is well mixed, the plume will reach to the surface, but since the ambient water is strongly stratified for most of the year, the plume usually does not rise higher than mid-depth and may even be held close to the bottom for periods. The South Californian coast experiences upwelling of nutrient-rich water at certain times of the year, and studies of several outfalls could not show differences in levels of nitrate and ammonium between stations beside sewage outfalls and other coastal stations (Eppley *et al.*, 1972). The dissolved material from the Santa Monica outfall was carried out to sea, along with much of the fine particulate matter, but there was some accumulation of the latter, and about 3 km of the sea-bed in the upper part of the canyon was altered by settlement from the sludge. This produced sediment with a high BOD, but the dissolved oxygen in the water did not drop low enough to make the area unfavourable to fish.

## B. EFFECTS ON LIVING ORGANISMS

18. This section first discusses effects on the various main components of the biota, and then deals with more general considerations.

### *1. Phytoplankton*

19. In view of the nutrients added to surface waters by sludge dumping from ships, and of the rapid reactions exhibited by unicellular organisms, the most immediate effects might be expected from the phytoplankton. Indeed, it is well documented that enhanced phytoplankton growth (as shown by C production) is a general result of nutrient enrichment by sewage in shallow coastal waters, whether the input is treated or untreated. When the input is from an outfall at the bottom in deeper water, the nutrients may not rise to the euphotic zone and there may be little effect on primary production (Bascom, 1980) but in well-mixed areas increased growth is indicated by elevated levels of chlorophyll *a*, and values 2-3 times normal have been measured round Californian sea outfalls (Eppley *et al.*, 1972). Surface waters have even been turned bright green by a single-species algal bloom off New Jersey (Ryther and Dunstan, 1971). Effects have been recorded in terms of increased number of cells as well as reduced species diversity, for example, off the north Adriatic coast, where algal cells per unit volume, averaged throughout a year at a slightly polluted station, were more than double those at a clean station some 40 km distant (Stirn, 1973). Off Lebanon, Taslakian and Hardy (1976) recorded substantial changes in the structure of coastal phytoplankton communities arising from sewage pollution. There are many examples of significant effects of this kind in bays or coastal areas where input may be from several sources including land run-off, rivers and sewage outfalls. Most sludge dumping grounds are in more open areas and effects are either rather localised or difficult to detect. In the New York Bight, for example, high but not excessive summer chlorophyll *a* levels have been measured over the dumping ground (National Marine Fisheries Service, 1972) and at Garroch Head in the Clyde neither unusual blooms nor elevated chlorophyll levels have been recorded relative to nearby urbanised coastal areas. If primary production effects were of little significance on these accumulating grounds, they would not be expected on dispersing grounds, and in fact in Liverpool Bay any extensive blooms, such as *Phaeocystis*, are usually attributed to a combination of inputs to the area rather than solely to sludge dumping.

20. When changes in primary production are detected, it should be emphasised that more than just enrichment by nitrogen and phosphorus can be involved. Dunstan (1975) studied the effects of secondary treated sewage and sewage sludge on several species of phytoplankton and

trace metals, chelation potential, organic load and vitamins, can assume significance. He noted great variation in the reaction of different species to the same sewage, and of the same species to different samples from the same treatment plant. Although the immediate effect may be seen in an increase of the growth or standing crop, long-term effects can occur, leading to alteration in the species structure of the community - an ecologically important event which can take place without obviously changing the algal biomass. However, one result may be the establishment of a bloom of toxic dinoflagellates. This can produce mass mortalities of marine organisms (Adams *et al.*, 1968) and can also have serious public health implications, as discussed later.

## 2. *Zooplankton*

21. In relatively enclosed areas where enrichment from sewage leads to increased primary production, such as in the Lake of Tunis (Stirn, 1973) very rich zooplankton populations have also been recorded, but this is not usually the case for sludge dumping grounds, and the study of New York Bight already referred to reported that no effects could be detected on zooplankton populations, with regard to either species composition or distribution. In deeper water, where particle movement is less dominant, larger-particle feeders may be favoured, and in the Clyde, euphausiids and *Sagitta* were seen in large numbers (McIntyre and Johnston, 1975). The levels of organochlorine residues in zooplankton from inside the Firth of Clyde tend to be higher than in samples from outside (William and Holden, 1973), but no adverse effects on the zooplankton have been demonstrated, and it is not possible to link the residues directly with the sewage sludge, since there are other significant inputs of organochlorines in the Firth.

## 3. *Micro-organisms*

22. Micro-organisms transferred to the sea in sewage are discussed later. Here we are concerned only with the effects that added sewage has on those already present in the dumping area. Large numbers of bacteria are present normally in sea water and the sediments, and since they tend to be associated with surfaces, the more turbid the water and the finer the sediment, the higher the populations. It would be expected that the deposition of sewage, adding both particles and nutrients, would favour the growth of bacteria but, although this is confirmed by studies in sewage-polluted coastal harbour areas (Persone and De Pauw, 1968), little relevant information is available from sludge dumping grounds.

## 4. *Intertidal organisms*

23. Since sludge disposal sites are normally offshore, effects on intertidal organisms would

beach. Even then, impact would be slight because of dilution, and the kind of changes described by Littler and Murray (1975) for a domestic sewage discharge to a rocky shore in California - a marked reduction in species diversity and cover of plants - would not be likely. However, sewage from intertidal and shallow water outfalls will have effects in the littoral zone. In some parts of the Clyde, eutrophication of the shore appears to have produced a general enhancement of most animal populations, and this may provide an early warning of future changes (McIntyre and Johnston 1975). In the Forth estuary, on the east coast of Scotland, the input is more concentrated and the macrofauna have been reduced to a single-species, high-biomass population in patches near an outfall.

24. Before leaving the intertidal zone, reference may be made to salt marshes, since there is an expanding literature on the effects of sewage on such habitats. Experimental studies of salt marshes treated with sewage sludge (Valiela *et al.*, 1975) showed that sludge, in spite of introducing considerable levels of heavy metals and chlorinated hydrocarbons, fertilised the marsh, increasing the total peak standing crops of vegetation.

## 5. *Benthos*

25. Abiotic changes in the deposit on sewage sludge disposal grounds have already been referred to, and there are closely associated changes in the biota - largely, in this context, animal communities. Since it has been shown that enhancement of primary production is not usually more than local and slight, increased fall-out from this source would not be expected, and the main effects may be attributed to direct settlement of sludge. It follows therefore that effects on non-accumulating grounds would be negligible, and field observations bear this out. In both Liverpool Bay (DOE, 1972) and in the outer Thames estuary (Shelton, 1973), where London sludge is dumped in shallow water (up to 20 m deep) with tidal velocities sometimes exceeding $1 \text{ m s}^{-1}$, bottom deposits are mobile and this is the main factor influencing the benthos which in general is sparse. A slight local increase in polychaete worms may be attributed to sewage input, but even this is speculative. In the Southampton-Portsmouth area, the Needles spoil ground receives 22,500 tonnes of sludge per year, and here also, with rapid water movement, no effect on the benthos is obvious after many years of dumping (Jenkinson, 1972).

26. On accumulating grounds, on the other hand, significant effects are well documented. Sewage sludge from the city of Hamburg was dumped daily on a muddy ground in 20 m depth just NE of the Elbe I lightship in the southern North Sea, and Caspers (1975) reported massive concentrations of certain benthic bivalve molluscs (in particular *Abra alba* and *Nucula nitida*) and also the polychaete worm *Pectinaria koreni*. In Delaware Bay where dumping started in 1961, the benthic fauna in and around the dump site a decade later was diverse and abundant, and large numbers of species characteristic of pollution were not collected. However, the small bivalve mollusc *Nucula proxima* was very numerous (up to over 12,000 per $m^{-2}$) in soft sediments about

the sediments. Near the discharge point of the Los Angeles 7-mile outfall the benthic biomass ranged between 435 and 1,284 g $m^{-2}$ compared with 70 g $m^{-2}$ in the control area (Bascom, 1980).

27. The best documentation is probably the New York Bight (Pearce, 1972; Pearce *et al.*, 1976). Areas devoid of normal benthic life were found where sediments contained more than 10 per cent dry weight organic matter, and on the edges of this ground a community dominated by the burrowing anemone *Cerianthus americanus* was present. The adverse nature of the environment at the centre of the dumping ground is suggested by observations on lobsters and crabs (Young and Pearce, 1975), showing that some specimens were affected by 'shell disease' (skeletal erosion) and by a pathological condition of the gills. The cause of these abnormalities is not known, and although there is some experimental evidence to link them with sewage sludge, it was difficult to confirm this in the field because of the great variety of polluting inputs to the Bight. Further, in the absence of a record of the percentage incidence of the disease, it was not possible to assess its significance. In the Clyde the situation is comparable if less extreme (Mackay *et al.*, 1972; Topping and McIntyre, 1972). In the centre of the dumping ground, the biomass of the macrobenthos is several times higher than on clean ground nearby, and the number of individuals is greater by more than an order of magnitude. However, the area affected is quite small - about 10 km.

28. In the tropics, pollution problems may be more acute, since many organisms there live closer to their tolerance thresholds, particularly in relation to oxygen, than their counterparts in colder waters. Although there is no information directly on offshore dumping, there are some studies of sewage discharges. Coral reefs may be particularly susceptible. Moderate release of treated sewage seems to enhance reef production, but acceptable levels of enrichment have not been determined. It has been observed that oxygen reduction only slightly below the normal level of a healthy reef caused the death of many reef inhabitants. Johannes (1975) describes how the south end of Kaneche Bay, Hawaii, receives secondary treated domestic sewage in a shallow basin of about 800 hectares. More than 99 per cent of the coral in it is now dead, and transplanted living specimens died within a few weeks. Survival in other areas showed significant correlation with such sewage-related variables as phosphate content of the water and turbidity, but the exact cause of death was thought to be anaerobic conditions in the sediments leading to hydrogen sulphide release. A reef alga became dominant in the area and the ecosystem totally changed.

## *6. Fish*

29. The benthic invertebrates which flourish at the centre of long-established disposal sites - the few species of polychaete and oligochaete worms - are not usually consumed by fish, but active fish feeding takes place on the greater variety of species whose populations are enhanced on the edges of the grounds. In the New York Bight, 22 fish species, of which six were common, were recorded actively feeding on the edge of the ground, and several others such as herring and

mackerel were occasionally numerous (National Marine Fisheries Service, 1972). A study of this area (Pearce *et al.*, 1976) was not able to produce conclusive evidence that fisheries were adversely affected. In the Clyde, the rich concentrations of large plankton at the centre of the dumping ground attracted gadoids which could be observed by underwater television feeding above the bottom. Finally, the abundant invertebrates reported on the Hamburg sludge ground would be clearly attractive to demersal fish. No serious adverse effect on fish is therefore indicated. There are records, however, of fin rot disease in several species of flatfish caught in the Bight (O'Connor, 1976), and a connection between this and pollution was suggested. This was supported from studies of flatfish around the 7-mile Los Angeles outfall (Bascom, 1980) but, as with diseased crustaceans, the causes are not understood. The present position may be summarised as follows. While it may be accepted that fish-disease incidence is potentially a very sensitive index of incipient stress, the presence of a unique disease or pathological condition has not been specifically linked with a single pollutant or group of pollutants. However, there is much controversy. Some researchers (Dethlefsen, 1984; Vethaak, 1987) attribute the high disease prevalence in local areas to pollution, while others (Moller, 1986) cite natural phenomena. More recently McVicar *et al.* (1988) surveyed fish disease in a sewage sludge dumping ground, on a control area in the vicinity and in areas well clear of the pollution sources. No adverse effects from the sewage sludge could be detected, and indeed a correlation could be made between the dumping areas and lower levels of disease. The authors concluded that any such field correlations should be treated with caution.

## 7. *Sea-birds*

30. It is well known that many sea-birds are attracted to coastal sewage outfalls. A study in Scotland (Pounder, 1974) showed that the largest wildfowl flocks in the river Tay concentrated round those outfalls with the largest domestic flow rate and the highest ratio of domestic to industrial discharge. Some of the species, such as goldeneys, are known to feed on a variety of benthic invertebrates which would be enhanced at such outfalls, but they almost certainly feed also on some of the discharged material itself. There is at present no indication that species feeding round the outfalls, or those feeding downstream of the discharges, are adversely affected in the short term, although the possibility of long-term contamination by heavy metals or organochlorines cannot be dismissed.

31. Unfortunately, comparable studies of sea-bird feeding over sludge dumping grounds are not available, but feeding activities on the same scale as on coastal discharges are unlikely. Birds are quickly in evidence when material is ejected from ships at sea, but the mechanics of sludge dumping, and the nature of the material, make it unlikely that much of it would be available for long to feeding birds, although it has been suggested that birds are more attracted to vessels dumping primary sludge than to those carrying the more finely divided digested sludge. The main point at issue is probably whether contamination from sludge could ultimately affect birds via residues in their food, but in view of the low contribution of contaminants by sewage sludge in

comparison with other sources of input, the undoubted threat to birds from toxic chemicals (Bourne, 1976) is most likely to arise largely from those other sources. On the other hand, birds may constitute a threat to public health. It has been recognised that they are potential vectors for disease, and it is possible that micro-organisms picked up on sludge dumping grounds could be transferred to food processing establishments or inland reservoirs.

## 8. *Marine mammals*

32. As in birds, high body burdens of certain contaminants such as mercury and organochlorines have been recorded in seals and other marine mammals (Sergeant and Armstrong, 1973; Holden, 1972) and in some cases the high levels are associated with areas of urbanisation. However, marine mammals are even less likely than birds to feed directly on materials from sewage sludge, and build-up of body residues of contaminants is expected to occur via the food, presumably through fish in many cases. Since input of contaminants to the sea via sewage is usually relatively low compared with other sources, and the food chain is circuitous, it would be difficult to establish a significant link.

## 9. *General considerations*

33. The effects from sewage disposal will clearly be of two kinds, first those resulting from inorganic and organic enrichment in the water column, or major chemical and physical alteration of interstitial water and sediments, and second those related to potentially toxic components, chiefly metals and organochlorines. While there is evidence of some build-up of metals in the sediments of dumping grounds it has not been clearly shown that this has any adverse effect on the biota, although some aspect of contaminated sediments may be associated with shell disease in crustaceans and fin rot in flatfish. The main changes recorded can be attributed to the enrichment effect and to physical and chemical alteration of the sediments and interstitial water, leading to a species-poor but usually biomass-rich benthic community. The implications of this for the feeding of commercial fish have already been discussed, but it should be noted that for shellfish species such as *Nephrops*, which depend directly on mud to provide a burrow, the habitat itself is altered by the material deposited, and species are driven out.

34. Since the deterioration of sediment quality caused by dumping results in a progressive elimination of the more sensitive groups of organisms, it is possible to recognise stages in this progression and use these as indicators of deterioration. Thus it is generally accepted that reduction of a benthic community to a few species of worms, in particular the polychaete genus *Capitella* and the oligochaete *Peloscolex*, is an indication of gross pollution and that the next stage is total elimination of the fauna. In Kingston Harbour, Jamaica, organic pollution produced

polychaete *Spiochaetopterus oculatus*, and these zones were found to be increasing at the expense of normal mixed communities (Wade *et al.*, 1972). A lesser degree of enrichment leads not to reduction but to enhancement of the existing community, but often with one or two of the normally abundant species favoured. Thus, in organic-rich sediments in the New York Bight, Pearce *et al.* (1976) noted greatly elevated numbers of the anemone *Cerianthus americanus*, the polychaetes *Capitella capitata*, *Nephtys incisa*, *Mediomastus ambiseta* and *Pherusa affinis*, and the bivalve mollusc *Nucula proxima*. In European waters, another species of *Nucula* along with *Abra alba* and the polychaete genus *Pectinaria* have been found enhanced on the edge of dumping grounds.

35. While the use of macrobenthos as indicators is well documented, the possibility of using meiobenthos (small metazoans passing through a 1/2 mm mesh) has also been investigated. On a sandy beach where the macrofauna was enhanced, probably by pollution, the meiofauna, particularly the benthic copepods, was reduced in overall species and the distribution of those which remained was confined to the less polluted areas (McIntyre and Johnston, 1975). Similarly, on a sewage polluted beach in the Baltic, Arlt (1975) found the copepod *Nitocra spinipes* to be abundant in the most polluted zone, and the normally common *Tachidius discipes* absent. Although meiofauna has not been extensively studied on sewage sludge dumping grounds, a preliminary sampling at Garroch Head suggested much reduced numbers at the centre of the ground.

36. It does seem clear that benthic organisms can give a useful indication of the extent of pollution, but it is probably important to examine the overall pattern and structure of the community rather than to rely on a single species.

37. Sediment characteristics may also be used, and Johannes (1975) suggests that an expanding anaerobic layer, increasing sulphide levels, and dropping redox potentials are a rapid means of tracing impact of sewage on benthic communities.

# II. PUBLIC HEALTH

## *1. Pathogens*

38. A wide variety of pathogenic organisms is present in sewage and sewage sludge, including viruses, bacteria, protozoans and parasitic worms. The numbers and types of these organisms vary greatly, depending primarily on the health of the community which generates the sewage. Treatment of sewage before disposal does help to reduce the risk of infection, since anaerobic digestion inactivates bacteria such as salmonellas, shigellas and vibrios as well as many viral parasites, and kills cysts of *Entamoeba histolytica*, although the eggs of roundworms are able to survive. It is known that sea water is to some extent bactericidal and virucidal, but there are antagonistic factors capable of lowering those properties, and the fate of specific micro-organisms in the sea is difficult to determine. Coliform bacteria have perhaps been most extensively studied because they are used as indicators of sewage pollution, although there are other sources such as land drainage, rivers and seagulls, some of which can be more important than sewage. The die-off of coliforms in the sea has been studied both experimentally and *in situ* (Gameson and Gould, 1975) and it is known that while they survive well in the dark, solar radiation is a major source of mortality. Increasing attention is being directed to viruses, and the fact that enteric viruses are known to survive in the marine environment and that only a few virus particles are needed to initiate infection (Scarpino, 1975) would seem to justify this. However, there is at present a divergence of opinion on the factors responsible for viral inactivation in the sea, and further research is needed (Akin *et al.*, 1975).

39. In nearshore waters, contamination by sewage of edible species, such as filter-feeding shellfish, is a recognised problem, and techniques of cleansing and certification have been developed to deal with it. Contamination is unlikely to extend from offshore disposal grounds to coastal areas, and on offshore grounds the possibility of bacterial contamination will make exploitation of shellfish unattractive even if the ground is sited where stocks are available. New disposal grounds will probably be selected to avoid shell fisheries, but in the New York Bight, a circular area of 11 km radius round the sewage sludge dump site was closed to shellfishing in 1970 by the U.S. Food and Drug Administration, and this was later extended to include the Long Island and New Jersey shorelines where coastal outfalls and river input contributed sewage contamination (Stanford *et al.*, 1981). What little information is available on fish from dumping grounds does not suggest any public health problem. Further microbiological research is, however, required. The environment of dumping grounds is thought to be favourable to the building up of strains of micro-organisms resistant to pollutants such as heavy metals, and it has been suggested from work

(Koditschek and Guyre, 1974; Timoney *et al.*, 1978).

## 2. *Toxic chemicals*

40. Contaminants such as heavy metals, chlorinated hydrocarbons and oil are usually present in sewage sludge and build-up of at least some of these in the sediments on disposal grounds has already been discussed. Lear *et al.* (1981) recorded the uptake of certain metals by scallops and clams on a sewage sludge dumping ground off the Delaware-Maryland coast. The work of Halcrow *et al.* (1973) shows that while there is enrichment of some metals in the sludge dumping ground in the Clyde, this is limited to a relatively small area, and levels of metals in fish do not seem significantly greater than in other parts of the U.K.. Elsewhere, surveys of the concentration of metals in commercially fished species of fish and shellfish from around the U.K. have revealed (particularly in the case of mercury in fish) increased levels in two areas used for the disposal of large quantities of sewage sludge (HMSO, 1971 and 1973). However, inputs of heavy metals via sewage sludge are only one of a number of possible inputs, and it is difficult to establish the degree to which sewage sludge dumping contributes to this situation. Production of input budgets may assist our understanding in this respect, and in the Clyde (Topping, 1974) it was shown that the sewage sludge input for cadmium, copper, lead, and zinc made up only 29 per cent of the total, and the contribution from atmospheric deposition was apparently greater. However, contamination of marine produce from whatever source does require the attention of public health authorities.

## 3. *Toxic blooms*

41. Extensive blooms ('red tides') of certain planktonic organisms, in particular of the dinoflagellate genera *Gonyaulax* and *Gymnodinium* can lead to mass mortalities of marine animals (particularly sedentary filter feeders) and to the accumulation of a toxin in molluscan shellfish which has serious public health implications, the main hazard being paralytic shellfish poisoning (PSP). Red tides are a regular occurrence in certain parts of the world, but seem to be increasing in intensity and spreading to new areas.

42. Experimental work suggests that dinoflagellates thrive best under conditions of low salinity and high organic enrichment. Some authorities consider that the presence of biologically active substances of terrigenous origin are required to initiate a bloom, and red tides have been linked with sewage discharges, which may aid the development of blooms by providing a favourable environment for the dinoflagellate and a less favourable one for its competitors and predators. Specific work by Doig and Martin (1974) on *Gymnodinium breve* which causes red tides along the Florida coast showed that enrichment by municipal waste increased its cell concentration significantly.

# III. GENERAL DISCUSSION AND CONCLUSIONS

43. Detecting, evaluating and monitoring the effects of sewage disposal present several problems. As is clear from the above discussion, it is often difficult or impossible to isolate effects due to such disposal from others resulting from river outflow, land run-off or other dumping. Both New York Bight and Liverpool Bay are examples of areas with multiple inputs where this difficulty has been encountered. Even when a sludge dumping site is relatively isolated, the normal variability of natural populations often makes it difficult to attribute observed changes to specific causes without a long series of studies. Thus, changes detected on a ground where dumping is comparatively recent may not be unequivocally attributed to sewage, and it is only on grounds such as the Clyde, which have been in use for many years, that alterations in the sediments and benthos can be related unambiguously to dumping.

44. In order to detect sewage in the presence of other contaminants, or to trace its spread from the source of input, some means of recognition is required. The usual approach is to identify some component of the sewage load, which would not have been in the area already, or at least not in significant quantity. Tomato seeds have been found useful in this context, and coliform bacteria are routinely used as indicators. Steroids, which are minor components of plants and animals, have also been suggested, since due to their taxonomic specificity they can provide distinctions between sources. Thus steroids unique to mammalian faecal material could be useful indicators, and of those, coprostanol has been suggested. It has been used to identify sewage contamination in New York Bight sediments (Hatcher *et al.*, 1977) and in sea water and sediments off Japan (Kanazawa and Teshima, 1978). However, for success in this context, resistance to breakdown is important, and the degradation of coprostanol was studied experimentally by Bartlett (1987) who showed that in sludges it decayed to less than 15 per cent of initial levels after 30 days, but that in sediments the levels remained essentially unchanged after 54 days. He concluded that continuous or intermittent sewage discharges could be traced in sediments, making coprostanol a useful tool in assessing the area of impact, and representing an additional detection device which can be used with advantage together with other techniques. This confirms the work of Yde *et al.* (1982) who compared coprostanol with bacterial indicators of faecal pollution and showed that, while the determination of coprostanol is less sensitive than the detection of bacterial faecal indicators, it is preferable in some circumstances, such as when the samples cannot be analysed immediatley after collection, since coprostanol can be preserved and analysed later. A note of caution was sounded however by Venkatesan *et al.* (1986), whose work on Antarctic sediments showed that the presence of coprostanol could be attributed to the faeces of marine mammals.

45. It is worth considering the ultimate fate of sewage at sea. This will clearly depend on a number of factors, including its dispersion and decomposition in the water column and its rate of settlement and degradation on the bottom, all of which will determine its eventual rate of accumulation in the sediment. As already noted, there may be accumulation of the minerals or refractory substances which settle quickly (depending on the local hydrography). But the bulk of the material is volatile or degradable and while much of this will reach the bottom, its rate of accumulation will be a balance between input, dispersion and degradation. Duedall *et al.* (1975) have pointed out that in the New York Bight, although total organic carbon values are high, they are not anomalous compared with some areas outside the dump site, and the C:N ratios in the sediments are low, indicating nitrogen is not limiting. These facts suggest that processes exist for significant and perhaps rapid breakdown of the sludge. In support of this, it is known that a lipid biopolymer, cutin, can make up 12-28 per cent of the organic matter in sewage sludge, and that certain bacteria and fungi can grow on cutin as the sole source of carbon (Kolattukudy and Purdy, 1973). Finally, laboratory studies by Grunseich and Duedall (1978) show that the availability of oxygen is the major factor controlling the decomposition rate of sewage, and they estimate that the lower limit for the residence time of sewage sludge in the New York Bight apex is about 19 weeks.

46. From the information reviewed here, it would appear that sewage disposal in a properly selected marine area would not normally produce unacceptable effects. Indeed, on a dispersing ground, significant effects would probably not be measurable. On an accumulating ground, short-term localised effects may be detectable in the water column, but these are likely to be of little significance. After a long period of dumping on such a ground, a marked change in the sediment and the benthos will result, but there is not likely to be a public health risk unless commercial shellfish are contaminated, and any damage to benthic stocks will be localised and quantifiable. While these conclusions are based on the best available data, they should not be taken to imply that no additional information is required. Further research on existing dumping grounds, and a greter understanding of the physical and biological processes involved in contamination are required to produce a better assessment of the problems. It should also be remembered that the existence of the sludge implies the existence of a liquid fraction from the treatment of the original sewage, and this must be disposed of. It will be high in nutrients but, if discharged into well-mixed turbulent waters, need not create a problem.

47. The major concern is thus probably not from offshore dumping but rather from shoreline disposal of untreated sewage and from coastal discharges at the shoreline or in shallow water, where the near-field environmental processes do not provide adequate dilution (Carter, 1975).

48. For the latter, there may be a significant 'floatable' content and therefore an amenity problem (Ludwig, 1975), but the greatest risk will be to public health. It has already been suggested that for sludge disposal in the open sea such risks are slight, but when coastal towns discharge untreated domestic sewage to shallow water, especially during the holiday season when the population may be increased by orders of magnitude, minor faeces-borne upsets to eyes, nose and

and viral hepatitis can be disseminated. The influence of different degrees of treatment on the overall environmental effects of the resultant material merits further study, but at the other extreme it must be recognised that in many parts of the world sewage is disposed of directly on the shore in areas where no plumbing or drainage systems are available.

49. While each individual operation of pipeline disposal or ocean dumping will have local effects, if it can be shown that sewage is causing detectable degradation of a significant number of estuaries and other sea areas throughout the world, then we may have a problem which is regional or even global. Some satisfaction may be derived from the knowledge that there is an increasing degree of control of sewage discharges thanks to various international pollution conventions. The London Dumping Convention gives global coverage and various regional conventions (Barcelona, Helsinki, Oslo, Paris) are concerned with dumping at sea and/or land-based discharges in defined sea areas. Thanks to these conventions, input data should increasingly become more detailed, more accurate and more available, and programmes of surveillance should be under way to detect and keep under review the effects of the inputs.

50. However, opposition to sewage sludge disposal at sea is strengthening. In the United States, the 12-mile dump site, at 27 m depth in the New York Bight apex, which had been in use since 1924 (Santoro, 1987), and which received 6.6 million metric tonnes of sewage sludge in 1985 was closed at the end of 1987. In its place, the 106-mile site is utilised, which lies off the edge of the shelf in depths ranging from 1,430-2,800 m. At this site, benthic impacts are unlikely since significant amounts of sludge will probably not reach the bottom - it has been estimated that it would take about four months for sludge to reach the sea floor in that area (O'Connor *et al.*, 1985).

51. Similar attitudes are strong in western Europe. The United Kingdom is the only country still dumping sewage sludge in the North Sea, and there is serious discussion on a project to transport sewage sludge in a 100,000 t tanker to a site on the abyssal plain some 400 km west of France in 4,000 m depth of water (Davies, 1988). The waste would be passed down a discharge pipe to just above the sea-bed, thus greatly lengthening potential pathways back to man, and avoiding the surface plume which is the main threat from the deep-sea mining of metalliferous muds and manganese nodules. While there are moves in some quarters to stop all sea disposal, a proper compromise may be achieved by using the very deep sea, and ensuring that there are no pathways for toxic substances or pathogens back to man; that bottom-living species and ecosystem processes are not threatened; and that the effects can be adequately monitored.

# IV. SUMMARY

52. Sewage enters the sea mainly by two routes - via marine outfalls, or by dumping at the surface from ships. The sewage may be disposed of raw, or it may receive treatment which will divide it into a liquid phase and a sludge. In addition, when sewage is disposed of in fresh water, its components may reach the sea via rivers.

53. Effects of sewage are related in particular to the impact of its nutrient and carbon content on the ecosystem and to public health effects of sewage pathogens via recreational activities and consumption of contaminated seafood. Sewage will also contain, depending on its origin, variable amounts of toxic substances, chiefly metals, synthetic organic compounds and oil.

54. When delivered from outfalls, sewage effects, if the outfall has been adequately designed and properly sited, are largely on the sea-bed in the vicinity of the effluent point.

55. When dumping at sea, a decision has to be made between an accumulating or a dispersing site, and there are arguments for and against each. The accumulating site confines the impact and facilitates monitoring but damages a limited area. The dispersing site dilutes contamination and spreads the fertilising effect, but obscures the long-term impact.

56. Properly controlled sea disposal should ensure that sewage does not damage public health, reduce amenity or adversely affect ecosystems. The advantages of deep-ocean sites are under discussion and one is already in use, although a vociferous lobby opposes ocean disposal.

57. Alternative options to sea discharge are incineration or land disposal of one type or another. The costs of each of these options are readily calculated but it is difficult to quantify the relative environmental effects and risks.

TABLE. EXAMPLES OF ANALYSES OF SEWAGE SLUDGE DISCHARGED FROM SHIPS AND OUTFALLS

(A) Composition of sludge discharged at Garroch Head, Scotland
(Topping and McIntyre, 1972)

| Constituent | Range of values |
|---|---|
| | *(percentage of dried solids)* |
| Volatile matter | 65-76 |
| Nitrogen (as N) | 2.5-3.1 |
| Phosphorus (as $P_2O_5$) | 1.3-1.4 |
| Fat and grease | 10.7-20.9 |
| Silicon (as $SiO_2$) | 13.5-19.6 |
| Heavy metals | *(ppm dried solids)* |
| Lead | 500-630 |
| Copper | 320-740 |
| Cadmium | 4-14 |
| Iron | 11,400-22,900 |
| Chromium | 100-1,300 |
| Manganese | 300-700 |
| Nickel | 70-110 |
| Zinc | 1,200-1,900 |

(B) Average concentrations of constituents discharged through the 7-mile outfall (1971-1979) as measured by the Hyperion Treatment Plant laboratory (H. Schafer).
Average flow was 4.6 million gallons per day.
(Bascom, 1980)

| Constituent | Average value |
|---|---|
| | *$mg\ l^{-1}$ dry weight* |
| Total solids | 8,700 |
| Oil and grease | 710 |
| Ammonic nitrogen | 290 |
| Total phosphorus | 240 |
| Cyanide (CN) | 0.37 |
| Phenols | 0.41 |
| Silver | 0.72 |
| Arsenic | 0.22 |
| Cadmium | 0.96 |
| Chromium | 10.0 |
| Copper | 13.2 |
| Mercury | 0.25 |
| Nickel | 3.4 |
| Lead | 2.42 |
| Zinc | 21.0 |
| Selenium | 0.63 |
| | $\mu g\ l^{-1}$ |
| DDT | 3.7 |
| PCB | 20.4 |

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# ANNEX XIII

# DEVELOPMENT OF COASTAL AREAS

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# INTRODUCTION

1. Marine scientists responsible for categorizing various substances and activities, as these have affected the water quality or health of coastal and estuarine habitats, have ranked *coastal development* as the single, most degrading item (Smith *et al.*, 1988). In doing this scientists have recognized that development *per se*, or urbanization and industrialization, contribute significantly to both the toxic contaminant load and to eutrophication, as well as to the physical degradation of coastal areas. The Club of Rome's Report, "The Limits to Growth" (Meadows *et al*, 1972), notes that development in itself results in increased use of resources, more contamination, and far more physical degradation and reduction in carrying capacity. The issue of development therefore logically includes many items formerly categorized individually, for instance, toxic contaminants; nutrients and concomitant eutrophication and hypoxia; fresh water diversions for drinking, irrigation, and industry; and dredging of harbours and building of piers, or harbour development.

2. The World Commission on Environment and Development (WCED) was created by the United Nations (UN) General Assembly at the 38th Session of the UN in the fall of 1983 and four years later released its report entitled "Mandate for Change". In the latter several issues were identified to be covered in a report entitled, "Our Common Future", commonly cited as the "Brundtland Report" after its chairperson, Gro Harlem Brundtland, of Norway (Brundtland *et al.*, 1987). The report discusses the concept of *sustainable development* , provides narrative which links industrial development and urbanization with degradation of habitats, and raises the promise of producing more with less as well as managing the commonly held resources of the world's oceans. The Brundtland Report emphasized that: "The major land-based threats to the oceans require effective national actions based on international co-operation." Such efforts would be predicated on inventories of coastal and marine resources (and their present uses); establishment of priorities and goals; review of the legal and institutional requirements for integrated management of the EEZs, as well as the terrestrial developments which compromise these; and the rapid clean-up of industrial and municipal pollution which threatens estuaries, coastal waters, and the continental shelf fisheries and habitats. The requirement for co-operation and co-ordination as needed for the management of the UNEP Regional Seas was stressed. Perhaps most important, the Brundtland Report stressed that today increasing numbers of persons "...live beyond the World's ecological means..."

3. Until recently, about the time of the first international Earth Day (April 1972), there was relatively little reported on the total or overall effects of development on habitat quality or "health"

habitat quality due to development. Although there are some earlier references, one of the first studies which evaluated contaminant loading from an urbanized area (Providence, RI, U.S.A.) located on a well defined estuarine system (Narragansett Bay) was carried out by the Marine Ecosystem Research Laboratory of the University of Rhode Island (Oviatt, 1980). In addition to field measurements, the same researchers also studied the fates, behaviours, and effects of selected contaminants in experimental "mesocosms" in order to verify observed field phenomena.

4. This annex is based on the review of several hundred published papers and widely distributed reports as well as on the material contained in the 15 draft "Regional Seas Reports" available at the time this paper was being prepared. Without exception these Reports indicated that coastal development was having a major effect world-wide.

5. Very recently, workers investigated changes in the flux of several major ions as these are carried from rural and urbanized areas through catchments leading to estuaries in Great Britain (Prowse, 1987). Nichols *et al.* (1986) summarized the effects of several activities associated with development and how these have diminished the quality of habitats in San Francisco Bay. The introduction to the paper stressed the changes in fish populations as these were related to the early ecological changes which occurred at the turn of the century. Shifts in the Bay fisheries presumably resulted from "a combination of overfishing, elimination of essential habitats, and changes in water quality." San Francisco Bay is a classic example of an estuary affected by a multiplicity of man's activities. Various toxic substances have been and are being introduced through agricultural developments, industrial discharges, and point and non-point sources of domestic wastes. As with other West Coast "estuaries" (San Diego Bay; Commencement Bay/Tacoma; Elliot Bay/Seattle; and San Pedro Bay/Los Angeles), San Francisco Bay sediments had intermediate to high levels of contaminants; parallelling these findings, the incidence of liver lesions in demersal fishes "tended to vary with the degree of urbanization - highest prevalences occurred at sites with the highest levels of chemical contamination" (Varanasi *et al.*, 1988).

6. However, the Bay has been even more drastically affected by reduced river inflows; these have occurred as a direct consequence of the construction of dams, reservoirs, and canals used to increase storage of water, as well as increased annual export rates of waters moved to other regions to support urbanization and agriculture (see Halim, this report, as well as Rozengurt, Herz, and Josselyn, 1987).

7. Very recently aquatic habitat managers and scientists have turned their attention to assessing the *cumulative impacts* of land development on coastal resources. Dickert and Tuttle (1985) have reviewed the literature and have identified critical issues that must be addressed in formulating any planning system that might incorporate cumulative impact assessments. This was done using a case study of a coastal wetland basin, Elkhorn Slough, California. Methodologies were developed and implemented to be used in the assessment of the *cumulative* effects of erosion and sediment deposition in a wetland.

8. Most of the draft UNEP Regional Seas Reports commented on how the changes in land use patterns were affecting coastal water quality. For instance, near the Black Sea agricultural lands were rapidly being industrialized and urban centres concomitantly grew in size. This was true in Bulgaria, Romania, and Turkey (Balkas *et al.*, 1987). Even the finest and most valuable Turkish tobacco-growing areas and coastal waters are now industrialized and affected by industrial wastes and urban pollution.

9. The aforementioned reports indicate that in recent years the scientific community has begun to measure the chemical and physical effects resulting from "development" and to use the data so obtained to assess the events caused by several stresses. More important, a number of research groups have begun to develop centralized, compatible historical data bases to be used in the analyses of water quality and management of aquatic habitats (Stoddard *et al.*, 1986). Yet other scientists have persisted in developing the concept of *assimilative capacity* of coastal and oceanic waters for pollutants. Assimilative capacity was first discussed extensively as an issue during a NOAA Workshop at Crystal Mountain, Washington, U.S.A. , 29 July - 4 August 1979 (Goldberg, 1979). While the participants found it difficult to define "assimilative capacity" precisely, there was a consensus that"... it is the amount of material that could be contained within a body of sea water without producing an unacceptable biological impact". One of the objectives of the Workshop was to produce "endpoints" which would show when an unacceptable impact had occurred. General deliberations revolved around problems related to estuaries as well as coastal and oceanic waters. Four panels studied the site-specific problems of an oceanic disposal site (the so called 106 mile site), Puget Sound, New York Bight, and the southern California Bight. The four studies were done in a case study mode; the results led to a workshop conclusion that "... the waste capacity of U.S. coastal waters is not now fully used." However, in two of the areas, Puget Sound and the New York Bight, there was evidence presented that "... the assimilative capacities for some substances or in some areas have been exceeded." In these two instances the areas affected were in close proximity to major developed urban areas which discharged many wastes in point and non-point modes.

10. The concept has been further discussed by ICES and GESAMP, sometimes as *environmental capacity* or *accommodative capacity.* The application of the assimilative capacity concept will be discussed in the concluding paragraphs of this section. In a similar vein, the Brundtland Report (Brundtland *et al.,* 1987) emphasized *sustainable development.* Industrial operations must be conducted in a far more efficient manner in terms of resource use (and recycling of worn out products), generating less pollution and wastes, using renewable rather than non-renewable resources, and using far less energy.

11. A factor most important in managing development, *economic cost,* is now being dealt with in a more systematic, realistic manner. Using spills of oil and other hazardous materials as a starting point, Yang, Dower, and Menefee (1984) have reviewed the use of economic analyses in

(long-term distributed costs) are now being analyzed in case studies of marine habitats (Fontaine, personal communication).[1]

12. Finally, the U.S. National Research Council, Board on Basic Biology, and the Canadian Environmental Assessment Research Council held a binational workshop on cumulative environmental effects. The proceedings included extensive sections on effects in marine systems. In his paper Dayton (1986) stressed that total, cumulative effects must be considered in any management scheme or programme of assessments. Moreover, legislation, regulations, and management protocols must emphasize the additive effects of various forms of development. Generally, it is not individual insults which degrade marine ecosystems but, rather, their cumulative effects.

13. Following up on Dayton's remarks, Waldichuck (1986) points out that most assessments in British Columbia (and probably other areas) have *not* considered cumulative effects. He stressed the *immediate* need for a conceptual framework within which "... the cumulative environmental impacts from contaminants and development in estuaries can be evaluated." Moreover, in his conclusions he noted that "... the environmental impact of *expansion* of an existing structure (or level of contaminant) can be approximated by examining the impact of the existing facility (or generic information)." This is an extremely important thought; the consequences of an existing stress on a habitat, an invertebrate population, or a fish or shellfish stock, can be projected to other habitats or populations, especially where increases in contamination or degration are proposed. The formulation of management plans for terrestrial developments that affect coastal zone and riparian habitats leading to estuaries must be predicated upon generic or existing historical data and information.

14. The remainder of this annex will describe the several factors known to be associated with development and pollution and which have a clear bearing on the health of aquatic habitats. By using a number of *case studies* from tropical as well as temperate waters, the paper documents the consequences of various types of development, their effects on living resources and other amenities, as well as certain economic costs which might be associated with continued or intensified development in pristine as well as already highly urbanized areas.

15. Recognizing that land use is the principal factor, after population size, in determining how harbours and the coastal zones in general have been, and will continue, to be used, NOAA has recently compiled its National Estuarine Inventory Data Atlas, Volume 2: Land Use Characteristics (NOAA, 1987). This volume summarizes, by estuarine drainage systems, the various uses to which coastal regions are put on a national basis.

16. Finally, recognizing that development has affected the coastal seas, international bodies (Brundtland *et al.*, 1987), the U.S. Congress (1989), and other agencies and organizations have called for immediate remedial action and have provided recommendations for action. For instance, the U.S. Congress (1989), based on a two-year study, has recommended that the United States

should expand pollution control efforts in coastal ecosystems, using a regional approach; control development and wastes; and protect and manage the marine fisheries far more rigorously and efficiently.

# I. FACTORS ASSOCIATED WITH DEVELOPMENT

## A. DEMOGRAPHIC PATTERNS

17. Scores of reviews have been written on the relationships between increases in human populations, urbanization, and effects on the environment (see Meadows *et al.*, 1972, for example); many of the early ideas on the subject were developed at the turn of the century, refined during the '20s and '30s, and popularized in volumes such as "The Road to Survival" (Vogt, 1948) and "Our Plundered Planet" (Osborne, 1948). Increasingly, marine scientists have recognized that improvements in marine aquatic habitats will be made principally by managing terrestrial and riverine habitats, especially through the use of planning and zoning ordinances which will control human population densities and activities as well as development in the more sensitive portions of the coastal zone.

18. There have been numerous studies of harbour and port development; most major maritime cities have been studied from an economic point of view and appropriate reports and popular articles prepared, based on the resulting data. One such study was written on New York City (Koebel and Krueckeberg, 1975) and is of special interest because it details, in a case study, urban and subsequently suburban development, and suggests relationships between population increase, increased pollution, and environmental decline, especially in the coastal zone. Moreover, their paper considers New York City in the context of *megalopolis*, an extended 800 km urbanized tract that connects Richmond, VA and Washington, DC with Philadelphia, PA; Newark, NJ; New York, NY; Bridgeport and New Haven, CT; Providence, RI; Boston, MA, and Portland, ME. One of the real hallmarks of the late 20th Century, *megalopolis*, and their central urban areas, have had measurable impacts on principal estuaries and coastal zones associated with the Northeast coastline of the United States, as well as those of several other regions of the United States and other nations.

19. As pointed out by Koebel and Krueckeberg (1975), New York City grew from a village of 270 people in 1628 to a gotham of 19 million in 1970; this was due initially to its fine port and later to its proximity to a canal and rail system which allowed manufactured merchandise to be shipped inland and agricultural and forestry products, as well as hard minerals, to be shipped outward to Europe or to the southern colonies of the U.S.. In later decades, growth was fostered by continued human migration to the city to support elaborate and profitable mercantile and industrial activities. The Ports of New York, Newark, Philadelphia, Baltimore, and Norfolk have

urbanization. Likewise, ports in Europe, the United Kingdom, Latin America, Asia, and Africa have also grown tremendously in recent decades. Gross increases in pollution, continued decreases in estuarine water quality and effects on living marine resources accompanied such growth in each port that has been investigated. Some of the first recognized pollution effects on living resources were due to tainting of shellfish with petroleum products (Goode, 1987) and metallic substances, i.e. copper, released from early 20th Century refineries and smelters located on the shores of the Port of New York and the adjacent Raritan Bay. Similar effects were reported early on in Great Britain (Roberts, 1926).

## B. PORT AND HARBOUR DEVELOPMENT AND MAINTENANCE

20. The importance of New York's excellent port in the early years of the development of the city is well documented in "The Rise of New York Port" (Albion, 1939) and its development parallels that seen in other countries. Since the 1930s several studies have further documented the role of port development and harbour maintenance in bringing about changes in habitat quality; the most recent example for our case study would be "Port Facilities and Commerce", by A. Hammon (1976). Again, while this volume is concerned with New York City *per se*, the narrative is relevant to many other temperate and subtropical port cities. As it is not possible to detail changes in every major port or harbour, New York is described as a case study with events seen in other harbours added as appropriate.

21. At the time Hammon's paper was written, New York was "America's busiest cargo port." The paper detailed the extent of the various shipping channels, their length and depth, and also documented the amounts of several types of cargoes moved through the harbour areas. Commodities carried included petroleum products, "basic" chemicals, liquified gases, a range of ores and minerals, and manufactured goods. To provide an example, the amounts of foreign refined petroleum products imported to the port in 1963 and 1972 were 19.3 and 30.5 million short tons, respectively. Foreign raw oils imported increased from 9.0 to 13.1 million short tons during the same period. Shipment of domestic matter also increased greatly in the same decade; materials and commodities moved *within* the harbour, best referred to as intraharbour commerce, increased from 39.5 to 54.6 million short tons, and regional waterborne commerce changed from 10.5 to 22.7, a doubling of the original amount.

22. Associated with shipping is a significant amount of pollution. In a study commissioned by the U.S. Maritimne Commission (Harris, 1973), it was projected that 15.8 million litres of "oily wastes" would be generated in the Port of New York alone each day in 1980. While much of this waste is treatable at shore-side facilities, an undetermined amount escapes to harbour waters. Tanacredi (1981) considered just one component of oily wastes, drained automotive crankcase oil. He found it present in Jamaica Bay, which flows to the New York Bight, in significant amounts,

carrying with it large amounts of wastes. The situation in smaller, developing, countries is similar or worse. Coastal cities in East Africa annually receive 6.5 million tonnes of crude oil, along with the associated consequences of discharged oily ballast and flush-outs conducted in coastal waters; no modern methods for clean discharges are in place (Bryceson *et al.*, 1987).

23. In the Kuwait area, a major centre for the production of oil, a principal cause of oil pollution is, again, the discharge of ballast and bilge wastes. Linden *et al.* (1987) cite estimates of 150,000 Mt of oil polluting the area annually.

24. Human wastes released from shipboard have always represented a major form of pollution released to harbours. Beginning with the Federal Water Pollution Control Act of 1972, the U.S. Government took steps to require the use of, and promulgate, performance standards for marine sanitation devices (MSDs). These are capable of storing and/or treating human wastes while vessels operate *within* harbours having restrictive discharge regulations. Vessels with MSDs discharging wastes to harbours after 30 January 1980 were to meet U.S. EPA standards of 2 faecal coliform bacteria per millilitre and 150 mg suspended solids per litre. Again, it has not been established how much effect such contaminants have on the overall water quality, nor how such wastes affect living marine resources or amenities of interest to society. Several descriptive reports suggest, however, that this pollution does have an effect, especially in non-maritime or local ports. Unfortunately, very few ports have installed the necessary facilities, including MSDs.

25. As will be covered in other contaminant sections of this report, contaminants enter harbours via many conduits including: run-off from streets, roofs, and parking areas; rivers and streams; point discharge from pipes and point dumping; the atmosphere; and vessels themselves. Most forms of contaminants are then carried from harbour areas by ebbing tides or are circulated throughout an estuary where they eventually combine with suspended matter to settle to the sea floor (see Kullenberg, 1986). Once in place and associated with sediments, many forms of contaminants may remain sequestered with the sediments until they are resuspended by wave and current action, as during storms, or until the sediments are removed by dredging carried out to create or maintain depth of shipping channels, turning and berthing areas, or dockside facilities.

26. In our present case study, the urbanized New York harbour, the U.S. Corps of Engineers constructs and maintains 78 different waterways; rates of siltation vary by waterway and thus frequency of required maintenance dredging varies as well. Between 1965 and 1970 the principal disposal area, the "mud grounds" in the New York Bight, received 25.5 million $m^3$ of dredged spoils. The smallest volume (1.1 million $m^3$) occurred in 1970, the largest (8.8) was generated in 1968. An undetermined but substantial amount of these volumes is heavily contaminated with organic and inorganic toxicants; therefore, the amount of contaminant inputs to the coastal waters, as well as the release of materials at the dredging sites, varies from year to year. This obviously makes precise measurements and assessments of the consequences difficult.

27. The physical effects of dredging and deepening of channels to foster increased volumes of vessel traffic and make possible the entrance of larger ships to harbours also have been shown

to have immediate significant biological consequences. Durand *et al.* (1985) noted that channelization of a harbour system of the Ivory Coast of West Africa had a major effect. The harbour at Abidjan was originally largely limited to, and characterized by, relatively fresh waters present in lagoons generally isolated from the ocean. With the opening of the Assinie Pass and the digging of the Vridi Canal, the interior waters, as was the case of those of San Francisco Bay, became increasingly saline. With an expansion of harbour facilities came rapid increase in human population and industrialization; similar rapid increases in human population in temperate habitats, i.e., Scotland, U.K., have been reported following industrial growth and improved harbour facilities (Currie and MacLennan, 1984). Direct discharge of untreated sewer wastes from the city of Abidjan to its harbour has resulted in anoxia and eutrophication. Changed patterns of circulation resulting from dredging allowed saline waters to move to fjord-like bays; these salty waters then became isolated at the bottom, contained high levels of "sulphur-bearing" matter, and were rarely renewed or exchanged. The resulting anaerobic conditions led to formation of hydrogen sulfide inimical to fish life.

28. The consequences of physical modification for Abidjan were exacerbated by the effects of domestic pollution; 400 thousand persons contribute to the loading of a sewer network which discharges into a lagoon *without* treatment (Dufour *et al.*, 1985). Bryceson *et al.* (1987) report similar conditions in the Eastern African region, Mauritius, where dredging and land reclamation have affected coastal lagoons. Lagoon systems in Latin America (Sierra de Ledo *et al.*, 1985) and other continents are also measurably affected. Sen Gupta and Qasim (1987) note that in India only 50 percent of the total population bordering the Indian Ocean is provided with proper sanitation arrangements; only 20 percent of 3.8 million people living in coastal cities in eastern Africa are properly sewered (Bryceson *et al.*, 1987). The result of these conditions is that there are high coliform counts on beaches and in coastal waters. In many instances fish have been completely eliminated from coastal streams and estuaries; fish and shellfish exposed to raw sewage are sold for human consumption in Tanzania's capital. Continued urban and agricultural development in many third world countries, without provision for proper sanitation and waste treatment, will exacerbate already existing problems.

29. Even in the Kuwait area, which has considerable resources for use in pollution abatement, dredging of harbours as well as aggregate mining interact with discharge of sewage in a way that causes local damage. Present sewage treatment in the area covers about 58 percent of the population of coastal urban centres (Linden *et al.*, 1987).

30. In many harbours the accumulation of leaking or released cargos, discharged wastes, and eroded silts and sands requires regular dredging with the result that dredged spoils must be deposited in other habitats. Siltation due to erosion and transport of mine tailings are issues in the South Asian Seas region (Sen Gupta *et al.*, 1987) as well as in northern Europe, the Mediterranean (Jeftic *et al.* 1987), North America, the Southeast Pacific region (Escobar *et al.*, 1987), and other regions. Several countries are now considering techniques to isolate dredged materials and dumped

prevent the escape and movement of contaminated sediments (Bokuniewicz, 1983). Several national agencies are now investigating how to identify and categorize harmful organics, for instance petroleum hydrocarbons, and to identify those substances of petroleum origin which should be considered in the management and regulation of ocean disposal of toxic substances. Workshops and teams have begun to review chemicals in terms of (a) which chemicals should be regulated or isolated by capping or other means and (b) the recommendations subsequently used in fisheries resource management (Schmick *et al.*, 1986).

31. In regard to managing contaminated wastes various other schemes have evolved. In the early 1970s a review was commissioned on the possible effects of an enlarged supertanker terminal in the New York metropolitan area. The authors of the final report stated that the operation and construction of such a terminal would be more harmful to the fisheries and other marine life if located *within* an enclosed estuary, i.e., the Raritan Bay, than if placed *offshore* in the open Bight (McHugh *et al.*, 1972). Increasingly, it will be necessary to depend upon such assessments to manage our coastal waters. This is especially true where large-scale industrial endeavours are involved in development, or where numerous small projects will have collective effects. The latter emerges as the more difficult issue to resolve.

32. Cronshaw, in his introduction to a case study on Swansea Bay, a temperate habitat, (Collins *et al.*, 1980), lucidly states the need for baseline information in regard to principal harbours and developed areas, especially as these might be affected by physical changes or chemical contamination. A series of generic questions are posed and must be understood and addressed by all managers of aquatic habitats if future degradation is to be avoided.

33. Similar concerns and questions have been raised by aquatic scientists working in semitropical and tropical areas. Browder *et al.* (1986) have documented how development has degraded, or potentially degraded, the wetlands along the Gulf of Mexico between Florida and Texas, using a case study of changes in one semitropical estuarine system affected principally by physical changes (channelization), compared with other systems not so affected. These authors used relative abundances in numbers and species of fishes as a basis for comparison.

34. Gomez *et al.* (1987), Escobar *et al.* (1987), Sen Gupta *et al.* (1987), Halim *et al.* (1987), and Bryceson *et al.* (1987) all reported concern for effects of contaminated dredged materials and other toxic wastes associated with harbour development and operation in the tropical Pacific and Indian Oceans. Sen Gupta *et al.* (1987) reported that in India 105 ports are dredged daily to maintain channel depths; they noted that fisheries in areas affected by dumping have been depleted or have disappeared.

35. In the tropical Kuwait and Red Sea areas, Linden *et al.* (1987) and Halim *et al.* (1987) reported special problems with channel sediments heavily contaminated with oily wastes. Problems of sediments contaminated with metals are identified in the Red Sea.

## C. INTRODUCTION OF CONTAMINANTS

36. In a volume entitled "Urbanization, Water Pollution, and Public Policy," Carey *et al.* (1972) alerted regional managers to the dangers associated with continued urban development. The authors interviewed scores of scientists and managers about the issues that could be identified and for which solutions could be developed. One of the major concerns expressed was the use of then (late 60's) existing or historical data to project the effects of urbanization on the declining water quality in the New York Metropolitan area. The authors suggested that the consequences of "...continued economic growth has produced its own antithetical need to limit and even reverse growth...". Subsequently many other studies have been done to determine the relations between growth (development), contamination, and effects.

37. In the NOAA volume on Land Use Characteristics (NOAA, 1987) several graphs show the dominant land use in estuaries and the percentage of land-use categories, and pointed out that most major U.S. estuaries have at least one highly urbanized environment situated on their shores. In another related report, NOAA has found that many of these urbanized coastal environments have been contaminated with, and degraded by, toxic contaminants (NOAA, 1987a and 1988), especially in the northeast region of the U.S. and at major west coast embayments. The highest concentrations of trace elements (metals) in sediments were found in the Salem and Boston harbour areas, in the western Long Island Sound, and in Raritan Bay, all located near heavily populated and industrialized areas or in the path of waters moving from such developed areas.

38. Considering organic contaminants, such as aromatic hydrocarbons (PAHs), DDT, and PCBs, the same east coast estuaries were generally reported to be contaminated, often well above "typical values" for estuaries or harbours, as measured by the NOAA Status and Trends Program. Casco Bay, the relatively small harbour of Portland, MA, also had high values of PAHs, as did the San Diego harbour. Chlorinated hydrocarbons were abundant in sediments collected from the Boston harbour and Raritan Bay, but also in sediments taken from sites off Los Angeles; this probably reflects the effects of "well-documented waste discharge", including entrained DDT within the region.

39. It is not surprising to find elevated levels of contaminants in sediments and target organisms from urbanized estuaries, or coastal waters receiving materials from such areas. Off-shore sediments and organisms collected over broad geographic areas (Nova Scotia to the Delmarva Peninsula) were recently surveyed for the presence of organic and inorganic contaminants. Ocean quahogs (*Artica islandica*), a bivalve shellfish, from Georges Banks and off Nova

habitats in the New York Bight, Rhode Island Sound, and Buzzards Bay, however, were more contaminated, with values up to 25 ppb (Steimle *et al.*, 1986). The same study reported PAH values from non-detectable to 55 ppb, with the highest values from the New York Bight stations. The authors also reported that in regard to toxic trace metals, "elevated trace metal levels were also usually associated with known areas of inputs, e.g. waste dumpsites, or adjacent to heavily industrialized coastal areas."

40. In a very recent review and evaluation of available data on the distribution of contaminants in tissues from living marine resources taken in New England waters, an excellent case study (Capuzzo *et al.*, 1987) concluded that "there are serious contamination problems along the New England coastline." Moreover, their review of a large data set, involving several species and covering extensive geographic and temporal scales, led them to state: "Urban harbors are repositories of many wastes and the most serious evidence of chemical contamination in fish and shellfish populations is found among samples collected from harbor locations." Similarly, Varanasi *et al.* (1988) hypothesized that increased contaminant loading and prevalence of disease in fish were found in urbanized areas.

41. To date, the relatively high levels of chlorinated hydrocarbons in livers of demersal fish from the above sites have *not* been necessarily correlated with high values in sediments. Future investigations at these sites will attempt such correlations using "models" based on other contaminants and fish species.

42. Highly urbanized coastal environments represented by sites in Boston Harbor, Salem Harbor, and Raritan Bay all had high levels and counts of the organic contaminant coprostanol and the bacterium *Clostridium perfringens*, both, again, highly indicative of municipal domestic sewage.

43. Similar research and monitoring in temperate areas affected by urbanization, waste disposal, agricultural run-off, and other forms of development show similar loadings of contaminants to river and estuarine situations. In a detailed study (Prowse, 1987), demonstrated terrigenous export of elements (ions) from urban areas (Southampton, Hampshire, UK) up to 104 times (average 10 times) those from rural areas. In part, the enrichment of ions in urban settings is attributable to disturbances and exposure of deeper layers of exposed sediments to atmospheric acidity and consequent mobilization. Other anthropogenic sources of inputs of ions include building materials; industrial and domestic pollution; atmospheric deposition from surrounding industry; and highways, railways, and airports, as well as garden and lawn fertilizers (Prowse, 1987).

44. In a recent study of inputs of nutrients, trace elements and toxic organic substances, researchers at the University of Rhode Island report that most nutrients ($NH_4$, $NO_2$ and $NO_3$, $PO_4$, and silicates) are carried into Narragansett Bay by the Providence and Pawtuxet rivers. Copper was largely introduced from sewage treatment plants, and most lead had its origins from combined sewage outfalls (CSO); the latter are systems where run-off from streets, roads, parking

as Narragansett Bay. During periods of heavy rainfall, sewage wastes are flushed directly to waterways, along with roadwastes and general debris.

45. The increased nutrient loadings to Narragansett Bay were found to have caused summer hypoxic conditions in the Providence River, as well as in the upper reaches of the Bay and the salt wedge. In the United Kingdom, 1972 levels of nitrate reported in small rivers (i.e. the Frome in Dorset) were 42 percent greater than levels over a preceding period of seven years. Although farmers were not exceeding Ministry of Agriculture recommendations, 25 percent of the applied nitrogen in a basin was being carried to reservoir waters (Green, 1984). Similar amounts are undoubtedly exported to coastal waters where they may increasingly exacerbate hypoxia, cause reduced summer oxygen values, and prove lethal to fishes. In regard to several contaminants, development in the Merseyside in the U.K. has been a slow, centuries-long, steady process, accompanied by continued degradation of the Mersey estuary (Handley, 1984). As reported for other coastal waters, the BODs associated with heavy domestic and industrial pollution loads have caused oxygen depletion and odour problems. Similar concerns have also been raised in relation to other U.K. estuaries, including Swansea Bay (Collins *et al.*, 1980).

46. In the temperate Orient there have been numerous reports of the effects of contaminants which result from development and industrialization. One of the first reports (Nakai *et al.*, 1972) provided details as to how paper pulp mills had a "cascading" effect on Suruga Bay; organic residues from discharges of pulp mills resulted in reduced oxygen levels and clogged shipping routes, thus requiring dredging and the placing of contaminated dredge spoils in other parts of coastal waterways. Once more, Japanese scientists noted that human and industrial wastes have affected both phyto- and zooplankton, bottom dwelling (benthic) animals, and finfishes as well as the fisheries *per se.* Most effects were measured at mouths of rivers draining developed or urbanized areas (ports) leading to Suruga Bay and similar embayments. The effects included biological uptake of organic and inorganic contaminants, changes in the abundance of planktonic forms and fishes, and the presence of benthic indicators of pollution including bacteria and certain polychaete worms, i.e., *Capitella capitata.* These are now recognized as classic signals of pollution-induced changes.

47. The developing economies of the European countries bordering on the Black Sea have resulted in the introduction of organic and inorganic contaminants, however, little is known of the associated effects. Measurements of contaminants from Turkey are virtually unavailable (Balkas *et al.*, 1987).

48. Regional Seas reports from all locations or Regions in the tropical Pacific and Indian Ocean also indicated that major categories of contaminants are released to coastal waters. Generally, however, there are few data on consequences or effects.

49. In its most recent general report, the Government of Japan's Environment Agency

inhabited districts (DID), which stood at 41 million persons in 1960 "...soared to 70 million in 1980". This was accompanied by a significant change in ownership of personal vehicles (from 7 million to 26 million between 1970 and '84), a principal indicator of the spread of urban lifestyles and a major source of contamination to terrestrial and aquatic ecosystems associated with the DIDs. With increases in urbanization and the use of automobiles, there has been a concomitant increase in road construction and other components of the Japanese urban infrastructure necessary to transportation and vehicular movement.

# II. EFFECTS OF URBAN SUPPORT FACILITIES, POWER PLANTS, ROADWAYS, AND OTHER INFRASTRUCTURE

50. It is obvious that, with some exceptions, i.e., recreational and domicile development, most coastal development and urbanization occurs around principal coastal cities, ports, and harbours. The nature of ports varies; there are large ports with various kinds of development, much of which serves international commerce, and there are smaller ports (non-maritime) which tend to function in coastal (local) commerce and transport (Fawcett and Liffmann, 1985). All ports, however, are generally dependant on a wide range of services and support facilities. Railways, roads and highways carry commerce and commodities to and from ports. Steam electric and nuclear power plants are almost invariably located on harbour, local rivers, or nearby coastal waters. In the tropics and sub-tropics, Ocean Thermal Energy Conversion (OTEC) is being considered as a local, "unpolluting" form of energy development (Myers *et al.*, 1986). Many of the workshops and meetings on port development and associated activities address a range of economic concerns but few can be found which address the *collective* impacts of such development on habitat quality and the fisheries (Fawcett and Liffmann, 1985).

51. There are reports and published papers on specialized aspects of, say, OTEC development and the fisheries (Myers *et al.*, 1986). In this instance the authors suggest that effects on fisheries are not projected to be "significant enough to deter the early development of OTEC"; however, the effects of conventional power plants, nuclear and steam electric, are thought to be significant, especially where thermal additions may exacerbate other stresses (see Jensen, 1976; Jones *et al.*, 1975; and Krenkel and Parker, 1969).

52. The collective effects of major highway and railroad developments on aquatic systems are very poorly known, but are deemed, intuitively, to be important factors. In their recent paper, Colwill and Thompson (1984) assessed the impacts on terrestrial habitats due to trunk roads and motorways developed in the U.K. over the past 25 years. They note that factors contributing to the *total* environmental impacts of highway systems are numerous, including visual intrusions (human and wildlife), noise, emissions from exhausts, contaminants from vehicle movement (dust, salt, spray, air turbulence), and contaminants from cargoes *per se*. Not identified specifically are the direct and indirect consequences of highway construction: sediment erosion and siltation, damming or clogging of waterways, elimination of wetlands and bottom habitats, interference with movements and colonization of aquatic life, introduction of contaminants from asphalt and other construction materials, and so on.

53. Three items were noted to be of special importance: tetraethyl and tetramethyl lead

pollution, gaseous emissions (carbon monoxide, hydrocarbons, oxides of nitrogen), and salt used for de-icing roadways (Colwill and Thompson, 1984). The full significance of these three factors in relation to aquatic habitat quality is, again, not fully appreciated. However, several reports have suggested that principal sources of lead and petroleum hydrocarbons involved roadways, parking areas, and garages. Likewise, de-icing salts applied to roadways are seen as inimical to freshwater and upper estuarine ecosystems. While these factors do not affect directly marine offshore habitats, they certainly affect anadromous species and many estuarine-dependant marine fishes. The authors suggest that the effects of highway construction and operation"... can be minimized by careful attention to detail during the planning and construction and subsequent maintenance stages."

54. One other important factor likely to be associated with harbour development and urbanization in the future is the matter of resource recovery systems. As landfills and other terrestrial and aquatic waste disposal sites become "filled" or unusable, greater attention will be given to incineration or sorting of wastes and recycling. Once more, there are few hard data or assessments on how resource recovery facilities affect habitat quality, but future urbanization and related mitigation should consider observed as well as possible effects.

## A. RECREATIONAL DEVELOPMENT

55. The use of coastal waters for bathing, diving, boating, fishing and recreation in general is increasing. In Florida, 65 percent of the *residents* over 18 years of age, or 5.2 million persons, use beaches annually. This results in over 36,000 jobs (with an annual payroll of $240 million) and total "sales" of $1.1 billion (Bell and Leeworthy, 1986). These authors noted that development of marinas, fishing piers, and general tourism adds even more to the economy; *tourist* uses of swimming beaches, alone, adds $3.4 billion and 142,638 jobs.

56. The support of a large sector of an economy obviously results in an increased pressure to develop further urban coastal areas for recreation and tourism purposes. Areas viewed as wetlands or swamps are under pressure to be modified as harbours for small vessels, as walking trails, as well as for recreational beaches. The location of coastal construction setback lines may be governed by the value of beaches (per square foot) for recretional purposes as opposed to their use for habitat to living resources or for fisheries purposes.

57. Beaches are often identified for modification, i.e., beach sand nourishment, widening of beaches, or beach stabilization. Rarely are the effects of such activities (or the long-term distributed costs) factored into an evaluation of how development or urbanization affects the quality of the habitats needed for the fisheries and the health of coastal seas. Recently, studies have been

commissioned to determine how beach restoration (Nelson, 1985) or creation of beaches might be accomplished. Nelson (1985) has provided guidelines for meeting certain biological or habitat criteria.

## B. DEVELOPMENT AND LOSS OF WETLANDS

58. Perhaps the one single most debilitating activity affecting the well-being of the coastal zone is the continued alteration of wetlands. In temperate zoogeographical provinces, wetlands that are being filled, dredged, or physically removed are characterized by the presence of grasses and other rooted vegetation such as *Spartina* or *Zostera*. In the tropics, coastal wetlands are often formed by hardwood trees and bushes; mangrove stands are especially important in protecting and stabilizing subtropical and tropical coastal zones. Browder *et al.* (1986) have detailed the changes in semitropical wetlands and associated fauna when channelization occurs.

59. While there are hundreds of papers on the subject of the importance of wetlands, especially mangroves, few have provided the basis for long-term coastal monitoring and management. Herz (1987) reported on the use of a form of remote sensing, aerial photography, to verify rapid change due to the alteration or removal of mangrove ecosystems along the coasts of Brazil. These ecologically crucial species are being removed for fuel and to provide new lands for agriculture. Their exploitation for such purposes is most prominent along the northern Brazilian coasts where mangrove and related vegetation are the key species; some 84 percent of the mangroves of Brazil are on the "north coast." In the industrialized eastern and southern coasts, however, mangroves occur in smaller pockets and clusters and therefore are extremely vulnerable to domestic and industrial pollution (Herz, 1987). Herz, and others, have documented the areas of temperate and tropical wetlands lost to urban, industrial, and agricultural development and the provision of cheap fuels.

# III. CONCLUSIONS

60. The foregoing indicates that there is a wide range of ecologically stressful factors associated with development, be it urbanization and industrialization or the implementation of more intensive agriculture or changes in agricultural practices. New syntheses and assessments, such as the NOAA Land Use Characteristics Data Atlas (Donovan and Tolson, 1987) or the compilation of data from remote sensing studies of change in mangroves (Herz, 1987), will be used for benchmarks against which future changes can be compared. These, and other techniques will, in the future, provide the data bases for enhanced zoning and management of sensitive coastal zone habitats. It is important now, however, to be cognizant of how measured change affects habitat quality and living marine resources. To date, few programmes have attempted the difficult task of relating (a) changes due to development to (b) field and laboratory measurements of biological effects or to (c) changes in the distribution and abundance of important living marine resources (fisheries stocks) or human health, as affected by the totality of development.

61. Using certain macropollution (dredging, contaminants, reduced dissolved oxygen) histories and indices, Summers *et al.* (1986) have demonstrated the effects of various forms of pollution or consequences of development on fish and shellfish stocks in several North American estuaries. Using robust computer and statistical techniques these authors related long-term habitat quality to the relative abundances of certain key resource species. This application of historical or generic data to real, current, resource and habitat management questions is most important if steps are to be taken to reverse the documented changes which are occurring in fishery resource species as a result of coastal development. Moreover, it has been suggested that, by considering certain aspects of a species' life history, it is possible to predict the consequences (additive) of pollution on an already stressed species (Schaaf *et al.*, 1987).

62. In addition to the use of historical or generic data to provide assessments useful in conserving aquatic habitats, *models* have been used in legislation, litigation, and zoning to resolve long-festering environmental issues. For instance, in a case where the actual impacts of electric power generation on estuarine resources was unknown, relatively "simple" models were used to evaluate alternatives for mitigation, thus allowing a negotiated settlement in a court case (Barnthouse *et al.*, 1984). Several investigators, however, have cautioned on unreasonable overdependence on models and modelling.

63. In few instances multijurisdictional management of the seas is now underway, usually spurred by measured effects of man's activities. Responsibility for man's impact in the Wadden Sea is not confined to any one country. Therefore mitigation and abatement must become multi-

national in their implementation, monitoring, and enforcement. Mineral exploration, mining, shipping, waste discharge, fisheries, industrial and domestic development, and recreational activities have all come into conflict, and proper aquatic habitat management rapidly becomes a very complex endeavour. Even differing kinds of fishing compromise one another; yet continued multiple-use management depends upon consensus as to the conflicts and issues, resolution of conflict(s) (based on agreed upon standards or criteria), monitoring to ensure proper evaluation of mitigation and abatement efforts, and feedback to the citizenry to ensure future support by the citizens and their elected representatives. Some of these matters were discussed by Wolff and Zijlstra (1980), Essink (1981), and others; their concepts as well as those of others have recently been incorporated into broader-scale management plans for the North Sea (Netherlands Ministry of Transport and Public Works, 1986). The "Water Quality Management Plan - North Sea" provides an overview of the factors that have led to the formulation of the plan and the steps necessary to implement a multijurisdictional, ecosystem-wide strategy for management of the North Sea.

64. Other areas have been proposed for multijurisdictional management; in the U.S.A., Chesapeake Bay is bounded by three states and the District of Columbia. Several major cities including Norfolk and Richmond, VA, Washington DC, and Baltimore MD, either sit on its shores or affect its water quality by virtue of river inputs, tides and currents, or canal systems. Scores of smaller cities are sited on the Bay's shores. Largely on the basis of focused studies of principal issues (low dissolved oxygen, increased contaminant burdens, etc.) which began in the mid 1970s, the U.S. Environmental Protection Agency (1983) developed "A Framework for Action" which specified steps to be taken in future decades to rectify the consequences of decades of unplanned development. This led ultimately to the writing of the "Chesapeake Bay Restoration and Protection Plan," a plan prepared by the States of Maryland, Pennsylvania, and Virginia as well as the U.S. Environmental Protection Agency (1985). Capitalizing on the various concerns and recommendations contained in these volumes, several interagency studies were initiated, which resulted in reports to be used in planning future development, as well as in the implementation of management steps. One recent paper from the Chesapeake Bay Living Resources Task Force, "Habitat Requirements for Chesapeake Bay Living Resources" (Maryland Department of Natural Resources, 1987) provides a base for developing *standards* and *criteria* for water quality necessary to manage the main fish species threatened by development on the shores of Chesapeake Bay. A second paper, "Vegetated Filter Strips for Agriculture Runoff Treatment" (Magette *et al.*, 1987) provides information on one way to manage the agricultural landscape so as to reduce erosion, manage runoff, and thus prevent excessive nutrient and contaminant loading.

65. Additional guidance is now forthcoming from management programmes, such as the Chesapeake Bay Program, which have been in place for over a decade. In some instances, i.e., the Thames estuary in the U.K., first steps were taken over two decades ago and improvements are now highly visible. Many national and state governments now recognize the need for coastal and estuarine management, and are taking steps to introduce interagency, inter-jurisdictional manage-

multiplicity of resources (King and Kendall, 1987). Since several management plans, which consider the totality of effects from development, are now in place, they should be considered as case studies which will provide useful guidance for the future. Equally important, however, is the matter of judging when aquatic habitats are being significantly degraded. To arrive at decisions in regard to the social importance of habitat degradation it is necessary to establish standards or criteria for recognizing unacceptable change. Recently, several agencies have worked together to develop *indices of degradation*. Such indices are now being tested and evaluated. O'Connor and Dewling (1986) have recently reviewed a set of indices applicable to marine habitat degradation, which result from a preliminary interagency consensus towards an approach to future assessments and management.

66. Finally, countries should, as emphasized by Waldichuck (1986) in his recommendations, begin to develop immediately a conceptual framework within which the cumulative, multi-jurisdictional environmental effects of contaminants and physical degradation can be evaluated. Without such an approach, bolstered by appropriate standards for water quality, and the criteria to judge these by (Passino and Smith, 1987), there will be no effective control of the adverse aspects of development.

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# ANNEX XIV

# SELECTED CONTAMINANTS: RADIONUCLIDES

A. SALO

FINNISH CENTRE FOR RADIATION AND NUCLEAR SAFETY
HELSINKI
FINLAND

## TABLE OF CONTENTS

# RADIONUCLIDES

## I. SOURCES, LEVELS AND QUANTITIES OF RADIONUCLIDES IN THE OCEANS

### A. NATURALLY OCCURRING RADIONUCLIDES

1. Radioactive substances have always been present in the oceans. Radionuclides of 16 of the 91 naturally occurring elements are present in measurable concentrations in sea water and there are probably trace quantities of radionuclides of an additional 16 [1]. The radionuclides of natural origin fall into two groups. The first comprises those with such long half-lives (greater than 8 $10^8$ y) that they still remain from the creation of the solar system; these are the primordial radionuclides. The major radionuclides within this group are $^{40}K$, $^{87}Rb$, $^{235}U$, $^{238}U$ and $^{232}Th$, of which the latter three are parents of decay series whose shorter-lived daughter members (such as $^{234}U$, $^{230}Th$, $^{226}Ra$, $^{210}Pb$, $^{210}Po$, etc.) are all present in sea water. The second group, the cosmogenic radionuclides, comprises those produced by the interaction of high-energy cosmic radiation with the atmosphere. Depending on their half-lives and chemical properties, varying proportions of the radionuclides produced in this way reach the oceans. The major radionuclides within this group are $^{3}H$ and $^{14}C$.

2. The distribution of the natural radionuclides in the oceans depends on their sources, half-lives and the degree of their involvement in physical, geochemical and biological processes. The radionuclides $^{40}K$, $^{87}Rb$, $^{235}U$ and $^{238}U$ are fairly uniformly distributed, with the greatest variations occurring in the source regions, such as the river estuaries which supply the greatest part of these radionuclides from the weathering of terrestrial rocks. Thorium-232, which is also derived from weathering and land run-off, is rapidly scavenged from sea water by particulate matter and transported to the sea-bed; the range of concentrations of this radionuclide which have been measured in open ocean waters reflects the relative intensities of this process. The main source of $^{226}Ra$ to the oceans is the decay of $^{230}Th$ in ocean sediments where it accumulates after being scavenged from the water column following production through the decay of $^{234}U$. A proportion of the $^{226}Ra$ produced in the sediment diffuses back into the overlying water [2] where its distribution is governed by water movements, radioactive decay and by involvement in biological and geochemical processes [3]. The cosmogenic radionuclides are not produced uni-

formly in the atmosphere and their deposition into the oceans, mainly by rain, is also non-uniform. Estimates of the concentrations and the total quantities of some of these radionuclides in the oceans are given in Table 1. For the first four radionuclides, the inventory has been estimated from the measured concentrations and a total ocean volume of 1.37 $10^{21}$ litres; for $^{226}Ra$, measured concentration-depth profiles have been integrated over the volumes of the depth zones of the oceans; and for $^{3}H$ and $^{14}C$ the data have been taken from reference 4. Reasonable accuracy in the total inventories can be obtained only for those radionuclides for which the details of the spatial distributions are well known.

3. Table 2 shows concentrations of some natural radionuclides in sediments, beach sand and common rocks.

4. In order to give an idea of the comparative concentrations of naturally occuring radionuclides in marine organisms, Table 3 is reproduced from reference 5. It should be noted that the $^{210}Po$ and $^{226}Ra$ concentrations could probably vary by about an order of magnitude either way.

## B. FALLOUT FROM NUCLEAR WEAPONS' TESTS

5. The major series of atmospheric nuclear test explosions were carried out during the years 1954-1958 and 1961-1962. The radioactive debris generated by these explosions has been delivered to the oceans from the troposphere and the stratosphere in both northern and southern hemispheres. The spatial and temporal variations in the deposition of $^{90}Sr$ have been studied in detail using data from a worldwide network of sampling stations. Although there have been many measurements of the concentrations of fallout radionuclides in the oceans, the fact that the distributions are still evolving makes it very difficult to estimate the present, time-dependent inventories. Estimates of the total input of certain of the longer-lived fallout radionuclides have been made by the Expanded Expert Panel of the London Dumping Convention (LDC) [6] and the resulting values will probably, due to decay that has taken place since their introduction, be overestimates of the actual oceanic inventories. Inventories of selected radionuclides in the oceans were also compiled by an IAEA group [72,73,74,75] based on observed data.

6. The radionuclides of primary interest from the radiation dose point of view are $^{3}H$ (tritium) from thermonuclear explosions, $^{14}C$ produced by the capture of neutrons in atmospheric nitrogen, the long-lived fission products $^{90}Sr$ and $^{137}Cs$, and plutonium generated by activation of $^{238}U$ or from unfissioned material. All of these radionuclides are also of interest in regard to wastes from the nuclear fuel cycle.

7. The total quantity of tritium produced by weapons tests has been estimated to be 2.4

$10^8$ TBq [4]. On the assumption that the deposition of tritium, as tritiated water, had the same dependence on latitude as that of $^{90}Sr$, and that the direct input to the oceans was directly proportional to the oceanic surface area in each latitude band, then the estimated total direct inputs are as given in Table 4 [6]. These values will underestimate the amount that has entered the oceans because a proportion of that deposited on land will have been carried into the sea by run-off.

8. It has been estimated that 2.2 $10^5$ TBq of $^{14}C$ have been produced during weapons' tests [4]. The available information indicates that at least 16.7 $10^4$ TBq has already entered the oceans, and that the greater part of the remainder will eventually do so [72]. The oceanic distribution of bomb $^{14}C$ in surface water is far from uniform. Generally, concentrations are higher in low to mid-latitudes, particularly in those areas with a relatively shallow thermocline. Concentrations are lower in areas of upwelling and where there is strong convective mixing with deep ocean waters as in the oceans surrounding Antarctica. Over the next few centuries present variations in oceanic $^{14}C$ distribution will become smooth and will ultimately reflect the age structure of ocean waters.

9. The input of $^{90}Sr$ to the oceans from fallout has been estimated as indicated above for tritium, and the input of $^{137}Cs$ has been calculated on the assumption that the empirical value of 1.6 for the $^{137}Cs/^{90}Sr$ activity ratio in deposition at many terrestrial sites can generally be applied to oceanic fallout [4]. Also, an overview of $^{137}Cs$ inventories by sea area (Table 5) has been made recently [73], based on observed concentrations.

10. The inputs of $^{239}Pu$, $^{240}Pu$ and $^{241}Pu$ have been estimated in the same way as those of tritium, assuming that the total quantities originally produced were 7.8 $10^3$, 5.2 $10^3$ and 1.7 $10^5$ TBq, respectively [4]. The results of these calculations are summarized in Table IV [6]. For $^{239,240}Pu$ an inventory based on observed concentrations in the oceans resulted in 16 $10^3$ TBq [74]. The difference between the estimates based on measurements of the ocean samples and input data are assumed to be due to an under-estimate of the rainfall and dry fallout and thus of the plutonium deposition over the oceans, or due to inadequate sampling.

## C. DISCHARGES FROM NUCLEAR FUEL CYCLE AND OTHER ACTIVITIES INVOLVING RADIONUCLIDES

11. There are authorized routine releases of low-level radioactive wastes to the environment from the normal operations of installations such as nuclear power reactors, reprocessing plants and other fuel cycle facilities. These releases from civilian and commercial installations are generally well documented as part of the monitoring and surveillance programmes imposed by national au-

thorities. Similar releases also occur from installations associated with the military nuclear fuel cycle, but data concerning these are not usually made public.

12. Releases from land-based operations may be both to the atmosphere and surface waters. The discharges to the atmosphere are usually dominated by radioactive noble gases which do not result in significant contamination of the oceans. Releases to rivers, estuaries and coastal waters give rise to local and regional contamination of the seas and oceans.

13. Actual discharges vary considerably, both between installations and over time. Given the number of installations potentially contributing radionuclides to the marine environment, the varying periods of operation and the number of different radionuclides involved, the estimation of present-day, decay-corrected, inventories would be a tremendous task. In addition, the results obtained would not, of themselves, be particularly enlightening. To provide some perspective, however, some attempts have been made by the Expert Panel to the LDC to obtain a maximum estimate of the annual releases of radionuclides in liquid effluents from land-based installations [6].

14. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) provides average values of the annual quantities of radionuclides released in the liquid effluents from a variety of reactor types normalized to unit electrical energy production per year [4]. Assuming that:

(a) these data can be generally applied to all installed nuclear power generating plants;

(b) the average load factors which can be derived from the data given apply to each reactor type worldwide;

(c) all liquid effluents can, potentially, reach the sea; and

(d) all $^{14}C$ discharged to the atmosphere also reaches the sea,

then the estimated annual inputs of certain radionuclides to the oceans (rounded up to one significant figure) are as given in Table 6 [6]. Also given are the average total annual discharges of certain radionuclides from reprocessing plants [based on reference 4]. It should be noted that not all fuel is reprocessed. In fact, only 5 per cent of fuel currently used for nuclear electricity production is reprocessed. The remainder is stored, pending decisions to reprocess or dispose.

15. For comparison with some actual cases, Table 7 gives the total releases from Sellafield, U.K., between 1952-1986 and from Cap de la Hague, France, 1966-1985 taken from reference [7].

of the tailings and surrounding rock or unconsolidated sediments, on the integrity of any retaining structure, and on the permeability of any capping material. These factors are likely to change with time. Close to a tailings site the concentrations of these radionuclides may be sufficiently high to be distinguished by measurements from the natural levels of these radionuclides, but further away the contributions could only be calculated. The radionuclides presented in Table 8 are those released to the environment according to UNSCEAR estimates [4] from conversion, enrichment and fuel fabrication facilities in the nuclear fuel cycle. These activities, at the front end of the nuclear fuel cycle, do not significantly alter the picture given before of the inventories of natural radionuclides in the oceans.

## D. DUMPING OF LOW-LEVEL PACKAGED WASTE

17. The bulky form and low quantities of radionuclides present in the majority of dumped waste precludes accurate estimates of the quantities of each radionuclide involved. However, estimates of the total quantities sent for disposal have been made, and these are summarized in Table 9 [6,8,9]. Small quantities of unpackaged and liquid wastes were also dumped in the western Atlantic Ocean by the United States [8]. The amounts of radioactive material that are shown in Table 9 are best-estimate values. It is not practical to state specific values for uncertainties. It can be noted in this regard that a recent re-evaluation by the United States of all available data for past United States dumping, which stopped in 1970, resulted in an increase in the estimate of the quantity of radioactivity dumped by about 25 per cent from earlier estimates [6].

18. For assessments of the consequences of past dumping at the sites in the North-East Atlantic Ocean, a more detailed breakdown of the composition of the waste by radionuclide has been estimated (Table 10) [9].

## E. ACCIDENTS

19. Four major nuclear accidents with radionuclide releases to the environment have taken place: Sellafield (Windscale), U.K., 1957; Chelyabinsk, U.S.S.R., 1957; Three Mile Island, U.S.A., 1979 and Chernobyl, U.S.S.R., 1986. Besides these, in 1968 a B-52 airplane from the U.S. Strategic Air Command crashed on the ice 11 km west of Thule Air Base, Greenland, contaminating the environment with plutonium and americium from the nuclear weapons carried by the plane. A few re-entries of nuclear powered satellites and losses of nuclear submarines have also occurred.

of $^{137}Cs$, 12 TBq of $^{106}Ru$ and 1.2 PBq of $^{133}Xe$. The principal route for human irradiation has been shown to be iodine in milk [4]. At Chelyabinsk in southern Urals, owing to a fault in the cooling system used for the concrete tanks containing highly radioactive nitrate acetate wastes, a chemical explosion occurred and 74 PBq radioactive fission products were released. The main radionuclides released were $^{144}Cr + ^{144}Pr$ 66 %, $^{95}Zr + ^{95}Nb$ 25%, $^{90}Sr + ^{90}Y$ 5.4 % and $^{106}Ru + ^{106}Rh$ 3.7 %. The main exposure pathways to man were ingestion of $^{90}Sr$ with food, in particular milk, and the external exposure from $\gamma$ emitting radionuclides [78].

21. In the Three Mile Island accident about 370 PBq of noble gases, mainly $^{133}Xe$, and some 550 GBq of $^{131}I$ were released to the atmosphere [4].

22. Rough estimates of the radionuclide releases from the Chernobyl reactor accident indicate that approximately 3 - 6 per cent of the core inventory of non-volatile radionuclides (e.g. $^{95}Zr$ - $^{95}Nb$, $^{141,144}Ce$, $^{103,106}Ru$, $^{140}Ba$-La, and transuranic radionuclides) settled out in the immediate vicinity of the plant in the U.S.S.R. [10,13]. Data from other countries indicate that relatively little of the available core inventory of the non-volatile radionuclides was transported outside the U.S.S.R., so the total release is not likely to have exceeded 4 per cent. Regarding the volatile radionuclides $^{131}I$, $^{132}Te$, $^{134}Cs$ and $^{137}Cs$, considerable amounts were transported outside the U.S.S.R.. It has been estimated [13] that roughly 20 - 25 per cent of the core inventory of $^{131}I$, of $^{134}Cs$ and of $^{137}Cs$ may have been released, and approximately equally distributed in the European part of U.S.S.R., the rest of Europe and the remainder of the Northern Hemisphere [11,13]. Regarding radio-caesium this would mean that the release could have been of the order of 100 PBq; UNSCEAR [13] has recently estimated the release of $^{137}Cs$ at 70 PBq. As the bulk of the released amounts has been found in the terrestrial environment in the U.S.S.R. and the rest of Europe, only a fraction of released caesium was deposited on to the ocean surface or is likely to reach the oceans with run-off. The deposition was very unevenly distributed within the U.S.S.R. and the rest of Europe. However, the deposit per unit surface area decreased strongly when moving further from the source, the maximum country average outside U.S.S.R. being 23 kBq $m^{-2}$ in Austria, decreasing to 0.14 kBq $m^{-2}$ in Japan and to 0.03 kBq $m^{-2}$ in the U.S.A. [12]. Thus, as the $^{137}Cs$ inventory in the oceans in the Northern Hemisphere was 410 PBq [6] and the contribution from Chernobyl, according to UNSCEAR [13], was 4.7 PBq, an addition of the order of one per cent to the inventory in the oceans in the Northern Hemisphere is implied.

23. The consequences of the Thule accident have been intensively studied by several expeditions to Thule. Most of the radioactive contamination was removed from the surface of the sea ice. However, approximately half a kilogramme of plutonium went to the sea bottom. Figure 1, reproduced from reference 14, shows the plutonium inventories in various compartments of the environment at Thule.

# F. OBSERVED CONCENTRATIONS OF SOME ARTIFICIAL RADIONUCLIDES IN WATER, SEDIMENTS AND ORGANISMS

24. For radionuclides from man's activities the variation according to the origin and the site of measurement vary to such an extent that it is not possible to give representative concentrations of radionuclides in the various ocean compartments. However, the only human practice that, to date, has resulted in measurable contamination of whole ocean basins has been nuclear weapons' tests. It can be mentioned that in deep ocean water the concentrations of $^{90}Sr$ and $^{137}Cs$ are generally less than 2 Bq $m^{-3}$ and those of actinide elements considerably lower [6,15].

25. Figure 2 shows $^{137}Cs$ concentrations in water (dpm /100 kg should be divided by 6 to get Bq $m^{-3}$) in North Atlantic and Arctic Oceans, originating from both nuclear weapons' tests and releases from reprocessing plants at Sellafield and Cap de la Hague [16]. $^{137}Cs$ and $^{134}Cs$ have served as water tracers as far downstream from their input as the Arctic ocean on account of their soluble properties and high discharge levels during a few years. The discharges from Sellafield peaked in 1974 to 1978. The releases from Cap de la Hague were substantially lower. The $^{137}Cs$ contribution originating from nuclear weapons' tests was only a few Bq $m^{-3}$ in the Atlantic surface water. In the North Atlantic and Arctic Oceans the absence of a systematic variation in $^{239,240}Pu$ concentrations with latitude indicates that fallout plutonium dominates, i.e. that the contribution from Sellafield is negligible relative to the fallout component. The concentration of $^{239,240}Pu$ was around 0.013 Bq $m^{-3}$ at the beginning of 1980s. However, somewhat higher levels of $^{239,240}Pu$, up to 0.026 Bq $m^{-3}$, were observed in the area between the Faroe Islands and Bergen, Norway, and in the waters immediately south of Norway. Half of this was considered to originate from Sellafield [17].

26. In the Mediterranean, $^{137}Cs$ concentrations originating from nuclear weapons' tests vary from about zero near the bottom to up to 5 to 10 Bq $m^{-3}$ at the surface [76,77].

27. In some closed shallow brackish-water sea areas, such as the Baltic Sea, the concentrations of $^{90}Sr$ and $^{137}Cs$ from nuclear weapons' tests reached 20 - 40 Bq $m^{-3}$, and those of $^{239,240}Pu$, 0.002 - 0.004 Bq $m^{-3}$ [18].

28. In a study area in the North Pacific 900 km southeast of Tokyo, the $^{90}Sr$ and $^{137}Cs$ concentrations range from 2.7 to 3.7 Bq $m^{-3}$ and from 4.4 to 5.1 Bq $m^{-3}$, respectively, and they are almost identical throughout the surface layer (0 - 500 m). The vertical distribution of $^{137}Cs$ is shown in Figure 3 [19]. Plutonium profiles at some North Pacific sites

*al.* and their results are summarized in Figure 5 [22]. As shown in this figure, relatively high concentrations of $^{239,240}Pu$ were found in the surface water near the Far East and South East Asia. A decrease of the surface plutonium concentration with respect to time was studied at similar locations ($\sim$30 $^{o}$N, $\sim$147$^{o}$E) during the KH-82-5 cruise of the R/V Hakuho-maru (22 Nov, 1982 to 25 Feb, 1983). Samples of surface oceanic waters from the North East Pacific and the East Tropical Ocean were analyzed for concentrations of $^{239,240}Pu$ and of $^{241}Am$ [22]. As compared with the previous data obtained by the Meteorological Research Institute of Japan, it is obvious from Figure 6 that the plutonium contents in surface oceanic water has decreased over a period of several years owing to the sedimentation of plutonium from the surface layer to a subsurface layer and the effect of dilution by circulating sea water.

29. Notwithstanding the low concentrations of fallout radionuclides in deep ocean water, significant quantities have accumulated in the superficial deep ocean sediment. For instance, Noshkin has measured concentrations of $^{239,240}Pu$ in the range 0.015 to 1.3 mBq $g^{-1}$ in the North-East Atlantic sediments [23]. In the Japanese research area in the Pacific 900 km southeast of Tokyo, at the depth of about 6000 m in the topmost 2 cm, concentrations of $^{90}Sr$, $^{137}Cs$ and $^{239,240}Pu$ originating from nuclear weapons' tests were of the order of 0.04 mBq $g^{-1}$, 0.4 mBq $g^{-1}$ and 0.07 mBq $g^{-1}$, respectively [19]. For comparison it can be mentioned that in the depths of the open Baltic Sea, (around 200 m, the deepest being 400 m) concentrations in sediments of up to 5 mBq $g^{-1}$ for $^{90}Sr$, 100 mBq $g^{-1}$ for $^{137}Cs$ and 5 mBq $g^{-1}$ for $^{239,240}Pu$ have been measured [18].

30. Typical values for $^{137}Cs$ in deep-sea fish muscle are of the order of 0.1 to 0.5 Bq $kg^{-1}$ [15] and those for $^{239,240}Pu$ are somewhat more variable, but generally of the order of $10^{-3}$ to $10^{-4}$ Bq $kg^{-1}$. In the Baltic Sea $^{137}Cs$ concentrations in the muscle of migratory fish (*Clupea harengus*) vary from 1 to 4 Bq $kg^{-1}$ due to global fallout [18].

31. Figure 7 shows $^{239,240}Pu$ concentrations originating from Sellafield releases in *Fucus vesiculosus* and *Mytilus edulis,* as a function of distance along the Scottish coast [17].

32. The Chernobyl accident has changed the concentration values found in particular closed sea areas such as the Baltic Sea, especially for $^{137}Cs$. Concentrations of $^{137}Cs$ in the surface layers of the Baltic Sea water were a few to some ten times higher than before the accident [24]. In 1987 no clear differences between the $^{137}Cs$ concentrations from coastal and offshore stations were found any more, and the concentrations in the surface water varied between 100 and 400 Bq $m^{-3}$. A marked difference still existed in the $^{137}Cs$ contents between the surface layer and the bottom water of the Baltic Proper [25]. In the Mediterranean, the rapid removal of the radionuclides originating from Chernobyl which entered the Mediterranean as a single pulse has been studied. A rapid removal of radionuclides from surface water and transport to 200 m in a few days by zooplankton grazing was observed [26].

# II. MODELLING OF THE DISTRIBUTION OF RADIONUCLIDES IN THE OCEANS AND THEIR TRANSPORT TO MAN

## A. OCEAN MODELS

33. Modelling in the framework of radiation protection can be split into three phases:

(a) ocean models for radionuclide transport;

(b) model for transfer of radionuclides in food chain;

(c) models for dose calculation.

34. Regarding ocean radionuclide transport, the mathematical models of water movement serve as the framework on which the effects of geochemical and biological processes are superimposed. A division of deep ocean models to near and far field can be used, although for certain processes the division between such regions is not always distinct. For the near field, high spatial resolution is required, at least close to the source. For this reason, analytical models are useful because their spatial resolution is not limited by any considerations of grid size, although caveats about correct parameterization of small scales apply. However, the inclusion of more processes of interest (e.g. scavenging) makes it more and more difficult to construct analytical solutions, so there is a natural progression from very simple analytical models to more complex and ultimately numerical models.

35. Far-field models are distinguished by the fact that the finiteness of the ocean is likely to be important, but the details of the near field are not. The progression in complexity for this case is essentially from low to high in the spatial representation with increasing realism. The choice of an appropriate model for a particular contaminant will depend on its natural lifetime (due to e.g. radioactive decay), on its reactivity with various forms of particulate material, and on the purpose of the calculation. Suitable models for deep-sea conditions are recommended in GESAMP report no. 19 [27] for use in time-dependent and steady-state problems and for the near and the far field. They are listed in Table 11.

36. International work on coastal modelling is now under way in the GESAMP framework. Most work has so far been on hydrodynamic models, less on sediment transport, geochemistry, biological transport and benthic transport models.

## B. PATHWAYS TO MAN

37. The calculation of the concentration field of a contaminant in the marine environment needs to include specific consideration of transport by biological processes only if these are significant compared with parallel physical and chemical processes. The hazards to man and other organisms may nevertheless arise because of exposure via pathways (e.g. food chains) which, whilst trivial in mass transport terms, create a specific linkage between a region of high concentration and a living organism somewhere else. The estimation of contaminant concentrations in such pathways, and therefore the hazards they create, demands careful consideration, particularly since marine food chains are often long (5 trophic levels) compared with terrestrial ones (3 trophic levels).

38. A concentration factor approach is adequate for living organisms that are more or less stationary and remain in an area where contaminant concentrations are homogeneous. This assumes that the contaminant concentration in the organism is proportional to that in the water in which it lives, so that if the water concentration doubles, so (eventually) will the concentration in the organism. It does not matter whether the contaminant actually reaches the organism indirectly (e.g. via sediment or plant material) or not, so long as there is a one-to-one correspondence in concentrations over some appropriate averaging time. The exact mechanisms need not be studied, provided field data for deriving the actual concentration factor can be obtained. Such stationary organisms may thus be treated as part of the marine environment, since the contaminant concentrations in them may be directly estimated using an appropriate concentration factor from the local water concentration. It should be noted that some organisms take up radionuclides from particulate matter. The use of concentration factors in such cases implies assumptions of equilibrium not only between the organism and water but also between particles and water.

39. This simple method does not apply to organisms that undertake major migrations across regions of high concentration gradient (e.g. in the vertical). Their contaminant concentration cannot be simply related to the water concentration in the region where they are likely to be found (and eaten), since they may have been feeding elsewhere, where the concentration is quite different, although it can be related to the average concentration experienced during the migratory movement, if the contaminant is absorbed directly from the water. More detailed consideration of the mechanisms involved is therefore required in this case.

40. Studies of concentration factors have shown that the concentration of certain contaminants increases along the food chain, although in many cases the opposite effect occurs. There have been many attempts to construct food-chain models which show this effect - for example that

of gamma and log-normal distributions [30]; equilibrium models for diverse food chains [31]; and models specifically designed to estimate the transfer of radionuclides from the deep-sea disposal of waste to man [32].

41. As pathways of radionuclides to man from the marine environment the following can be listed:

Actual pathways

Surface fish consumption
Mid-depth fish consumption
Crustacea consumption
Mollusc consumption
Seaweed consumption
Salt consumption
Desalinated sea water consumption
Alginate consumption
Fish meal consumption
Suspended airborne sediments
Marine aerosols
Handling fishing gear
Boating
Swimming
Beach sediments
Deep-sea mining

Hypothetical pathways

Deep-sea fish consumption
Plankton consumption

42. During the past few years, one particular pathway from sea to man has received special attention in the surroundings of Sellafield, U.K., namely sea-air-man. Study of this pathway was prompted by the observation by Peirson *et al.* (1974) [33] and by Pattenden *et al.* [34] of trace elements from the North Sea that revealed a mechanism of transfer from sea to air. An important finding of the studies [35] was the effect of transfer from sea to land of the actinides, but less notably of the fission products. The evidence came from the inland transects along which $^{239,240}Pu$ and $^{241}Am$ could be seen to decrease with distance from the coast, and also from the anomalously low caesium-to-plutonium ratio at the coast compared with that observed inland. Further confirmation of this effect came from the measurements by Pattenden *et al.* [36,37] of [illegible] The contribution in 1978 of the sea borne plutonium to the coastal

region of Cumbria has been roughly estimated by Eakins *et al.* [38] as 0.01 per cent of the total discharged to the sea from Sellafield, which makes it broadly comparable to the atmospheric emission.

43. Having observed and quantified the effect of sea-to-land transfer, effort has been made to elucidate its mechanism. The injection of sea water and sediment bearing the actinides into the air by waves breaking in the surf zone, and by onward transmission to land when the wind is onshore [39,40], is considered likely [41].

44. Direct measurements of the radioactivity in samples of sea spray from breaking waves have also been made. In the Irish Sea, the sea-surface microlayer and artificially generated marine aerosols have been observed to be enriched in actinides. The particulate phase is thought to be largely responsible for the phenomenon [42]. In the same study there is no evidence that $^{137}Cs$ is enriched in marine aerosols. In Figure 8, deposition of $^{239,240}Pu$ and $^{137}Cs$ relative to estimated weapon fallout are shown as a function of distance from the sea near Sellafield [43].

45. Regarding the radiological importance of the sea-to-land transfer it can be mentioned that in the Sellafield case it is much less than that of the marine food pathway [44]. The peak doses due to sea-to-land transfer to an average person living some kilometres from Sellafield have been calculated to have been 24 μSv in 1973 and would have fallen to 4 μSv in 1985. The corresponding doses to a member of the most exposed group at the same site peaked at 35 μSv in 1973 and was calculated to have decreased to 20 μSv in 1985. The most exposed group is considered to consist mainly of agricultural workers who, compared with the average person, are assumed to inhale more activity resuspended from the soil and to consume more locally produced food [45].

## C. DOSIMETRIC MODELS

### *1. Man*

46. Radiation dose is the relevant quantity in radiation protection for assessing health consequences. The procedure for calculating doses to man follows two alternatives according to the nature of the pathway.. For external exposure pathways, the dose to individuals from concentrations of radionuclides in air, water or on the ground is obtained by applying the appropriate dosimetric models and taking into account shielding effects, annual rate of occupancy and any other factors characterizing the behaviour of the individual in question.

47. For internal exposures, inhalation rates, absorption rates, or ingestion rates of food and

metabolic models.

## 2. *Living marine resources*

48. To assess the consequences to other species a similar approach to that for man is needed. Generalized models have been developed to allow the estimation of the general magnitude of the radiation exposures likely to be experienced by different types of organisms, both from radionuclides uniformly distributed internally and from contaminated water and sediment. The organisms that were considered by an IAEA advisory group are fish, large and small crustaceans, and molluscs. In each case the geometry adopted was an ellipsoid tissue of unit density. The types of species were selected to indicate the effect of body size (large and small crustaceans), concentration factor (usually highest for molluscs), external dose from sediments (benthic and bathypelagic forms), and of the dose rate due to $\alpha$, $\beta$ and $\gamma$ emitters [15].

49. Available data on concentration factors, developed almost entirely from surface or coastal water organisms, are often sufficient to provide an approximate estimate of whole-body burdens of radionuclides with the assumption of uniform distribution. At present there is insufficient information concerning the metabolism of elements, in particular deep-sea organisms, to predict the detailed internal distribution of radionuclides. Reasonably accurate estimates of the dose rates to particular organs or tissues that may be considered to be radiosensitive, such as the gonads, are therefore difficult to derive.

# III. INTERNATIONAL RECOMMENDATIONS ON CONTROLLING THE RELEASES OF RADIONUCLIDES TO THE ENVIRONMENT

## A. BASIC PRINCIPLES

50. Radiation protection is concerned with the protection of man, while still allowing justified activities from which radiation exposure results. The basic objectives of radiation protection are to prevent such detrimental effects as tissue or organ failure, and to limit the probability of cancer and hereditary effects to levels deemed to be acceptable. A system of radiation protection considered adequate to protect man as an individual has been assumed to protect other species as populations, although not necessarily as individuals.

51. The system of dose limitation by the International Commission on Radiological Protection (ICRP) [46] provides the basic principles for radiation protection. These principles are:

(a) no practice shall be adopted unless its introduction produces a positive net benefit ("justification of practice");

(b) all exposures shall be kept as low as reasonably achievable, economic and social factors being taken into account ("optimization of protection");

(c) the dose equivalent to individuals shall not exceed the limits recommended for the appropriate circumstances by the ICRP ("individual dose limitation").

52. The basic principles of radiation protection determine the requirements for limitation of releases to the environment. Justification of a practice refers to the introduction of the practice as a whole (a given application of radionuclides, electric energy production by nuclear means, etc.) and not to individual portions of the practice such as the management of radioactive effluents. Acceptance of a practice or the choice between practices will depend on many factors, only some of which are associated with radiation protection. The role of radiation protection in justification procedures is to ensure that the detriment from the practice is fully considered in assessing its net benefit.

53. The optimization requirement implies that the detrimental effects of radiation from a

be optimized, i.e. the resulting doses must be kept "as low as reasonably achievable, economic and social factors being taken into account". Methods for optimizing releases involve the choice between various control options. Of the available options, only those can be considered which result in acceptably low individual doses. The criterion for 'acceptably low' can be derived from the basic dose limit, which for the members of the public is presently 1 mSv $y^{-1}$ [47]. Since the dose limit is individual-related, irrespective of source,[1] account must be taken of the presence of other sources, the continued operation of all these sources in the future, and the eventual introduction of new sources. For this reason a source-related limit, lower than the dose limit and called the source upper bound, must be set by the authority as the boundary condition for optimizing. Under these conditions it was considered that the average dose to the population would be less than a fraction of a millisievert and would entail an average risk of individual harm in the range of $10^{-5}$ to $10^{6}$ $y^{-1}$.

## B. RELEASE LIMITATION SYSTEM [48]

54. The primary dose limits, which apply to the sum of all controlled exposures of an individual from all sources, cannot be used directly to control the dose to an individual from one particular source. Instead, a lower, source-specific, dose limit, the upper bound (UB), applies. This is the boundary condition of any optimization assessment of radiation protection, and limits the exposure of the most exposed individuals (the critical group).[2]

55. The derivation of the release upper bound proceeds as follows. The starting quantity is the individual dose limit. The dose upper bound will be less than this by the amount of the contribution of present and foreseen regional and global sources of radionuclides and radiation exposure that are subject to the dose limitation system. A further decrement from the dose limit will possibly be the extent to which a competent authority may reserve some margin for future developments of the practice (source) in question or of others. The possible longevity of the practice being considered is also relevant.

56. The competent authority might set this margin by specifying that the maximum annual dose in the critical groups with regard to a particular practice (e.g. nuclear power production) should not exceed a fraction, F, of the primary dose limit. This maximum annual dose to a critical

1 The dose limit does not apply to natural radiation, to doses to patients from medical uses of radiation, nor to accidents.

2 A critical group is selected so that it is representative of those individuals in the population expected to receive the highest dose. The dose limit for individual members of the public is applied to the mean dose in the critical group.

group, which consists of three components, will be limited by:

$$H_{local} + H_{regional} + H_{global} = FH_{limit}$$

where the suffixes refer to the components of the critical group doses and $H_{limit}$ is the primary dose limit.

57. Depending on the choice of F and the estimates of the contributions expected from regional and global exposures, the competent authority will arrive at the source-specific dose upper bound ($H_{UB}$) that would limit the local contribution from the sources which are under its control. That is,

$$H_{UB} = FH_{limit} - H_{regional} - H_{global}$$

58. Any control of regional and global contributions would have to be exercised through international agreements. At present, such agreements do not exist.

59. The upper bound for annual releases may be derived from the dose upper bound by using the overall transfer factors that relate the doses to critical groups to the activities of the radionuclides released by all modes. The competent national authorities then derive optimized release limits below release upper bounds, and finally decide upon an authorized release limit.

60. As regards the dumping of packaged radioactive wastes at sea, no dose upper bound has yet been set internationally, but guidance has been given by the IAEA to national authorities to use as an upper bound a value substantially lower than 1 mSv $y^{-1}$, the dose limit recommended by ICRP for members of the public. For the purposes of the Convention on the Prevention of Marine Pollution by Dumping of Wastes and other Matter, and in accordance with the requirements laid down, in its Annex I, paragraph 6, and Annex II, Section D, respectively, the IAEA [49] has defined high-level radioactive wastes or other radioactive matter unsuitable for dumping at sea, and made recommendations to contracting parties about the issue of permits for dumping radioactive waste or other radioactive matter.

61. The basis for the quantitative definition is a set of limits on the rates of release of radionuclides to the deep ocean. These release-rate limits are calculated from dose limits to individual members of the public by using models of radionuclide dispersion in the ocean. It has been emphasized that although a dose of 1 mSv $y^{-1}$ to individuals was used for the purpose of arriving at the quantitative definition of matter that shall not be dumped, this does not imply that 1 mSv $y^{-1}$ from actual dumping operations is acceptable. The upper-bound concept should be used by national authorities when actually granting permits for sea dumping. Presently the IAEA is further developing its recommendations on setting global and regional upper bounds. Regional upper bounds for practices and sources will need to be established through agreements between countries. At present $^{14}C$ is the only radionuclide which calls for a global upper bound because of its dispersion throughout the whole environment when discharged.

# IV. MONITORING OF RADIONUCLIDES

62. The principles of monitoring for the radiation protection of the general public have recently been discussed by ICRP [50] and IAEA [51]. The three primary objectives that have been identified are: to assess actual or potential doses to critical groups arising from the presence of radioactive materials in the environment; to demonstrate compliance with authorized limits and legal requirements; and to check the condition of the source, the adequacy of operation of the plant and the effectiveness of effluent control. Some subsidiary objectives of monitoring have also been identified. They include e.g. more basic research work to improve the knowledge of environmental behaviour and transfer of radionuclides, and to verify the predictions of any model used.

63. Regarding releases from major land-based nuclear installations it is customary to have both release monitoring (source monitoring) and environmental monitoring in the surroundings of the plants. Nuclide-specific analyses of releases and environmental materials combined with modelling are normally required to meet the primary objective.

64. Compliance with authorized limits and legal requirements - in the case of dumping at sea - is carried out prior to the wastes being taken to sea (compliance with authorized limits) and during the actual dumping operation (compliance with site requirements). Source monitoring of release from dumped packed waste is impractical because of the depth of the dump site (4000 m) and the long time scale of the releases from the dumped packages. The most important primary objective - the assessment of actual or potential doses to critical groups - can only be achieved by the application of suitable modelling techniques, as the expected resultant dose to man will not occur until some time in the future. This applies in most cases to land-based releases as well. The modelling approach does not totally preclude the simultaneous implementation of monitoring or research work which involves performing radioanalytical measurements of environmental materials. This type of monitoring information will also serve to validate some of the model predictions as time passes, but validation of models at the present time is probably best achieved by appropriate research coupled with inter-model comparisons. Since 1981, an international Research and Environmental Surveillance Programme (CRESP) is carried out under the auspices of OECD/NEA with the objective of reinforcing the scientific basis of assessments of the continued suitability of the North East Atlantic dumping site. The programme deals with geochemistry, physical oceanography, biology, modelling and radiological surveillance. Recently the terms of reference of the programme were expanded to cover also research related to coastal releases in the Paris convention area.

# V. RADIATION DOSES AND THEIR BIOLOGICAL EFFECTS TO MAN AND LIVING RESOURCES

## A. LIVING RESOURCES

65. The simple dosimetric models previously mentioned have been combined with available information concerning the distributions of the naturally occurring radionuclides in the deep sea to obtain estimates of the magnitude of the natural radiation exposure of the fauna. The results are given in Table XII and are taken from the 1985 NEA site suitability review [9].

66. The radiation background to the deep-sea fauna has been reviewed by Woodhead and Pentreath [52], from which it appears that the dose rates received are just as high and as variable as those experienced by shallow-water organisms. The major external source in the water arises from $^{40}K$ and $^{87}Rb$ that, being present at concentrations that vary in direct proportion to salinity, are essentially the same in deep water as at the surface. The dose rate to the larger fauna from $^{40}K$ is of the order of $10^{-3}$ mSv h$^{-1}$. External doses from sedimentary materials - for the fauna in direct contact with bottom sediments - vary in relation to the nature of the sedimentary material and arise primarily from the radionuclides of the $^{238}U$ and $^{232}Th$ decay series. Because the concentrations of uranium and thorium in the deep-sea sediments vary approximately in an inverse way relative to the calcium carbonate content, they also vary from one site to another. There are also differing degrees of disequilibria between the members of each decay series. Excess $^{230}Th$ arises from scavenging in the overlying water column where it is produced by $^{234}U$ decay. Further disequilibria exist between $^{230}Th$ and its daughters.

67. With regard to internal sources, again $^{40}K$ and $^{87}Rb$ are ubiquitous in living organisms, and little difference would be expected between deep-sea and shallow-water organisms. Of particular interest, however, is the $\alpha$-emitting $^{210}Po$. The importance of this nuclide in marine organisms has long been known. More recent information has demonstrated that not only are $^{210}Po$ concentrations in the viscera of shallow-water organisms particularly high, but that deeper-living species also contain comparable concentrations. This has even been demonstrated for fish; for example, concentrations of $^{210}Po$ in *Coryphaenoides armatus* tissues are not dissimilar from those in species living in shallower waters; and, of equal significance, the concentrations in the tissues of individuals of the same species can vary by almost an order of magnitude. Doses to the main organisms from dumping of radioactive waste during the past years have been recently reviewed by the NEA

the dose rates to deep-sea organisms from natural background and peak doses from the dumping of low-level radioactive wastes.

68. For many aquatic animals, mortality and induction of histopathological changes occur only after exposure to high dose rates or to large doses of radiation. In general, early-life history stages are more sensitive than adults, and organisms occupying successively higher positions in the phylogenetic tree require progressively lower doses or dose rates to elicit these effects. However, it is expected that these responses will be limited to individuals and not populations, unless the area of the marine environment impacted by radiation encompasses all or almost all of a particular species′ domain.

69. An important consideration in assessing the effects of radiation is the impact on the reproductive success of a species. Reduced success may result from the killing of germ cells, from a decrease in fecundity, from the induction of dominant and recessive-lethal mutations, from damage to developing embryos and larvae, or from any or all combinations of these factors.

70. Information on the effects of radiation on the gonads of fishes and invertebrates are available, but comparisons of the results from one study to another are often not valid because the irradiated stages in gametogenesis were not always the same. Furthermore, even when the same stage was irradiated, it was not established that the capacity for repopulation of damaged cells was the same between organisms. However, some indication of the relative sensitivities of different invertebrates and fish can be obtained by comparing the total doses or dose rates at which significant changes in fertility occurred or at which sterility ensued (Table 13). Dose rates between 0.2 and 5 mGy $h^{-1}$ appear to define a critical range in which detrimental effects on fertility are first observed in sensitive organisms. This is of particular interest because it shows that effect levels in some aquatic animals are comparable to those observed in some mammals, and indicates that germ cells from some fish and invertebrates are not more radio-resistant than those of mammals.

71. Total doses and dose rates that result in sterility are generally higher than those reducing fertility but lower than those causing mortality. For example, in *Neanthes arenaceodentata* the total dose resulting in reduction of fertility, induction of sterility, and increased mortality are about 0.5, 50, and 500 Gy, respectively. These kinds of results provide further evidence that germ cells are more sensitive than most somatic cells.

72. Examination of the data available on total doses and dose rates resulting in fertility reduction and sterility in aquatic organisms indicates that there are large differences with species (Table 13). For example, for *Daphnia pulex* [53] the dose rate causing sterility is only about twice as great as that affecting fertility, whereas for *Neanthes arenaceodentata* it is about 100 times greater.

73. The information that would be more relevant than that on effects on fertility and sterility for assessing the impact of radiation on populations is the effect of radiation on the intrinsic rate

(r) of natural increase. The parameter r includes the birth rate and the death rate. To interpret or predict the ecological consequences of increased radiation, the effects of radiation on these properties of populations must be better understood. Unfortunately, data on the effects of radiation on r are available only for *Daphnia pulex*, which is a planctonic freshwater crustacean. Marshall [53] reported that the dose rate resulting in the reduction of r to zero was in the order of 700 mGy $h^{-1}$, whereas a sharp decrease in birth rate occurred at about 500 mGy $h^{-1}$ and the birth rate was zero at 1400 mGy $h^{-1}$.

74. Results from the irradiation of early life stages indicate that some embryos are very sensitive to radiation. Mortality of developing fish embryos from acute irradiation has been observed at a dose as low as about 0.16 Gy [52]. Increased developmental abnormalities were documented by Donaldson and Bonham [54] after exposure of about 0.21 mGy $h^{-1}$. They initiated these 80-day experiments with one-cell chinook-salmon embryos. Similarly, developmental effects have been observed in mammals at dose rates as low as about 0.83 to 1.25 mGy $h^{-1}$. In the U.S. National Research Council report on exposure to low levels of radiation [55] studies are cited indicating increased pre- and post-natal mortality from exposures of about 1 mGy $h^{-1}$ in rats irradiated from conception to term.

75. Very little work has been done to examine the effects of radiation on the genetic material of aquatic invertebrates. However, in both fish and invertebrates, the types of alterations in the genome are similar to those induced in mammalian chromosome, and the data obtained to date on genotoxic effects indicate that the sensitivity of some aquatic animals with respect to cell killing and DNA damage is not less than that observed in mammals.

76. One of the more subtle effects of radiation exposure is to increase the mutation rate in the gene pool. The impact of these can vary from dominant lethality through recessive lethality to changes in the proportions of alleles at a locus present in the population. The effect of dominant lethality, which is expressed as embryo or larval mortality, is included in the radiation-induced reduction in reproductive performance in the first generation. Similarly, recessive lethals would be included in changes in the reproductive success over many generations.

77. The overall effect of non-lethal mutations is much more difficult to assess. The distribution of alleles at a locus in a species is, presumably, more or less at an equilibrium, determined by the selection pressures imposed by the environment. Changes in this distribution might affect the fitness of the population and its ability to adapt to changes in its environment. The size of the gene pool, i.e., its total inherent variability, might influence the response of the population to an increased mutation load.

78. For deep-sea organisms, little is known about the number, size and shape of their chromosomes, of the amount of DNA that is present, or of the ability of cells to repair damage to DNA. Radiosensitivity of terrestrial organisms has been related to the quantities of

and to the nature of the centromeres.

79. A factor important in radiosensitivity is the rate at which cells are cycling. It has been well estalbished that organisms in a dormant or hibernating state are more resistant to radiation than those organisms that are in a metabolically active state.

## B. MAN

80. Individual doses via ocean pathways from practices involving radionuclide releases are normally low compared with the dose limit 1 mSv $y^{-1}$ defined by ICRP for members of the public. UNSCEAR [4] has estimated a representative annual dose via all pathways to most exposed individuals from a reprocessing plant be about 0.2 mSv $y^{-1}$; from a nuclear power plant the corresponding dose is only a few microsieverts per year. According to the NEA estimate [9] of the total dose summed over all radionuclides from past dumping via the most significant pathway, the individual dose would be less than 0.03 mSv $y^{-1}$.

# VI. CONCLUSIONS

81. It can be concluded that global oceanic contamination has only occurred as a result of nuclear weapons' tests. Although the inputs from the tests have been considerable in quantity, their widespread dispersal has resulted in low-level ambient concentrations. Internationally recommended and widely used approaches to authorized disposal of radionuclides to the sea from man's activities have efficiently restricted marine contamination to limited areas and to low concentrations and consequently small radiation doses. There is, however, one exceptional case where releases from the reprocessing of nuclear fuel (Sellafield, U.K.) have been transported long distances and, after a peak release period, detected up to the Arctic Ocean. This case has been studied carefully by scientists from a number of countries, as the radionuclides released offered an opportunity to trace water and particulate movements. Doses to the most exposed group vary from year to year, the highest being about twice the average natural background.

82. Dumping of low-level radioactive waste into the deep sea is governed by the LDC and presently, since 1985, there is a moratorium on dumping radioactive waste at sea. An Expert Panel in the LDC framework stated, however, that "no scientific or technical grounds could be found to treat the option of sea dumping differently from other available options when applying internationally accepted principles of radioprotection to radioactive waste disposal". The present and future risk to individuals from past oceanic dumping of radioactive waste is considered extremely small. The doses to marine organisms are not expected to cause any detectable harm at population level.

TABLE 1: ESTIMATED QUANTITIES OF CERTAIN NATURAL RADIONUCLIDES IN THE OCEANS

| | Half-life years | Emissions | Concentration ($Bq\,l^{-1}$) | Quantity (TBq) | Reference |
|---|---|---|---|---|---|
| Primordial radionuclides | | | | | |
| $^{40}K$ | $1.3\ 10^{9}$ | beta/gamma | $1.0\ 10^{1}$ | $1.6\ 10^{10}$ | Woodhead *et al.* [56] |
| $^{87}Rb$ | $6.2\ 10^{10}$ | beta | 0.11 | $1.5\ 10^{8}$ | Pentreath [5] |
| $^{235}U$ | $7.0\ 10^{8}$ | alpha | $(1.6\text{-}1.9)\ 10^{-3}$ | $2.4\ 10^{6}$ | Woodhead *et al.* [56] |
| $^{238}U$ | $4.5\ 10^{9}$ | alpha | $(3.6\text{-}4.4)\ 10^{-2}$ | $5.6\ 10^{7}$ | Ku *et al.* [57] |
| $^{232}Th$ | $1.4\ 10^{10}$ | alpha | 0.4-20 | $(0.6\text{-}40)\ 10^{3}$ | Woodhead *et al.* [56] |
| Daughter radionuclides | | | | | |
| $^{226}Ra$ | $1.6\ 10^{3}$ | alpha | $(1.3\text{-}3.1)\ 10^{-3}$ | $4.7\ 10^{6}$ | Woodhead *et al.*, [58] |
| | | | | | Broecker *et al.*, [59] |
| | | | | | Chung *et al.*, [60] |
| $^{210}Pb$ | 22 | beta | $(0.37\text{-}5.0)\ 10^{-3}$ | $2.9\ 10^{6}$ | Cherry *et al.* [75] |
| $^{210}Po$ | 0.4 | alpha | $(0.08\text{-}3.6)\ 10^{-3}$ | $\sim 3\ 10^{6}$ | Cherry *et al.* [75] |
| Cosmogenic radionuclides | | | | | |
| $^{3}H$ | $1.2\ 10^{1}$ | beta | $(2.2\text{-}11)\ 10^{-2}$ | $8.5\ 10^{5}$ | UNSCEAR [4] |
| $^{14}C$ | $5.7\ 10^{3}$ | beta | $(5.9\text{-}6.7)\ 10^{-3}$ | $8.0\ 10^{6}$ | UNSCEAR [4] |

TABLE 2. CONCENTRATION OF NATURAL RADIONUCLIDES IN BEACH SAND, COMMON ROCKS AND DEEP OCEAN SEDIMENTS [56]

| Material | U ($\mu$g kg$^{-1}$) | $^{238}$U (Bq kg$^{-1}$) | $^{234}$U (Bq kg$^{-1}$) | $^{230}$Th (Bq kg$^{-1}$) | $^{226}$Ra (Bq kg$^{-1}$) | $^{235}$U (Bq kg$^{-1}$) | Th (mg kg$^{-1}$) | $^{232}$Th (Bq kg$^{-1}$) | K (%) | $^{40}$K (Bq kg$^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|---|
| Beach sand | 3.0 | 37 | | | | 1.9 | 6.4 | 26.0 | 0.33 | 100 |
| Granite | 5.0 | 63 | | | | 3.0 | 18.0 | 74.0 | 3.8 | 1200 |
| Shale | 3.7 | 44 | | | | 2.2 | 12.0 | 48.0 | 1.7 | 520 |
| Limestone | 1.3 | 16 | | Equilibrium assumed in the | | 0.7 | 1.1 | 4.4 | 0.2 | 63 |
| Sandstone | 0.45 | 5.6 | | $^{238}$U series | | 0.4 | 1.7 | 6.7 | 0.6 | 190 |
| Basalt | 0.50 | 6.3 | | | | 0.4 | 2.0 | 8.1 | 0.5 | 160 |
| Red clay | 0.9-2.4 | 11-30 | 11-30 | 590-2400 | 420-1720 | 0.7-1.5 | 5.5-24 | 22-96 | No data | |
| Globigerina ooze | 0.6 | 7.4 | 7.4 | 150 | 85 | 0.4 | 3.3 | 13 | No data | |

TABLE 3. ORDERS OF MAGNITUDE OF CONCENTRATIONS OF THE PRINCIPAL NATURALLY OCCURRING RADIONUCLIDES IN MARINE FOODSTUFFS [5]

| Radionuclide | Dose per unit intake (Sv $Bq^{-1}$) | Concentration (Bq $kg^{-1}$ wet) | | | |
|---|---|---|---|---|---|
| | | Fish [a] | Crustaceans [b] | Molluscs | Algae |
| $^{14}C$ | 5.6 $10^{-10}$ | .15 | 15 | 15 | 15 |
| $^{40}K$ | 5.1 $10^{-10}$ | 100 | 100 | 100 | 500 [e] |
| $^{87}Rb$ | 1.3 $10^{-9}$ | 1 | 1 | 1 | 1 |
| $^{210}Po$ | 4.3 $10^{-7}$ | 1.5 | 25 | 50 [c] | 4 [f] |
| $^{210}Pb$ | 1.4 $10^{-6}$ | 0.04 | 0.2 | 3 [c] | 0.2 [b] |
| $^{226}Ra$ | 3.0 $10^{-7}$ | 0.1 | 0.02 | 0.3 [d] | 0.02 [b] |
| $^{234}U$ | 7.0 $10^{-8}$ | 0.012 | 0.12 | 0.3 [d] | 2.5 [g] |
| $^{238}U$ | 6.3 $10^{-8}$ | 0.011 | 0.11 | 0.27 [d] | 2.2 [g] |

[a] Based largely on data in Pentreath (1977) [61] and Pentreath *et al.* (1979) [62].

[b] Values adjusted, relative to fish, on the basis of CF data in IAEA (1985) [63], except for $^{210}Po$ (Pentreath and Allington, 1988) [64].

[c] Based on data in Bangera and Patel (1984) [65], McDonald *et al.* (1986) [66], and Pentreath and Allington (1988) [64].

[d] Based on data in Bangera and Patel (1984) [65].

[e] Based on data in Thompson *et al.* (1982) [67].

[f] Based on data in Hodge *et al.* (1974) [68].

[g] Based on data in Holm and Persson (1980) [69].

TABLE 4. INPUT OF CERTAIN FALLOUT RADIONUCLIDES INTO THE OCEANS [6] AND ESTIMATED INVENTORIES BASED ON OBSERVED CONCENTRATIONS [73, 74]

| Radionuclide | Half-life years | Input (TBq) | | | Estimated Inventories [73, 74] (TBq) |
|---|---|---|---|---|---|
| | | Northern Hemisphere | | Southern Hemisphere | |
| $^{3}H$ | 1.2 $10^{1}$ | 1.0 $10^{8}$ | | 4.8 $10^{7}$ | |
| $^{14}C$ | 5.7 $10^{3}$ | | not less than 4 $10^{4}$ | | 1.67 $10^{5}$ |
| $^{90}Sr$ | 2.8 $10^{1}$ | 2.5 $10^{5}$ | | 1.2 $10^{5}$ | 4.39 $10^{5}$ |
| $^{137}Cs$ | 3.0 $10^{1}$ | 4.1 $10^{5}$ | | 2.0 $10^{5}$ | 6.37 $10^{5}$ |
| $^{239}Pu$ | 2.4 $10^{4}$ | 3.3 $10^{3}$ | | 1.6 $10^{3}$ | 1.6 $10^{4}$ a |
| $^{240}Pu$ | 6.6 $10^{3}$ | 2.2 $10^{3}$ | | 1.1 $10^{3}$ | |
| $^{241}Pu$ | 1.4 $10^{1}$ | 7.2 $10^{4}$ | | 3.4 $10^{4}$ | |

a $^{239+240}Pu$.

TABLE 5. $^{137}Cs$ INVENTORIES FOR ALL WORLD OCEANS [73]

| Ocean | Inventory (PBq) | Sediment inventory (PBq) | Areal inventory (GBq $km^{-2}$) |
|---|---|---|---|
| N. Atlantic | 119 (106-122) | 6.1* | 3.43 |
| S. Atlantic | 28.2 ± 5.4 | 3.6 | 0.43 |
| Carribean | 10.5 ± 0.6 | 0.6 | 2.41 |
| Mediterranean | 10.2 ± 1.2 | 0.5 ± 0.2 | 4.1 |
| Black sea | 1.8 | 0.2 | 3.91 |
| N. Pacific | 210 ± 32 | 4.8 | 2.66 |
| Bering sea | 4.6 | 0.2 | 2.0 |
| S. Pacific | 102 (63-142) | 4 | 1.07 |
| Indian | 86 (44-155) | 3.7 | 1.11 |
| North sea | 0.85 | 1.0 | 1.48 |
| Baltic sea | 0.49 | 0.049 | 1.28 |
| Norwegian sea | 6.75 ● 0.72 | 0.1 | 4.88 |
| Greenland sea | 4.74 ± 0.83 | 0.1 | 3.93 |
| Arctic (1979ff) | 47 ● 34 | 5.2 | 5.55 |
| Totals | 637 ± 70 PBq | 30 PBq | |

* 2.1 PBq from fallout and 4 PBq from ocean dumping.

TABLE 6. APPROXIMATE ESTIMATES OF THE MAXIMUM ANNUAL INPUT OF CERTAIN RADIONUCLIDES INTO THE SEA FROM LAND-BASED SOURCES [6]

| Radionuclides | Half-life years | Annual releases (TBq) Nuclear power plants | Reprocessing plants |
|---|---|---|---|
| $^{3}H$ | 1.2 $10^{1}$ | 4 $10^{3}$ | 2 $10^{3}$ |
| $^{14}C$ | 5.7 $10^{3}$ | 4 $10^{1}$ | 4 |
| $^{60}Co$ | 5.3 | 1 | 1 |
| $^{90}Sr$ | 2.8 $10^{1}$ | 2 | 6 $10^{2}$ |
| $^{106}Ru$ | 1.0 | 2 $10^{-1}$ | 2 $10^{3}$ |
| $^{137}Cs$ | 3.0 $10^{1}$ | 9 | 5 $10^{3}$ |
| $^{239+240}Pu$ | 2.4 $10^{4}$ and 6.6 $10^{3}$ | - | 5 $10^{1}$ |
| $^{241}Pu$ | 1.4 $10^{1}$ | - | 2 $10^{3}$ |

TABLE 7. APPROXIMATE INTRODUCTION AND INVENTORIES (TBq) OF A NUMBER OF RADIONUCLIDES ARISING FROM SELLAFIELD AND LA HAGUE. CALCULATED BY PENTREATH [7] LARGELY FROM DATA REPORTED BY STATHER *et al.* [70] AND BY CALMET AND GUEGUENIAT [71]

| Radionuclide | Sellafield (to 1986) | | La Hague (to 1985) | |
|---|---|---|---|---|
| | Total [a] | Cumulative inventory [b] | Total | Cumulative inventory |
| $^{3}H$ | 25230 | 16735 | 10190 | 8470 |
| $^{106}Ru$ | 27545 | 285 | 4905 | 800 |
| $^{137}Cs$ | 41080 | 33480 | 940 | 760 |
| $^{90}Sr$ | 6120 | 4690 | 755 | 675 |
| $^{239/240}Pu$ | 680 | 680 | - | - |
| $^{238}Pu + ^{239/240}Pu$ | - | - | 3 | 3 |
| $^{241}Am$ | 535 | 815[c] | - | - |
| $^{125}Sb$ | 170[d] | 65[d] | 1000 | 795 |
| $^{99}Tc$ | 305[d] | 305[d] | - | - |

[a] the sum of annual releases.
[b] decay corrected inventories.
[c] includes in-growth from $^{241}Pu$.
[d] since 1978.

TABLE 8. NORMALIZED EFFLUENT DISCHARGES FROM THE MODEL FUEL CONVERSION, ENRICHMENT AND FABRICATION FACILITIES (UNSCEAR) (MBq [GW(e)a] $^{-1}$) [4]

| Isotope | Atmospheric | | | Aquatic | | |
|---|---|---|---|---|---|---|
| | Conversion | Enrichment | Fabrication | Conversion | Enrichment | Fabrication |
| $^{238}U$ | 74 | 3.7 | 0.74 | 814 | 370. | 370. |
| $^{235}U$ | 2 | 0.07 | 0.22 | 20 | 7.4 | 7.4 |
| $^{234}U$ | 74 | 3.7 | 7.4 | 814 | 370. | 370. |
| $^{234}Th$ | 74 | 3.7 | 0.74 | - | - | 370. |
| $^{230}Th$ | 0.74 | - | - | 56 | - | - |
| $^{226}Ra$ | 0.07 | - | - | 126 | - | - |
| $^{222}Rn$ | 8140 | - | - | - | - | - |

TABLE 9. SUMMARY OF THE QUANTITIES OF PACKAGED RADIOACTIVE WASTES DUMPED IN THE OCEANS [6]

| | Number of sites | Years operational | Number of containers | | Total activity (TBq) |
|---|---|---|---|---|---|
| United States | | | | | |
| Pacific Ocean | 16 | 1946-1970 | 56,991 | | $5.5\ 10^{2}$ |
| Atlantic Ocean | 6 | 1949-1967 | 55,020 | | $4.0\ 10^{3}$ |
| Gulf of Mexico | 2 | 1955-1958 | 79 | | 0.4 |
| European countries | | | Mass | | |
| NE Atlantic Ocean | 10 | 1949-1982 | 142,275 tonnes | alpha-activity:<br>beta/gamma-activity:<br>$^{3}H$: | $6.8\ 10^{2}$<br>$3.8\ 10^{4}$<br>$1.5\ 10^{4a}$ |
| Japan | | | | | |
| Pacific Ocean | 1 | 1955-1969 | 1,661 | $^{60}Co$ | 15 |
| Republic of Korea | 1 | 1968-1972 | 115 | Not available | |

a Data for $^{3}H$ only apply for 1975-1982; for previous years $^{3}H$ was included with the beta /gamma activity.

TABLE 10. RANGE OF ASSUMED ANNUAL RADIONUCLIDE COMPOSITION OF WASTE DUMPED AT THE NORTH EAST ATLANTIC OCEAN SITE [9]

| Range of annual beta-gamma composition (percentage) | | Range of annual alpha composition (percentage) | |
|---|---|---|---|
| $^{3}H$ | 39 - 82 | $^{210}Po$ | 0.4 - 0.8 |
| $^{14}C$ | 0.2 - 1.5 | $^{226}Ra$ | 0.3 - 5.0 |
| $^{35}S$ | 0.05 - 1.1 | $^{234}U$ | 0.01 - 0.2 |
| $^{54}Mn$ | 0.0001 - 0.3 | $^{235}U$ | 0.6 - 1.8 |
| $^{55}Fe$ | 0.0001 - 1.0 | $^{238}U$ | 0.01 - 0.2 |
| $^{58}Co$ | 0.001 - 1.5 | $^{237}Np$ | 0.00007 - 1.2 |
| $^{60}Co$ | 1.3 - 8.7 | $^{238}Pu$ | 6.0 - 12 |
| $^{90}Sr$ | 1.2 - 2.6 | $^{239}Pu$ | 40 - 66 |
| $^{125}I$ | 0.09 - 1.2 | $^{240}Pu$ | 12 - 23 |
| $^{134}Cs$ | 0.1 - 1.3 | $^{242}Pu$ | 0.3 - 0.6 |
| $^{137}Cs$ | 1.5 - 3.7 | $^{241}Am$ | 13 - 24 |
| $^{241}Pu$ | 12 - 47 | $^{244}Cm$ | 0.1 - 0.2 |

Note: This table should not be taken as a definitive composition of the wastes. The 24 radionuclides in the list have been chosen on the basis of their relative abundance in wastes, half-lives and potential radiological significance in terms of individual dose. Some radionuclides initially present in the waste have been excluded.

TABLE 11. RECOMMENDED MODELS [27]

| | | |
|---|---|---|
| *Near field* | (a) | Simple finite ocean diffusive model (Appendix IV) |
| | (b) | Modified for finite source size and scavenging (Appendix VII) |
| | (c) | Plume solutions if the size of the near field exceeds the scale $K_H/U$ within which diffusion dominates (Appendix IV) |
| *Far field* | (a) | Well-mixed box (for contaminants with a long residence time) |
| | (b) | The one-dimensional scavenging models of Appendices VI and IX |
| | (c) | The Simple 3-D diffusive model with scavenging (Appendix VII) |
| | (d) | A medium-resolution box model |
| | (e) | Finite-difference models in 2- or 3-D |

TABLE 12. ESTIMATES OF THE RADIATION DOSE RATES (mSv $h^{-1}$) TO DEEP SEA ORGANISMS FROM THE NATURAL BACKGROUND AND PEAK DOSES FROM THE DUMPING OF LOW LEVEL RADIOACTIVE WASTE [9]

| | Fish | | Molluscs | Large crustaceans | | Small crustaceans | |
|---|---|---|---|---|---|---|---|
| | Bathypelagic | Benthic | Benthic | Bathypelagic | Benthic | Bathypelagic | Benthic |
| *Natural background* | | | | | | | |
| | $2.8\ 10^{-5}$ to $4.6\ 10^{-4}$ | $8.5\ 10^{-5}$ to $1.4\ 10^{-3}$ | $9.3\ 10^{-4}$ to $3.4\ 10^{-3}$ | $9.3\ 10^{-4}$ to $3.6\ 10^{-3}$ | $9.9\ 10^{-4}$ to $4.6\ 10^{-3}$ | $2.5\ 10^{-4}$ to $2.9\ 10^{-3}$ | $3.3\ 10^{-4}$ to $4.2\ 10^{-3}$ |
| *Past dumping:* | | | | | | | |
| site [a] | $5.2\ 10^{-5}$ | $7.3\ 10^{-5}$ | $8.0\ 10^{-3}$ | $3.6\ 10^{-4}$ | $3.8\ 10^{-4}$ | $1.2\ 10^{-3}$ | $1.3\ 10^{-3}$ |
| nested [b] | $1.2\ 10^{-5}$ | $1.2\ 10^{-5}$ | $9.4\ 10^{-4}$ | $6.9\ 10^{-5}$ | $6.9\ 10^{-5}$ | $2.3\ 10^{-4}$ | $2.3\ 10^{-4}$ |
| region [c] | $0.2\ 10^{-6}$ | $0.2\ 10^{-6}$ | $1.8\ 10^{-5}$ | $1.0\ 10^{-6}$ | $1.0\ 10^{-6}$ | $3.3\ 10^{-6}$ | $3.3\ 10^{-6}$ |

[a] 40 x 120 x 0.075 km

[b] 250 x 250 x 0.5 km

[c] 2500 x 3500 x 1 km

TABLE 13. EFFECTS OF CHRONIC AND ACUTE RADIATION ON THE REPRODUCTIVE SUCCESS OF AQUATIC ORGANISMS [a] [15]

| Organism[b] | Chronic radiation Dose rate ($mGy\ h^{-1}$) | | Chronic radiation Total dose (Gy) | | Acute radiation Dose rate ($Gy\ min^{-1}$) | Acute radiation Dose (Gy) | |
|---|---|---|---|---|---|---|---|
| | Fertility | Sterility | Fertility | Sterility | | Fertility | Sterility |
| **Invertebrates** | | | | | | | |
| 1 *Neanthes arenaceodentata* (worm, single generation) | 0.2 | 20 | 1 | 100 | 5 | 0.5 | 50 |
| 2 *Physa heterostropha* (freshwater snail, adults) | - | 100 | - | 320 | - | - | - |
| 3 *Daphnia pulex* (water flea, multiple generations) | 550 | 1400 | - | 105 | - | - | - |
| 4 *Gammarus duebeni* (amphipod) | - | - | - | - | - | 2.2 | - |
| 5 *Artemia salina* (brine shrimp, juveniles) | - | - | - | - | - | 9 | 21 |
| 6 *Diaptomus clavipes* (copepod, embryos) | - | - | - | - | 70 | 10 | - |
| 7 *Crepidula fornicata* (slipper limpet, larvae) | - | - | - | - | 2 | 20 | - |
| **Fish** | | | | | | | |
| 1 *Gambusia affinis* (mosquitofish, multiple generations) | 0.25 | - | - | - | - | - | - |
| 2 *Ameca splendens* (---, single generation) | <0.6 | -0.6 | ? | ? | - | - | - |
| 3 *Poecilia reticulata* (guppy, single generation) | 1.7 | 13 | ? | 72 | - | - | - |
| 4 *Oryzias latipes* (medaka, adult males) | 2.8 | 840 | 8 | 100 | - | 5 | - |
| 5 *Oncorhynchus tschawytscha* (chinook salmon, embryos) | 4.2 | - | 8 | - | - | 2.5 | - |
| 6 *Gambusia affinis* (mosquitofish, fry) | 13 | - | 5.6 | - | - | - | - |
| 7 *Salmo gairdnerii* (rainbow trout, 29 day old embryos) | - | - | - | - | - | 6 | - |

[a] Because all units of radiation exposure were converted to gray to facilitate comparison of data, they are approximate values.

[b] In parentheses are provided the common name, life stage irradiated and duration of exposure.

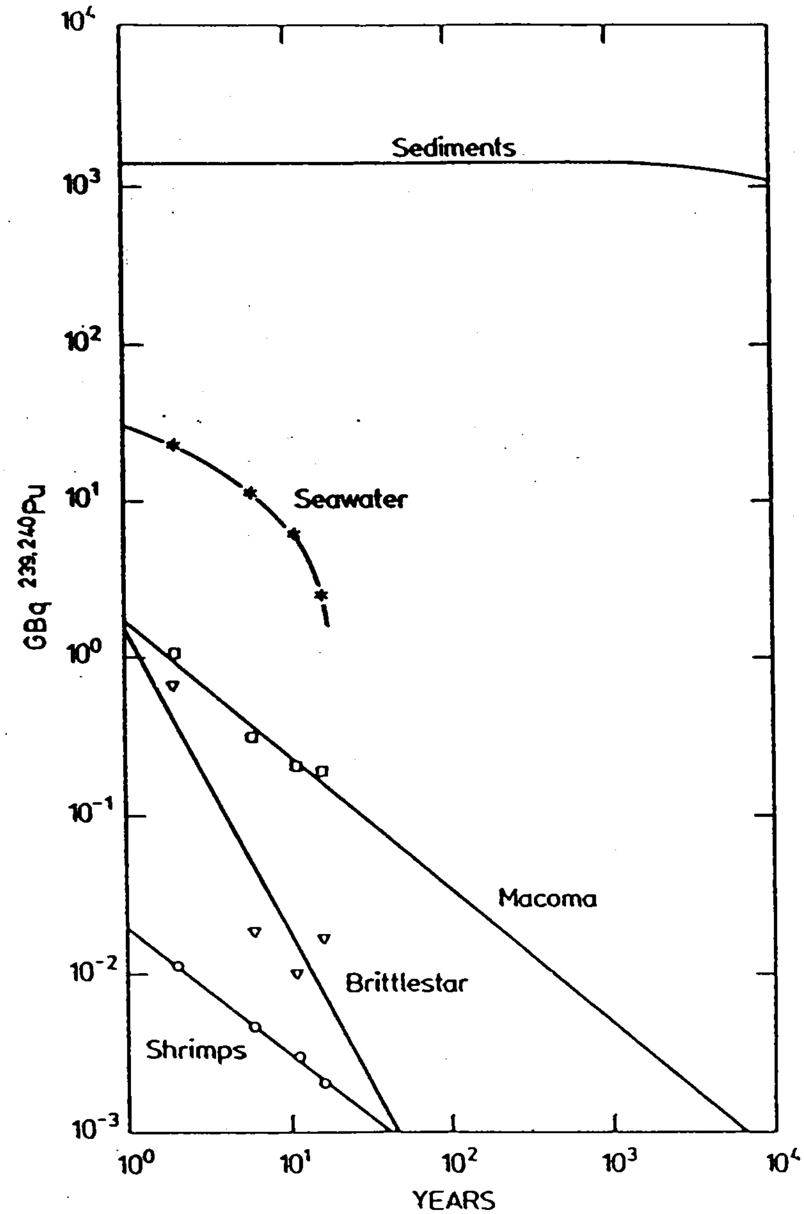


FIGURE 1. $^{239,240}Pu$ inventories in environmental samples collected at Thule 1970-1984. Within the contamined area ($3.25 \times 10^9$ $m^2$) the fresh weight biomass of shrimps was $0.11 \times 10^9$ kg, of Brittlestar: $0.062 \times 10^9$ kg, and of Macoma: flesh: $0.32 \times 10^9$ kg, and shell: $0.26 \times 10^9$ kg. The seawater mass was $3 \times 10^{14}$ kg and the mass (dry weight) of the 0-15 cm sediment layer was $3 \times 10^{11}$ kg. The abscissa is the time in years since the accident in 1968. (Redrawn from A. AARKROG *et al.* : [14])

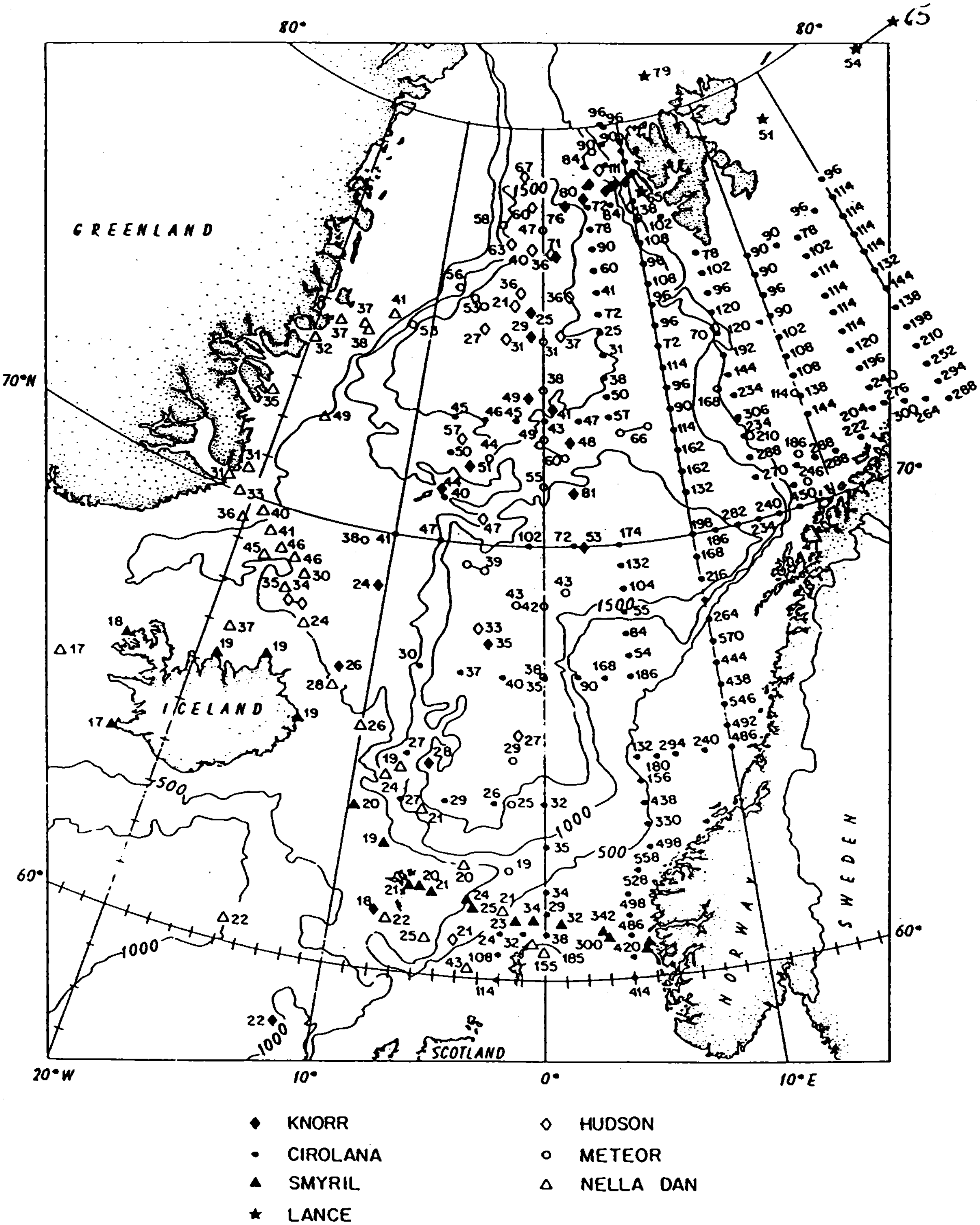


FIGURE 2. Observed $^{137}Cs$ concentrations (in dpm 100 $kg^{-1}$) in 1981-1982 in North Atlantic and Arctic Oceans [16]. (Divide values in figure by 6 to get Bq $m^{-3}$).

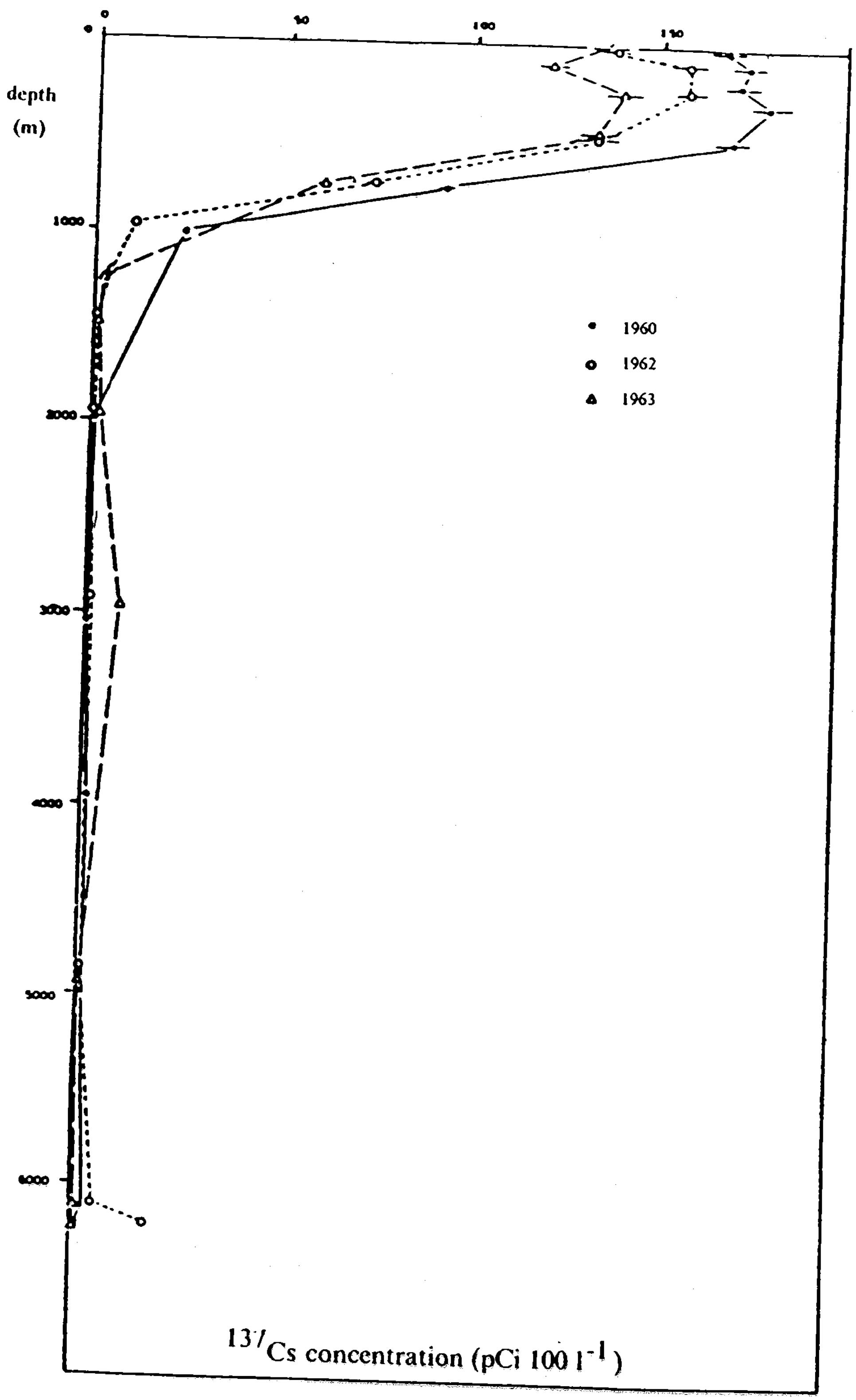


FIGURE 3. The vertical distributions of caesium 137 at the proposed site of low level radioactive wastes in North Pacific [19] (1 pCi = 37 mBq).

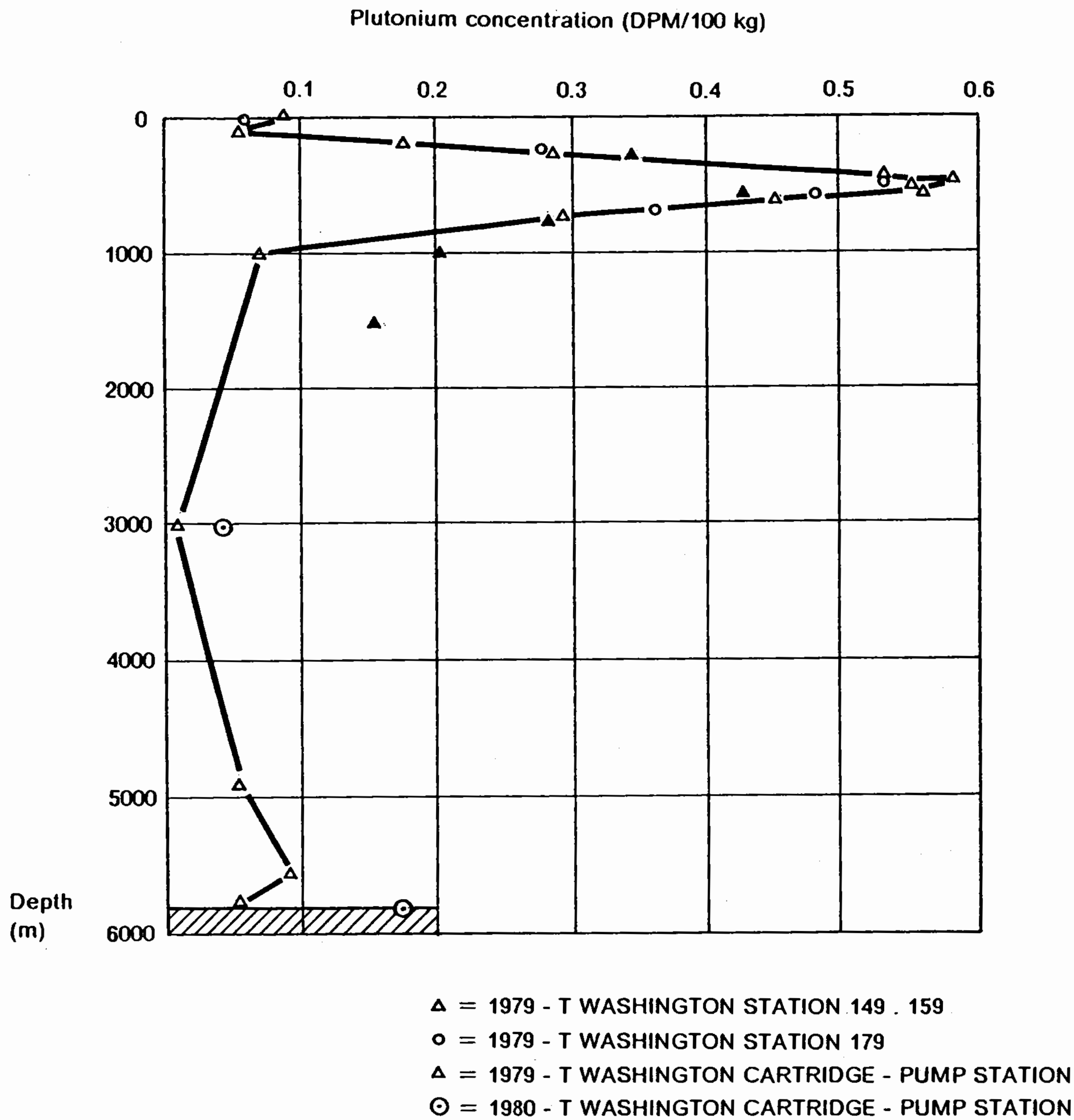


FIGURE 4. N-Pacific water profiles for plutonium. (Redrawn from Bowen, 1982) [20]. (Divide values in figure by 6 to get Bq $m^{-3}$).

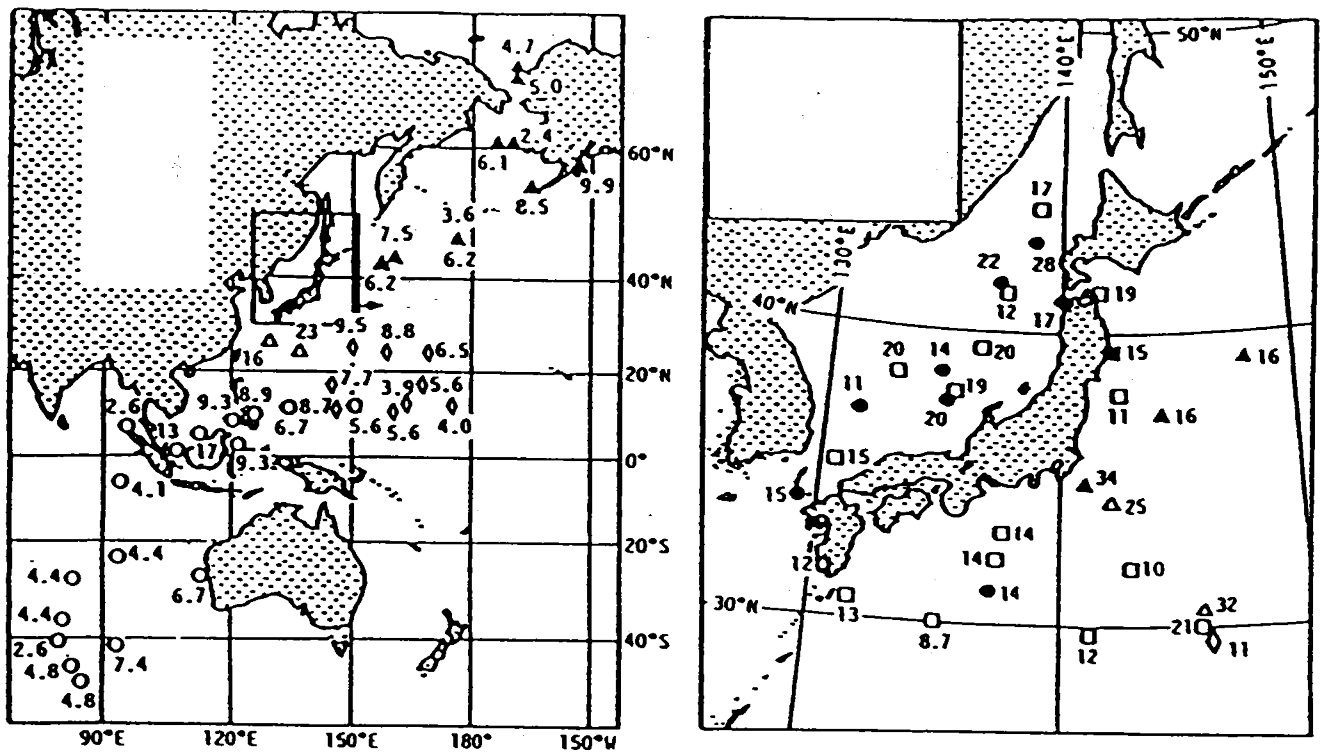


FIGURE 5. Location of seawater sampling stations and concentrations of $^{239,240}Pu$ in surface sea water in 1976-1982 (unit: $\mu Bq\ l^{-1}$) [22].

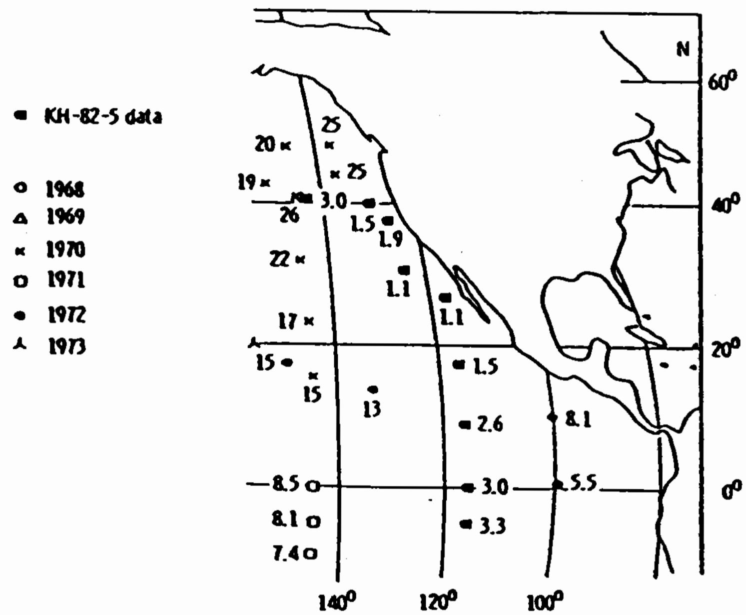


FIGURE 6. $^{239,240}Pu$ contents (in $\mu Bq\ l^{-1}$) in East Pacific surface water [22].

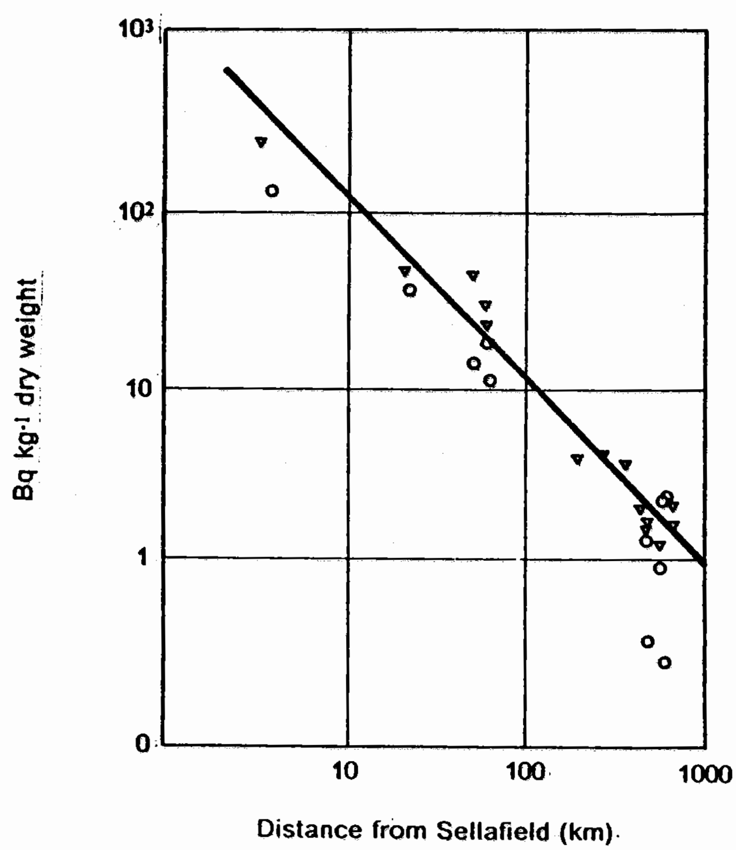


FIGURE 7. $^{239,\,240}Pu$ in *Fucus vesiculosus* (v) and *Mytilus edulis* (0) as a function of distance along the Scottish coast [17].

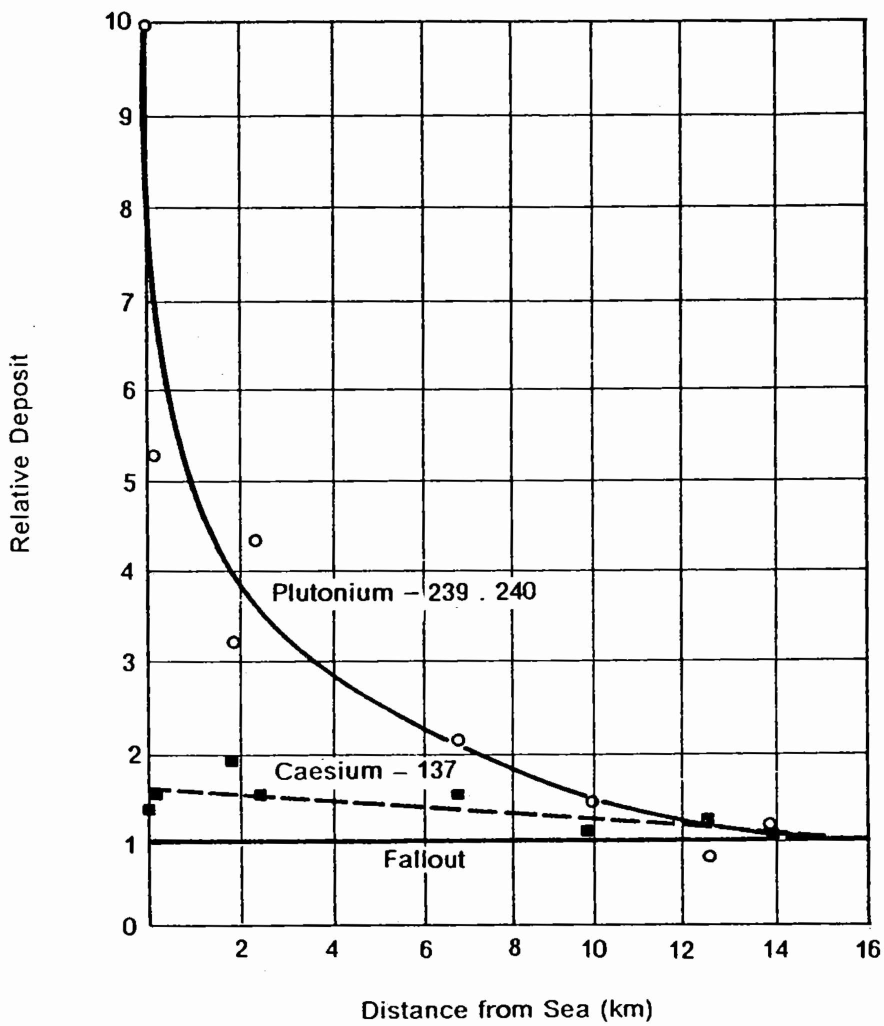


FIGURE 8. Deposition of $^{239,\ 240}Pu$ and $^{137}Cs$ relative to estimated weapon fallout at St Bees/Ennerdale in 1977 (Cambray and Eakins, 1980) [43].

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# ANNEX XV

# MARINE TRANSPORTATION OF OIL AND OTHER HAZARDOUS SUBSTANCES

C. WALDER

INTERNATIONAL MARITIME ORGANIZATION
LONDON
U.K.

## ACKNOWLEDGEMENTS

This report could not have been compiled without the considerable assistance and willing co-operation of a number of organizations, to whom grateful thanks are due and whose help is hereby acknowledged.

*Statistics on Oil Consumption, Import and Export*

All the figures included in these tables were extracted from the BP Statistical Review for the years 1977 to 1986 inclusive, together with some supplementary information provided by the BP Corporate Planning Department.

*Statistics on Oil Spills Worldwide and their Locations*

All the information on this subject was provided by the International Tanker Owners Pollution Federation, which made available all the details from their data bank and provided the maps for Figure 1 and Figure 2.

*Statistics on the World Oil Tanker Fleet*

The detailed breakdown of the position of the World Oil Tanker Fleet by size categories, in relation to compliance with the requirements of MARPOL 73/78 was provided by Clarksons Research Studies Ltd.

*Statistics of Bulk Chemical Movements Worldwide*

The information on this subject included in the report was extracted from a report prepared by Drewry Shipping Consultants Ltd., entitled "Seaborne Trade & Transportation - Report 87 / 4 - Liquid Chemicals" one of a series of reports on world shipping movements produced by that Company.

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# INTRODUCTION

1. The past ten years have seen major changes in the oil industry and in the movement of oil by sea. The reasons for these changes have been both economic and political. In the early part of the decade, the pricing policies introduced by OPEC had a marked effect on world consumption of oil as economies in the use of oil and the substitution of other fuels became, for many, a necessity for survival. More recently, the hostilities in the Gulf area have also had a restraining effect on movements of oil by sea.

2. World consumption of oil rose from 3.0 billion tonnes in 1977 to 3.1 billion tonnes in 1979, but thereafter, the rapid increases in the price of oil as a result of OPEC decisions caused consumption to fall very considerably to a low point of 2.8 billion tonnes in 1985. With the fall in the price of oil in 1986, consumption resumed its upward with 1986 consumption reaching 2.9 billion tonnes.

3. These changes in oil consumption reflected the rapid upward movement in the price of oil. For example, the price of a barrel of Saudi Arabian Light - generally regarded as the "marker" crude in the Gulf area - was $12.09 on 1 January 1977, but by 1 July of that year, this had increased to $18.00 and by 1 January 1980, the price had reached $26.00. Prices continued to escalate until 1983, by which time a barrel of Saudi Arabian Light cost $34.00. Thereafter, however, the price declined slightly to $29.00 until 1986, when the price structure collapsed following considerable over-production, and some OPEC countries were obliged to introduce netback pricing arrangements for their exports.

4. In 1987 world oil prices appear to have stablised, due mainly to the acceptance by most of the OPEC countries of a voluntary quota system for production which has resulted in a price of around $18.00 per barrel for Saudi Arabian Light.

5. The reduction in the price of oil to its present level from the previously existing extremely high prices obtaining in the early 1980s has encouraged renewed growth in the consumption of oil as the figures for 1986 demonstrate. This is in direct contrast to the situation in the late 70s/early 80s when the increase in the price of oil encouraged economy in consumption, and the exploration for additional supplies of oil and their rapid exploitation. Offshore drilling even in deep water and drilling in hostile environments became economic at the high prices then ruling and this resulted in marked changes in the pattern of movements of oil by sea.

6. Table 1 shows that whilst world oil consumption showed a reduction of 3.5 per cent, the tonnage of oil moved by sea fell by 31 per cent by the end of 1985 and even after the recovery of consumption in 1986, movements by sea were still 25 per cent below those of 1977.

7. The decline in the price of oil in 1986 however, whilst encouraging increases in consumption, has had the effect of slowing down the development of some of the smaller discoveries of oil, the economics of which, at these prices, verge on the marginal. The increased consumption has, initially at least, been supplied mainly from the Gulfs area, which had borne the brunt of the reductions of the previous years.

# I. WORLDWIDE MOVEMENT OF OIL BY SEA

8. The transport of oil by sea, as indicated earlier, has undergone some significant changes. The percentage of oil consumption moved by sea has dropped dramatically. Whereas in 1977 nearly 58 per cent of world oil consumption was moved by sea, by 1986 this figure had fallen to 44 per cent. Perhaps of greater significance is the fact that 430 million tonnes less oil was moved by sea in 1986 compared with 1977 - a reduction of 25 per cent.

9. The other significant change in oil movements has been the steady and continuous increase in the transport of oil as finished products – most of it non-persistent – rather than as crude oil, as shown in Table 2. In 1977 the transport of finished products represented 15 per cent of total exports but by 1986 this figure had increased to 25 per cent whilst crude oil exports in 1986 were 33 per cent less than in 1977.

## A. MAJOR EXPORTING AREAS

10. Table 3 shows the volume of oil shipped from the major exporting areas of the world in the ten years 1977 to 1986. This shows quite clearly that virtually all the reduction in exports has been borne by the Middle East, the 1986 exports from that area being only 53 per cent of those achieved in 1977 - a reduction of 485 million tonnes - whilst other areas were relatively less affected, though West Africa and North Africa both experienced some reduction.

## B. MAJOR IMPORTING AREAS

11. The major importers of oil, not surprisingly, are the more economically developed areas of the world.

12. *Western Europe* is still the largest importing area, though the advent of North Sea production has reduced its dependence on imports by some 33 per cent. Table 4 shows that total imports have declined from 650 million tonnes to 435 tonnes in the 10-year period, though, within these figures, finished product imports increased by 34 million tonnes, or 66 per cent in the same period.

13. Table 5 shows imports into Western Europe by source for the same period. This again demonstrates that virtually all the reduction in imports has been from the Middle East, from where 1986 imports were only 40 per cent of those a decade earlier, with an even more significant reduction in the tonne/miles of tanker transportation. Imports from Latin America and from North Africa and West Africa were relatively unaffected and, in fact, showed some increase over the 10-year period, oil from these sources being an important factor in product quality control.

14. *United States* is the second largest oil importing area, though it, too, has substantially reduced its dependence on imports - mainly due, in this case, to reduction in consumption (1986 746.8 million tonnes versus 865.9 million tonnes in 1977 - a reduction of 14 per cent).

15. The source of imports to the United States is shown on Table 6. The reduction in imports over the decade has been 32 per cent from 432 million tonnes to 297 million tonnes. Again, imports from the Middle East have fallen by no less than 83 per cent and, despite some recovery in 1986, were only 35 per cent of their 1977 level.

16. Imports from North Africa were also substantially reduced, the 1986 level being only 20 per cent of that in 1977, reflecting the deteriorating political situation between the United States and Libya.

17. *Japan* is the third largest major importing area. In this country also, consumption of oil has fallen from 260.4 million tonnes in 1977 to 204.4 million tonnes in 1986 - a reduction of 22 per cent. Since Japan has no significant domestic oil production, imports have fallen in line with reduced consumption.

18. Table 7 sets out the sources of imports into Japan in the last ten years, which shows that whilst Japan has endeavoured to widen its sources of supply in order to reduce its dependence on the Middle East, the majority of its oil imports continue to originate from that source. In 1977, 75 per cent of Japan's oil came from the Middle East, and by 1986 this figure had only fallen to 65 per cent.

19. *Latin America* is the fourth largest importing region, and is the only other area where imports have exceeded 100 million tonnes during the 10-year period. Latin America is the one area of the world where oil consumption has increased through the period (183.7 million tonnes in 1977 to 215.4 million tonnes in 1986 - an increase of 17 per cent). The increase in consumption reflects the economic growth in the region, but due to increased local production, imports have fallen from 119.3 million tonnes in 1977 to 80.2 million tonnes in 1986 - a decrease of 33 per cent. Table 8 shows imports in the region by source, and demonstrates yet again that the reduction has fallen on the Middle East.

20. All the changes in oil production, consumption and the resulting changes in import patterns, which have been dealt with in detail in the previous paragraphs, have produced a balance of supply and demand for the major regions of the world which is shown in Table 9, for the year

a fair indication of the present position of world oil consumption and the amount which is moved by sea.

## C. ACCIDENTAL SPILLAGES OF OIL INTO THE SEA

21. The number of oil spillages into the sea, of all sizes, has shown a steady and continuous decline. The most comprehensive statistics of such spillages are those accumulated by the International Tanker Owners' Pollution Federation, whose data go back to the year 1974. Table 10 gives a summary of their information which shows that accidental oil spillages have decreased substantially over the period. Most encouraging is the low level of major spills and the decline in the frequency of major catastrophic accidents.

22. The reduction in spillages into the sea, and particularly the improvement in preventing spillages in areas of high population density - where the effects can have far-reaching consequences and cause serious temporary dislocation of industry and social welfare - is shown on the two maps (Figure 1 and Figure 2) which show the location of accidental spills over 5,000 barrels in the two periods 1974 to 1979 inclusive, and 1980 to 1985 inclusive.

23. The reduction in serious spillages in the North East of the United States and in North West Europe is especially noticeable as is the reduction in incidents in the Mediterranean and the Caribbean/U.S. gulf area. Further evidence of the reduction in incidents affecting tankers is given in the report of the IMO Steering Group on Casualty Statistics, which gives an analysis of the Lloyds Register of Shipping's casualty reports. This shows a continuation in the overall downward trend in the serious casualty rate for tankers of over 6,000 grt, although 1986 was marginally higher than the record low attained in 1985.

The incidence of serious casualties in tankers over 6,000 grt for the years 1977 to 1986 is:

| *Year* | *Serious Casualty Rate per Hundred Ships* |
|---|---|
| 1977 | 2.31 |
| 1978 | 2.38 |
| 1979 | 3.11 |
| 1980 | 2.08 |
| 1981 | 2.74 |
| 1982 | 1.76 |
| 1983 | 1.88 |
| 1984 | 1.87 |
| 1985 | 1.69 |
| 1986 | 1.83 |

# D. INTERNATIONAL CONVENTION FOR THE PREVENTION OF POLLUTION FROM SHIPS 1973 AND THE PROTOCOL THERETO OF 1978

24. This Convention, generally referred to as MARPOL 73/78, entered into force in October 1983 with further amendments which entered into force in January 1986. Annex I of that Convention, which regulates the carriage of oil, has undoubtedly had a major impact on the reduction of operational pollution of the seas, as well as contributing to the reduction of accidental spillages. The requirement contained in Regulation 13 of that Annex, that (a) all new construction ordered after 31 December, 1975 or completed after 31 December, 1979, for crude oil carriers of 20,000 dwt or above, must be fitted with segregated ballast system (SBT); (b) all existing crude oil carriers over 40,000 dwt must be equipped with a crude oil washing (COW) system; (c) all tankers be fitted with effective oil/water interface detectors and overboard discharge monitors and (d) all other vessels of 10,000 grt be fitted with oily-water separating and oil discharge monitoring, as well as the establishment of a number of Special Areas in which there is total prohibition of all discharge other than clean water, have together resulted in a major diminution of operational pollution, not only from tankers, but from all other types of vessel.

25. These measures will have an increasing and progressive effect as the existing fleet of tankers is scrapped and replaced with new tonnage, since all new crude oil tankers of 20,000 dwt and above are required to be fitted with both segregated ballast and crude oil washing systems.

26. The number of vessels, by size category, which were equipped with these systems at the end of 1986, is shown on Table 11, from which it can be seen that 32 per cent of all ships in the world tanker fleet are equipped with segregated ballast and that 39 per cent of all tankers are equipped for crude oil washing. Of particular significance is the fact that the larger vessels - over 160,000 dwt - are all equipped with one or other of these systems, and that the most recent additions to the fleet are equipped with both.

27. The figures in Table 11 were extracted from statistics provided by Clarksons Research Studies, and represent minima in each category. This is because they represent positive answers from owners to a questionnaire of what equipment is installed on their vessels and not all owners reply to the questionnaire.

28. The column headed classification total represents the total number of ships in that category in the world fleet. In some cases the total of SBT ships plus COW ships exceed the total for the classification. This is because new construction is usually fitted with both systems.

29. In addition to the foregoing improvements in preventing operational pollution and re-

safety requirements and in the number of traffic separation schemes in areas of high traffic density. The most important means by which these improvements can be achieved is by ensuring that the Convention requirements are enforced. In recent years there has been a major strengthening of enforcement procedures and vessel inspections by the responsible authorities in many of the Contracting States, particularly those that are major importers of oil. Agreement has also been reached between certain States to exchange information on vessels found to be deficient in meeting the requirements of the Convention, in an endeavour to obviate, as far as possible, the substandard operator.

30. The significant reduction in the carriage of oil by sea in the past decade, coupled with the entry into force of MARPOL 73/78 with its more stringent standards, and the greater emphasis on enforcement and punishment of offenders, has undoubtedly resulted in a significant diminution of operational pollution.

31. The diminution in the number of accidental spills and the low level of catastrophic accidents have also resulted in a marked reduction of oil entering the sea from these sources. The expansion of Traffic Separation Schemes in areas of concentration of shipping and the parallel improvements in safety equipment required under the SOLAS (Safety of Life at Sea) Conventions now in force, have undoubtedly made a major contribution to the reduction in accidents and particularly those which cause pollution.

32. The continuous increase in tankers fitted with segregated ballast and crude oil washing systems as older vessels are scrapped and replaced by new tonnage, together with the greater emphasis on inspection and enforcement, and improvements in the safety of navigation should ensure that discharges of oil and oily mixtures to the sea should continue to decline.

## II. SHIPMENT BY SEA OF OTHER HAZARDOUS PRODUCTS

33. Unlike the movements of Crude Oil and Finished Products, for which there is excellent and reliable statistical information, similar information on the movement of hazardous polluting substances by sea is virtually non-existent. This is not altogether surprising, since these products are varied in character, are produced by many different industries and number several thousand different substances.

34. Hazardous polluting substances - the vast majority are classified as chemicals - are transported by sea in three different modes:

35. By *dry-bulk cargo carriers* - There is a limited movement of certain chemical products, such as sulphur, salt and some fertilizers which are carried in bulk carriers.

hazardous chemicals such as pesticides, weed-killers, tetraethyl lead etc., are transported in packages or in portable tanks in containers, the packages being in accordance with the requirements of the IMO International Dangerous Goods Code. Whilst this provides a considerable measure of safety in the event of an accident, there have been a number of cases where damage to containers following a casualty, or loss of packages overboard, have caused pollution and have required a major salvage and clean-up operation to be mounted. One of the problems encountered in these accidents has been the inadequacy of marking and a lack of description of the product carried, which has caused delay in assessing priorities and methods to be adopted in the salvage and clean-up operations.

37. By *liquid bulk tanker* - The movement by sea of petrochemicals, caustic soda solution, sulphuric acid and phosphoric acid is frequently carried in bulk by special tankers, and whilst there are some statistics available for this mode of transport, even these are forced to include some assumptions and to be hedged with the caveat: -

> "-----these trades are notoriously difficult to analyse and the Chemical Industry, in particular, is renowned for its secrecy."

## A. BULK MOVEMENT OF CHEMICALS BY SEA

38. Whilst the bulk movement of chemicals in tankers is increasing at a rapid rate - it has more than doubled in the last ten years, at a 10 per cent compound annual growth rate - its volume is small in relation to movements of oil. The latest estimates available, are those for the year 1985 and show a total movement of approximately 25 million tonnes. This figure includes the movement of several hundred different commodities, but detailed examination of the information reveals that some 22 products account for more than 80 per cent of the total movement. Table 12 gives a breakdown of the 1985 exports of these 22 products.

39. The pattern of trade in these 22 major products (rounded to the nearest 5,000 tonnes) for the year 1985 is shown at Table 13. This reveals that the United States is the leading exporter of chemicals in liquid form followed by Western Europe. Major importers are Asia/Oceania, Western Europe, North America and Latin America in that order.

## B. ACCIDENTAL SPILLAGES

40. The dearth of information on the movement of chemical products applies also to details of accidents involving vessels carrying hazardous substances. Most of the well-documented accidents are those relating to packaged goods or containerised cargo, there being several incidents involving the loss of drums of tetraethyl lead overboard, notably one in the North Atlantic and another off Italy. There was also one major incident involving drums of pesticides off Southern Spain and one which involved the loss of nearly one hundred drums of various chemicals in the English Channel. In this latter case again, inadequate marking of the contents presented major problems when the drums were recovered or washed ashore.

41. The effects of a major accident to a container ship carrying a cargo of many different types of chemicals, is demonstrated by the wreck of the ARIADNE at Mogadiscio, a brief summary of which is as follows:

42. The ARIADNE was an Oil/Bulk Carrier/Container vessel, built in 1979, of 16,186 grt, flying the Panamanian flag and manned by a Filipino crew. At the time of the accident she carried 73 containers and 30 drums as cargo. The containers carried 105 different chemical compounds, mostly in drums, some of which were highly toxic and some of which, if immersed in water, give off toxic fumes. The vessel was on a voyage from Northern European ports to various ports in East Africa.

43. On 24 August 1985, whilst in the process of leaving Mogadiscio harbour, Somalia, the vessel went hard aground on a rocky shore, close to a centre of population which the residents used for their washing and recreation. At the time of the incident some of the containers fell overboard and were washed ashore in the strong onshore swell. Efforts were made to refloat the vessel by a local salvage firm, but these proved to be unsuccessful, even after lightering some of the containers.

44. The vessel started to break up just forward of the after accommodation, which caused some bunker oil to escape. Heavy oil pollution of the shore was experienced but, in the light of other problems, a major shore clean-up was not mounted.

45. Some drums of toluene, which were deck cargo, were crushed and leaked into the sea and a container of nitric acid spilled which caused a fire. Salvage of containers was impeded as the acid ate through the deck. The fire was extinguished by the shore fire-engine, which was transported to the scene on a lighter.

the hull. It was possible to pump out the forward bunker tanks to road vehicles mounted on a lighter, thereby avoiding a possible further pollution.

47. In order to reach certain of the more toxic chemicals, it was necessary to remove the forepart of the wreck, which was eventually towed out to sea and sunk.

48. By April 1986 all of the dangerous chemicals which could be salvaged had been removed from the wreck. The local authorities required that the sea-bed in the vicinity of the wreck, which was contaminated, should be removed and a hulk was brought in from Mombasa, onto which the contaminated material was loaded. The hulk was towed out into deep water and sunk. The beach, which had been heavily contaminated by oil, is now completely clean as a result of the natural forces of the sea.

## C. INTERNATIONAL CONVENTION FOR THE PREVENTION OF POLLUTION FROM SHIPS 1973 (MARPOL 73/78 - ANNEX II)

49. Annex II of this Convention, generally referred to as MARPOL 73/78, which regulates the carriage by sea of hazardous substances in bulk, entered into force in April, 1987.

50. In addition to the requirements which affect all ships under Annex I, this Annex brings into effect new and very stringent requirements for the disposal of slops following tank washing after the carriage of cargoes which are regulated by Annex II. The Annex also requires strict compliance with the construction criteria laid down for vessels involved in these trades. These new requirements, together with those introduced by Annex I, should reduce still further the possibility of contamination of the marine environment from accidental spillage or operational discharge, despite the increase in volume and variety of cargoes of hazardous substances being transported by sea in bulk.

TABLE 1. WORLDWIDE OIL CONSUMPTION, PRODUCTION AND EXPORTS TEN YEARS 1977 TO 1986 *(Million tonnes)*

| Year | Consumption | Production | Exports | Exports as Percentage of Consumption |
|---|---|---|---|---|
| 1977 | 2,985.8 | 3,066.9 | 1,724.0 | 57.73 |
| 1978 | 3,082.7 | 3,092.9 | 1,681.1 | 54.53 |
| 1979 | 3,123.9 | 3,225.4 | 1,751.6 | 56.07 |
| 1980 | 2,988.4 | 3,081.9 | 1,588.2 | 52.97 |
| 1981 | 2,899.5 | 2,903.7 | 1,423.1 | 49.08 |
| 1982 | 2,823.7 | 2,787.3 | 1,269.8 | 44.97 |
| 1983 | 2,803.7 | 2,766.5 | 1,206.1 | 43.02 |
| 1984 | 2,831.4 | 2,839.0 | 1,229.9 | 43.44 |
| 1985 | 2,809.6 | 2,806.1 | 1,189.0 | 42.32 |
| 1986 | 2,881.0 | 2,932.8 | 1,292.9 | 44.88 |

TABLE 2. WORLDWIDE EXPORTS OF CRUDE OIL AND FINISHED PRODUCTS
TEN YEARS 1977 TO 1986 *(Million tonnes)*

| Year | Crude Oil | Finished Products | Total | Percentage Products of Total |
|---|---|---|---|---|
| 1977 | 1,458.6 | 265.4 | 1,724.0 | 15.39 |
| 1978 | 1,429.8 | 251.3 | 1,681.1 | 14.95 |
| 1979 | 1,494.7 | 256.9 | 1,751.6 | 14.67 |
| 1980 | 1,317.5 | 270.7 | 1,588.2 | 17.04 |
| 1981 | 1,164.4 | 258.7 | 1,423.1 | 18.18 |
| 1982 | 998.4 | 271.4 | 1,269.8 | 21.37 |
| 1983 | 935.9 | 270.2 | 1,206.1 | 22.40 |
| 1984 | 933.8 | 296.1 | 1,229.9 | 24.08 |
| 1985 | 886.7 | 302.3 | 1,189.0 | 25.42 |
| 1986 | 981.1 | 311.8 | 1,292.9 | 24.12 |

TABLE 3. MAJOR EXPORTS OF CRUDE OIL AND FINISHED PRODUCTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

| Year | Middle East | Latin America | North Africa | West Africa | South East Asia | USSR, Eastern Europe and China |
|---|---|---|---|---|---|---|
| 1977 | 1,024.4 | 162.3 | 157.1 | 118.0 | 91.9 | 101.0 |
| 1978 | 974.9 | 177.8 | 161.0 | 105.5 | 85.9 | 102.7 |
| 1979 | 1,009.6 | 188.6 | 163.8 | 131.4 | 92.0 | 87.8 |
| 1980 | 869.3 | 196.7 | 140.2 | 120.5 | 83.6 | 99.6 |
| 1981 | 725.4 | 217.7 | 107.9 | 85.9 | 82.5 | 106.8 |
| 1982 | 578.7 | 204.3 | 106.3 | 74.6 | 73.3 | 111.6 |
| 1983 | 512.9 | 200.3 | 108.2 | 70.6 | 70.3 | 130.5 |
| 1984 | 489.4 | 202.4 | 113.7 | 83.3 | 79.2 | 141.1 |
| 1985 | 446.6 | 182.8 | 115.8 | 88.3 | 123.4 | 100.6 |
| 1986 | 539.3 | 175.7 | 122.0 | 98.4 | 76.2 | 148.2 |

TABLE 4. MAJOR IMPORTS OF CRUDE OIL AND FINISHED PRODUCTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

| | UNITED STATES | | | LATIN AMERICA | | | W. EUROPE | | | JAPAN | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Crude | Products | Total | Crude | Products | Total | Crude | Products | Total | Crude | Products | Total |
| 1977 | 323.7 | 108.5 | 432.2 | 110.4 | 8.9 | 119.3 | 603.0 | 54.2 | 657.2 | 241.7 | 28.9 | 270.6 |
| 1978 | 306.9 | 102.3 | 409.2 | 121.7 | 11.1 | 132.8 | 588.2 | 60.0 | 648.2 | 235.2 | 27.4 | 262.6 |
| 1979 | 320.2 | 99.6 | 419.8 | 125.3 | 9.5 | 134.8 | 598.7 | 48.4 | 547.1 | 244.1 | 31.5 | 275.6 |
| 1980 | 259.3 | 77.7 | 337.0 | 131.2 | 15.5 | 146.7 | 514.9 | 74.0 | 588.9 | 220.2 | 25.6 | 245.8 |
| 1981 | 218.3 | 75.6 | 293.9 | 110.4 | 9.7 | 120.1 | 441.8 | 71.6 | 513.4 | 196.5 | 23.5 | 220.0 |
| 1982 | 172.5 | 76.1 | 248.6 | 90.1 | 14.6 | 104.7 | 383.0 | 83.1 | 466.1 | 180.8 | 25.4 | 206.2 |
| 1983 | 164.3 | 81.4 | 245.7 | 82.3 | 15.1 | 97.4 | 342.2 | 87.8 | 430.0 | 177.7 | 28.0 | 205.7 |
| 1984 | 169.7 | 94.7 | 264.4 | 78.5 | 11.6 | 90.1 | 337.7 | 90.2 | 427.9 | 183.4 | 31.2 | 214.6 |
| 1985 | 159.9 | 88.4 | 248.3 | 71.8 | 19.2 | 91.0 | 311.6 | 100.9 | 412.5 | 169.4 | 32.8 | 202.2 |
| 1986 | 208.8 | 88.3 | 297.1 | 61.7 | 18.5 | 80.2 | 346.9 | 88.5 | 435.4 | 165.8 | 38.6 | 204.4 |

TABLE 5. SOURCE OF IMPORTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

## *WESTERN EUROPE*

| Exported From | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 |
|---|---|---|---|---|---|---|---|---|---|---|
| U.S.A. | 4.1 | 4.5 | 5.6 | 8.2 | 7.2 | 12.6 | 10.6 | 11.4 | 9.2 | 9.9 |
| Latin America | 15.1 | 18.3 | 14.5 | 31.1 | 44.3 | 37.2 | 40.5 | 43.3 | 41.4 | 30.9 |
| Middle East | 436.0 | 429.3 | 430.6 | 362.3 | 289.7 | 229.8 | 169.2 | 151.6 | 144.2 | 174.5 |
| North Africa | 82.4 | 84.5 | 89.1 | 71.2 | 67.0 | 74.5 | 83.0 | 82.1 | 82.6 | 88.7 |
| West Africa | 42.2 | 38.9 | 50.0 | 49.0 | 32.6 | 34.9 | 37.5 | 48.2 | 48.3 | 52.7 |
| South East Asia | 1.0 | 1.0 | 1.2 | 0.7 | 2.4 | 1.2 | 1.2 | 1.6 | 2.1 | 0.2 |
| South Asia | - | 0.2 | - | - | - | 2.3 | 0.2 | 0.9 | 0.9 | 0.7 |
| U.S.S.R., Eastern Europe and China | 76.4 | 71.3 | 56.1 | 64.7 | 69.5 | 72.9 | 87.8 | 88.7 | 83,3 | 77.5 |
| Other | - | 0.2 | - | 0.2 | 0.7 | 0.7 | - | 0.1 | 0.5 | 0.3 |
| TOTAL IMPORTS: | 657.2 | 648.1 | 647.1 | 588.9 | 513.4 | 466.1 | 430.0 | 427.9 | 412.5 | 435.5 |

TABLE 6. SOURCES OF IMPORTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

## *UNITED STATES OF AMERICA*

| xported From | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 |
|---|---|---|---|---|---|---|---|---|---|---|
| anada | 27.8 | 22.9 | 22.6 | 18.6 | 21.2 | 23.1 | 26.5 | 30.5 | 37.7 | 32.8 |
| atin America | 113.8 | 114.3 | 125.8 | 105.4 | 96.2 | 100.7 | 106.2 | 110.7 | 102.6 | 107.5 |
| 'estern Europe | 12.4 | 18.5 | 17.5 | 18.1 | 26.1 | 28.7 | 27.0 | 30.5 | 24.1 | 30.3 |
| liddle East | 125.1 | 113.6 | 104.6 | 77.2 | 61.5 | 38.2 | 28.4 | 30.3 | 21.0 | 45.0 |
| orth Africa | 62.9 | 62.6 | 64.8 | 50.8 | 32.5 | 12.1 | 13.8 | 15.5 | 10.2 | 13.4 |
| 'est Africa | 60.7 | 48.6 | 58.9 | 46.8 | 35.8 | 30.2 | 23.5 | 22.1 | 25.2 | 31.8 |
| uth-East Asia | 28.3 | 27.4 | 25.5 | 20.1 | 19.6 | 13.4 | 17.2 | 18.7 | 20.3 | 26.0 |
| ustralasia | 0.5 | 0.5 | 0.4 | - | 0.1 | 0.2 | 0.2 | 1.8 | 2.0 | 1.6 |
| .S.S.R., Eastern urope and China | 0.7 | 0.5 | - | - | 0.9 | 2.0 | 2.9 | 4.3 | 5.2 | 3.3 |
| pan | - | 0.3 | - | - | - | - | - | - | - | - |
| uth Asia | - | - | - | - | - | - | - | - | - | 1.0 |
| **OTAL IMPORTS:** | 432.2 | 409.2 | 419.8 | 337.0 | 293.9 | 248.6 | 245.7 | 264.4 | 248.3 | 297.1 |

TABLE 7. SOURCES OF IMPORTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

*JAPAN*

| Exported From | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 |
|---|---|---|---|---|---|---|---|---|---|---|
| U.S.A. | 1.7 | 1.6 | 1.8 | 0.5 | 2.1 | 4.0 | 6.0 | 5.5 | 6.0 | 1.6 |
| Latin America | 0.5 | 0.8 | 0.5 | 4.0 | 7.6 | 7.3 | 8.9 | 9.2 | 7.5 | 10.4 |
| Western Europe | - | - | - | - | 0.3 | - | - | - | 0.2 | 0.8 |
| Middle East | 203.6 | 198.1 | 205.7 | 176.1 | 148.2 | 137.0 | 135.5 | 140.8 | 129.6 | 134.4 |
| North Africa | 1.2 | 0.5 | 0.7 | 2.7 | 3.5 | 2.2 | 2.1 | 1.2 | 2.6 | 1.6 |
| West Africa | - | - | - | 0.7 | 1.1 | - | 0.1 | - | - | - |
| East & South Africa | - | - | - | - | 0.1 | - | - | - | - | 0.1 |
| South Asia | - | - | 0.5 | 0.2 | 0.3 | 0.1 | 0.7 | 0.7 | 0.5 | 0.6 |
| South East Asia | 54.2 | 52.1 | 58.2 | 52.5 | 47.3 | 44.1 | 39.6 | 43.6 | 38.4 | 39.2 |
| Australasia | 2.1 | 1.8 | 0.7 | 0.5 | - | 1.1 | 1.3 | 0.3 | 3.2 | 2.2 |
| U.S.S.R., Eastern Europe and China | 7.3 | 7.7 | 8.0 | 11.1 | 9.5 | 10.4 | 11.5 | 13.3 | 13.6 | 13.5 |
| TOTAL IMPORTS: | 270.6 | 262.6 | 275.6 | 245.8 | 220.0 | 206.2 | 205.7 | 214.6 | 202.2 | 204.4 |

TABLE 8. SOURCES OF IMPORTS - TEN YEARS 1977 TO 1986 *(Million tonnes)*

## *LATIN AMERICA*

| Exported From | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 |
|---|---|---|---|---|---|---|---|---|---|---|
| U.S.A. | 5.8 | 4.8 | 12.1 | 14.8 | 13.2 | 16.7 | 13.1 | 13.2 | 15.1 | 11.9 |
| Latin America | 5.4 | 18.8 | 11.5 | 13.8 | 10.8 | 20.7 | 21.1 | 20.9 | 18.5 | 7.7 |
| Western Europe | 0.5 | - | - | - | - | 1.4 | 1.1 | 1.9 | 0.7 | 1.3 |
| Middle East | 77.7 | 74.4 | 74.5 | 80.4 | 69.2 | 44.5 | 42.4 | 34.8 | 30.4 | 28.5 |
| North Africa | 7.4 | 9.6 | 6.0 | 6.7 | 3.5 | 6.3 | 3.0 | 4.1 | 4.7 | 0.1 |
| West Africa | 13.6 | 15.0 | 19.5 | 20.8 | 14.2 | 5.1 | 5.8 | 5.2 | 8.4 | 7.5 |
| South East Asia | - | - | 1.2 | - | - | - | - | - | - | - |
| U.S.S.R., Eastern Europe and China | 8.9 | 10.2 | 10.0 | 10.2 | 9.2 | 10.0 | 10.9 | 10.0 | 15.2 | 23.2 |
| TOTAL IMPORTS: | 119.3 | 132.8 | 134.8 | 146.7 | 120.1 | 104.7 | 97.4 | 90.1 | 91.0 | 80.2 |

TABLE 9. SUPPLY AND DEMAND FOR OIL - YEAR 1986 *(Million tonnes)*

| AREA | SUPPLY | | | TOTAL | DEMAND | | |
|---|---|---|---|---|---|---|---|
| | Production | Imports | Stock Reduction | | Consumption | Exports | Stock Build |
| U.S.A. | 485.7 | 297.1 | 1.3 | 784.1 | 746.8 | 37.3 | - |
| Canada | 83.8 | 21.1 | - | 104.9 | 67.1 | 33.0 | 4.8 |
| Latin America | 330.4 | 80.2 | - | 410.6 | 215.4 | 175.7 | 19.5 |
| Western Europe | 198.2 | 435.4 | 7.1 | 640.7 | 587.1 | 53.6 | - |
| Middle East | 639.4 | 9.6 | - | 649.0 | 108.3 | 539.3 | 1.4 |
| North Africa | 139.6 | 5.1 | 18.3 | 163.0 | 41.0 | 122.0 | - |
| West africa | 111.2 | 2.4 | 2.9 | 116.5 | 18.1 | 98.4 | - |
| East & South Africa | - | 22.4 | 0.4 | 22.8 | 22.6 | 0.2 | - |
| Southern Asia | 34.5 | 21.2 | 3.4 | 59.1 | 56.3 | 2.8 | - |
| South East Asia | 108.8 | 97.5 | - | 206.3 | 117.3 | 76.2 | 12.8 |
| Japan | 0.6 | 204.4 | 0.4 | 205.4 | 204.4 | 1.0 | - |
| Australasia | 34.0 | 9.3 | - | 43.3 | 31.3 | 5.2 | 6.8 |
| U.S.S.R., E. Europe and China | 766.6 | 55.8 | - | 822.4 | 665.3 | 148.2 | 8.9 |

TABLE 10. INCIDENCE OF OIL SPILLS WORLDWIDE (1974 – 1986)

| Year | Total spills all sizes | 50 – 5000 bbls | 5000+ bbls |
|---|---|---|---|
| 1974 | 1450 | 91 | 26 |
| 1975 | 1350 | 98 | 23 |
| 1976 | 1099 | 66 | 25 |
| 1977 | 956 | 66 | 20 |
| 1978 | 746 | 57 | 24 |
| 1979 | 695 | 54 | 37 |
| 1980 | 554 | 48 | 13 |
| 1981 | 401 | 48 | 5 |
| 1982 | 247 | 44 | 3 |
| 1983 | 216 | 52 | 11 |
| 1984 | 146 | 25 | 7 |
| 1985 | 137 | 25 | 8 |
| 1986 | 118 | 24 | 6 |
| Total | 8115 | 698 | 208 |

## TABLE 11. WORLD OIL TANKER FLEET STATISTICS

as at 31st December, 1986

| DWT Classification | With Segregated Ballast Tanks | With Crude Oil Washing & I.G. | Classification Total |
|---|---|---|---|
| 10 – 16,000 | 58 | 3 | 247 |
| 16 – 25,000 | 99 | 12 | 462 |
| 25 – 35,000 | 117 | 41 | 506 |
| 35 – 45,000 | 137 | 74 | 299 |
| 45 – 60,000 | 108 | 97 | 184 |
| 60 – 80,000 | 128 | 136 | 223 |
| 80 – 90,000 | 73 | 124 | 175 |
| 90 – 120,000 | 67 | 101 | 147 |
| 120 – 160,000 | 44 | 136 | 196 |
| 160 – 200,000 | 11 | 18 | 27 |
| 200 – 255,000 | 31 | 136 | 147 |
| 255 – 320,000 | 37 | 173 | 188 |
| Over 320,000 | 3 | 55 | 56 |
| | 913 | 1,106 | 2,857 |
| | 31.96% | 38.71 % | 100.00 % |

TABLE 12. SEABORNE TRADE IN MAJOR LIQUID CHEMICALS
Year 1985 *(Thousand tonnes)*

| | |
|---|---|
| Acetone | 112 |
| Benzene | 949 |
| Butanol | 56 |
| Cyclohexane | 211 |
| Dodecylbenzene | 100 |
| Ethanol | 750 |
| Ethyl benzene | 77 |
| Ethylene dichloride | 863 |
| Ethylene glycol | 1,011 |
| Isopropylbenzene | 229 |
| Methanol | 2,459 |
| Perchlorethylene | 103 |
| Phenol | 170 |
| Propanol | 108 |
| Propylene glycol | 82 |
| Styrene | 910 |
| Toluene | 675 |
| Xylene | 1,189 |
| Sub-total – Petrochemicals | 10,054 |
| Phosphoric acid | 4,950 |
| Sulphuric acid | 1,170 |
| Caustic soda (50% Solution) | 4,990 |
| TOTAL | 21,164 |

TABLE 13. SEABORNE TRADE IN MAJOR LIQUID CHEMICALS - YEARS 1985 *(Thousand tonnes)*

| FROM: | North America | Latin America | Western Europe | E.Europe/ U.S.S.R. | Africa | Middle East | Japan | Asia/ Oceania | Other - Unspecified | TOTAL |
|---|---|---|---|---|---|---|---|---|---|---|
| TO: | | | | | | | | | | |
| North America | 330 | 515 | 1,450 | 85 | 5 | 140 | 240 | 200 | 10 | 2,975 |
| Latin America | 1,980 | 10 | 100 | 5 | 20 | - | - | - | 30 | 2,145 |
| Western Europe | 1,280 | 220 | - | 1,450 | 1,380 | 480 | 45 | 425 | 250 | 5,530 |
| E. Europe/U.S.S.R. | 1,005 | 30 | 660 | 40 | 40 | 40 | - | - | 10 | 1,825 |
| Africa | 175 | - | 360 | - | - | 5 | - | 20 | - | 560 |
| Middle East | 110 | - | 315 | 10 | - | 20 | 25 | 15 | - | 495 |
| Japan | 710 | 275 | 110 | - | - | 245 | - | 440 | - | 1,780 |
| Asia/Oceania | 1,720 | 80 | 1,105 | 20 | 1,240 | 410 | 605 | 415 | 100 | 5,695 |
| Other - Unspecified | 10 | - | 50 | 60 | 10 | - | - | 20 | 10 | 160 |
| TOTAL: | 7,320 | 1,130 | 4,150 | 1,670 | 2,695 | 1,340 | 915 | 1,535 | 410 | 21,165 |

FIGURE 1. OIL SPILLS OVER 5,000 BBLS OCCURRING FROM TANKERS, 1974-1979 (inclusive)

FIGURE 2. OIL SPILLS OVER 5,000 BBLS OCCURRING FROM TANKERS, 1980-1985 (Inclusive)

# ANNEX XVI

# QUALITY ASSURANCE OF CONTAMINANT DATA FOR THE MARINE ENVIRONMENT

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# INTRODUCTION

1. The importance of evaluating the quality of contaminant data from the marine environment has been fully recognized only within the last two decades. While quality assurance programmes have been a standard practice in the chemical and pharmaceutical industries for some time, their implementation in the environmental sciences, with respect to chemical analysis, has occurred relatively recently.

2. Quality assurance, in the case of chemical analyses, refers to the procedures used to evaluate whether measurements are good enough or adequate for their intended purpose (Taylor, 1981; 1985). Taylor (1985) defines quality assurance as consisting of two distinct, but related, activities:

*quality control - those procedures and activities developed and implemented to produce a measurement of requisite quality;*

*quality assessment - the procedures and activities utilized to verify that the quality control system is operating within acceptable limits and to evaluate the quality of the data.*

3. For marine environmental contaminants, data quality control includes sampling as well as analytical procedures. The methods used must be adequate for the intended purpose (i.e. minimal contamination during sampling and adequate sensitivity and specificity of analysis) and properly employed.

4. The assessment of the quality of data requires knowledge of the expected value. Data quality assessments are often made using standard reference materials, intercomparison of techniques and/or statistical tests (Taylor, 1985).

5. Even if quality control and assessment procedures are in place, the success of the quality assurance programme depends on the commitment of all concerned. And, as Taylor (1981) suggests, "analysts must aspire to produce high quality data and must be their own most severe critics". But in any case, quality assurance programmes should be subjected periodically to external checks (e.g. intercalibration).

# A. ASSESSMENT OF THE QUALITY OF MARINE ENVIRONMENTAL CHEMICAL DATA

6. Probably the best assessment of the quality of marine environmental data can be made for trace metals. Unlike many synthetic organic contaminants, trace metals have been analyzed by marine scientists for decades in virtually all substrates (i.e. water,organisms, sediments, etc.) because of their geochemical as well as of their environmental importance.

7. Problems with the quality of marine trace metal data have been well documented over the past two decades. Although similarly detailed documentation does not exist for organohalogens and hydrocarbons, the other commonly monitored marine contaminants, there is no reason to believe that the situation has been any better.

## *1. Water column*

8. One of the earliest assessments of the poor quality of trace metal data from the marine environment was for water samples. In 1970 an international intercomparison of trace metal analyses of three large sea-water samples was conducted by the Woods Hole Oceanographic Institution (Brewer and Spencer, 1970). Twenty-six laboratories participated and reported results which varied by an order of magnitude, clearly indicating that data from these laboratories were probably not comparable. Additional intercalibration exercises were subsequently initiated to assess the ability of participants to agree on the analysis of standardized samples for individual metals such as lead and mercury (Participants, 1974; 1976; Olafsson, 1982). Unfortunately, the results of these intercomparisons also demonstrated the non-comparability of data produced by most of the participants.

9. Among the organizations most concerned about intercomparability of water column trace metal data have been the IOC and ICES, mainly because of their interest in multi-national monitoring programmes. Results of more recent intercalibration exercises conducted by these organizations, when compared to results reported by Brewer and Spencer (1970) (Figure 1), demonstrate the improved agreement among a growing number of laboratories. The results of the most recent intercalibration are also in agreement with the most recent consensus on trace metal concentrations in sea water (Bruland, 1983) which have been reported in the scientific literature and which satisfy the criteria of oceanographic consistency.

10. Although there have been considerable improvements in the accuracy and precision of trace metal analyses of sea water by a growing number of laboratories, the ability to produce data of sufficient quality for most purposes is still not widespread. This is demonstrated by the results

of a more recent ICES intercomparison (Berman *et al.*, 1984) which indicated that generally less than half of the participating North American and European laboratories reported results for common contaminant metals on the same sample that were within 33 per cent of the value determined by isotope dilution mass spectrometry (Table 1).

11. The most recent experience, based on results of intercalibration exercises for organohalogen compounds leads to conclusions regarding data quality that are similar to those for trace metal quality (IOC, 1984). Problems with data quality for these compounds, especially polychlorinated biphenyls, are associated with both sampling and analytical procedures.

## 2. *Biological tissues*

12. From the first intercalibration exercise conducted for the purpose of assessing the quality of trace metal analyses in marine organisms (IDOE, 1971; Uthe *et al.*, 1972) variations between results submitted by participating laboratories were never as large as those reported for sea-water analyses. This is also true for organochlorine residues (Holden *et al.*, 1983). Nevertheless, the agreement between laboratories for contaminant analyses of biological samples have never been particularly good. As pointed out by Holden *et al.* (1983), who reviewed several intercalibration exercises for biological tissues, "with few exceptions the levels of agreement between participating analysts are not good enough to allow them all to take part in multi-laboratory monitoring programs".

13. The results of a recent ICES intercalibration of trace metals in biological tissues (Berman, 1984) can be used to assess the present state of quality of these data. A significant number of the results submitted by participants in this exercise were rejected on the basis of a statistical analysis of the data (i.e. t-test) (Table 2), and for those results that were not rejected the coefficient of variation for some metals (e.g. As and Pb) was still quite large. It must also be pointed out that these results do not include variations due to sampling, preparation or preservation procedures, since these steps were performed by a co-ordinating laboratory.

14. The quality assessment of data on marine biological tissues, based on intercalibration exercises, can be best summed up by the conclusion of Holden *et al.* (1983) who states:

> "Assuming the accuracy and precision of routine analyses of biological samples such as fish and shellfish are no better than for the intercomparison samples and possibly worse, the results of co-ordinated monitoring involving the same or similar laboratories in different countries cannot give a true indication of the relative levels of contamination in fish and shellfish samples of similar type from these countries. The best that can be achieved is a distinction between areas of high and low contamination."

### 3. *Marine sediments*

15. It has generally been assumed that the marine substrate most easily analyzed is the sediment column. Implicit in this assumption is that the quality of chemical data on marine sediments is expected to be good.

16. With the increased interest in using sediments as indicators of pollution in the nearshore region, it is surprising that few assessments of data quality with regard to this substrate have been made. Although some intercomparisons of sediment analyses have been conducted on national and regional scales, the only one conducted on an international scale was an ICES Sediment Working Group intercalibration exercise for trace metals (Loring and Rantala, 1988). Forty laboratories from North America and Europe participated. Since most of these laboratories were experienced in trace metal analyses of sediments, the results clearly give an assessment of the quality of data in a situation that is better than that prevailing on a global scale.

17. This intercomparison for sediments included results of analyses of samples after total digestion (HF) and strong and weak acid leaches. The relative standard deviation (rsd) of the results ranged from 1.5 to 106 per cent, depending on the metal and the extraction procedure.

18. Loring and Rantala (1988) assumed that an rsd of 20 per cent or less should constitute an acceptable inter-laboratory agreement. Based on this criterion, it was observed that there was no general agreement of any metal for all samples and all digests. They concluded that most of the laboratories could produce precise but seldom comparable results.

## B. STATUS OF QUALITY CONTROL OF MARINE ENVIRONMENTAL CHEMICAL DATA

19. The poor quality of analytical data from the marine environment indicates clearly the need for improved quality control measures to validate measurements if they are to be of any use. It has been pointed out by Goldberg and Taylor (1985) that existing environmental data banks are of little value because they contain both valid and non-valid data which cannot be distinguished.

20. In the future it will be possible to assess the changing state of the marine environment with regard to contaminant levels only if the data used to do so are validated through quality assurance programmes. The increased awareness of the problem by a growing portion of the international marine science community has led to a general increase in the attention to, and improvements of, quality control measures.

21. In recent years considerable attention has been given to establishing quality assurance

is the U.S. Mussel Watch Program (Galloway *et al.*, 1983) where intercomparison of participating laboratories and development of reference materials are integral parts of the quality assurance programme.

22. International organizations such as ICES, IOC and UNEP have ongoing intercalibration programmes to assess the quality of laboratory analyses. In addition, these organizations have also begun to sponsor exercises that aim to assess the comparability of various collection and sample preparation techniques (Brewers *et al.*, 1985; Brewers and Windom, 1982; Windom and Byrd, 1988).

23. A major aid in the improvement of quality control procedures in the analysis of marine samples has been the development of standard reference materials (SRMs) similar in nature to marine environmental substrates. New SRMs include standard sea water, sediments and biological tissues (Cantillo, 1986).

24. In recognition of the fact that many developing countries lag behind in the implementation of quality assurance programmes, some international organizations are sponsoring regional training programmes on sampling and analytical procedures. These organizations also have ongoing programmes to develop and publish standard analytical methods that meet typical quality assurance standards.

## CONCLUSION

25. The inability to validate most past data in the scientific literature and in data banks, coupled with the documented inability of many laboratories to produce comparable data on the same samples, makes it virtually impossible to estimate how concentrations of contaminants have changed in most regions, regardless of the substrate considered. However, the increased attention paid by national and international organizations to establishing quality assurance programmes to validate data should minimize this problem in the future. But, as has been pointed out by Taylor (1985), the problem cannot be solved once and for all. Instead, it requires ongoing recognition, remedial action and continual reassessment.

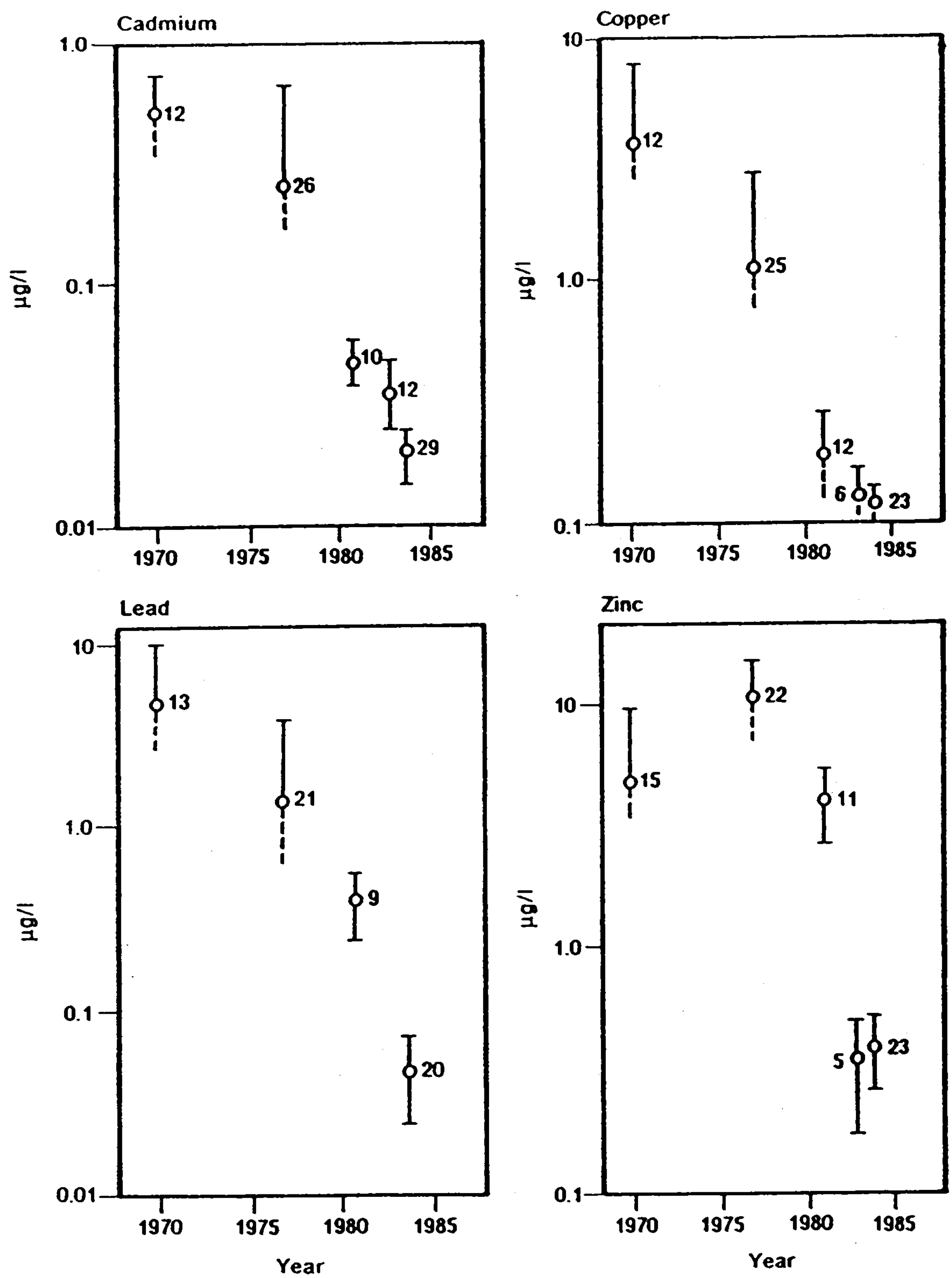


FIGURE 1. Comparison of results of intercomparison exercises for trace metal analysis of seawater samples conducted during the past two decades (Brewer and Spencer, 1970; Jones, 1977; Bewers *et al.*, 1981; Bewers and Windom, 1983; Berman *et al.*, 1984). Number of participating labs shown by mean.

TABLE 1. SUMMARY OF RESULTS FROM ICES FIFTH ROUND CALIBRATION FOR TRACE METALS IN SEA WATER*

| | Lab Reporting Results | Lab Reporting Results within 33% of IDMS+ |
|---|---|---|
| Cadmium | 48 | 25 |
| Copper | 46 | 20 |
| Lead | 40 | 19 |
| Nickel | 29 | 15 |
| Zinc | 39 | 12 |

* Berman *et al.*, 1984.

\+ Isotope dilution mass spectrometry.

TABLE 2 SUMMARY OF ICES SEVENTH ROUND INTERCALIBRATION FOR TRACE METALS IN BIOLOGICAL TISSUES*

| | Sample | Mean and CV of unrejected data | | No. Labs returning data | No. Labs whose data was not rejected |
|---|---|---|---|---|---|
| | | $\mu g\ g^{-1}$ | CV per cent | | |
| Arsenic | A | 25 | 21 | 21 | 21 |
| | B | 7.1 | 29 | 20 | 19 |
| | C | 4.6 | 35 | 20 | 20 |
| Cadmium | A | 26 | 8 | 48 | 37 |
| | B | 0.75 | 13 | 48 | 37 |
| | C | 0.06 | 38 | 44 | 31 |
| Copper | A | 331 | 10 | 49 | 45 |
| | B | 3.7 | 16 | 48 | 41 |
| | C | 3.1 | 27 | 49 | 42 |
| Lead | A | 25 | 21 | 21 | 21 |
| | B | 0.29 | 41 | 39 | 28 |
| | C | 1.98 | 41 | 38 | 31 |
| Mercury | A | 0.25 | 19 | 37 | 36 |
| | B | 0.081 | 10 | 36 | 31 |
| | C | 0.050 | 16 | 37 | 30 |
| Zinc | A | 179 | 6 | 49 | 38 |
| | B | 58 | 9 | 48 | 39 |
| | C | 93 | 10 | 48 | 41 |

* Berman *et al.*(1984)

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