REGIONAL SEAS

UNITED NATIONS ENVIRONMENT PROGRAMME

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

UNEP Regional Seas Reports and Studies No. 114 /1

Prepared in co-operation with

United Nations FAO UNESCO WHO WMO IMO IAEA

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*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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ANNEX I

THE PROBLEM OF PERSISTENT PLASTICS AND MARINE DEBRIS IN THE OCEANS

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INTRODUCTION

1. There has been growing concern in recent years among fishermen, scientists, seafarers, conservationists, government officials and others about the increasing amounts of persistent plastics and marine debris found at sea and on beaches. Since the 1940s, there has been an enormous increase in the use of plastic and other synthetic materials which have replaced many natural and more readily degradable materials. Because this debris tends to be buoyant and persistent, it presents the likelihood of interaction with marine organisms. This debris can be loosely classified in three groups: (1) fishing gear and equipment, such as nets and lines; (2) packing bands and straps and synthetic ropes and (3) a variety of other forms of non-degradable plastic debris and litter from both ships and land-based sources. These are generally materials that are likely to be disposed of, or that have a high probability of being lost.

2. All of these, both singly and in combination, present a threat to the environment. A major reason for this is that some of the most valuable qualities of synthetic material are also the least beneficial to the marine environment. Plastics tend to be extremely durable, lightweight, and inexpensive: qualities much desired by users. However, these qualities also mean that the products will persist and float in the water after loss or discard, and they are most readily discarded.

I. BACKGROUND

3. Plastics do not degrade as rapidly at sea as on land (Andrady, 1988). Ultra-violet light from the sun causes most plastic to embrittle and eventually disintegrate. However, water filters the ultra-violet light, thus weakening its effect. It also keeps the plastic cool and reduces thermal degradation. If algae or other marine organisms cover the surface, this further reduces the rate of degradation. If the added weight of the fouling causes the plastics to sink, the degradation will be even slower, because of cooler temperatures and reduced light.

4. No one who uses rivers or oceans will contest the presence of large amounts of plastic debris in the aquatic environment. The quantities are documented in a number of studies: benthic sediment surveys off the coast of the United Kingdom showed 2,000 pieces of plastic per square metre, and beach surveys in New Zealand showed even greater concentrations of plastic particles on its beaches (Coleman and Wehle, 1984). A Mediterranean study revealed that 60-70 per cent of surveyed debris was plastic (Morris, 1980), and a recent ocean sample of debris in the North Pacific found the level to be higher than 80 per cent (Dahlberg and Day, 1985). Plastic debris is even appearing on the remote shores of Antarctica (Torres and Gajardo, 1985) and has been found in every sector of the world's oceans (Pruter, 1987). Plastics are being used more often, and because of their durability, are remaining in the environment longer than most other man-made substances. As ever-increasing amounts of plastic are being produced, the threat to the marine environment will continue to grow.

5. Plastics and other types of marine debris come from a wide variety of sources. The ocean sources include commercial fishing, boats, merchant vessels, recreational boats and fishing vessels, cruise and tour ships, military vessels, and oil and drilling platforms. Land-based sources are also responsible for debris in the oceans. These include plastic manufacturing plants, sewer and sewage treatment overflows, solid waste dump sites, and litter.

6. Although plastic debris is ubiquitous in the marine environment, it does tend to be concentrated in certain areas. These include obvious locations, such as coastlines and beaches, shipping corridors, river estuaries, fishing areas, and dump sites, as well as less apparent areas, such as ocean gyres (Galt, 1985), upwellings or water-mass convergences (Dahlberg and Day, 1985). These are unfortunately also the areas where marine organisms concentrate, and therefore the debris poses an ever-increasing threat to them (Carr, 1987). Many marine organisms are attracted to floating objects and debris. This may provide a source of cover, may lure other prey (Laist, 1987), or may resemble natural prey. All of these attractions increase the opportunity for entanglement or ingestion.

II. FISHING GEAR AND EQUIPMENT

7. Debris from fishing vessels cause a major problem. Nets, which are either discarded, cut away when no longer useful, or lost, can continue to catch or trap marine life. They can trap marine organisms while floating on the surface, when sunk or snagged on the bottom, or while drifting at some intermediate level. Marine mammals (including endangered great whales), fish, sea birds, and turtles are among the animals caught. A recent Food and Agriculture Organization report estimated the world-wide loss of fishing gear in 1975 to have been 150,000 tonnes (FAO, 1985). In the North Pacific, an area of significant fishing activity, the major gill-net or drift-net operations use about 170,000 km of netting annually (Uchida, 1985). While there are no precise estimates of the total amounts of netting which are lost or discarded, even a very low percentage of these would result in a significant amount of free-floating netting, taking into account the large amount of fishing efforts in the North Pacific (Eisenbud, 1985; Laist, 1987) and the durability of the plastic netting. Another study estimates that trawlers fishing in the United States Exclusive Economic Zone off of Alaska have lost between 50 and 65 trawl nets during each of the last three years from a fishery involving 5,500 km of net annually (Low, 1985).

8. In the past, fishing nets were made of natural fibres, which degraded fairly readily when lost or discarded. However, beginning in the 1940s, and by the early 1970s, the fishing industry had converted to plastic netting, which was cheaper, lighter and more buoyant. Because this gear is not degradable, it can continue to trap marine life long after its loss or release. The estimates are imprecise at best as to the durability and "active life" of these nets or debris once at sea, although studies are now being conducted. Some initial results suggest that plastic nets and traps may continue to catch fish for several years either at the surface or at deeper levels or from the ocean floor (High, 1985). Even when balled up on the ocean floor as a result of tidal currents and wave action, nets may ensnare crustaceans and other benthic organisms. Lost or discarded fish traps or pots or parts thereof also contribute to the problem.

9. Fishing nets may become free-floating debris in a number of ways. They may be intentionally discarded or abandoned when they are in need of repair, or well-worn parts of nets may be discarded during repair. Nets may also be lost through entanglements or conflicts with other types of fisheries. Vessels conducting fishing operations or transiting through areas where stationary fishing gear has been set may find their nets badly tangled and be forced to cut them free. Fisheries conducted around wrecks or debris-collecting points are extremely vulnerable to loss through snagging, thus creating an even larger hazard. Finally, fishermen operating illegally will abandon their nets if confronted with surveillance or enforcement authorities.

10. Once the net is discarded or lost, it poses a continued threat to fish, marine mammals, turtles and sea-birds. Entangled animals may drown, or may be wounded or injured in a manner which will later affect their chances of survival. There are no precise estimates of the total losses incurred as a result of discarded and lost nets, but they clearly affect present and future fisheries. There have been numerous reports of the impact of these nets on marine mammal populations from all over the world (Coleman and Wehle, 1984; Laist, 1987). Drift nets, when used by fishermen, are known to trap and kill large numbers of Dall's porpoises in the North Pacific, and they presumably continue to do so when discarded or lost. The North Pacific fur seal population has been declining by about 6 per cent annually for a number of years, and entanglement in netting and debris is thought to be a significant factor in this decline (Fowler, 1987). Populations of northern sea lions in the Eastern Aleutian islands are also declining, and entanglement is thought to be a factor (Laist, 1987). The entanglement of the endangered Hawaiian monk seal in nets washed ashore has been reported, as have marine mammal deaths caused by entanglement in unattended nets, including recent examples of whales entangled in the Gulf of Alaska (Froetschel, 1986) and in the Tyrrhenean Sea (Di Natale, 1983). Roughly half of all right and humpback whales photographed during a recent study of New England whales showed scars of contact with fishing lines, nets, or gear (Weinrich, 1987). Netting is also known to entangle and kill large numbers of sea-birds (Bourne, 1977; De Grange and Newby, 1980) and turtles, including some endangered species (Balazs, 1985; Carr, 1987).

11. Lost and discarded netting and other debris also seriously affects shipping, because this debris can entangle and foul propellers, damage propellers and drive shafts, and clog sea intakes and evaporators. While the evidence has yet to be carefully documented and presented, the loss in productive time at sea and the costs of repair represent a clear economic loss to the maritime industries. There have also been reports of divers being entangled in abandoned netting.

III. PLASTIC STRAPPING AND PACKING BANDS

12. Plastic strapping and packing bands are used to hold material on pallets, secure cargo, strap boxes and crates, or reinforce packing cases. When pulled off rather than cut, the discarded bands are more likely to encircle marine mammals or large fish in a girdle or band which becomes progressively tighter as the animal grows. The jaw and limbs, as well as the neck and body, can also become entangled and thereby restrict the ability to eat or move. Plastic yokes for six- or four-pack beverage containers, perhaps best characterized as "litter", present a similar threat to birds and smaller fish.

13. Examples of fish, birds, and marine mammals being caught in this fashion have been reported (Coleman and Wehle, 1984). In the North Pacific, a survey of entangled fur seals found 20 per cent of them entangled in plastic packing bands (Scordino, 1985). There have been reports of this type of entanglement of other marine mammals from other areas of the Pacific and from locations worldwide, including New Zealand (Cawthorn, 1985), South Africa (Shaughnessy, 1980) and Antarctica (Bonner and McCann, 1982).

IV. PLASTIC LITTER

14. The third category of polluting debris consists of plastic litter: plastic bags or sheeting, packing material, raw plastic wastes, and containers and other plastic litter. There are numerous studies of turtles, whales and other marine mammals which were apparently killed by ingesting plastic bags or sheeting. Turtles are perhaps the most vulnerable to this type of pollution. One recent study found that plastic bags and sheeting were the most common type of debris ingested by turtles (Balazs, 1985; Carr, 1987). The turtles may mistake it for one of their normal food sources (e.g. jellyfish or salps), or eat it because it has become covered or encrusted with marine growth. Ingestion of this type type of material by whales, dolphins and porpoises and other marine mammals has also been reported (Coleman and Wehle, 1984; Center for Environmental Education, 1987).

15. A potentially more significant problem may be created by the increasing quantities of small plastic particles in the ocean. Pellets or spherules of raw plastic used in plastics production, and as insulation and packing material, enter the ocean from land-based sources such as plant and sewage outfalls, from rivers, and from ships. The major threat of this type of pollution is ingestion, by fish and sea-birds, which can affect the animals' feeding and digestive processes. This type of

pollution represents perhaps the most widespread threat to the marine environment. In a recent study, plastic particles were found in the digestive tracts of 25 per cent of the world's sea-bird species (Day *et al.*, 1985). More alarmingly, certain species seem more prone to plastic ingestion. In one species of albatross, plastic particles were found in 90 per cent of the chicks examined (Fry *et al.*, 1987).

16. Although the presence of countless bits of plastic in the ocean may not raise the same public concern as does an oil spill or a chemical hazard, it is none the less a visible reminder that the ocean is being used as a dump for plastics and other wastes. In beach areas where plastic debris accumulates, the resulting degradation of the environment often requires costly clean-up efforts. In addition, a littered shoreline poses a clear threat to tourism. Recent incidents in the United States involving plastic medical and personal hygiene debris, such as hypodermic needles and tampon applicators found on Atlantic beaches, emphasized this problem. In 1975, the U.S. National Academy of Sciences estimated that 6.4 million metric tons of litter were being discarded annually by the shipping and fishing industries annually.

V. RELEVANT LEGAL INSTRUMENTS AND CURRENT ACTIVITIES OF INTERNATIONAL ORGANIZATIONS

17. Both the Convention on the Prevention of Marine Pollution by Dumping of Waste and Other Matter (LDC) and the International Convention for the Prevention of Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78), address the problems presented by the disposal of persistent plastics. Annex I of the LDC prohibits the dumping of "persistent plastics and other persistent synthetic materials, for example netting and ropes, which may float or remain in suspension in the sea in such a manner as to interfere materially with fishing, navigation or other legitimate uses of the sea". Clearly, the dumping of the types of materials discussed above is in violation of the LDC.

18. Annex V of MARPOL 73/78 in Regulation 3, addresses the problem of pollution by ship-generated plastic waste. It states that "the disposal into the sea of all plastics, including but not limited to synthetic ropes, synthetic fishing nets and plastic garbage bags, is prohibited". However, Regulation 6 of that Annex provides that an exception be made with regard to "accidental loss of synthetic fishing nets or synthetic material incidental to the repair of such nets, provided that all reasonable precautions have been taken to prevent such loss". With the approval of Annex V by the United States in 1987, the percentage of the world's maritime fleet required for adoption of the Annex was reached, and it entered into force on 31 December 1988.

19. The issue of plastic debris in the oceans was discussed at the Ninth and Tenth Consultative Meetings of Contracting Parties to the London Dumping Convention, where it was

determined that, whether the deliberate disposal of persistent plastics and other synthetic material is covered by Annex V of MARPOL 73/78, or by Annex I of the London Dumping Convention, it is a source of marine pollution which the London Dumping Convention calls upon all Contracting Parties to control. It was noted that living resources and marine life may be harmed, and legitimate uses of the sea may be impaired, by the presence of persistent plastics and synthetic materials. The Tenth Meeting also published a report on this subject, which is an earlier version of this annex (Arnaudo, 1986).

20. This issue has also been discussed regularly since 1986 at meetings of the Marine Environment Protection Committee (MEPC) of the International Maritime Organization, which is responsible for the administration of MARPOL 73/78. At its 26th Session (September, 1988), the MEPC adopted the Guidelines for the Implementation of Annex V (MEPC, 1988). The guidelines provide comprehensive instructions and guidance concerning the prevention of pollution by garbage, including plastic, from ships. Two citations from the guidelines are particularly relevant:

1) "Fishing gear, once discharged, becomes a harmful substance". (Paragraph 3.5).

2) "Plastic garbage must be retained aboard ship for discharge at port reception facilities unless reduced to ash by incineration". (Paragraph 4.3.1).

21. The guidelines also stress the need for training and education of operators and seafarers and for public information programmes for the public. They detail handling and processing procedures and urge research into the problem of plastics in the marine environment. Finally, the guidelines outline measures for ensuring compliance with Annex V.

22. The United Nations Food and Agriculture Organization (FAO) also considered the issue of protection of living resources from entanglement by nets and plastic debris at the Seventeenth Session of the FAO Committee on Fisheries (FAO, 1987). From responses from member countries and a review of the literature on the subject, FAO reported on the scope of the problem and on the national and international responses to it. The report discusses existing and proposed methods to reduce entanglement of marine organism in fishing nets and equipment, whether active, lost or discarded.

23. In addition to the activities of the LDC, IMO and FAO described above, there are also a number of organizations which administer regional agreements to control pollution. These regional organizations might be appropriate fora in which to pursue the problem of lost and discarded nets, and other persistent plastic marine debris. The Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area calls upon contracting parties "to take all appropriate legislative, administrative or other relevant measures in order to prevent and abate pollution and to protect and enhance the marine environment of the Baltic Sea Area" (Article III). It also prohibits the disposal of "all plastics, including but not limited to synthetic ropes, synthetic fishing nets and plastic garbage bags" (Annex III). As with MARPOL 73/78, an exception is made for the accidental loss of synthetic fishing nets.

24. The Oslo Convention for the Prevention of Marine Pollution by Dumping from Ships and Aircraft, whose membership consists of the maritime states of Western Europe, also includes a prohibition on the dumping of persistent plastics and synthetic materials in a manner similar to the LDC. Article 1 of the Oslo Convention calls upon Contracting parties "to take all possible steps to prevent the pollution of the sea 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". Article 5 of the Convention prohibits the dumping of certain substances listed in Annex I, among them: "persistent plastics and other persistent synthetic materials which may float or remain in suspension, or sink to the bottom, and which may seriously interfere with fishing or navigation, reduce amenities, or interfere with other legitimate uses of the sea".

25. The Paris Convention for the Prevention of Marine Pollution from Land-Based Sources, comprised of the same membership as the Oslo Commission, addresses the issue of pollution by raw plastics, which may be ingested by birds, fish, and other marine organisms. Article 4 of the Convention calls for the elimination of pollution of maritime areas by substances listed in Annex A, among them "persistent synthetic materials which may float, remain in suspension or sink, and which may seriously interfere with any legitimate use of the sea".

26. Under the aegis of the UNEP Regional Seas Programme and in co-operation with the relevant international and regional organizations, seven regional intergovernmental agreements on the protection, management and development of the marine environment have been adopted. While these do not specifically address the issue of lost and discarded fishing gear and persistent plastics, they do call for appropriate measures to prevent pollution by ships. Problems specific to individual areas could be raised and discussed under these agreements. The agreements involve the regional seas of the Mediterranean, the Wider Caribbean, the West Coast of South America and Panama, East and West Africa, the Kuwait Action Plan sea area and the Red Sea and Gulf of Aden. Of these UNEP Regional Seas agreements, the Barcelona Convention for the Protection of the Mediterranean Sea Against Pollution is the most specific in addressing this problem. In Article 4 it enjoins contracting parties "to take all appropriate measures in accordance with the provisions of this Convention and those protocols in force to which they are party, to prevent, abate and combat pollution of the Mediterranean Sea Area and to protect and enhance the marine environment in that Area". The attached protocol for the prevention of Pollution of the Mediterranean Sea by Dumping from Ships and Aircraft, prohibits the dumping of "persistent plastic and other persistent synthetic materials which may materially interfere with fishing or navigation, reduce amenities, or interfere with other legitimate uses of the sea".

27. The Commission for the Conservation of Antarctic Marine Living Resources has since 1984 requested members to report annually on marine debris and entangled animals in Antarctic waters. Countries have been urged to take all possible steps to prevent loss or discard of fishing nets and other plastic debris. In 1988, the Commission distributed brochures for the public and placards for vessels describing the problem and suggesting ways to avoid pollution. The North Pacific Fur Seal Commission, no longer in existence, was perhaps the first international organiza-

tion to recognize the impact of marine debris on a marine mammal population. It reported annually on entangled fur seals from 1967, and circulated posters to alert the fishing industry and the public to the problem. Other marine resource conservation and management organizations have also focused on relevant aspects of the plastic debris problem. For instance the International North Pacific Fisheries Commission has reported on marine mammal entanglement since 1978.

28. In addition, several international bodies have addressed the problem. The Working Committee on the Global Investigation of Pollution in the Marine Environment (GIPME) discussed the plastic debris problem and the work on a manual to monitor the problem at its sixth session (October 1986). The issue wa. also raised at the October 1985 meeting of the International Council for the Exploration of the Sea. Finally, the marine debris problem has also been discussed at regional bodies of international organizations. A group of experts from the International Oceanographic Commission (IOC), the UN Environment Program and the Food and Agriculture Organization met in October 1987 to examine problems specific to the Mediterranean Sea, and another group of experts, organized by the IOC, met in May 1988 to examine marine debris in the Caribbean Sea.

29. It should also be noted that several international conferences have been held on this subject. In April 1986 the United States hosted the Sixth International Ocean Disposal Symposium in Pacific Grove, California, at which the disposal of persistent plastics and fishing nets and their effects on living resources and marine trasnportation was an important theme. The United States also hosted the International Workshop on the Fate and Impact of Marine Debris, in Honolulu, Hawaii, in November 1984 and organized a second International Workshop that was held in April 1989. Finally, a consortium of Pacific fishing organizations hosted the North Pacific Rim Fisherman's Conference on Marine Debris, in Honolulu, Hawaii, in October 1987.

VI. POSSIBLE SOLUTIONS

30. There are several possible solutions available to countries concerned about the overall problem of entanglement, and most of these solutions demand some degree of international co-operation. While there will be little argument that the discarding of nets, fishing gear, packing bands and other persistent plastic litter is harmful to the ocean environment and legitimate ocean uses, including recreation, there is little agreement about the degree of harm and appropriate steps to solve the problem. It has been argued that the problem may be as pressing as other marine pollution issues, such as oil spills and disposal at sea of chemical and radioactive waste (Laist, 1987). It has also been pointed out that the prevention of introduction of plastic debris into the ocean environment is extremely difficult.

31. Estimates of loss of marine life due to plastic pollution are difficult because there are seldom reports of large concentrations of entangled marine mammals, sea-birds or other marine organisms. Furthermore, the incidents of entangled or disabled animals are likely to be spread over large areas, and the carcasses may decompose or be quickly consumed by predators. By contrast, reports on effects of oil spills or hazardous waste disposal are frequent, and quickly focus public attention.

32. Another difficulty in quantifying the effects of discarded and lost nets and other plastic debris is that the problem varies from area to area. In the North Pacific, large quantities of fishing nets and equipment are found drifting at sea or washed up on beaches (Merrell, 1985; Merrell & Johnson, 1987), while in some area of the North Atlantic, plastic containers discarded from ships and pleasure boats appear to be the main problem (Dixon and Dixon, 1981). A recent study of beach litter along the German Bight showed that fishing gear accounted for 6-11 per cent of the litter collected by volume, while other plastics comprised 12-13 per cent, indicating both types of materials contribute significantly to the problem (IMO, 1986). As noted earlier, debris tends to concentrate around fishing grounds, shipping lanes, and water-mass convergences.

33. There is general agreement that more research and study must be done to define and monitor the magnitude of the problem and to identify solutions. Better estimates are needed of the number of marine animals killed and the impact of this problem on the fishing and shipping industries. More research is also required on the technologies and procedures available for storing and disposing of synthetic and non-synthetic materials at sea, and on the nets and plastic debris themselves, to determine if they can be constructed in a manner less harmful to the environment or can be recovered more readily. Finally, more information is needed on the national and international response to date to ascertain what co-ordinated action is necessary and possible. As mentioned earlier, most international bodies which address ocean and fishery issues are considering

aspects of this problem. It would be most useful if one of these organizations were to co-ordinate international efforts.

34. National and local governments are also becoming more aware of the problem associated with persistent plastics and marine debris. The entry into force of Annex V of the MARPOL, 73/78 Convention will lead to domestic legislation in various countries, such as the U.S. Marine Plastic Pollution Research and Control Act of 1987, passed in December 1987. Other countries have implemented legislation which addresses specific aspects of the marine debris problem. For example, Bulgaria has regulations which prevent disposal of fishing nets at sea (FAO, 1987), Japan has passed legislation which requires marking of high-sea fishery drift nets (Takehama, 1987), and several individual states in the United States require degradable carriers of yokes for beverage containers (Bean, 1987). As a first step, all countries should be urged to ratify Annex V of MARPOL, and to draft necessary domestic implementing legislation.

35. There are also practical actions which can be pursued to alleviate some aspects of the growing problem of marine pollution by persistent plastic debris. These may be undertaken by individual countries, regional arrangements, or international bodies.

36. The most obvious, and probably the most effective, solution to this problem is public education and awareness. Fishermen, vessel and port operators and crew, and the public in general, must be advised of the environmental problems caused by deliberate disposal and accidental loss of plastic marine debris. This should include appropriate surveillance of those potentially responsible for land-based marine pollution, such as plastic manufacturers, sewage plant operators, and land-fill managers. Countries should take all appropriate steps to inform their constituencies of the impact of this type of pollution and urge cooperation in efforts to prevent improper disposal and to help with clean-up efforts. Countries may also wish to consider active educational techniques, such as incorporating information on the problem into the curricula of schools, distributing posters or brochures which illustrate the threat of persistent plastics and encourage proper methods of disposal, holding symposia or workshops to focus attention on the problem, and media campaigns directed towards user groups such as recreationists, fishermen, and merchant seafarers.

37. There are also a number of direct steps that can be taken. Packing bands and beverage container holders or yokes which trap and ensnare marine animals can be cut after use. Manufacturers might consider making a small part of the band or ring easily degradable. All hazardous plastic products could be labelled to indicate their potential damage to the environment and marine life. Fishing net manufacturers might wish to consider making parts of their nets, or the lines connecting them to floats or anchors, more easily degradable, to reduce the damage done by ghost-fishing after the nets are discarded or lost. Geographic areas which are known to cause net entanglements or snags, and thus result in further net loss, can be marked on charts to warn the fishermen. Finally, beach clean-up campaigns, such as those recently conducted in the United States, (Center for Environmental Education, 1987), have the practical impact of removing litter, and also serve to raise public awareness, while compiling data on the magnitude of the problem.

38. Steps which require regulatory or government action may also be considered. In the fisheries area, consideration might be given to developing greater control of the use of fishing nets, such as restrictions on the placement of unattended nets. Fishermen also could be required to tag or mark their nets, to ensure identification of the disposer or loser. Similarly, small transmitters could be attached to the nets. Finally, consideration might be given to establishing a fund to provide a reward or bounty paid for the return or salvage of abandoned nets.

39. Countries may also wish to consider improvements for waste disposal facilities on board vessels and at reception facilities and ports. Mechanical devices such as compactors, comminuters, and incinerators can greatly reduce the amount of waste. The guidelines recently approved by the Marine Environment Protection Committee of the International Maritime Organization for implementing Annex V provide a clear guidance and suggestions for improvements in the area of plastic litter handling and disposal (Marine Environment Protection Committee, 1988). Countries will also need to increase the number and expand the capacity of port reception facilities. Finally, countries may wish to explore greater use of degradable plastic materials and recycling technologies. The technology now exists to use bio-degradable plastic products, which would be broken down by organisms, and photo-degradable plastic products, which would embrittle and break down after exposure to sunlight for a fixed time (Andrady, 1987; Department of Commerce, 1988). Recycling of plastics, especially fish netting, should also be considered. At least one country, Japan, has demonstrated success in one type of plastic recycling (Matsunaga, 1987).

VII. CONCLUSION

40. The ever-increasing amount of persistent plastic and marine debris in the oceans is a worldwide problem, which presents a serious threat to the environment and to marine organisms. The problem will continue until actions are taken to reduce the amount of plastic and debris lost or discarded at sea. The issue needs the increased attention of the plastic industry and all ocean users.

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ANNEX II

EXPLOITATION OF NON-LIVING MARINE RESOURCES: MINERALS OTHER THAN OIL AND GAS

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I. OIL AND GAS

1. Technology employed by the offshore petroleum industry has changed dramatically over the past 20 years, allowing the international petroleum industry to explore and produce in environments that were considered almost prohibitive two decades ago. This technology development which has revolutionized the offshore petroleum business is a result of adaptation, innovation, and integration.

2. Today more than one-quarter of the world oil production is from offshore regions (see Table 1). That portion has been growing at a rate of nearly 10 percent per year for the past decade, and major exploration activities continue off the East, West and Gulf Coasts of the United States; in offshore Alaska; in the Asia-Pacific, especially the China Sea; off Latin America, especially Brazil; in the northern North Sea; and off Canada. Several of these regions could be categorized as hostile environments because of storms, severe waves and currents, deep water, or Arctic or sub-arctic conditions.

3. For example, exploration has been underway for several years under the severe ice conditions of the Beaufort Sea off the United States and Canada; in iceberg conditions along Greenland and eastern Canada; and under severe wind, wave, current, and deepwater conditions along the eastern Canadian and U.S. coasts, in the North Sea, and off southern Australia. Outside the United States, the major offshore production experience in very hostile environments has been in the North Sea. The major offshore exploration experience in hostile waters (without production to date) has been off the coast of Canada.

A. EXPLORATION AND DEVELOPMENT

4. Oil exploration offshore involves basically the carrying out of geophysical surveys followed by drilling. Environmental effects of geophysical surveys are of a minor and transient nature. Deep penetration seismic surveys used to rely on the detonations of explosives with considerable disruption due to the shock wave and the gas bubble discharge. Explosives are now rarely used, and the seismic waves are generated by air guns or electro-mechanical devices which have a considerably lesser effect. Confirmation of the presence of oil, however, requires the drilling of holes. Impacts may therefore be expected from the use of drilling platforms and from the drilling activity itself. Exploration drilling platforms vary in construction, based mostly on water depth and weather conditions. Floating platforms affect the environment, if anchored, through the'

physical impact of the anchors and chain. Dynamically positioned platforms do not impose measurable impacts on the environment. Fixed-leg or jack-up platforms or guyed towers, the latter used for production only, impact also at the points of contact with the bottom, generally fairly small areas. More significant impacts result from the construction of drilling islands or large gravity structures, a method widely used in frontier areas.

1. Island construction

5. In frontier areas, only very large oil fields will be economic and islands are generally of large capacity. They range from massive transient structures such as concrete island drilling systems (CIDS) shown in Figure 1, to islands constructed of gravel dredged nearby; or, in the arctic, constructed of ice, and sprayed into place. Platforms have increased in size as water depths have increased (Figure 2), some being now taller than a 50-storey building, and have employed many innovative designs in frontier areas (Figure 3 and 4). In general, the effect of the platform structure on the environment is not as severe in these areas as the effect of the environment on the structures (Figure 5). In the case of the large gravel islands however, there is a double impact due to the formation of the dredged gravel borrow pit as well as the island itself. Impacts in these cases, which may be severe over several tens of acres, must be examined on a case-by-case basis.

2. Drilling muds

6. During oil- or gas-well drilling, a mixture of fluids, chemicals, clays and other fillers make up the drilling mud which is circulated downhole through the drill bit and back to the surface. It is used for: (1) lubricating and cooling the bit; (2) circulating cuttings to the surface for geological examination; (3) forming a supportive wall or mud casing (generally called a mudcake) to the drill-hole wall; and (4) preventing blow-outs.

7. In cold waters no discernible effects of mud spillage have been reported in studies done in the U.S. However, concern has been expressed for the possible effects of the muds on corals. In the U.S. no drilling is permitted within one mile of coral reefs but reef studies were carried out in the South China Sea by Philippine Cities Service Oil Co. for the U.S. Minerals Management Service. To evaluate the effects on coral growth rate before and after drilling, a series of cores were taken from living *Porites lutea* heads along a transect from the drill hole, and observations were made on the general appearance of branching and foliose corals, algal cover, and the presence of drill cutting in the bottom sediments. Whether drilling muds had actually settled on the bottom to any great degree is not known. No drill mud was present on the surface of the reef or within the reef sand at the time of study (15 months after completion of Well no. 2). If mud did settle on the bottom, it is evident that it rested there only temporarily. Laboratory studies indicated that drill mud diluted by sea water to concentrations of about 10 ppm suspended solids or less has limited effect on coral growth. Plume studies conducted in the field by Shinn and others (1980)

showed that plumes from exploratory drilling were diluted to less than 10 ppm suspended solids within 26 m of the source.

B. PRODUCTION AND ABANDONMENT

8. An important component in an offshore petroleum system is the completion or wellhead unit. These devices sit on the sea floor over the well to prevent blow-outs and control the flow of oil as well as capping the well. Deployment of these devices, in many instances, requires divers or submersible operations. Some wellhead units have the capability for men to work within them; such completion systems are especially important in deep-water work where it is difficult, if not impossible, for divers to reach them from outside. The installation of these devices is another depth-limited factor in exploitation.

9. As an oil field is being developed, adequate transport, storage and movement facilities also must be developed. Usually a field has a series of pipelines along the bottom that will collect the oil and transfer it to a central platform or loading area where it can then be picked up by tankers. Alternatively, if land is nearby and there are refinery or storage capabilities, a pipeline to land may be installed. Pipelines may be buried in the sea bed to prevent damage by ships or fishermen. A good knowledge of the sedimentary and environmental characteristics of the region is necessary for the safe installation of pipelines. One shortcoming of pipelines is the difficulty and expense in laying them in deep water.

10. Transfer of oil to tankers is usually done from some type of mooring buoy that takes the oil from a production platform or special tanks. In many instances, the ship being loaded has a deep draft and thus loading has to be done offshore. In some areas deep-water mooring facilities are used to transport oil from tankers to land facilities (Figure 6). They are very effective since the tanker is moored to the buoy at one point and can freely rotate with the winds or currents. Large storage tanks can also be used and are fairly common in the Persian Gulf and North Sea. Some production platforms have storage capacities of over half a million barrels of oil. New, innovative storage systems include large bladders made of synthetic material that can be towed as well as being used for storage.

11. Numerous environmental concerns are associated with offshore oil production including: oil spills; effluent discharges; effects of natural or man-made hazards; and ecological damage. The petroleum deposits themselves are under considerable pressure. If there is a failure in this system, the oil or gas can escape or blow out and may catch fire. Relief may be achieved by drilling side wells to reduce the pressure and stop the fire. During blow-outs large amounts of oil and gas can escape; two of the most spectacular blow-outs were the Ekofisk in the North Sea in April 1977 and the IXTOC in the Gulf of Mexico.

12. A major problem in any marine operation is storms and high waves. Winds of 300 km h^{-1} or more can occur during hurricanes and waves can exceed 20 m. Such conditions can subject any offshore operations to critical stresses. Most platforms are designed to survive such conditions, but conscientious operators will remove personnel and shut down operations when and if a storm is even near their region. An accurate satellite weather system is especially valuable for offshore operations.

13. Other problems associated with platforms are scour of sediment around the legs of jack-up or production platforms, and therefore their possible collapse, and the growth of algae on their legs, which can increase drag. In cold areas, metal fatigue and ice can cause hazardous conditions for drilling platforms. One method used to drill in areas of extreme cold is to build artificial islands then drill from the islands, using them as platforms. Such operations are commonly used in the Beaufort Sea.

1. Spills

14. Technology and techniques for oil spill containment and clean up are an important environmental consideration. While the oil and gas industry is genuinely concerned with preventing oil spills, the industry's capability to contain and clean-up spilled oil in hostile environments has not been proven under actual conditions. Although some deep-water oil spills may occur as the oil and gas industry moves further offshore, and although current capability to clean-up such spills is limited, the equipment and methods for combating deep-water spills are essentially no different from those used for nearshore areas. Most deep-water spills are likely to be of less concern than shallow water, near-shore spills because: 1) they generally occur in less biologically sensitive areas; 2) natural processes may often work to dissipate and degrade deep-water spills before significant damage can be done; and 3) greater distance from shore allows more lead time in which to consider what (if anything) is to be done.

2. Effluent discharge

15. The discharge of contaminated effluents such as oil fluids, sewage and deck drainage from platforms is strictly controlled in the U.S.A. The problem is basically the same as that for the control of effluent discharges from ships.

3. Platform removal and abandonment

16. Abandonment of platforms generally requires their removal and this can be a problem. Fixed-leg platforms may be cut off below the sea bed by explosives and removed for scrap. The operation is expensive and some platforms have been tipped over to use as artificial reefs. Strict

control is required so that abandoned platforms do not become navigational hazards.

C. ENVIRONMENTAL IMPACT CONCERNS

17. The development of offshore oil and gas resources and protection of the environment are potentially conflicting objectives and the subject of continuing debate. Major environmental considerations relate to the status of marine mammals, and the adequacy of oil spill containment and clean-up techniques. Many species of fish, marine mammals, including whales, and birds may be affected by oil and gas development. Although the risk of catastrophic oil spills from offshore operation is believed to be low, effective containment and clean-up measures are essential in light of the potential harmful effects of any such spill. Industry has invested large amounts of funds and effort in engineering technology to prevent blow-outs and other catastrophic rig accidents and some claim that there is little market incentive for developing oil-spill countermeasures compared to spill avoidance. For the most part, oil-spill containment and clean-up technology has been developed for spills in nearshore and temperate regions. In deep-water areas, high-sea states may be encountered, and greater distances from shore may create logistical problems for clean-up of oil spills. To date, it has not been demonstrated in a real situation that industry will be able to use effectively the existing oil-spill equipment and countermeasure strategies in hostile environments.

18. With regard to the effects of drilling muds on corals, the Philippine study showed that, although the coral growth rate data did not statistically show detrimental effects of drilling within 20 m, other effects, such as dead branching corals and rusty brown staining, were evident and related to the drilling, although exactly how drilling caused these effects is not understood completely. In addition to the effect of a temporary cutting pile, shading from the 60-m-long drilling vessel may have affected life in the shaded zone. Similar brown staining and lack of branching corals beneath 100-y-old lighthouses is commonly observed along the Florida reef tract.

19. All of the above concerns are addressed in the U.S. by the Department of Interior, through its Minerals Management Service prior to leasing. Table 2 shows the normal procedures for environmental impact mitigation which are built in to the pre-leasing process under the U.S. Outer Continental Shelf Lands Act.

II. MINERAL DEPOSITS OTHER THAN OIL AND GAS

A. THE MINERAL DEPOSITS OF THE OCEANS

20. Mineral deposits in the oceans can be characterized either as unconsolidated, capable of being collected directly by dredging, or consolidated, requiring additional energy to fragment the deposit before the collection (Cruickshank, 1962; Table 3). Either type may occur at or beneath the sea floor.

1. Unconsolidated deposits

21. Unconsolidated deposits include construction materials such as sand, gravel, and shells; heavy mineral placers, containing materials such as titanium, tin, and gold; metalliferous muds such as those under development in the Red Sea; manganese nodules; and oozes of silica and calcium carbonate.

2. Consolidated deposits

22. Consolidated deposits include bedded deposits, such as coal and iron ore; crusts, such as the cobalt-rich manganese oxides found on Pacific Ocean sea mounts, massive sulfide deposits in the form of mounds and stacks occurring at certain spreading centers; and essentially tabular veins or mineralized channels in consolidated host rocks.

3. Fluids

23. Fluids may be considered special cases. They are represented by dissolved salts traditionally recovered by evaporation ponds; slurries of fine grained, loosely consolidated materials; and hydrothermal solutions.

24. Most of the known deposits that could be economically mined are comprised of unconsolidated or weakly consolidated materials. Possible exceptions are the consolidated, cobalt rich, ferromanganese crusts and polymetallic sulfides considered for leasing now in the United States. Consolidated deposits of phosphorite may also be of interest in the future, but potential markets exist now for unconsolidated deposits of sand and gravel and heavy mineral placers (Bureau of

Mines, 1987a and 1987b).

25. Most economically recoverable deposits are found on the continental shelves at depths of 200 m or less. Individual mines in the U.S.A. are expected to cover between 8-100 km^2 or less than 0.01 percent each of the roughly 1,500,000 km^2 of continental shelf. Mines would last 20 years or more. The most extensive operations in U.S. waters probably would involve manganese crust mining. In the near term, such a mine would be in water depths of about 1,000-2,000 m and could require 500-600 km^2 for a 20-year operation. However, one such mine could provide half the nation's demand for cobalt and substantial amounts of other strategic metals.

B. MINING METHODS FOR OCEAN MINERALS

26. There are four basic methods of mining solid minerals: scraping the surface, excavating a pit or trench, removal through a borehole in the form of a slurry or fluid, and tunneling into the deposit (Figure 7). All deposits on land are mined by one or more adaptations of these methods, and offshore coastal shelf mining is amenable to the same basic approaches (Table 4). Each mining method has variations that may be tailored to a specific situation, and most of the deposit types can be mined by more than one method. Similarly, any one method can be applied to more than one deposit type (Cruickshank, 1978).

27. Marine mineral deposits, whether consolidated or unconsolidated, may occur at or beneath the sea bed. Deposits at or near the surface of the sea bed can be gathered by mechanical devices that scrape the sea bed and gather the ore for lifting to the surface or for other treatment. In its simplest form, the action is like raking or shoveling or, in some cases, vacuum cleaning. Where a thick layer of hard material is present, the scraping action may be preceded by ripping or cutting. Mineral deposits lying wholly or partially beneath the sea floor may be removed by excavation. The applicable mining methods range from those designed to recover free-flowing materials to those for excavating solid rock. Certain types of deposits of solid rock may be mined by borehole methods, whereby fluid is added and ore is transformed into a fluid or slurry and removed by pumping, or, in other cases, a mineral may be selectively leached and recovered in solution from beneath the sea-bed. Deposits buried too deeply to mine from the sea bed may also be mined by conventional underground mining methods using shafts and adits (tunnels that lead into mines) for access and ventilation, either from shore or from natural or artificial islands. The sea-bed location of these mines only slightly adds to the conventional problems of access, safe overhead cover, and ventilation. The effect on the environment is much the same as an onshore mine.

28. This section further explains each of the mining systems listed on Table 4, describes the manner in which each may disturb the marine environment, and notes types of deposits with which

it is normally used. For further reading, references are provided to literature describing commercial operations using these mining systems (Cruickshank *et al.*, 1968; Cruickshank and Marsden, 1973 and Cruickshank, 1973). An estimate of the time frame in which marine mining could take place in the U.S.A. is given in Table 3a.

1. Scraping

29. Although there are many variations, there are basically four mining systems that may be used to scrape the surface of the sea floor. Two are wholly mechanical and two involve hydraulic action to lift the mined rock.

(a) Drag-line dredge

30. This system is used in offshore mining and deep seabed sampling, as well as in construction. Its use has been advocated for the recovery of sea-floor nodules and slabs of phosphorites (Mero, 1965). The material would be recovered by large dredge buckets that scrape slabs and nodules from the surface of the deposit and feed them into barges for transportation to shore. Annual production at such mine for phosphorite could be of the order of 400,000 tonnes (roughly 250,000 m^3 of ore). This rate of production could involve 40 m^3 buckets scraping an average of 20 tonnes of phosphorite from the seafloor every 20 minutes in a water depth of about 200 meters.

31. At an average abundance of 100 kg m^{-2}, nearly 4 km^2 of sea floor would be mined annually. At a mine life of 20 years, a total of 80 km^2 would be affected if the deposits were confined to the surface of the sea-bed. Thicker deposits would allow smaller areas of disturbance.

32. Assuming the presence of some fine-grained sediment on the sea floor, as well as some breaking up of the phosphorite, a benthic turbidity plume would be created by the scraping action. Some fine material would inadvertently be introduced into the dredge buckets and washed out either en route to the surface or at the surface as a result of washing of the phosphorite in the barge. The fine sediments would eventually rain to the sea floor, creating a blanket of fines.

(b) Trailing suction dredge

33. A suction hopper dredge uses a pump to draw a slurry of bottom water and sea floor sediment into a riser or pipe leading to the mining vessel. As the sediment accumulates in the hopper, the water weirs over-board. This system is used primarily for maintaining harbour channels. However, it is also used extensively for mining sand and gravel in water depths up to 40 m in the North and Baltic Seas (Padan, 1983), and new vessels, such as the ARCO Avon, launched in 1986, are designed to extend mining capability to depths of 50 m (Drinnan and Bliss, 1986).

As its name implies, this type of dredge mines, while in motion, creates numerous shallow trenches in the sea floor commonly about 1 m wide and 0.3 m deep.

34. The suction dredge uses one of several types of drag head with a coarse-grid steel framework across the opening of the suction head to prevent large rocks from entering the suction pipe. Coarser particle sizes are screened out and rejected after passing through the pump. Fine materials are washed overboard with the slurry overflow. In some cases, vibrating screens allow part of the sand fraction to be dumped back into the ocean, since the ratio of sand to gravel mined may differ from the desired marketable mix.

35. Environmental disturbances that may result from this system include the extraction of trenches in the sea floor, creation of a turbidity plume by washing overboard clay particles taken up in the dredge pipe, and production of a blanket of fines covering the sea floor down-current from the dredge. However, measures have been developed in Japan and Great Britain to either avoid or mitigate these problems. The use of sand and gravel as construction material, for example, requires that stringent measures be taken to avoid mining any clay layers since the entire shipload would be considered contaminated and unmarketable. Regulatory options include mandating bottom discharge of overflow waters to reduce the size of trenches or other excavations. Further details on environmental effects and the technical and administrative means the British have used to control them are given by Drinnan and Bliss (1986) and Pasho (1986). Additional detail on the equipment used and extensive photographic documentation are given by Hess (1971).

36. The volume of material that may be mined includes the sum of the marketable material, the fraction to be rejected, and the overburden initially stripped away. For example, in moving 1 million tonnes of product to shore annually, an equal amount of sand might be rejected to reduce the sand/gravel ratio from 70/30 to a more marketable 40/60 (NAS, 1975). The 2 million tonnes of annual excavation would result in a gradual lowering of the ocean floor by about 60 cm over an area of 2.5 km^2. As mining proceeds, the rejected sand released above the mine area would partially smooth the bottom contours.

37. Some silt and clay will typically be mixed in the mined material despite avoidance of distinct clay layers. Discharge of these fine materials, suspended in the sea water overflowing from the hopper, would cause turbidity plumes at the depth of discharge. The daily increment of new plume in this example would consist of 60,000 tonnes of water and about 200 tonnes of material finer than 74 microns (200 mesh). Because the fines will stay suspended for days, the impact of daily activity of surface discharges could be considered cumulative. The blanket of fines would settle gradually to the sea floor as it travels with the currents, building up a very thin veneer over a large area. Discharge near the sea floor would result in a thicker layer over a smaller area.

38. Various methods exist to control turbidity and others are being developed. Existing methods usually rely on controlling the depth and direction of discharge but the Marine Mining Panel of the U.S. - Japan Cooperative Program in Natural Resources recently tested a technique to control the formation of bubbles in the discharge streams to reduce the settling time for

particulates (Marine Mining Panel, 1984). This Japanese-developed Anti-Turbidity Overflow System (ATOS) was tested at a typical dredge site off the coast of Japan to determine its potential for control of both surface turbidity and sediment dispersion. The system appears to work well in course-grained sediment. Data from test in fine-grained sediment are still being evaluated.

39. Trailing suction dredges, if modified for great depths, can also be used to mine deep-sea manganese nodules. Operating characteristics and environmental data for this application are well documented (NOAA, 1981).

(c) Crust miner

40. Recent interest in manganese oxide crusts containing relatively high values of cobalt, and in some cases platinum, has led to proposals to develop the deposits. The manganese crusts vary in thickness from mere stains to about 40 cm thick and cover a variety of substrate rocks ranging from hard basalt to weak hyaloclastite. The physical properties of the crusts are similar to a hard coal. The crusts occur extensively on the Pacific seamounts and submarine ridges at depths between 800 and 2,400 m. The mining system primarily proposed for this work is a vessel equipped with a hydraulic lift system with an active bottom miner (Halkyard and Felix, 1987). The miner would be a self-propelled tractor, controlled from the surface vessel. It would be capable of breaking and removing the thin crust from the underlying rock and feeding it to the hydraulic lift system through a hydrocyclone to separate entrapped substance. The roughly cleaned ore would be pumped to the surface vessel for further cleaning and transport to shore.

41. Potential environmental effects, as discussed in the EIS for the proposed Hawaii-Johnson Island lease sale (DOI and HI, 1987), are expected to be much the same as the effects of the drag-line dredge.

(d) Continuous line bucket dredge

42. This system has been tested for possible future use in recovering manganese nodules (Masuda *et al.*, 1972) and is described in NOAA's EIS for deep seabed mining (NOAA, 1981). The continuous line bucket (CLB) may also be used for mining phosphorite nodules and slabs, and its use has also been proposed for cobalt crust mining.

43. The CLB's environmental disturbances would be similar to those of the drag-line dredge, except that the latter would involve discrete episodes of bottom disturbances and the CLB system would be continuous.

2. *Excavating*

44. Mineral deposits that are located mostly within the sea bed may be removed by excavation. These deposits include thick deposits of sands; metalliferous muds; layered or disseminated deposits of unconsolidated placer minerals or overlying bedrock; and deposits of consolidated minerals in vein, tabular, or massive form, which may extend for considerable distances into the bedrock. The mining system used will depend largely on the ease with which the material may be excavated and removed from its surrounding environment, on the water depth, and on the climate in the area of operations (Cruickshank *et al.*, 1969; Cruickshank, 1987). Six examples of seabed mining operations are presented here that range from the excavation of free flowing materials to the excavation of hard rock.

(a) Clamshell bucket

45. Clamshell buckets have been used to mine sand and gravel in Japan and tin in Thailand; and to sample phosphorite off New Zealand. The buckets are mechanically actuated to bite into the sea bed and remove material. The need for multiple cables to actuate the grabs can cause complications, particularly in heavy seas where wave-compensating devices may also be needed. Moreover, the clamshell is inefficient in clearing bedrock of fine materials. It is best suited for excavation of large-size granular material where accuracy of positioning is not important. The size of buckets may range from about 0.1 m^3 to as much as 7.6 m^3.

46. The action of the grab on the bottom stirs up any fine materials present and these also tend to wash out during the lift to the surface. Where wash-out is of major significance, it may be prevented or reduced by placing a canvas or plastic hood over the bucket.

(b) Bucket-ladder dredge

47. The bucket-ladder dredge is most efficient for excavation of deposits containing boulders, clay, and/or tree stumps and weathered bedrock. Dredges of this type have been used successfully all over the world for mining gold, tin and platinum placers and diamonds, although their use offshore has been limited to gold and tin. They are frequently used for clearing harbours because of their capability for digging into broken rock and coral. The bucket ladder delivers a virtually water-free product to the mineral dressing plant on board the dredge. Discharge of water from the shipboard operations is limited to that needed to concentrate the valuable constituents by techniques based on the use of flowing water to remove the less dense materials. In case of gold, the bulk of concentrate recovered is only a few parts per million so that virtually all the material removed from the deposit is returned to the sea-bed. Considerable turbulence accompanies these operations, which may involve up to 5 million m^3 of material per year. Bucket-ladder dredges are limited to depths of about 50 m and rarely operate at depths over 20 m. Few of these

dredges have been built for offshore mining in the last 30 years, and it is likely that they will be superseded by the bucket-wheel suction dredge.

(c) Bucket-wheel suction dredge

48. Bucket-wheel dredges use a small-diameter bucket wheel mounted on the suction ladder to excavate material. This combines the best aspects of the bucket ladder and suction dredges. Very high torque or digging power can be applied to the wheel, which can deliver the excavated material directly into the mouth of the suction pipe for transport to the sea surface. Digging capability is equal to the bucket ladder with respect to ease of digging and bucket capacity, while the depth capability is greatly increased (Anonymous, 1984).

49. The combination of simultaneous digging and suction at the sea floor reduces bottom turbulence and provides the option to either treat the ore on the vessel or pipe it to shore. Other disturbances would be similar to those from other suction dredges.

(d) Anchored suction dredge

50. Anchored suction dredges are widely used in Japan for mining sand and gravel at depths less than 30 m. These dredges have been used in Britain as well, although the vessels built since 1980 are virtually all trailing suction dredges (Drinnan and Bliss, 1986).

51. In contrast to the trenches left by trailing suction dredges, anchored suction dredges leave pits in the ocean floor. These tend to fill much more slowly than the trenches, and are a greater source of problems for bottom trawlers. However, since fish often aggregate at scraps and other irregularities on otherwise flat bottoms, isolated pits may be considered assets by recreational fishermen. The rate at which the pits fill is a function of the size of the hole, the type of material remaining, and the currents. Holes in gravel, for example, are estimated to take 25 years to fill, while trenches in sand may fill in a matter of hours or months (Drinnan and Bliss, 1986).

52. Anchored suction dredges have also been tested for mining metalliferous muds at depths of 2,000 m in the Red Sea. This deposit consists of about 700 million tonnes of zinc, copper, and silver-rich muds in a small, enclosed basin. The mud has the consistency of shoe polish and is about 10 m thick at the mine site. Mining involved the conversion of a deep drilling oil exploration vessel (the SEDCO 445) to carry a specially designed mud pump and delivery pipe that was lowered to the sea bed in a manner similar to the lowering of a drill pipe. The pump was vibrated into the mud, with a water jet to fluidize the material, and the ore pumped to the ship for treatment. Approximately 15,000 m^3 of muds were retrieved from four separate sites over a period of about six weeks. The muds were de-watered and subjected to froth flotation on board the vessel. The effluent from both operations was discharged through a pipe at a water depth of 400 m. All op-

erations were fully monitored to observe effects on the water column and living organisms.

53. In their report of the experiments, Mustafa and Amman (1981) observed build-up of a highly diluted diffusion cloud at about 900 m. They concluded that dilution would be so extensive that a concentrated pile of tailings on the deposit would not occur, especially if a disposal depth of 500-700 m were chosen. Commercial mining operations in the Red Sea using the anchored suction dredge may involve the removal of as much as 3.5 million tonnes of material per year of which over 90 per cent would be returned to the sea. Environmental effects are further discussed in Thiel *et al.* (1986).

(e) Cutterhead suction dredge

54. Cutterhead suction dredges are typically used to excavate fairly compacted, granular materials in water less than 30 m deep. The rotating cutterhead is usually an open basket with hardened teeth or cutting edges somewhat like an oversized dentist's drill. The end of the suction pipe is normally located within the basket. In standard practice, the dredge is swung back and forth in an arc pivoted from a large post or spud attached to the stern. The dredge cutterhead cuts downward a short distance with each swing. Because the cutter rotates in one direction only, the bite is much stronger on one swing than the other. In mining for heavy minerals, the action of the cutter tends to disintegrate the material, allowing heavy minerals to separate, fall below the cut, and be left on the sea floor. Cutterhead suction dredges have never been used successfully for mining gold, although they have been widely used for mining cassiterite (tin placers) in west Thailand, where the deposits are rich enough to economically sustain the inefficient clean-up.

55. Suction dredges circulate large quantities of slurry that must be decanted on board the dredge or pumped ashore by pipeline. In either case, there is a significant discharge of water containing fine particulate materials. Treatment of the decanted solids may be unnecessary for construction sand and gravel, but may be required for heavy minerals. The valuable constituent or concentrate from these ores will rarely amount to more than a few per cent of the materials mined. Therefore, as much as 95 per cent of the material dredged from such placers must be disposed of at either the mine or the shoreside treatment site.

56. The cutterhead suction dredge may also be equipped with a multiblade ripper to cut into moderately consolidated rock. Present use is limited to the excavation of soft rock, such as coal and shale. However, advances in rapid tunneling technology suggest that rock cutter heads could be designed for medium strength rocks, such as sandstone and limestone (Hignett and Banks, 1984).

(f) Drilling and blasting

57. Deposits that are too hard to excavate by dredging must be broken by other means. The normal system for excavating hard deposits is to drill into the deposit and blast with explosives. Fracturing using highly pressured fluids is also possible, but would represent very special and rare cases, and are not considered here. Blasting of the material is only an intermediate step, and would be followed by the gathering and lifting of the ore by one of the methods previously described. Blasting operations are designed to expend as much force as possible on fragmenting the ore, and the water column effects are much lower than those from equivalent, unconfined explosions.

58. Drilling and blasting for offshore mining is rare, but may be exemplified by the Castle Island Mine operations in the Gastineau Channel in southeast Alaska. That deposit was a massive vein structure of barite which outcropped on the island. Mining involved deepening the mine pit to as much as 30 m under water using conventional drilling and blasting, followed by excavation of the broken ore with a barge-mounted clamshell dredge. Blasting took place every few weeks at the most. Localized fish kills were reported. Traces of fine barite were noted in the bottom sediments as much as 1 km downcurrent from the operations, but no indication of any effect on the marine organisms was apparent (Thomson and Smith, 1978). With respect to the possibility of mining deep sea-bed deposits that require fragmentation, many more aspects need to be examined. Technically acceptable means of drilling and fragmenting hard rock in deep water have not yet been developed. However, methods of gathering and lifting the fragmented material may be assumed to be similar to the methods developed for deep sea-bed nodule mining.

3. Borehole mining

(a) Fluidizing (slurrying)

59. Under proper conditions, certain types of unconsolidated or marginally consolidated mineral deposits may be mined as a slurry pumped through a drill hole penetrating the sea floor. Sub-sea-bed sand was mined in this way in shallow waters offshore Japan in 1974 (Padan, 1983), and recent onshore experiments in Florida proved the capability of this approach in recovering phosphate from beneath thick overburden (Savanick, 1985). In this instance, a borehole was drilled from the land surface to the base of the ore body, then a water-jet cutting system inserted through the borehole was used to fragment the loosely consolidated phosphate. At the same time, a downhole slurry pumping system recovered the phosphate through the borehole, thereby creating a water-filled cavity about 6 m in radius. After mining was completed, the cavities were backfilled with sand to prevent ground subsidence. A U.S. Bureau of Mines study found this system to be cheaper than conventional land mining systems if the overburden is at least 50 m feet thick (Hrabik, and Godesky, 1985). In the recovery of sulphur, superheated water is pumped into the deposit to

melt the sulphur so that it may be pumped out of the ground.

60. Environmental effects are primarily associated with the installation and movement of mobile drilling platforms, with the disposal of drill cuttings and any waste rock that cannot be returned to subsurface, and with the release of any slurry or processing waters that are not recycled. Recycling of slurry waters is anticipated to occur to a substantial degree.

(b) Fluidizing (solution mining)

61. Hard-rock deposits that are amenable to hydrometallurgical treatment of their ores are potentially extractable by fluidizing methods. The valuable constituent is dissolved in place and the solution is removed through a borehole. For complex ores, there are major problems in dealing with toxic or corrosive solvents used to enlarge fractures to provide a flow path for the solvent through the deposit, and to selectively extract the desired metals. These problems are being overcome on land, however, and the methods developed there should be applicable in more sophisticated form to sea-bed deposits. Environmental effects should resemble those of slurry mining during normal operations.

62. Effects of accidental spills of solvents would depend on the nature of the solvents and the sites impacted, but should be localized so long as the solvent is water soluble, due to the rapid dilution prevailing in the oceans and the buffering capacity of sea water.

4. Tunneling

63. Underground mining by tunneling is commonly practiced in sub-surface hard rock. In certain cases, sub-sea-bed deposits of bedded coal, potash and ironstone, as well as veins of lead, copper and tin have been mined by conventional underground methods. Entry to these mines is either from the shore or from natural or artificial islands in shallow waters. The location of the mines in the sea-bed only slightly adds to conventional problems of access, safe overhead cover, and ventilation. The effect on the environment is similar to that for any shoreside mine. The possibility of developing underground access through sea bed air locks has been considered for special cases but is not considered further here.

C. NORMAL OPERATING ENVIRONMENTAL DISTURBANCES

64. Although there are many types of marine mineral deposits, mining methods, and environments in which mining may occur, mining operations will affect the marine environment in only a few ways, the principal ones being those shown in Table 5. These forms of disturbance are not new. Individually or in combination they result from trawling operations, harbour dredging, military manoeuvres, construction at sea and, in some cases, natural causes. However, mining will create new sources and new combinations of these disturbances, and the locations, durations, and intensities of these effects may be new.

1. Sea-bed

65. On the sea-bed, one can routinely expect disturbance from ore collection. The sea floor in the path of the collection machinery will be raked, fragmented, suctioned up, or compacted as the ore is gathered. Should fragmentation by explosives be required, additional disturbances would be introduced into the marine environment by shock waves. Noise can be assumed to be a by-product of mining operations. The attenuation of the noise will depend on its frequency spectrum as well as the properties of the water masses through which it moves. In some forms of mining, the area near the mining machinery will be illuminated. Movement of overburden or excavation of ore may create trenches or pits in, or mounds on, the sea floor. These mounds and depressions have implications for potential use conflicts and other environmental impact concerns that may persist after mining ceases.

66. Sub-surface mines and slurry mining cavities have the potential of causing sea floor subsidence or collapse, but these phenomena should be avoidable by proper mine design. In the case of borehole mining, involving the removal of ore in a slurry, the resulting cavity can be backfilled with waste rock as in subsurface mines before the site is abandoned.

2. Water column

67. Sea-bed mining creates a turbidity plume at the sea floor to the extent that fine material is present above or in the deposit, or is created by fragmenting or grinding rock as part of mining. The largest sea-floor turbidity plumes may be created by mining systems designed to reject fine material during pick-up of the ore.

68. Surface turbidity plumes will be created by those systems designed to pump ore in a slurry to a surface vessel and then allow the excess water to overflow to the sea. The size of the plume will vary in proportion to the amount of fine material unavoidably recovered with ore. The fines will tend to remain in suspension in the mining vessel and be discharged with the slurry water. Water-column environmental concerns generally focus either on these suspended particulates or

on dissolved substances. Turbidity plumes move away horizontally from their sources with current and are rapidly diluted, but the existence of plumes can often be detected far from the mine site.

69. The fine particulate material suspended in plumes eventually rains to the sea floor, resulting in a thin layer of fine material over a large area. The area affected can be minimized by sub-surface discharge or other techniques that force the particulate material to settle closer to its source, resulting in a heavier accumulation over that smaller area.

D. ENVIRONMENTAL IMPACT CONCERNS

70. The environmental impacts of marine mining encompass both the effects of mining itself, as depicted in Table 5 (Cruickshank *et al.*, 1987), and the effects of transport, beneficiation and refining of the ore mined. These latter, nearshore and onshore, sources of impact will often be the more significant sources of impact, but they are beyond the scope of this document with its focus on impacts in the sea.

1. Near-field effects

71. The effects of fragmentation, collection, excavation, and subsidence on organisms can be characterized as near field, that is, they are essentially restricted to the mine site, although exceptions are possible. Species extinctions, for example, seem umlikely, but significant impacts on regional populations of some species could result if breeding grounds were mined. The potential for significant impacts is carefully considered during the preparation of environmental reviews for mining proposals, and should be avoidable.

2. Far-field effects

72. Effects of the plumes and sedimentation - unlike the effects of fragmentation/collection, excavation, and subsidence - can be characterized as far field since they may be felt well beyond the mine site. The probability and severity of resultant biological consequences depend on the characteristics of the specific mining operation, the geologic setting and the geographical location of the mining activity. It is possible, however, in spite of such uncertainty, to use the data base created by past projects to identify those far-field effects that are more likely to be of concern. These include NOAA's Deep Ocean Mining Environmental Study (DOMES), NOAA's New England Offshore Environmental Study (NOMES), the Corps of Engineers' 5-year Dredge Material Research Program, studies of offshore sand-and gravel-mining operations in the United Kingdom, and some basic research studies of factors affecting impact and recovery from specific disturbances.

73. The sensitivity and natural resiliency of the benthic community is important in determining the recovery of the affected ecosystems (ICES, 1975). Some species are likely to be more affected than others because of feeding mode (e.g., filter feeders), life habit (e.g., surface dwellers), degree of mobility (e.g., tube dwellers), or sensitivity of life stage (e.g., larvae).

74. Those organisms living in an environment with episodes of high turbidity and sedimentation are likely to withstand some disturbance. For example, oysters, which are filter feeders, are able to endure some increases in sedimentation (Macklin, 1961; Dunnington, 1968; Loosanoff, 1962), and benthic deposit feeders can burrow out from some increase in deposition (Nichols *et al.*, 1978; Hirsch *et al.*, 1978; Maurer *et al.*, 1978). However, for both deposit and filter feeders, there are redeposition thicknesses and rates beyond which the animals cannot survive.

75. Several general, and numerous specific, factors have been found to be critical in determining the rate at which a disturbed area is recolonized by species that were previous residents. A mobile adult stage allows faster recolonization of large disturbed areas, especially if the larval stage is non-mobile (Levin, 1984; Laurer and Simon, 1976). The season, duration, and areal extent of the disturbance can be critical, if the disturbance occurs during a period when stability of the sea floor is important to the survival.

3. Physical impact concerns

76. Physical impacts of concern will generally be those that impact marine organisms, fisheries, other commercial uses, or defense operations. Nearer shore, in territorial waters, potential impacts on coastal erosion will require careful appraisal. The major concerns regarding purely physical impacts will be space use conflicts and poor housekeeping. Fouling of nets on discarded mining gear, displacement of individual fishermen, unannounced activity by miners, and fouling of gear in holes left by mining have generally been the major concerns in Britain (Drinnan and Bliss, 1986). However, these concerns have administrative and technological solutions.

77. Ore collection will invariably be a source of physical impact. The sea bed in the path of the collection mechanism will be raked, broken, and/or compacted as the ore is gathered. If the ore needs to be fragmented by explosives, shock waves will create additional disturbances. In some forms of mining, the area near the mining machinery will be illuminated. Usually the effects of fragmentation and collection will be essentially limited to the mine site and the period of mining. Other effects such as the creation of trenches, pits or mounds on the sea floor may affect marine organisms and fishery operations for years.

78. Alternatively, boulders may be uncovered, forming permanent obstructions which snag fishing trawls. In this respect the French have experimented with abutting dredging tracks to obtain a flat sea bed. Also, in some cases, rejected materials can be guided back into the collector tracks immediately behind the mining machines. Use of trailing suction dredges, which interfere much less with bottom fisheries than do anchored suction dredges, reduces conflicts with com-

mercial fisheries (Drinnan and Bliss, 1986).

79. Large excavations can also lead to coastal erosion if the wave patterns and sediment movements near shore are changed sufficiently. These effects are being studied by the U.S. Army Corps of Engineers, but an interim guide to safe practices can be obtained from British experience. In their coastal zones, dredging seemingly causes no problems in water depths greater than half the normal wave length, or more than one fifth the length of extreme waves. Dredging in waters over 20 m (about 60 feet) deep is usually approved with only a desk review. Proposals for dredging in 10-20 m get more detailed review and may require site specific information. Proposals for dredging in waters shallower than 10 m may require substantial study (Drinnan and Bliss, 1986).

80. Both the scraping and excavating approaches to mining can also change the character of the substrate by exposing materials with different properties. For example, if exposure of silts and clays is a concern, it may be necessary to require that the bottom portion of the layer being mined be left in place to minimize such change. However, for sand and gravel mining and for some other minerals, the economics of mining may provide strong incentives not to expose silts or clays. If so, such regulations may be unnecessary.

81. Subsurface mines and slurry mining cavities may cause local sea floor collapse or subsidence, possibly leading to coastal erosion or changes in sediment patterns, but this phenomenon can be avoided by proper mine design. In the case of borehole mining, cavities are likely to be waterfilled during the mining and waste can be injected into the cavity before the site is abandoned. Both practices reduce the potential for collapse of the rock overlying the cavity, and reduce the volume of waste discharged at the surface.

82. Sea-bed mining will create turbidity plumes at the sea floor as fine materials are created during mining or are resuspended. The properties of the plume, particularly settling times of the particles, will depend on their size distributions, specific gravities and concentrations, which are also the properties of greatest biological significance. Size and specific gravity of the particles determine residence time in the water column and influence the potential for resuspension. The largest of these plumes may be created by mining systems designed to reject fine material during pickup of the ore. The chemical characteristics of the smaller, slower settling particles in these plumes generally will be similar to existing suspended sediment.

83. However, they may differ if mining exposes an anoxic layer, because the solubility of adsorbed metals can differ greatly between reducing and oxidizing environments. The effects of such differences are likely to be local and brief due to rapid dilution by oxygenated waters. Rock fragments created by mining are likely to consist of relatively inert material and are less likely than resuspended sediments to be a source of dissolved metals.

84. Surface turbidity plumes are created as wastes are discharged, and will vary in proportion to the amount of fine material discharged and the disposal technique. The fines will tend to remain in suspension on board the mining vessel and be discharged with the slurry water, and, in a con-

tinuous mining operation, a continually renewed plume is present. Sediment concentration in any one parcel of water decreases as the water mass moves away from the point of discharge, but each such parcel is replaced by another, and a steady state evolves in which high concentrations are always found near the discharge point even though the concentrations in each parcel are continually decreasing.

85. The fine particulate material suspended in plumes eventually falls to the sea floor, forming a thin layer over a large area. The area affected can be reduced by sub-surface discharge or other techniques that force the particulate material to settle close to its source, but this creates a heavier accumulation over that smaller area. The appropriate sedimentation pattern, and thus the appropriate disposal methods, will depend upon the characteristics of the site, including its ecosystems.

86. Field studies and modeling results indicate that plumes, both surface and benthic, can be detected over distances of the order of kilometres, and, in certain cases tens of kilometres. Plumes from coastal dredging typically are visible for only 1-5 km but may be visible up to 20 km (DOI, 1974). In the clearer oceanic waters beyond the continental shelf, the plumes can be measured up to 120 km from the point of discharge. However, detection of plumes at these distances requires the measurement of materials which are prominent in the discharge from the mining vessel, but are rare in natural waters. Manganese in the discharges of manganese-nodule mines is an example (Lavelle *et al.*, 1981). Reduction of light levels sufficient to measurably reduce the rate of photosynthesis may be present for 30 to 50 km downcurrent from such vessels (Chan and Anderson), 1981).

87. Measurable effects of sedimentation will be much more localized than those of plumes. Lavelle *et al.* (1981), for example, found that 90 percent of the sediments suspended by manganese-nodule mining were redeposited within 70 m, even though the mine site was covered with clays, which would be expected to form a plume of small-sized slowly-settling particles. Thickness of the sediment layers resulting from mining were less than 1-5 mm at 200 m from the disturbed area, about 2-4 mm at 50 m, and 8 mm at 25 m. Thickness of resedimentation may be considerably higher in some mining operations, but the pattern of rapid decline in sedimentation as distance from the source increases will hold. In addition to the effects shown in Table 5, mining will also be a source of light and noise. Since light is absorbed rapidly in water, and particularly so in turbid waters such as may occur near the mining equipment, the radius of the area affected will generally be a matter of metres or tens of metres. Noise will affect a larger area since sound propagates through water much more readily than does light, and may carry well beyond the mine site. However, noise is generally not expected to be a significant problem.

4. Biological impact concerns

88. Biological impacts will occur on the areas actually mined, as immobile and slowly moving organisms are destroyed by the mining machinery. However, the areas mined are expected to be small relative to the total area occupied by the affected species. The areas affected by the turbidity plume and resultant sedimentation are considerably larger but still small relative to the total area of the affected habitat. The small scale of mining operations means that the offshore impacts generally will be limited in areal extent. However, since exceptions can be expected, assessments of the potential impacts of individual projects will be required to insure minimal adverse impacts. The most certain impacts will probably be the unavoidable ones at the mine site as machinery moves across the sea floor to fragment, collect, or otherwise process ore or move overburden. The significance of that destruction will be addressed in the site-specific environmental analyses for proposed operations.

89. Detailed and independent assessments of expected effects of marine mining are available in an analysis published in 1987 by the U.S. Office of Technology Assessment (OTA). A workshop associated with this OTA study concluded that surface and mid-water effects should be minimal if appropriate precautions are taken. The workshop participants thought that the effects on bottom organisms would be the most pronounced. Extinctions, which are the severest form of impact, sublethal effects, and recovery rates were deemed to need continued study. These needs were felt to be particularly strong in the deep sea where the identities of the organisms and their life histories are generally unknown.

90. Effects on organisms living in the water column are likely to be minor in most areas. Effects at the sea floor - particularly those resulting from the resedimentation of the benthic turbidity plume, the actual destruction of organisms, and the change in sea floor topography - will generally be more important than effects of changes in the water column. This conclusion is based upon observations of dredging operations and laboratory studies of the effects of suspended materials concentrations. Given the relatively rapid dilution of suspended sediments and dissolved substances, contrations should fall within acceptable ranges near the point of discharge. Factors, both biological and non-biological, that may affect the validity of this conclusion are listed in Table 5, and each of these factors should be considered in the determination of potential environmental effects.

(a) Turbidity

91. The principal biological effects of increased turbidity from mining are the interference with filter feeding, clogging of gills, and inhibition of photosynthesis because of decreased illumination. Enough variables are involved to give seemingly contradictory results when individual phenomena or laboratory data are discussed in isolation, but generalizations are feasible if one examines the impacts of actual dredging operations. The effects of turbidity caused by mining in

oceanic waters are localized and relatively mild because of rapid dispersal of particles. Non-swimming organisms will drift with the plume as it rapidly dilutes. Therefore their exposure to high turbidity will be brief, while swimming organisms probably will avoid the areas of highest turbidity completely. If so, these organisms will be excluded from a portion of their geographic range, but it will be a very small portion. For example, mid-ocean discharges during tests of manganese-nodule mining equipment resulted in near-ambient concentrations within 4 km of the discharged point (NOAA, 1981).

92. Similar results generally prevail in the shallower waters of the continental shelf, although higher concentrations may occur. For example, measurements by the French (Cressard and Augris, 1982), during an experimental dredging of sand and gravel in the Bay of Seine (depth 20 m), showed concentrations of 5 to 25 g l^{-1} of unknown duration. However, high concentrations such as these tend to be highly localized and recolonization of such disturbed areas may be rapid. In the shallow waters of the Beaufort Sea, large dumping operations during construction of gravel islands produced downstream concentrations only about three times the ambient concentration of 8-12 mg l^{-1} within 340 m of the site. Concentrations were only twice ambient levels within 1,730 m. Similarly, concentrations of 100 to 300 mg l^{-1} at 300 m downstream of barge dumps were predicted for island construction in the Canadian Arctic (DOI, 1983b).

93. These concentrations harm the more sensitive individuals of the more sensitive species but not all individuals of these species are likely to be affected. Concentrations that led to the death of 10 percent of the tested organisms in the Beaufort were 10 g l^{-1} or more for tolerant species and less than 1 g l^{-1} for highly sensitive species. Even this lower threshold is three to 10 times the concentrations measured 100 m downstream of barge dumps. The predicted or measured sediment concentrations a few hundred metres away from the dump site where generally 10 to 100 times below the lethal levels for even the most sensitive 10 per cent of the tested populations (DOI, 1983b).

94. In addition to sediment concentrations, there are three other potential sources of concern. First, the presence of toxics in the sediments can greatly increase the adverse impacts of turbidity. However, the data showing such effects are based on studies of sediments from harbours and estuaries in which toxins, such as pesticides and heavy metals, are introduced to the marine environment and subsequently adsorbed to the surfaces of sediment particles. Such contaminants are rarely found in more than trace amounts in the sediments of the open ocean. Moreover, in natural mineral deposits exposed to sea water, the biologically active metals are in essentially insoluble forms and generally are biologically inert.

95. Second, fluid muds (fluff) may form near the water sediments boundary and persist for weeks if there is insufficient circulation to disperse them. Such a condition could be a problem for organisms that are not equipped to bore through this layer to reach the less turbid environment (Hirsch *et al.*, 1978). However, the significance of this layer is practically a function of potential

for fluid-mud formation when fine, non-cohesive materials are resuspended or discharged and, along with the resulting hazards, it will strongly depend on the site operating conditions. However, lack of circulation is rare in waters of the continental shelf although fluff may form in coastal waters. Shell dredging and the operations in Galveston Bay, for example, resulted in a near-bottom mud flow with particle concentrations from 20 to 150 g l^{-1} that lasted throughout the dredging operations (Masch and Espay, 1967, from Peddicord, 1976).

96. Third, in all cases where mining involves the mechanical fracturing of the rocks, the particles produced are more likely to have sharper, more angular edges than resuspended bottom sediments. These sharp fragments may damage organisms more than natural sediments can.

97. Both laboratory and field observations have been made of various organisms under conditions of increased suspended sediments for various durations. The results, when compared with increased concentrations expected in the water column, suggest that these effects may not be of concern if the dilution factor is high and there are no exceptionally sensitive bottom communities in the area. For instance, a study of animals from San Francisco Bay found that at least 90 percent of the animals in most of the 18 species tested were alive after 10 days exposure to concentrations of 100 g l^{-1} kaolin clay suspensions (Peddicord, 1976). Even the more sensitive species were only adversely affected by tens of grams per litre of processed bentonite clays over several days time.

98. However, these experiments did show that the sensitivity of some species to bentonite increased with higher temperatures (e.g., 18°C) or lower oxygen concentrations (Wakeham *et al.*, 1975), suggesting that, in the San Francisco Bay, seasonal changes in these two factors may be important in determining the severity of impact from dredging and disposal. However, Peddicord (1976) noted that the laboratory experiments did not evaluate physiologically sublethal effects which may be more important ecologically than direct mortality. Lunz *et al.*, (1984) in a recent summary of data for eggs, larvae, and adults of fish and shellfish harvested in the coastal zone found that several of the molluscs and crustaceans tested were sensitive to 100 mg l^{-1} or more of suspended sediments during multi-day exposures, although others were tolerant of higher levels of 100 g l^{-1} or more over 1 to 3 day exposures.

99. The results of laboratory studies of marine fish vary. Moore and Moore (1976) found that turbidities of 126 to 135 mg l^{-1} increased the time for the European flounder to see its prey.

100. Although it is likely to be of minor importance if the plume is short-lived, this non-lethal effect would surely reduce the feeding rate and, hence, the vigor of the affected animals. Effects of manganese-nodule mining plumes on captive yellow fine tuna and kawakawa, have been studied by exposing the fish to deep-sea clays and nodule fragments. No detectable effect was found in adults at concentrations from 9 to 59 mg l^{-1}. Feeding continued to occur in concentrations off 11 mg l^{-1} and no change in behaviour was noted in the presence of a turbidity cloud (Barry, 1978). However, preliminary results by Barry suggested that turbidities greater than 4 mg l^{-1} sometimes caused feeding inhibitions and coughing in tuna in the laboratory (Matsumoto, 1984).

101. Some studies of larval fish have also shown feeding behavioral modifications because of decreased illumination, a possible consequence of increased turbidity (Blaxter, 1980; Kawakawa and Hara, 1980; Riley, 1966; Wyatt, 1972; Houde, 1975; Saksena and Houde, 1972; Houde, 1977). However, Masumoto (1984) concluded from a review of existing literature that the rapid dilution of the plume from manganese-nodule mineship discharges would be rapid enough to prevent any significant adverse impact on tuna and bullfish eggs, larvae, and adults. A limited number of field observations have been made on the behavior of marine fish in areas of high turbidity. The results suggest that adults are less sensitive than the young, and that there are differences among species. Whitebait (immature herring) and sprats have been found to avoid areas where china-clay wastes are being discharged. Mackerel, however, do not seem to avoid these areas and exhibited no problems in feeding, despite being visual feeders (Wilson and Connor, 1976; Shelton and Rolfe, 1971; Shelton, 1973).

102. Fish are observed to be abundant off the mouth of the Columbia River where sediment concentrations may reach 10 to 100 mg l^{-1} (Pruter and Alvertson, 1972). Ritchie (1970) noted no adverse effects from overboard dredge spoil disposal on 44 species of fish from Chesapeake Bay. Observations at Hawaii showed that densities of fish larvae are negatively correlated with higher turbidities, whether natural or man-related, which Miller (1974) suggests may be caused by avoidance of turbid areas. Lunz *et al.*, (1984) summarized laboratory reports of significant mortality of white perch, yellow perch, and striped bass larvae at suspended sediment concentrations of 0.5 - 5.4 g l^{-1} after one to four days exposure. Larvae in the open ocean would not be exposed to such concentrations for such long periods, as they would drift with the diluting turbidity current, and the potential significance of exposures in the field may be overstated by these laboratory data.

103. The possibility exists that in certain cases the circulation of nutrients and sediments might affect productivity. In estuaries or closed areas this effect would likely be negative, perhaps triggering algal blooms. In the open ocean such effects were anticipated from deep sea bed mining, but there have been no indications in the literature of significant effects being measured.

104. An excellent review of potential, adverse effects on zooplankton from manganese-nodule mining plume in the deep sea bed was conducted by Hanson *et al.*, (1982). Many of these species are filter feeders which ingest particles on the basis of size. Research has shown that zooplankton would most likely ingest non-nutritive particles from a mining plume (Hirota, 1981), effectively reducing their food intake. Not surprisingly, large increases in the ratio of non-nutritive to nutritive particles are harmful. For example, concentration of 0.6 to 6 mg l^{-1} of clay wastes from aluminium extraction resulted in lower body weights of copepods (Paffenhoffer, 1972).

105. Some zooplankton can differentiate nutritive particles and reject low-quality food. This mitigates, but does not eliminate the effect. However, the effect to be expected in the presence of mining may not be readily distinguished from natural variability. Hanson *et al.*, (1982) concluded from their review of manganese-nodule mining wastes that the oceanic zooplankton community, would probably not be significantly harmed. Even though these wastes included trace metals

released from the nodule fragments and pelagic clays as well as suspended load, they noted that the plumes would be rapidly diluted, exposures would be brief and the animals would tolerate slight increases in trace-metal concentrations.

(b) Sedimentation

106. Feeding mode, life habit, degree of mobility, or sensitivity of life stage all affect the vulnerability of organisms to sedimentation. Some organisms are able to burrow out from tens of centimetres of sediment, although there seems to be a maximum depth below which no escape response is initiated (Nichols *et al.*, 1978; Hirsch *et al.*, 1978; Maurer *et al.*, 1978). Other species can tolerate a slight increase in sediment deposition above natural loads. Oysters have been found to be fairly tolerant of some increase in sedimentation although a large amount of deposited material can cause suffocation (Mackin, 1961; Dunnington, 1968). Adults react to slight increases through increased pumping or, in some cases, through closing their valves. However, Loosanoff (1962) found that some animals died when stress was prolonged for more than 48 hours. Larvae and eggs were most sensitive, with normal development being affected at 188 mg l^{-1} and stopped at 2 g l^{-1}. Clam larvae and eggs in general exhibited less sensitivity to increased concentrations of silt (Davis and Hidu, 1969).

107. The resiliency of animals may be much less if their typical natural environment has low turbidity, such as is found in much of the tropics. Hermatypic corals are known to depend on light for growth (Goreau, 1961), with growth being inversely related to the amount of resuspended sediment (Dodge *et al.*, 1974). Consequently, a substantial increase in suspended sediment in areas of coral growth could pose some concerns, even though many corals can withstand some sedimentation through active removal. Bak (1976) studied the growth rates of fringing corals before and after dredging in the Caribbean. Although light levels were reduced to approximately 1 percent of the surface illumination for only a few days, growth of *Madracis mirabilis* and *Agarlicia agaricites*, both efficient sediment rejectors, was reduced by one-third for more than 1 month. Colonies of *Porites asteroides*, however, were killed because when covered with sediment which they were unable to remove, they lost their zooxanthellae (symbiotic algae), and died.

108. Other areas that may not be able to withstand slight increases in sediment deposition are those used by bottom-spawning fish. For example, in the North Sea where gravel dredging resulted in large pits 3 to 5 m deep, water flow over these areas was slowed, resulting in finer sediments being deposited (Dickson and Lee, 1973a and 1973b). Many of these pits fill in slowly, with measured rates suggesting that decades may be required before they fill with silt and that they may never return to pre-mining conditions. Likewise, monitoring of dredge channels along the Atlantic coast of the United States has shown that silt-clay particles, rather than sand, fill in the channels, resulting in an altered substrate (Taylor and Salomon, 1967).

109. In Europe, recommendations have been made to avoid those areas where herring attach

their eggs to the sea floor as well as areas and times of sand-eel spawning (ICES, 1975; De Groot, 1979a). Similar bottom spawning species exist off both coasts of the United States (e.g., herring, winter flounder, sand eels, rocksole) and deserve attention, as the spawning areas of these species may be affected by increased sedimentation, especially if a fluid mud layer is created (Hirsch *et al.*, 1978). Also, there are special environments, such as the Arctic (DOI, 1938b), where recolonization may be extremely slow, but on which little information is available. Wright (1977) observed that full benthic recovery may take more than 12 years as a consequence of gravel island building for OCS oil and gas development in the Canadian Beaufort Sea.

110. Dunton *et al.*, (1982) stripped the organisms from a small area in a cobble-kelp community in the Arctic and found most of the areas still bare after three years. Such slow recovery of a community suggests that this environment may deserve special attention. Deep-sea environments probably would be equally slow to recover, since they are also very cold and food is limited. Sea-floor spreading areas with hydrothermal vents are another unusual environment that would deserve attention if any mining were anticipated near them.

111. In cases where most of the benthic community is adversely affected, recolonization will have to occur from populations outside the disturbed area. The rapidity of this process seems to be highly influenced by the similarity of the new substrate to the pre-mining condition (De Groot, 1979b). De Groot (1979a) established that recovery of the benthic fauna would take two to three years following sand and gravel mining. Recovery is aided if some of the sand and gravel deposit is left, so that the substrate remains similar. Pfitzanmeyer (1978) found that the upper Chesapeake Bay recovered to its original condition 18 months after dredge-spoil disposal when there was no major topographic or stratigraphic change and the sediment was similar. Kaplan *et al.* (1974) found that the number of species in an enclosed bay in Long Island before and after dredging differed as a function of sediment type, with sediment changes being related to the circulation changes that resulted from the dredging.

112. Other studies of dredged, inshore areas found rapid recovery (Cronin *et al.*, 1971; Harrison, 1967; Connor and Simon, 1979). However, in many cases, while the biomasses may reach pre-dredging conditions, the diversity and distribution of species does not always replicate the pre-mining environment (May 1973; De Groot, 1979b). Such a shift in community structure has been reported in experiments with recolonization boxes at depths greater than 1000 m (Desbruyers *et al.*, 1980; Grassle, 1977; Levin and Smith, 1984). Opportunistic species able to move rapidly into an unoccupied area, were the first organisms to settle in the boxes and had the largest populations. The number of those species, however, was low in comparison to adjacent areas. After 26 months, the species composition of the deep-water boxes still differed greatly from adjacent areas.

113. Shallow-water studies have shown that marine communities must evolve through successional stages (Thistle, 1981; Shelton and Rolfe, 1971; Gallagher *et al.*, 1983; Kaplan *et al.*, , 1974) before the redevelopment of the pre-dredging community begins. Gallagher *et al.* (1983)

identified certain species that had to move into an area before other species could recolonize. These results suggested that having a similar substrate increases recovery, but other steps must occur before the pre-mining condition is reached.

114. Species that produce a large number of pelagic larvae are frequently the first species that move into an area that has been depopulated. However, other qualities have also been found in opportunistic species. In intertidal environments, the first polychaete recolonizers were frequently species whose adult stages were mobile (Levin, 1984; Dauer and Simon, 1976). Levin also found that the scale of disturbance, the type of larval development (that is, pelagic or non- pelagic), the settlement patterns and the mobility of the species were important to the success of recolonization. Species with non-pelagic larvae rapidly recolonize small areas of disturbance. Polychaetes with pelagic larvae were more adept at moving into large disturbed areas, especially when the disturbance occurred during the peak of the larval dispersal period. Consequently, the recovery rate of an area where mining operation has resulted in small shallow pits may differ considerably from large areas that have been scraped in a contiguous manner.

(c) Explosive shock

115. Explosives were widely used at one time for deep seismic exploration offshore but have now been superseded by other methods such as air guns and sparkers. The present limited used of explosives underwater for mining is strongly regulated, in the U.S.A., by both State and Federal authorities to minimize environmental damage, particularly in areas where commerical fishing is practiced or protected species are present. In all cases of use, environmental analyses are required and all operations are subject to immediate shut down if unacceptable environmental effects on marine communities are detected.

116. The use of explosives to break up, overburden, and fragment the mineral deposit will generate shock waves, which generally will cause some deaths in the benthic invertebrate fish populations. The extent of the effects depend on the velocity of detonation. For example, research has shown that dynamite and TNT produce far more deaths in fish than black powder (Hubbs and Rechnitzer, 1952; Kearne and Boyd, 1965; Anonymous, 1947 and 1948). Thus, the effects cannot be predicted until the explosive material proposed for use in any specific case is determined. These explosives may be novel materials since conventional mining explosives, such as gelignite and ammonium nitrate, may not be suitable in deep water where the pressure might reduce the effectiveness of the explosion (DOI, 1983a).

117. Adverse effects from individual explosions should be brief and confined to a small area since explosive forces rapidly dissipate with distance from the source. Significant, permanent, adverse impacts on the biological community would be unlikely unless either rare or unusual species occurred near the mining operation, or the cumulative effects of repeated explosions differed greatly from the effects of single events. However, the presence of the normally prevalent species

in areas subject to explosions during war, naval training, and occasional public work activities shows lack of irreparable damage from repeated explosions. Rates of recovery probably will be similar to the rate of recovery from other forms of marine mining.

118. The organisms most susceptible to the shock waves generated by the detonation of explosives are those fish species that possess a swimbladder. The swimbladder is a membranous gas-filled sac that has a hydrostatic function in most fish that possess it. Fish without swimbladders are mostly deep-water species or predators, such as sharks, which swim at various levels in pursuit of prey. Young (1973) found that invertebrates living on or near the sea floor are more tolerant of explosives than fish with swimbladders.

119. The individual components of an underwater explosion important to impacts on fish are thought to be the amplitude (peak pressure), the type of compression wave, and the refraction wave (Hubbs and Rechnitzer, 1952). In an explosion, two types of positive pressure waves are produced, the initial compression or shock wave and the following bubble-pulse wave (Cole, 1948). The compression wave causes the primary disturbance to the water. Dynamite detonates instantaneously and produces a sharp shock wave with an abrupt front of intense pressure, which can seriously injure fish with swimbladders. Black powder burns more slowly and produces a less intense peak pressure of small amplitude, which is less injurious. The bubble-pulse wave from both dynamite and black powder explosions is of a relatively long duration and low maximum pressure. No major impact on fish was observed from this wave by Hubbs and Rechnitzer (1952), although some difficulties have occurred with shallow explosions in seismic surveys. These reported problems, however, may have been caused by refraction waves.

120. The rarefaction wave is produced when the compression wave is reflected from the water-air boundary and is transformed, with loss of energy, into a negative pressure pulse. These suddenly applied negative pressures appear to kill fish by explosion of the swimbladder. As a result, the refraction wave causes more damage near the surface of the water than at depth, where ambient pressure is higher. Dynamite produces much higher negative pressures than black powder and is again more harmful to fish. Baldwin (1953), for example, during his observations of the use of black powder in geophysical surveys in salmon fishing areas off California found no dead or injured salmon even though many were seen swimming in the blasting area before detonation. Rapid compression followed by the negative pressure pulse caused by rarefaction thus seems to be most responsible for injury to fish.

121. Several studies showed conflicting results of the effects of detonation at different depths, but the conflicts seemed to result from the different environmental contexts. Kearns and Boyd (1965), studying explosions in the upper water column, concluded that the areas within which fish would be killed increased with increasing depth of detonation. Goertner (1981), studying explosions at the seafloor, found that the probability of death was reduced with increasing depth of explosion, and, hence, higher ambient pressures. The differences apparently related to the location

of the explosions and, perhaps, the absolute depths involved. Studies of both behavioural and physiological effects on fish larvae and juvenile stages caused by pressure pulses from air guns and explosive removal of structures were conducted in the Santa Barbara Channel under the joint oversight of an energy company and of commercial fisheries. The reports are in preparation, and similar studies are in progress in the Gulf of Mexico with MMS support.

122. Research conducted in Japan on the relationship between explosion size, fish type and distance from the blast found that damage always occurs to the fish's stomach when the pressure is 5 kg m^{-2} or greater. A pressure of 3 kg m^{-2} is considered safe. It is equivalent to a distance of 200 to 500 m from a 1.5-ton explosion, the largest permitted by Japanese Law (Padan, personal communication). Mammals such as sperm whales, which regularly dive to and feed at depths greater than 1,000 m, could possibly be affected if they were in the immediate vicinity of the mine site during an explosion. However, use of small preliminary explosives to scare fish and mammals from the blast site may prove useful in reducing any such impacts if they are deemed likely to occur. Furthermore, as noted earlier, holes drilled in the sea floor for detonation would direct the blast downward and laterally and minimize adverse effects (DOI, 1983a).

(d) Light and noise

123. Light and noise generated by mining equipment may disturb nearby benthic organisms and possibly modify their behaviour. However, many, if not most, mobile, non-territorial organisms should be able to move away from the disturbance. Even though there is little natural light below 1 km, deep sea organisms have functional eyes and the light introduced during mining will attract some organisms and cause others to move away. Moreover, observations from submersibles indicate that fish exposed to bright, artificial light may be at least temporarily mesmerized, possibly blinded. The exact impact has not been determined (NOAA, 1981).

124. Not much is known about the intensity, frequency and duration of mining noises. The type of collector most likely to be used for mining manganese nodules may attract scavengers, especially species like the rat-tail fish, that communicate by sound (NOAA, 1981). Noise also affects marine mammals, but significance is not well known (DOI, 1983a; DOI, 1983b).

125. Most research has been conducted on the effects on endangered and/or threatened marine mammals from OCS oil and gas activities. Some researchers (Geraci and St. Aubin, 1980) have suggested that most animals become habituated to low-level background noise, such as ship traffic and onshore and offshore petroleum activities; however, some animals show abrupt responses to sudden disturbances. Responses to continuing abrupt noise disturbance of California sea lions, Stellar sea lions, and harbour seals are fairly well documented (DOI, 1983b). Gales (1982) found that possible auditory effects from a sonic boom (a pulse somewhat similar to explosive seismic pulses) include startle, flight, auditory discomfort, and hearing loss. Stationary dredging operations in the arctic did not seem to greatly disturb beluga and bowhead whales, al-

5. Conclusions

126. The following conclusions can be drawn from the research and operational experience cited in the preceding sections.

- Mining in the outer continental shelf will produce effects from both the mining itself and the transport, beneficiation, and refining of the ore mined.

- The nearshore and onshore effects associated with the transport, beneficiation, and refining of the ore are common in existing ports, and hence familiar.

- Site-specific studies and monitoring of environmental effects of marine mineral mining will often be necessary during at least pilot tests of new equipment and new operations. Knowledge of the method of mining, the type of ore, and the characteristics of the mine site are needed to be able to predict the effects at the level of detail needed in EISs.

- Marine minerals will be mined by the four basic methods of mining used for solid deposits on land, i.e. scraping the surface, excavating a pit or trench, tunneling, or drilling a borehole and removing the valuable constituent in a solution or as a slurry. Any of these mining methods is applicable to more than one type of mineral deposit.

- Near-field effects associated with fragmentation/collection on, or excavation of, the sea floor will be limited to the period of active mining and will be limited to the mine site.

- Far-field effects in deep, oceanic waters are the least known, with resedimentation being the main concern. Of particular concern is the effect on benthic organisms and the repopulation of mined areas.

- Because of the rapid dilution prevalent in the waters of the oceans, organisms living in the water column are unlikely to be exposed to adverse concentrations of suspended sediments. In the immediate vicinity of mining operation, exposure is expected to be brief due to the transient nature of the plume.

- Organisms living on the sea floor are the most likely to be affected because of the resedimentation of the benthic plume, the actual destruction of the organisms, and the change in the character of the sea bed.

III. OCEAN ENERGY DEVELOPMENT

127. Ocean energy development was a high priority item in the seventies but has become less of a perceived priority during the eighties for a number of reasons, including the short term, and artificial, glut of lower cost oil in the world marketplace. For this reason, funding for research and development of alternative fuels and energy sources has been reduced in many instances, particularly in the oceans, and the collection of data on the environmental effects of ocean energy production has consequently been very limited. However, because of the vital importance in the future of ocean energy development, the following brief notes are inserted for the record and to serve as a base from which to prepare continuing documentation on these concerns.

A. OCEAN THERMAL ENERGY CONVERSION (OTEC)

128. The following notes are abstracted largely from (Quinby-Hunt, and others, 1986). Ocean Thermal Energy Conversion (OTEC) is a power-generating system that uses the temperature difference between warm surface water in the tropical ocean and the cooler water at depth to run a Rankine-cycle heat engine. For the temperature difference typically available in the upper 1,000 metres in tropical and sub-tropical waters, 20-25°C, energy extraction efficiencies are low, two to four per cent (Dugger *et al.*, 1981), compared to conventional steam generation plants. Because of the low efficiencies involved, large flows of ocean water (the fuel in an OTEC system) are required: about 10 m^3 s^{-1} per megawatt (DOE 1979a). Two OTEC operating cycles are currently under development in the United States at Argonne National Laboratory, Argonne, Illinois and at the Solar Energy Research Institute at Golden, Colorado: closed- and open-cycle. Because the technology is highly experimental, most of the technical details have not yet appeared in the literature.

129. In closed-cycle systems a low-boiling point working fluid (ammonia or Freon) is evaporated by warm surface waters. The vapour is expanded to drive a turbine. The expanded vapour is then condensed by cooler deep ocean water and returned to the warm side.

130. In open-cycle operations warm surface water is used as the working fluid. Surface sea water is introduced into an evaporator under partial vacuum separating the sea water into steam and brine. The steam, after passing through a turbine, is condensed using cold ocean water.

131. Several configurations for OTEC plants have been considered to date: free-floating plants (grazing and moored), bottom-resting facilities (at various distances from shore) and shore-based plants. The floating and far-offshore configurations currently are not being actively pursued in the U.S. Program due to increased technical and safety risks associated with them (Lewis, 1983). Also, use of open-cycle technology for freshwater generation and mariculture requires land or proximity to land.

132. Numerous documents have addressed the environmental effects of closed-cycle OTEC (DOE, 1979a.b., 1980, 1981). Although the earliest OTEC experiments were on open-cycle systems (Claude, 1930), the environmental effects of open-cycle OTEC have not yet been thoroughly investigated.

133. Open-cycle OTEC uses warm ocean water as the working fluid. Prior to evaporation the water is degassed, removing dissolved oxygen, nitrogen, carbon dioxide, and trace gases from sea water. Once in the evaporator under partial vacuum, the fluid is separated into a somewhat more concentrated warm water effluent, more non-condensable gases, and water vapour. The vapour is used to drive a turbine. The spent vapour can be converted to liquid water in a conventional surface condenser using a heat exchanger, or by direct contact condensation.

134. Conventional condensation using a surface condenser is expected to result in higher parasitic power requirements. Conventional condensation will result in fresh water as a by-product. Because of the higher parasitic power requirements, conventional condensation may require larger flows than would condensation by direct contact heat exchange.

135. For open-cycle systems using direct contact heat exchangers, the cold water stream may be degassed prior to use in the condenser. Inside the direct contact condenser, the cold water will release more non-condensable and trace gases.

136. In addition to atmospheric releases, a number of fluid streams result from open-cycle operations:

OPERATION	STREAM GENERATED
Evaporation	Concentrated warm water effluent
Condensation, Surface	Fresh water, cold condenser water
Condensation, Direct contact	Direct contact, mixed effluent discharge

The fluid streams can be either discharged directly, or used first for other secondary economic purposes, either directly or in various combinations, and then discharged. Current research programs call for at least part of the sea-water discharges to be used for mariculture or solar ponds (SETS, 1983). The fresh water could be used for domestic, industrial, or agricultural purposes.

1. Paths to the environment

137. The nature of the releases to the atmosphere and to the marine environment will depend upon the type of heat exchanger used, any secondary uses, and the chosen discharge configuration.

2. Atmospheric releases

138. Carbon dioxide, oxygen, nitrogen, other atmospheric trace gases, and volatile metals will be released during degassing of the warm and cold water. Due to the difference in solubility among atmospheric gases in sea water, the ratio of N_2: 0_2: $C0_2$ = 28: 19: 1 dissolved in sea water differs from the ratio 2400: 630: 1 found in the atmosphere (Skirrow, 1975). The solubility for these gases also increases with pressure. Thus sea water from depth will degas naturally when brought to the surface. The volume of each gas released by degassing at any given pressure will vary inversely with the ambient temperature of sea water (Weiss, 1970), with warmer surface waters having less dissolved gases than colder deep waters. Further non-condensable and trace gases will be released from the warm water stream during evaporation and from the cold water during direct condensation.

3. Marine discharges

139. Depending on the heat exchanger, discharge configuration, and secondary uses chosen, a number of different discharges will result, each reaching the point of discharge by a different path. As the composition and other physical and chemical properties of these discharges differ, so will their potential impacts. Seventeen paths for releases to the environment are described by Quinby-Hunt and others (1986, op. cit.) and include discharge resulting from uncombined OTEC; combined OTEC; fresh water use; mariculture use; solar ponds; and other industrial uses.

4. Summary

140. A number of oceanographic and environmental studies need to be undertaken to assure that open-cycle OTEC operations are environmentally benign. Those that are also necessary for the development of closed-cycle OTEC have been discussed elsewhere (DOE, 1979b, 1980; Meridian, 1983). The studies required for the logical development of a viable open-cycle OTEC technology and for the development of OTEC in general include studies of:

(a) Terrestrial impacts

(i) design studies to assure that an open-cycle OTEC plant is aesthetically acceptable,

(ii) optimization of designs to minimize building size and mass and land use, and

(iii) quantification of solid wastes generated (that is, shell material from mariculture, carbonates from outgassing, etc.).

(b) Atmospheric impacts

(i) global and local climatic implications of carbon dioxide releases from open-cycle OTEC plants

(ii) trace constituent releases from open-cycle OTEC plants, and

(iii) effects of artificial upwelling on local atmospheric conditions.

(c) Marine impacts

(i) influence of degassing and various evaporation and condensation strategies on the pH and buffer capacity of intake and discharge waters,

(ii) effect of pH and alkalinity change on the solubility of salts dissolved in sea water,

(iii) effect of pH change on the chemistry of ocean water and on coral and benthic communities,

(iv) discharge of de-aerated water on marine communities, and

(v) properties of re-aerated sea water after degassing, and various evaporation and condensation operations.

5. Concerns for Secondary Uses: Mariculture and Solar Pond OTEC

141. For mariculture, environmental concerns include the addition of nutrients, biological oxygen demand (BOD), chemical oxygen demand (COD), detritus, non-native species, and diseases by the discharge water. Generation of solid waste may also result in terrestrial concerns. For solar ponds, there is the possibility of release of thermal brines, biocide, and working fluid. All these potential impacts need to be investigated.

6. Conclusions

142. Open-cycle OTEC can be designed as a relatively benign technology when compared with known problems related to conventional energy production. However, a number of new environmental concerns arise with the development of a new and innovative technology. Some concerns are the same as those associated with closed-cycle OTEC. Others, identified above, are unique to open-cycle systems. It is essential that all significant concerns be addressed to assure that open-cycle OTEC is an environmentally acceptable alternative to conventional energy and secondary product generation.

B. WAVE ENERGY

143. Wave energy is the utilization of the potential energy in ocean waves caused by wind blowing on the sea surface. Swell is the smoothed response of the sea surface to distant winds and storms. The energy density of waves has been measured at 40 times the density of wind energy per square metre (Masuda *et al.*, 1988).

144. Four types of wave energy devices are described by Masuda (1988, op. cit.) and these include navigational buoys, large wave power floating systems, backward bent duct buoy, and shoreline fixed wave power generators.

1. Navigational buoys

145. These small moored generators develop from 0.06 to 0.5 kw per unit and as many as 1,200 have been constructed and deployed from Japan. A small impulse, air operated turbine is used to generate current, with or without batteries.

2. Large-wave powered buoys

146. These systems in a number of variations were listed by the International Energy Agency (IEA) as a joint effort by Japan, USA, U.K., Ireland, Canada, Norway and Sweden. The test system has operated satisfactorily for 10 years off Japan.

3. Backward bent duct buoy

147. A more efficient energy collector was developed with a horizontal intake pipe, facing away from the wave direction. A 60 per cent conversion efficiency is claimed for this

buoy, against 30 per cent for the vertical pipe and 10-15 per cent for the large-waves powered buoy.

4. Shoreline-fixed wave-power generators

148. These devices, of which a number have been tested, are designed for use on rocky, wave-swept coasts and are ideally adaptable to Pacific Islands Terrain. Negotiations are underway to install a commercial-sized generator in the Kingdom of Tonga, using Norwegian technology.

5. Environmental aspects

149. Despite all the activities in testing wave energy systems, much of which was carried out during the period of high oil prices, documentation on the environmental aspects is very scarce.

C. TIDAL ENERGY

150. Tidal energy has been a long developing technology, involving the damming of estuaries and the controlled flow of the tidal waters through turbines. The major operating scheme on the river Rance in France involves major engineering. For other schemes in the Severn River (U.K.) and at Passamaquoddie in the U.S.A. engineering proposals have been quite substantial but were not available for inclusion in this report. They include such problems as altered estuarine mixing and siltation.

D. CURRENTS

151. Energy from ocean currents is a dream with substantial foundation. Obviously the contained energy in oceanic circulation must be very large, and engineering concepts to tap it range from large anchored turbine arrays to electromagnetic induction generators laid within the Gulf Stream between the U.S.A. and the Bahamas. Literature was not available for inclusion in this study.

E. GEOTHERMAL

152. Geothermal resources can be classified into three primary groups: hydrothermal, geopressurized or hot dry rock. The most common is a water-dominated hydrothermal system. The Geysers in California is an example of the geopressurized vapor-dominated resource. Hot dry rocks, though the least common, have been identified in Cornwall, U.K. but the technology needed to develop this source is still in the experimental stage.

153. Once a well is drilled, the geothermal steam is used to turn a turbine, located at the wellhead, or a near-by power plant, which creates electricity. The geothermal fluids are then cooled, and used for other purposes or reinjected into an unused, inactive, well.

154. The geothermal operations create by-products, in addition to heat, which are released into the atmosphere and the ground. These pollutants or by-products vary because the chemistry of wells vary. The ultimate emissions also depend on the use of pollution/emission control equipment. Geothermal wells can emit large amounts of particulates, hydrogen sulfide (H_2S), and sulfur dioxide (SO_2) into the air. The fluid brines can contain large amounts of silica which are not considered toxic under present federal standards. Although many of these pollutants can be dangerous at certain levels, H_2S, one of the more significant ones, is not covered by ambient air quality or emission standards at this time.

Marine geothermal

155. The presence of very significant geothermal anomalies are indicated along the axes of divergent tectonic plates in the deep oceans by the presence of hydrothermal discharges with measured water temperatures as much as 350 °C. The hot rocks are obviously very close to the surface in these areas, and temperatures ranging near 1,000 °C have been postulated within a few hundred metres from the seabed. The possibility of developing these energy sources is obviously well in the future, but in estimating possible thermal and chemical changes to the oceans, they should serve as useful data sources for modeling these very significant inputs.

156. The use of geothermal power in island countries has been established in Hawaii, U.S.A., but the surprising opposition to development has not come from pollutant or technical factors but from a general negative bias from a local public that does not want to accept change, and invokes even the sanctity of an ancient god (Pelé, Goddess of the Volcano), to avoid it.

157. This opposition can be very significant. As Table 6 indicates, as much of 52 per cent of Hawaii's energy needs could be produced from geothermal sources by the year 2000. Such a change could be environmentally significant because of the reduction in the burning of fossil fuels.

158. Other environmental effects from geothermal wells, affecting the oceans have not been well documented. There remains a significant amount of work to do to clarify these effects.

F. BIOMASS

159. Historically, biomass has provided the most significant contribution to the generation of electricity in some island communities of all of the alternate energy sources. Biomass, as defined by the Public Utilities Commission in Hawaii, for example, is any organic material which is not derived from fossil fuels. Forms of biomass used for the generation of energy include bagasse, wood, macadamia nut husks, molasses, algae, organic wastes and municipal refuse. Energy can be produced from these feedstocks by: (1) combustion for the creation of process heat; (2) combustion to transform water to steam and then to electricity; (3) distillation or hydropyrolysis into liquid fuels; or (4) anaerobic digestion of pyrolysis into gaseous fuels (DPED, 1981). Because of the wide range of sources and applications, biomass has been called the most flexible of Hawaii's renewable energy resources. The wide range of potential biomass feedstocks and methods of producing energy suggest the potential for numerous and varied impacts and, therefore, potential conflicts.

160. Most of the conflicts arise from the using of wood sources at a rate which depletes the original stock. Not much information was available on production from marine algae but it has been suggested as a viable energy source in California where giant kelp is harvested for commercial purposes.

161. Ocean farming of kelp is an attractive concept for producing low Btu, gas in that minimal impact on present resource use is expected. Questions remain as to the availability of sufficient nutrients and sunlight to provide the degree of growth necessary to make this biomass source economically viable. Other issues pertaining to interspecies competition and alterations in marine materials cycling have also been raised. Emissions from harvesting and transport of kelp appear to be minimal. Open- ocean farming, as opposed to nearshore production, may eliminate or at least ameliorate these problems. In either case, the technology appears promising enough to warrant further study.

162. Considerably more work needs to be done to examine the environmental effects on the oceans which would be specific for any particular site or species.

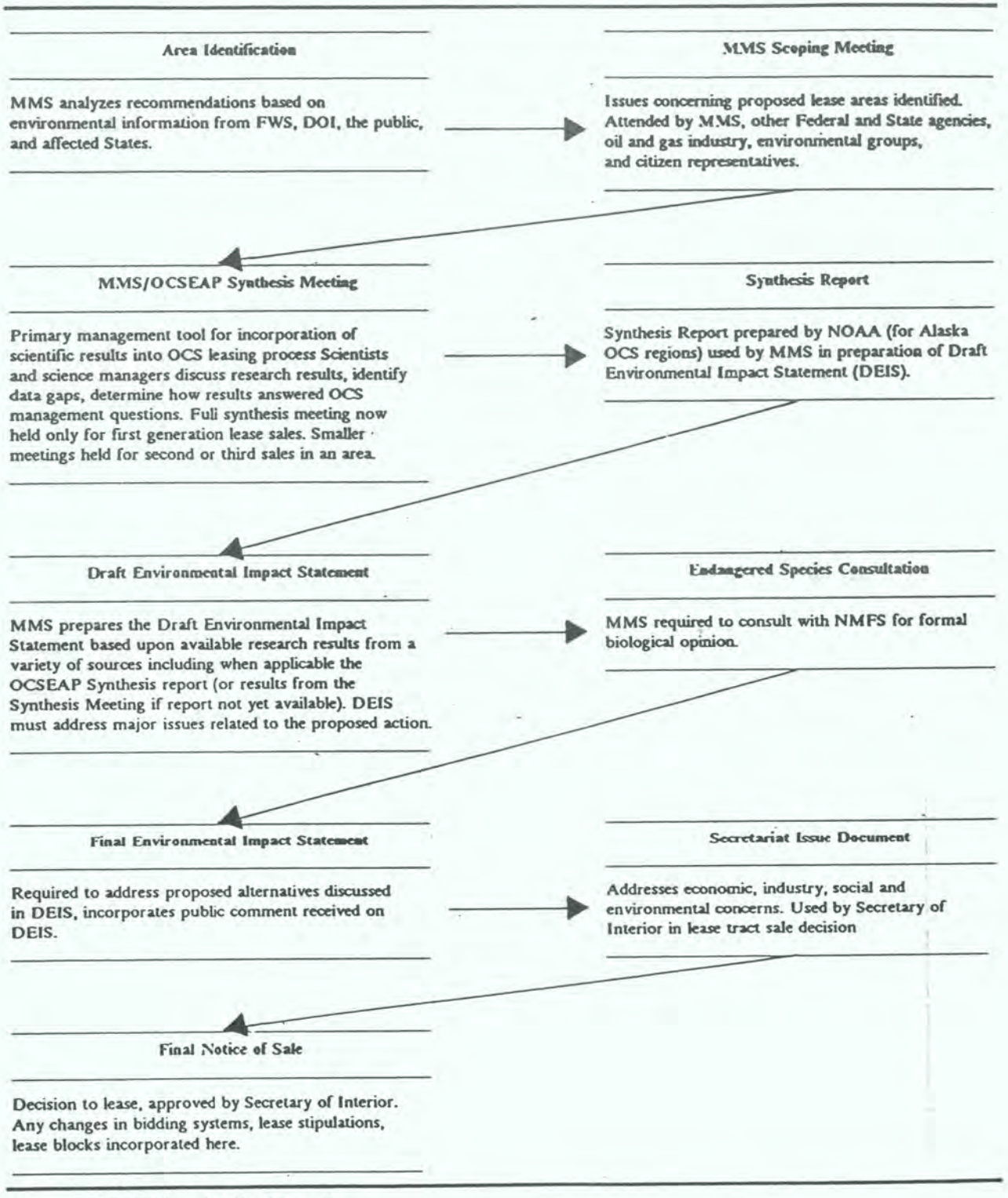
TABLE 1. WORLD OFFSHORE OIL PRODUCTION

Region or area	Oil Production (million bbl/day)	
	1983 (actual)	1984 (projected)
Middle East	3.74	3.88
Latin America/Caribbean[a]	3.24	3.36
North Sea	2.88	3.05
United States (GOM + Calif.)	1.68	1.78
Southeast Asia and others	1.53	1.56
West Africa	0.80	0.80
Soviet Union	0.18	0.17
Mediterranean	0.14	0.15
Total	14.19	14.75
Percent of onshore + offshore	26.6%	27.7 %

a Latin America /Caribbean includes Mexico, Venezuela, Trinidad, Brazil and Argentina as key producers.

SOURCE: Offshore Magazine, May 1984.

TABLE 2. USE OF ENVIRONMENTAL INFORMATION IN LEASING PROCESS*



Source: Office of Technology Assessment.

* Abbreviations:

DOI	Department of the Interior
DEIS	Draft Environmental Impact Statement
FWS	Fish and Wildlife Service
MMS	Minerals Management Service
NMFS	National Marine Fisheries Services
NOAA	National Oceanic and Atmospheric Administration
OCS	Outer Contential Shelf
OCSEAP	OCS-Environmental Assessment Program

TABLE 3. CLASSIFICATION OF MARINE MINERAL RESOURCES

Fluid		Unconsolidated		Consolidated	
Sea-bed	Subsea-bed	Sea-bed	Subsea-bed	Sea-bed	Subsea-bed
Seawater Magnesium Sodium Uranium Bromine and Salts of 26 other elements	*Hydrothermal Fluids*	*Industrials Materials* Sand & gravel Shells Aragonite *Heavy Mineral Placers* Magnetite Ilmenite, Rutile Chromite, Monazite *Nodules* Manganese Nickel, Cobalt Copper Phosphorite *Muds and Oozes* Metalliferous Carbonaceous Siliceous Calcareous Barite	*Heavy Mineral Placers* Gold Platinum Cassiterite Gem stones *Bedded Deposits* Phosphorites	*Crusts* Phosphorite Cobalt Manganese *Mounds and Stacks* Metallic sulfides	*Disseminated, Stratified, Vein or Massive Deposits* Coal Phosphates Carbonates Ironstone Limestone Metallic sulfides Metallic salts

TABLE 3a. ESTIMATED YEARS REQUIRED TO MOBILIZE FOR COMMERCIAL PRODUCTION OF EEZ HARD-MINERAL RESOURCES

Resources	Plausible development	Market favorable	Accelerated development	Factors that would tend to accelerate development
Construction	< 5	3	0.5	Critical construction need of importance to owner (State or Federal).
Cr and Pl placers (Oregon, Washington, Northern California, possibly Alaska)	> 20	5	3	Interruption of supply from South Africa and U.S.S.R. Disappointing results of exploration at Stillwater. Failure of research to significantly improve recovery from scrap. Demonstrations that environmental effects are not serious.
Au placers (chiefly Alaska)	10	4	2	High price gold.
Ti, Sn, Nb, Th placers (East Coast, Alaska)	> 20	5	3	High prices. Demonstration that environmental effects are not serious.
Phosphate	15	5	3	Pressure for land development, environmental conservation. High prices for imported ore.
MnO_2 crusts for Co Johnston Island	15	10	6	Interruption of supply from Zaire. Exhaustion of ore from domestic mines. Failure of research to significantly improve scrap.
MnO_2 nodules for Ni, Cu (Blake Plateau, Hawaii)	20	8	4	Interruption of supply of Mn from South Africa and of Ni from Canada. Disappointing production from Brazilian deposit. (Normal development unlikely because Blake Plateau nodules metal poor and best. Pacific nodules are not in the U.S. EEZ).
Sulphides in rift zones (Gorda Ridge)	> 20	12	7	High prices of Zn, Cu, Ag.

Source: U.S. National Research Council, 1987.

TABLE 4. MINING METHODS APPLICABLE TO MARINE MINERAL DEPOSITS

Mining Methods		Marine Mineral Deposit Types (with selected examples)									
		Unconsolidated Deposits					Consolidated Deposits				
Mining Approaches	Mining Systems	Construction Aggregates	Heavy Mineral Placers	Metalliferous Muds	Nodules and Slabs	Oozes	Bedded	Crusts	Massive	Mounds and Stacks	Veins
Scraping	Drag line dredge		A		A						
	Trailing suction dredge	P	A	A	P	F					
	Crust-miner							F			
	Continuous line bucket	F	F	F	A	F		F			
Excavating	Clam shell bucket	A	A						F	F	P
	Bucket ladder dredge	A	P								
	Bucket wheel dredge	A	A								
	Anchored suction dredge	A	A	A		F			F	F	
	Cutterhead suction dredge		P					A			
	Drilling & blasting								P	F	P
Fluidizing sub-sea-floor	Slurrying	A	A	P			F				
	Leaching								F		F
Tunneling beneath sea-floor	Shore entry						P		A		
	Artificial island entry						A		A		

P = Principal method used; A = Alternate method; F = Future use posible

TABLE 5. NORMAL OPERATING ENVIRONMENTAL DISTURBANCES [1]

Mining Methods		Sea-bed				Water Column		
Mining Approaches	Mining Systems	Fragmentation/ Collection	Excavation	Subsidence	Resedimen-tation	Turbidity Plume	Suspended Particulates	Dissolved Substance
Scraping	Drag line dredge	X			X	X	X	X
	Trailing suction dredge	X			X	X	X	X
	Crust-miner	X			X	X	X	X
	Continuous line bucket	X			X	X	X	X
Excavating	Clam shell bucket		X		X	X	X	X
	Bucket ladder dredge		X		X	X	X	X
	Bucket wheel dredge		X		X	X	X	X
	Anchored suction dredge		X		X	X	X	X
	Cutterhead suction dredge		X		X	X	X	X
	Drilling & blasting		X					
Tunneling beneath sea-floor	Shore entry			X				
	Artificial island entry			X				
Fluidizing sub-sea-floor	Slurrying			X				
	Leaching							

[1] X = Disturbance expected. Magnitudes of the disturbances are also dependent on the nature of the mineral deposits and the amounts of fine particles present on the seabed. Closed circulation systems are assumed for slurrying and leaching. Open circulation would result in plumes and resedimentation.

TABLE 6. HAWAIIAN ELECTRIC INDUSTRIES: ENERGY SCENARIOS FOR THE YEAR 2000

	Scenario 1	*Scenario 2*
Municipal Solid Waste	7.0	7.0
OTEC	5.0	5.0
Wind	6.0	6.0
Bagasse	0.1	0.1
Fossil fuels	82.0	30.0
Geothermal	0.0	52.0

FIGURE 1. MOBILE OFFSHORE DRILLING UNIT

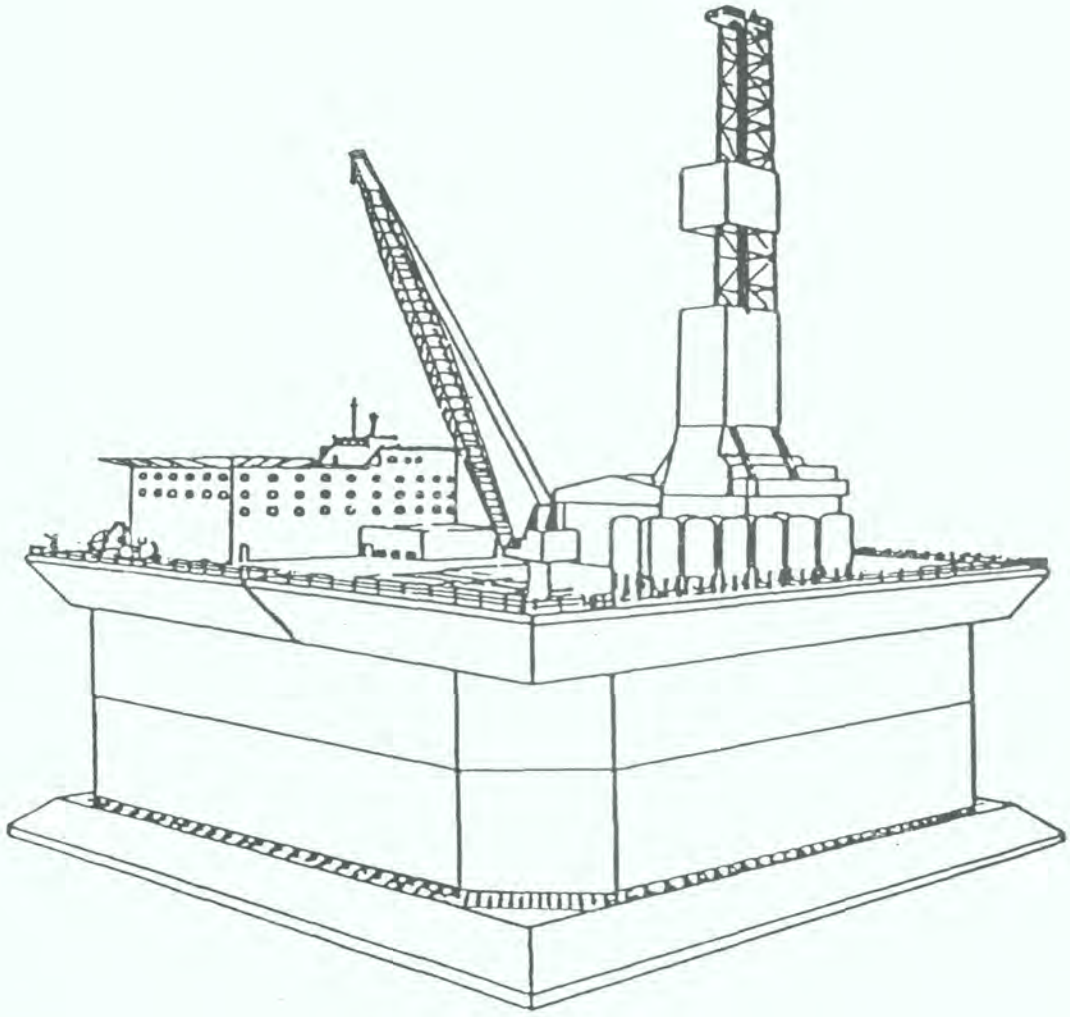
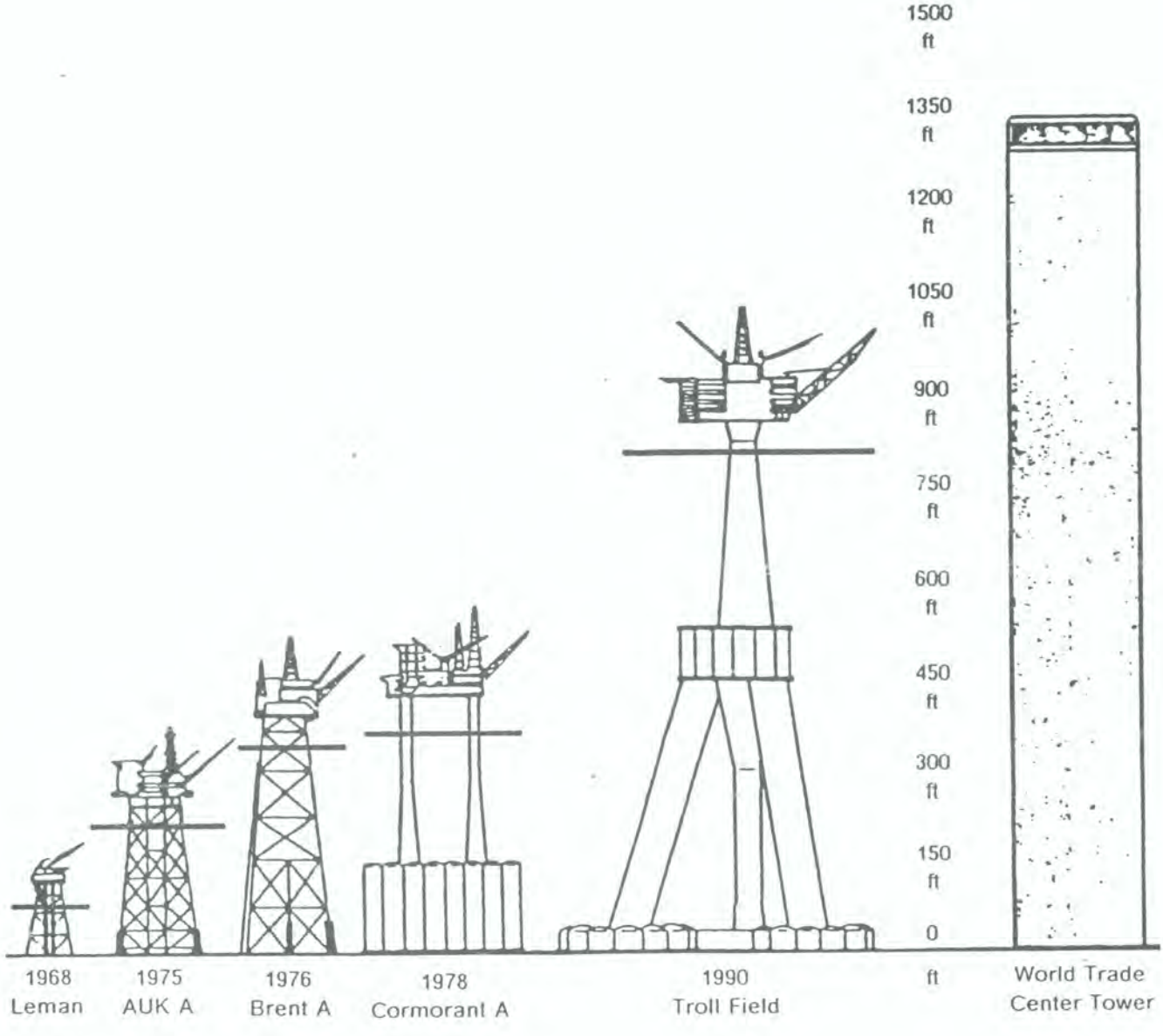
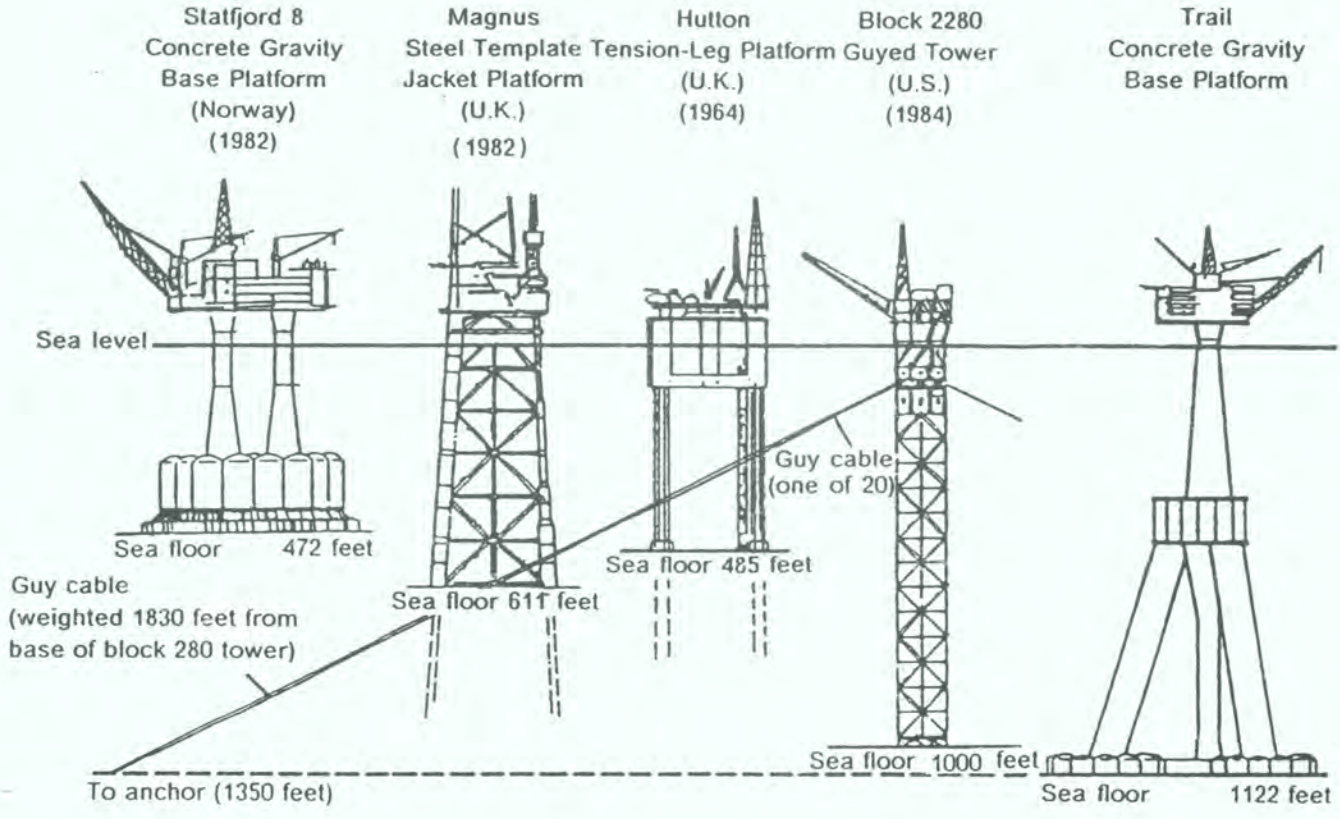


FIGURE 2. PROGRESSION OF PRODUCTION PLATFORMS FOR THE NORTH SEA



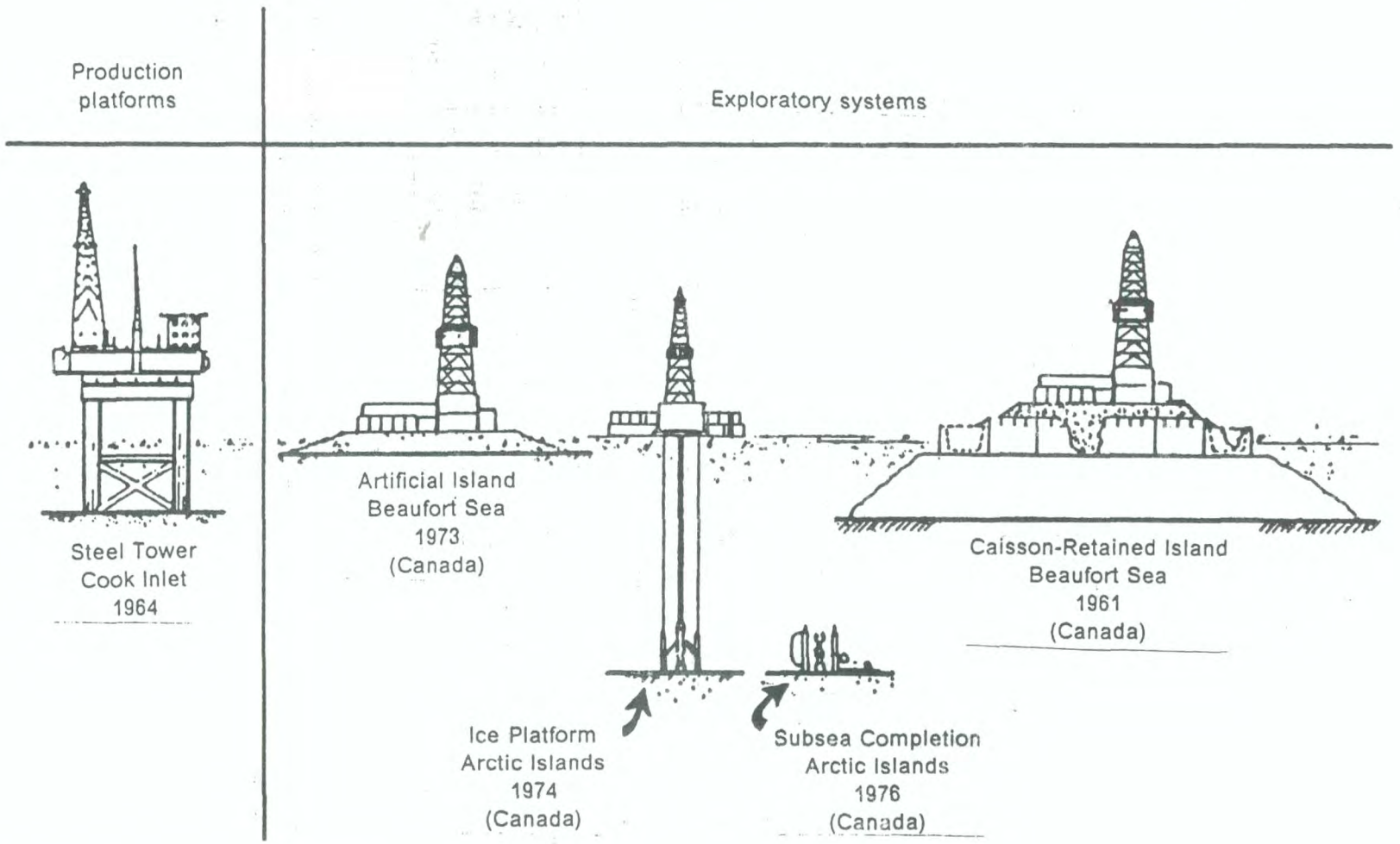
SOURCE: Shell Oil Co.

FIGURE 3. PRODUCTION PLATFORM TECHNOLOGIES FOR FRONTIER AREAS



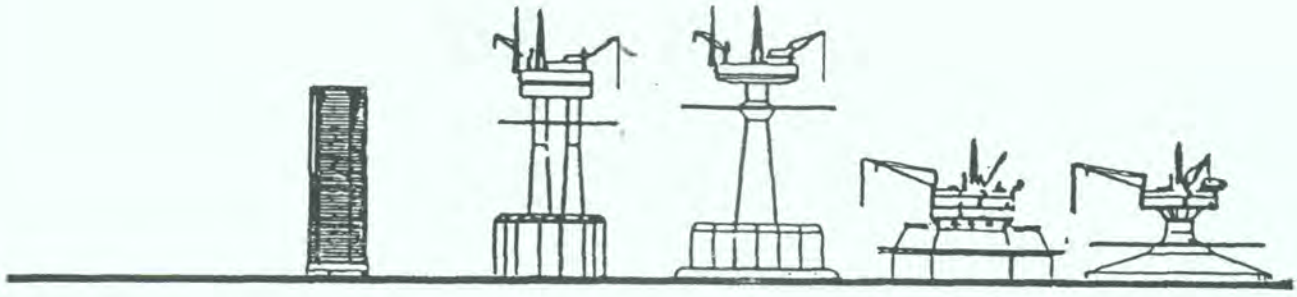
SOURCE: Redrawn from *Scientific American* April 1982

FIGURE 4. ARCTIC EXPLORATION AND DEVELOPMENT MILESTONES



Redrawn from Proceedings of DOI EEZ Symposium, November 1983

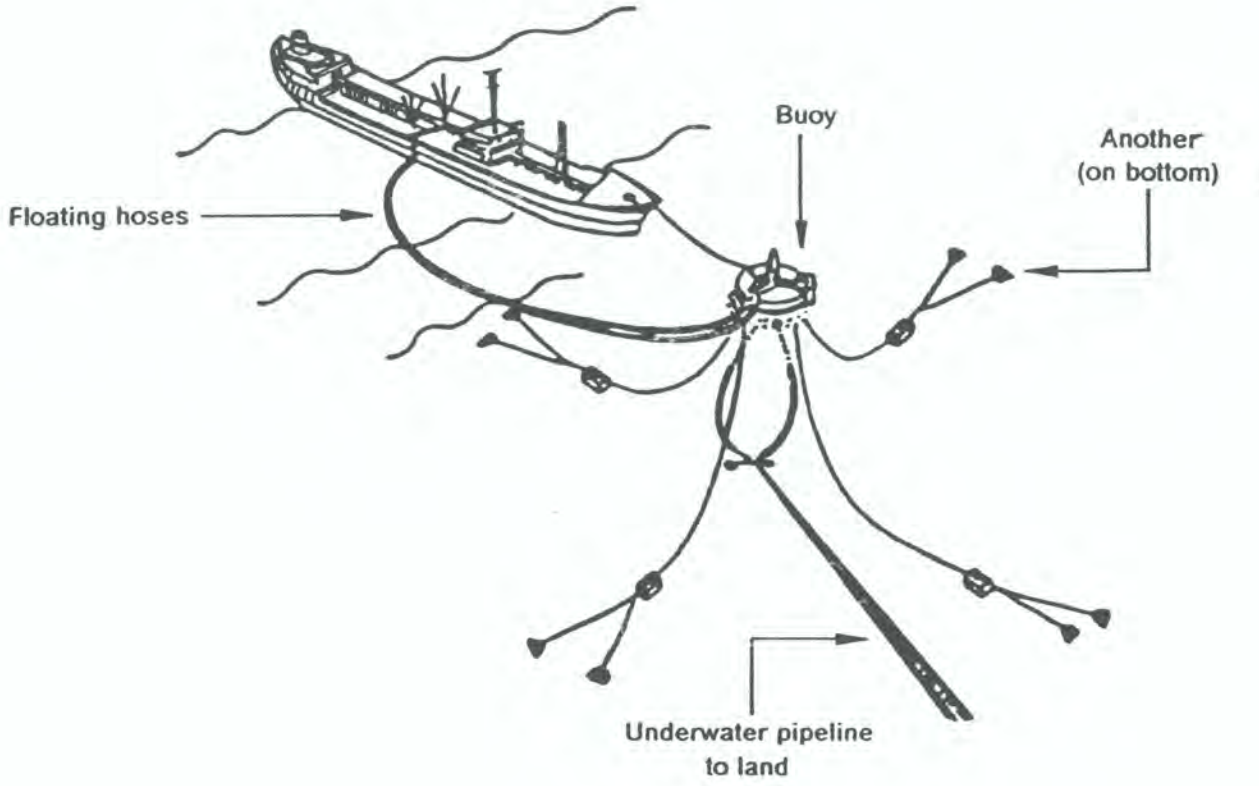
FIGURE 5. ENVIRONMENTAL LOAD COMPARISON FOR REPRESENTATIVE GRAVITY STRUCTURES



	50 story building	North Sea condeep	Bering Sea monotower	Norton Sound concrete island	Beaufort cone
Water depth (ft)	0	300 - 500	300 - 600	45 - 90	50 - 200
Typical horizontal loads (10^6 lbs)					
• Wind	1 - 5	1	1	1	1
• Earthquake	1 - 5	30	70	100	50
• Waves	—	100	70	100	30
• Ice	—	—	10	100	100 - 200

SOURCE: Redrawn from Hans O. Jahns, Offshore Outlook Technological Trends American Arctic. Offshore Mechanics and Arctic Engineering Symposium (Dallas, Texas. February 1985)

FIGURE 6. A TYPICAL CONFIGURATION OF AN OFFSHORE, SINGLE-POINT MOORING SYSTEM.
(Adapted from Draft Environmental Impact Statement, Maritime Administration Tanker Construction Program, Vol. 1, NTIS Report No. EIS-730-392D).



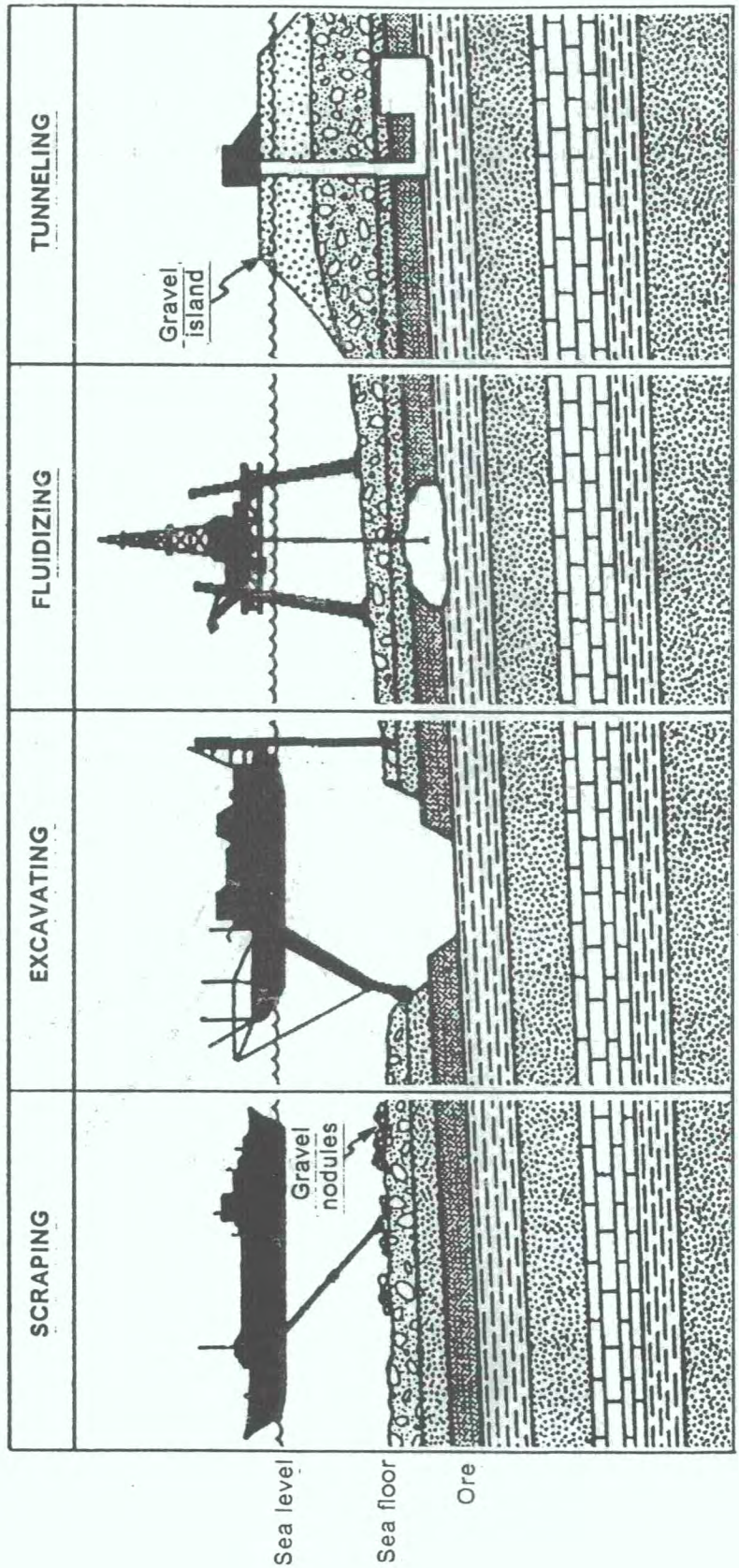


FIGURE 7. THE FOUR APPROACHES TO MINING SOLID MINERALS

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ANNEX III

DISPOSAL OF DREDGED MATERIAL

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WATERWAYS EXPERIMENT STATION
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INTRODUCTION

1. Navigable waterways of the world are vital to the economic growth of coastal nations and are the obvious link in international trade. Coastal ports and waterways are rarely naturally deep, and navigable depths to support shipping must be maintained by dredging. Annual dredging on a global basis results in hundreds of millions of tonnes of dredged material that must be disposed of and managed in an economically and environmentally sound manner. Because the annual cost of port and waterway maintenance ranges in the hundreds of millions of dollars, the least costly modes of disposal are sought, and sea disposal is often an economically disposal solution.

2. Many of the waterways are located in industrial and urban areas, and the sediment to be dredged is often contaminated with wastes from these sources. Consequently, disposal of the contaminated sediment has generated serious concern that such operations may adversely affect water quality and aquatic organisms. Much research into these effects has been completed over the last decade, and the results summarized in this annex. The Permanent International Association of Navigation Congresses (PIANC) has developed general technical guidance (PIANC, 1986) on sea disposal for the use of their global organization. Regulatory guidance, however, is promulgated through the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, 1972 (the London Dumping Convention - LDC), and is discussed later in this annex. LDC Member States then carry out domestic regulatory programmes that incorporate the constraints of the LDC.

3. Contaminated or otherwise unacceptable material accounts for only a small per cent of the total dredging, leaving a large quantity of material that may be disposed of at the broadest range of alternative sites, with concern only for its physical impacts. In the United States, significantly contaminated, or unacceptable, dredged material in the coastal zone accounts for less than 10 per cent of the total dredged (U.S. Congress, 1987). Global distribution of contaminated material probably follows a relatively similar pattern. The following report will primarily deal with contaminated material, as the greatest concern rests with that fraction. Acceptable or clean material can be used for numerous purposes, and the reader is referred to U.S. Army Corps of Engineers (1987) for a thorough discussion of beneficial uses of dredged material.

I. BIOGEOCHEMICAL COMPLEXITY

4. To appreciate the impact of sediment discharge and resuspension on water quality and aquatic organisms, one must understand how chemical constituents, which may have various effects on aquatic organisms, are associated with dredged sediments.

5. The following discussions only hint at the complexity of the chemical composition and at its variability within and among sediments. Detailed discussions of sediment chemistry, biology, and water quality interrelations are available in the literature (Lee *et al.*, 1975; Brannon *et al.*, 1976a; Chen *et al.*, 1976; Gambrell *et al.*, 1977; Burks and Engler, 1978; DiSalvo and Hirsh, 1978; Jones and Lee, 1978).

A. ELEMENTAL PARTITIONING

1. Sediment.

6. Sediments can be classified by their composition and mode of transport to the estuarine environment. Among the classifications are detrital and autogenic components, which are the most important in describing the sediment at a proposed dredging site.

7. Detrital components are those that have been transported to a particular area, usually by water. Detrital materials are derived from soils of the surrounding watershed and include mineral grains and rock fragments (soil particles) as well as stable aggregates, associated organic material, and culturally contributed components derived from agricultural run-off and industrial and municipal waste discharges.

8. Autogenic components are those that are formed in place or have not undergone appreciable transport. These materials are generally the products of aquatic organisms and include shell material ($CaCO_3$), diatom frustules (SiO_2), some organic compounds and products of anaerobic or aerobic transformations.

2. Interstitial water.

9. In considering the *in situ* association of trace elements in estuarine sediment with various sediment phases, the water contained in interparticle voids or interstices must be considered. This

is termed interstitial water (IW). In relation to the overlying water, chemical constituents may frequently be enriched in the IW by several mechanisms.

10. Metals and some nutrients are ionically bound to the sediment in several exchange locations, including exchange sites of the silicate phase, and are associated with organic matter or trace elements complexed with the organic phase. Man-made organics such as polychlorinated biphenyls (PCBs) may be physically attached to these highly active silicate materials. Only a small amount of these poorly, or slightly, soluble constituents is found dissolved in the IW.

11. Heavy metals are associated with hydrated manganese and iron oxides and hydroxides that are present in various amounts in sediment. Another location for heavy metals is in the sediment/organic phase. The metals are incorporated into living terrestrial and aquatic organisms and are relatively stable; however, they may be released to the water column during decomposition.

12. The greatest concentration of most inorganic chemical constituents is contained in the silicate mineral fraction (earth's crustal material) of a sediment.

B. SEDIMENT CHARACTERIZATION

1. Elements and their locations.

13. A particular element or molecule can be present (partitioned) in a sediment in one or more of several locations. Possible locations include the lattice of crystalline minerals, the interlayer positions of polysilicate (clay) minerals, adsorbtion on mineral surfaces, association with hydrous iron and manganese oxides that are hydroxides existing as surface coatings on discrete particles, absorbtion into and adsorbtion on to organic matter existing as surface coatings or discrete particles, and dissolved in the sediment IW.

14. These locations represent a range in the degree to which an element may become released from dredged material when it is discharged to the receiving water of a disposal site, falls through the water column, and settles on the bottom of the water body. The range extends from stable components in the mineral lattices, which are essentially insoluble, to soluble components in the sediment IW, which are readily mobile.

2. Characterization procedures.

15. A sediment characterization procedure to elucidate the phase distribution of contaminants in dredged material must be applicable to many types of marine and freshwater sediment, both aerobic and anaerobic. Electrochemical (Eh, pH) changes, that occur after anaerobic bottom

sediment is disturbed and resuspended, may result in possible solution or precipitation of many elemental species, so the sediment should be thoroughly characterized. Disturbance of samples to be used for sediment characterization must be minimal, so drying, grinding, and contact with atmospheric oxygen are undesirable.

16. The sediment phases are presented in the following paragraph in their relative order of mobility and bioavailability. IW is the most mobile and, consequently, the most available. When contaminants enter a body of water and subsequently enter sediment particulate matter, they normally enter two or three phases in various concentrations that cannot be distinguished from natural levels by bulk or total analysis (Brannon *et al.*, 1976a, Jones and Lee, 1978).

17. From the previous discussion of elemental partitioning and for analytical purposes, the following categories of sediment components should be considered (Brannon *et al.*, 1976a):

a. Interstitial water - IW, an integral part of sediment, is in dynamic equilibrium with the silicate and organic exchange phases of the sediment, as well as with the easily decomposable organic phase.

b. Mineral exchange phase - This is that portion of the element that can be removed from the cation exchange sites of the sediment using a standard ion exchange extractant (dilute HCl, NaCl, $MgCl_2$, etc.).

c. Reducible phase - The reducible phase is composed of hydrous oxides of iron and manganese as well as of hydroxides of iron and manganese, which offer relatively reducing (anaerobic) conditions. Of particular importance are the toxic elements (arsenic, cadmium, cobalt, copper, mercury, and nickel) that may be associated with discrete iron or manganese phases as occlusions or coprecipitates.

d. Organic phase - This phase or partition of elements is that considered to be solubilized after destruction of organic matter. This phase contains very thightly bound elements as well as those loosely chelated by organic molecules. An initial extraction by an organic chelate may be needed to differentiate between loosely bound and tightly bound elements.

e. Residual phase - The residual phase contains primary minerals as well as secondary weathered minerals that are, for the most part, a very stable portion of the elemental constituents. Only an extremely harsh acid digestion or fusion will break down this phase. By far the largest concentration of metals is normally found in this fraction.

C. EFFECTS OF GEOCHEMICAL CHANGES

18. A number of studies of chemical-constituent release mechanisms have evaluated conditions that enhance the release of toxic metals when the sediment/water geochemical environment is drastically changed (Burks and Engler, 1978). As an example, the significant release of zinc to the water-soluble phase was shown to occur at pH5 under oxidizing (Eh) conditions. It must be emphasized that such acid-oxidizing, pH-Eh conditions do not normally occur in open-water disposal because anaerobic sediment generally remains near neutral pH and the oxidation processes that occur in the water column are not such as to result in an acidic condition (Brannon *et al.*, Jones and Lee, 1978).

19. When dredged material settles after open-water disposal, it normally returns to an anaerobic and near-neutral pH condition. On the other hand, if this dredged material were placed in an upland containment area where oxidizing conditions could occur for a year or more, and if the dredged material were high in total sulphide (common in many fine-grained estuarine sediments), the pH could become acidic and result in significant release of some contaminants to the water-soluble phase (Burks and Engler, 1978). Therefore, judicious selection of the disposal mode (open water versus upland) and an understanding of the long-term implications of either disposal mode are very important.

20. As the geochemical environment changes from aerobic to anaerobic or vice versa, a somewhat orderly sequence of physicochemical events occurs. In oxygen-saturated (aerobic) conditions, the chemistry of various constituents is dominated by the oxides of the various chemical forms (e.g., nitrogen-nitrates, sulphur-sulphates, manganese-manganic, iron-ferric, carbon-carbonate, phosphorus-phosphate). If the environment becomes increasingly anaerobic, the chemical constituents are biochemically and chemically reduced to the forms most stable under oxygen-free conditions: nitrate-N_2, manganic-manganous, ferric-ferrous, carbonate-methane, and sulphate-sulphide. The last sequence represents the most intense anoxic conditions. This sequence is somewhat reversed when an anoxic sediment is aerated and oxidized.

II. TOXICOLOGY/BIOCHEMICAL ASSESSMENT

21. The toxicology of dredged material is much too site-specific to review on a global basis. Effects are dominated by local pollution-control measures or the lack thereof, local watershed characteristics, geomorphology, geochemistry, etc.. Consequently, this section will discuss the various toxicological techniques used in the U.S.A. to assess the hazard potential of dredged material. Various tests of pollution potential are required by legislative mandate (Committee on Public Works, 1973) and range from simple water leaches (U.S. ACE, 1976) to multi-organism benthic bioassays (U.S. EPA/U.S. ACE, 1977).

A. ELUTRIATE TEST

22. The Elutriate Test (U.S. ACE, 1977), which uses one part sediment to four parts leaching water, has been in use since 1973 and has been evaluated under an extremely wide range of conditions in marine, estuarine, and freshwater systems. Sediment is collected from the proposed dredging site, and water is normally collected from the proposed discharge site. These sites may be close enough to one another to be essentially the same area.

23. In a definitive review, Lee and Plumb (1974) concluded that the Elutriate Test was potentially useful for evaluating the short-term release of contaminants from dredged material discharged to open water. Further laboratory investigations (Lee *et al.*, 1975) pointed out that the oxygen status and the solid:liquid ratio during the test procedures were the most important factors influencing test results. It was found that the 1:4 ratio offered reliable results (Lee *et al.*, 1975), while aeration of the elutriate best simulates water-column conditions at most open-water sites, if it is known that anoxic conditions will not occur at the disposal site. Field verification investigations (Jones and Lee, 1978) have shown the Elutriate Test to be an excellent predictor of releases noted in the field, and to be environmentally conservative when used in conjuction with water quality criteria.

24. The usefulness of the Elutriate Test in projecting the long-term release of certain contaminants from resettled dredged material was demonstrated on sediment from United States harbours by Brannon *et al.*, 1978.

B. BULK OF TOTAL SEDIMENT ANALYSIS

25. Bulk sediment analysis refers to the harsh acid or solvent extraction of a sediment that results in a total or near total concentration of a chemical constituent to be measured. Lee and Plumb (1974) concluded that bulk sediment analysis was not adequate to assess water quality effects and would not result in any level of environmental protection. Numerous other reviewers and investigators have come to the same conclusion (Lee *et al.*, 1975; Brannon *et al.*, 1976a, b; Brannon, 1978; Brannon *et al.*, 1978; Burks and Engler, 1978; DiSalvo and Hirsh, 1978; Jones and Lee, 1978). Brannon *et al.* (1978) and Jones and Lee (1978) have shown conclusively that bulk sediment analyses cannot be used to predict long- or short-term release of contaminants, and DiSalvo and Hirsh (1978) demonstrated that no relationship exists between bulk sediment concentration and bio-accumulation by aquatic organisms. The bulk or total sediment analysis is, however, a useful tool in conducting an inventory of sediment contaminants. The inventory can provide a basis for further testing (bioassay/bioaccumulation) or to provide a relative ranking among sediment sample locations or various sites to be dredged. For natural constituents (e.g. metals and nutrients) a comparison can be made with natural background levels to estimate anthropogenic input.

C. BIOASSAY

26. The biogeochemical complexity of a sediment, the sediment fractions to which contaminants may associate, and the need for an effect-based ecological assessment that integrates multiple contaminants, necessitate the use of bioassay/bioassessment techniques. Moreover, an evaluation of contaminant toxicity, bioavailability and persistence on an acute and chronic basis is required by the LDC and is part of the U.S. evaluation process in determining acceptability for ocean disposal. The following paragraphs describe in general terms the types of bioassessments that may be carried out on dredged material.

1. Liquid phase.

27. After filtration through 0.45-μ filter or equivalent centrifugation, the filtrate or liquid phase may be subjected to bioassay by a relatively diverse group or organisms (U.S. EPA/U.S. ACE, 1977; Shuba *et al.*, 1977; 1978). Bacteria and protozoans have been found unsuitable for routine assays, and their use should be discouraged (Shuba *et al.*, 1977). Phytoplankton is not recommended as test material because of its natural dynamic variability and should only be used as a special case. Zooplankton was found to respond adequately and may be used to assess stimulation or toxicity. The liquid-phase bioassay should include a

plankton species, a crustacean or mollusc, and a fish as test organisms (U.S. EPA/U.S., ACE, 1977).

28. Experimental design and replication are required for this and all phase bioassays (U.S. EPA/U.S. ACE, 1977). An estimation of mixing and dilution that is expected to occur on disposal must be factored into the experimental design to simulate field conditions. Since water-column or liquid-phase effects are short-term and intermittent in nature, the time of exposure and liquid-phase concentrations should represent "real world" conditions (Lee and Plumb 1974; DiSalvo and Hirsh, 1978; Jones and Lee, 1978). Shuba *et al.* (1978) noted that when toxic sediment was assayed, the liquid-phase bioassay usually projected the earliest measure of toxicity. Furthermore, chemical-constituent comparisons to water quality criteria and the development of relationships among toxicity, bioaccumulation and water quality criteria should only be made with this phase (U.S. EPA/U.S. ACE, 1977).

29. Bioassays alone cannot be used to make precise estimates or predictions of what actual effect would result in the field from a specific discharge. Consequently, an evaluation of effects other than mortality would only add more uncertainty to interpretation of results. Mortality was therefore chosen as the indicator of potential environmental effects rather than sublethal considerations. There is, however, considerable research under way to develop procedures for sublethal or chronic bioassays for future use in the regulatory programme.

2. Suspended particulate phase.

30. Evaluation of this phase of discharged material may only be addressed through the use of a bioassay (U.S. EPA/U.S. ACE, 1977; U.S. EPA, 1977a). Results of chemical analyses on material in this phase would be no more than an inventory of constituents and would not be useful to project potential water quality problems. The bioassay approach can be used to assess impacts due to the physical presence of suspended particulates and to the biologically active chemical constituents associated with the particulates. Appropriate organism selection is discussed in the literature (U.S. EPA/U.S. ACE, 1977) and is similar to the liquid-phase bioassay. Phytoplankton bioassays with the suspended particulate phase should be discouraged because of the extreme difficulty in interpretation (U.S. EPA/U.S. ACE, 1977). The interpretation of the bioassay of this phase should be based on mortality as the measurable end-point. Sublethal effects should be noted, but could only be judged subjectively as to their potential for harm.

3. Solid phase.

31. The greatest potential for impact generally lies with settleable or solid-phase material that may cause some type of benthic organism impact. The impact might range from simple physical disruption to direct toxicity and bioaccumulation. This phase also has the greatest potential for long-term harm such as sublethal toxicity and bioaccumulation. On discharge this phase does not normally mix with the water column as do the other two phases, and bottom-dwelling organisms can live and feed on the deposited material for long periods (U.S. EPA/U.S. ACE, 1977; DiSalvo and Hirsh, 1978).

32. There are no solid-phase chemical analyses that have shown any promise in predicting potential for environmental harm (DiSalvo and Hirsh, 1978); consequently, the biological approach is used. Regulations require that bioassays be used to evaluate the potential for environmental harm from this phase, and that aquatic organisms be used as analytical tools to determine potential for biochemical impact (U.S. EPA, 1977a). Organism selection has been discussed elsewhere (U.S. EPA/U.S. ACE, 1977) and should include one filter-feeding, one deposit-feeding, and one burrowing species. Organism selection is further refined to comprise a crustacean, an infaunal bivalve, and an infaunal polychaete.

33. Mortality is chosen as the interpretative end-point because of its clear environmental significance. Sublethal effects should be noted if observed, but should not be considered subjectively in terms of potential effects.

34. The initial interpretation of organism mortality may be based on statistical significance at the 95 per cent confidence level rather than at a specific per cent mortality limit (U.S. EPA, 1977a) and is deemed environmentally conservative, as any statistical increase in mortality over controls is considered potentially undesirable. Professional judgement is then used to interpret the magnitude of mortality if the differences are statistically real, in order to formulate decisions on the acceptability of the material. Any mortality less than 10 per cent is considered to be lacking in ecological significance (U.S. EPA/U.S., ACE, 1977; Prater and Anderson, 1977a). This approach does not attempt to explain the ecological meaning of the toxicity, but it does assume that mortality may be adverse on extrapolation of laboratory results to field conditions.

35. This approach and subsequent interpretation to evaluate organism mortality is at the forefront of the state-of-the-art, and developing research has considered this and related techniques (U.S. EPA/U.S. ACE, 1977; Prater and Anderson, 1977a, b; DiSalvo and Hirsh, 1978; Shuba *et al.*, 1978; Swartz *et al.*, 1979; Tsai *et al.*, 1979; Prater and Hoke, 1980a, b; Rubinstein *et al.*, 1980, 1983; Engler *et al.*, 1981, Alden and Young, 1982; Roberts *et al.*, 1982; Seeley *et al.*, 1982; Peddicord and Hansen, 1983).

D. BIOACCUMULATION AND BIOMAGNIFICATION

36. Even though bioaccumulation may occur from exposure to any of the three phases, the liquid and suspended particulate phases are considered of secondary concern because of short contact time between these phases and aquatic organisms. The solid phase, however, must be assessed for the long-term bioaccumulation potential because of the long-term organism/sediment interaction that occurs after disposal. In the regulatory programme, the presence of an animal in or on the disposed material must be directly related to evaluation of body burdens of specific chemical constituents when compared to reference animals.

37. Because of the long-term concerns of bioaccumulation and the short-term nature of the laboratory bioassays (10-day duration), field evaluation of the bioaccumulation in site-specific organisms should be used wherever there is a historical precedent of disposal at the site in question (U.S. EPA/U.S. ACE, 1977). Most aquatic disposal sites have been used traditionally for discharge operations, and a valid historical precedent for disposal probably exists in most regions. Under these conditions, the animals have lived and reproduced on material from past dredging and disposal operations; if bioaccumulation is occurring, future disposal of similar contaminated material should be assessed carefully. For new disposal areas, or for dredged material recently contaminated, laboratory investigations must be designed to simulate field conditions as closely as possible, and to consider time of exposure.

38. Interpretation of bioaccumulation data is even more difficult than the interpretation of toxicity tests results. Many toxic metals are required nutrients, and several others are consumed with no apparent harmful effect to the organism (DiSalvo and Hirsh, 1978). On the other hand, the uptake or bioaccumulation of certain toxins by human food resource organisms is obviously detrimental, and the interpretation must have a margin of environmental safety. Interpretation on bioaccumulation is initially based on statistically significant (95 per cent confidence level) differences in the body burden of specific constituents between organisms at the dump site and the same species living on uncontaminated sediments of similar sedimentological characteristics (U.S. EPA/U.S. ACE, 1977). It must be realized, however, that a statistically significant difference cannot be presumed to predict the occurrence of an ecologically important impact. Professional judgement must be used to interpret the magnitude of bioaccumulation.

39. Research by de Kock and Marquenie (1982) has shown direct relationships between contaminated sediment in polluted estuaries to bioaccumulation by aquatic organism; however, the researchers did not separate test results according to the route of uptake (e.g., from sediment or from overlying waters contaminated by other sources or from the sediment itself). This research points out the significant loadings of various contaminants (especially metals) that are associated with sediment movement, either natural or man-induced.

40. Only a few toxins have established human-consumption limitations, and true biomagnification has not been clearly documented in aquatic systems. Biomagnification, an increase in body burden through trophic levels resulting in significant increases without relation to water or sediment concentration, has been suggested to be of concern in the aquatic system. After significant study, Kay (1984, 1985) concluded that, with the possible exception of mercury, biomagnification does not occur in the aquatic environment, but that it is a real concern in terrestrial systems. Significant research is under way in the U.S.A. to develop more meaningful association between sediment contaminants, body burdens and effects. The research has as a goal the development of accurate and reliable pre-dredge predictive testing protocols.

III. ENVIRONMENTAL EFFECTS ASSOCIATED WITH AQUATIC DISPOSAL

A. BACKGROUND

41. Waste and run-off from man's industrial, urban, and agricultural activities have contaminated sediment in many waterways and harbours. As these activities increased, concern developed that dredging and disposing of such sediment would adversely affect water quality or aquatic organisms. A number of local studies were made prior to 1970 to investigate the environmental impact of specific disposal practices and to explore alternative disposal methods (Boyd *et al.*, 1972). However, these early studies did not provide sufficient definitive information on the environmental impact of current disposal practices, nor did they fully investigate alternative disposal methods.

42. In the River and Harbor Act of 1970, Congress authorized the U.S. Army Corps of Engineers to initiate a comprehensive nationwide study of the environmental impact of dredging and dredged material disposal operations and to develop new or improved dredged material disposal practices. A detailed planning document for the Dredged Material Research Programme (DMRP), a description of the technical and management structure and a summary of the programme, and a DMRP publications index and retrieval system are available in Boyd *et al.* (1972), Saucier *et al.* (1980), and Herner and Co. (1980), respectively.

43. To those concerned with national or regional planning, regulation and policy formulation, there are two extremely important fundamental conclusion that can be drawn from the results of the DMRP. The first conclusion is that there is no single disposal alternative that presumptively is most suitable for a region, for a type of dredged material, or for a group of projects. Correspondingly, there is no single disposal alternative that presumptively results in impacts of such a nature that it can be categorically dismissed from consideration. Put in different terms, there is no inherent effect or characteristics of an alternative that rules it out of consideration from an environmental standpoint prior to specific on site evaluation. This holds true for open-water disposal, confined upland disposal, habitat development, or any other alternative.

44. Specific on site evaluation means that each project must be considered on a case-by-case basis. It is not technically sound, for example, to make the general statements that ocean disposal must be phased out or that all material in the Great Lakes area classified as

polluted must be confined behind dikes. To do this would be contrary to research results that have indicated that there can be situations where there is greater probability of adverse environmental impacts from confined disposal than from open-water disposal and other situations (such as when certain types of contaminants are present) where confined disposal might provide the greatest degree of environmental protection.

45. The implications of the first conclusion from a management point of view are fully recognized. Case-by-case evaluations are time-consuming and expensive, and may seriously complicate advanced planning and funding requests. Nevertheless, from a technical point of view, situations can be envisioned where tens of millions of dollars may have been or could be spent for alternatives that contribute to adverse environmental effets rather than reduce them.

46. The second basic conclusion is that environmental considerations are acting more strongly than possibly any other force in making long-range regional planning necessary as a lasting, effective solution to disposal problems. No longer can disposal alternatives be planned independently for each dredging operation for multiple projects in a given area. While each project may require different specific solutions, the interrelationships must be evaluated from a holistic perspective, and thought must be given to replacing particular disposal alternatives as conditions change. Regional disposal management plans not only offer greater opportunities for environmental protection, ultimately at reduced project costs, but also meet with greater public acceptance once they are agreed upon.

47. It is the general intent of the following discussion to demonstrate that no category of disposal alternatives is without environmental risk, reflects the best management practice, or offers the soundest environmental protection. It is therefore proposed that all specific disposal alternatives be fully investigated during the planning process, and be treated on an equal basis until a final decision is made that is based on all available facts (National Advisory Committee on Oceans and Atmosphere, 1981). Lastly, it is hypothesized that it is feasible, even for the most highly contaminated dredged material, to consider all disposal alternatives through application of adequate management plans, and that such controlled disposal would be legally acceptable under domestic regulations and international treaties.

B. LABORATORY INVESTIGATIONS

48. Investigations for the development of regulatory testing and criteria have shown conclusively that no relationship exists between sediment characteristics determined by the so-called "bulk" or "total sediment analysis" and the effects of aquatic disposal on water quality or aquatic organisms (Engler *et al.*, 1974; Lee and Plumb, 1974; Fulk *et al.*, 1975; Lee

et al., 1975; Blom *et al.*, 1976; Brannon *et al.*, 1976a, b; Chen *et al.*, 1976; Shuba *et al.*, 1976; Gambrell *et al.*, 1977; Jones and Lee, 1978). However, these investigations did show that the Elutriate Test (Keeley and Engler, 1974; Engler, 1976), can be used to predict water quality perturbations and water-column biological impacts (Lee and Plumb, 1974; Brannon *et al.*, 1976a, b; Jones and Lee, 1978).

1. Water quality.

49. Sources of comprehensive information on sediment chemistry, biology, and water quality interrelationships include Lee *et al.* (1975), Brannon *et al.* (1976a), Chen *et al.* (1976), Gambrell *et al.* (1977, 1978), Burks and Engler (1978), DiSalvo and Hirsh (1978), and Jones and Lee (1978).

50. Results of laboratory investigations of the impact of disposal on water quality showed that ammonium, iron, manganese and orthophosphate were released from anaerobic sediment during simulated disposal and initial mixing in the water column and after the sediment had settled to the bottom (Blom *et al.*, 1976; Chen *et al.*, 1976; Burks and Engler, 1978; Jones and Lee, 1978). It was found, however, that when contaminated fine-grained harbour sediment was dispersed in a water column, the sediment scavenged or cleaned the water column of numerous toxic heavy metals and nutrients.

51. Negligible release of chlorinated hydrocarbons to the soluble phase was detected during simulated open-water disposal of dredged material from a broad selection of marine, freshwater, and estuarine sediment (Fulk *et al.*, 1975; Chen *et al.*, 1976; Burks and Engler, 1978). After the sediment settled and formed a new sediment/water interface, several patterns of constituent release and immobilization were detected (Fulk *et al.*, 1975; Blom *et al.*, 1976; Burks and Engler, 1978). With the exception of iron, manganese, and some nutrients, release from the sediment to the water column was extremely small. Several toxic metals were released in concentrations less than one part per billion from both contaminated and uncontaminated sediment. It must be emphasized that these processes and transformations occur naturally in all sediments at somewhat similar levels and do not appear to be of a significant nature (Blom *et al.*, 1976; Chen *et al.*, 1976; and Burks and Engler, 1978).

52. Long-term release studies were conducted by Brannon *et al.* (1978) using sediment from 32 locations in marine, estuarine and freshwater areas that represented broad geographical and pollutional variations. The sediment/water systems were evaluated under various conditions. Under conditions likely to prevail at aquatic disposal sites, organic carbon, orthophosphate, and zinc exhibited consistent net releases to the water column. Even so, the sediment (some highly contaminated) would not be expected to cause significant water quality problems.

2. Sediment solid fraction.

53. Research has suggested little release of most chemical constituents from dredged material, further emphasizing the need for determining the biological effects of chemicals associated with the sediment solid fraction. Regardless of the chemical nature of the solid fraction, the physical effect on various organisms must also be thoroughly evaluated. Investigations to determine the effects of turbidity (suspended dredged material) on aquatic organisms, the ability of organisms to migrate vertically through deposits of dredged material, organism uptake of sediment-sorbed metals and pesticides, and the biological effects of sediment contaminated with a wide range of pollutants are discussed in the following paragraphs. Results of related field studies are included when appropriate.

(i) *Turbidity*

54. Turbidity studies conducted by Peddicord *et al.* (1975), Peddicord and McFarland (1978), and McFarland and Peddicord (1980) using adult marine, estuarine, and freshwater organisms have shown lethal concentrations of suspended dredged material to be an order of magnitude or more higher than maximum water-column concentrations observed in the field during dredging operations (Stern and Stickel, 1978; Wright, 1978). In laboratory investigations, the mortality of selected organisms was demonstrated at concentrations of suspensions of dredged material exceeding 2 to 20 g l^{-1} (2,000 - 20,000 ppm) at 21-day exposure times. Field observations following disposal operations have shown turbidity or suspended particulate levels to be less than 1 g l^{-1} (1,000 ppm) that persisted for exposure times of only hours. Based on these and other observations, it was concluded (1) that the physical effect of turbidity from dredged material discharge in open water would be of minimal impact and (2) that the primary impact of turbidity is of an aesthetic nature and must be controlled and treated as such. The only exception to this conclusion would be the sensitive coral reefs of tropical regions, where low concentrations of suspended particulates would significantly impact large areas.

55. Peddicord and McFarland (1978) conducted studies using harbour sediment chosen for physical similarity to bentonite suspensions used in previous turbidity studies (Peddicord *et al.*, 1975) in order to assay for impacts due to chemical properties of the sediment in suspension. Measurements were carried out using sediment from relatively uncontaminated reaches of the San Francisco Bay, and compared with measurements using more highly contaminated bay sediment.

56. Organism response did not differ greatly between pure mineral suspensions and uncontaminated natural sediment. The most sensitive species tested, striped bass (*Morone saxatilis*), survived only a few hours at levels of 0.5 g l^{-1} of contaminated sediment, a condition probably representing the worse-case turbidity generation associated with a dredged material disposal operation. Such conditions are very unlikely to occur in the field, where

motile organisms may escape turbidity maxima and where water currents disperse sediment as the sediment settles out of the water column.

57. Chemical analyses of several species for heavy metals, pesticides, and PCBs indicated minimal uptake of some of the contaminants, but none were accumulated to levels that appeared to be sufficient to influence the survival of the exposed organisms (Peddicord and McFarland, 1978). Moreover, body burdens or tissue levels did not exceed those levels generally considered acceptable for human consumption of the edible portions. However, in this study, Peddicord and McFarland experienced analytical difficulties that limited comparisons to relative differences among treatment variables and precluded absolute comparisons between any two of them. Such difficulties in interpreting chemical data argued for developing assay techniques to evaluate total toxicity of a sediment regardless of specific toxicants.

(ii) *Vertical migration*

58. Other physical-impact investigations have evaluated the ability of estuarine and freshwater benthic organisms to move vertically after being covered or smothered by various loadings of dredged material (Maurer *et al.*, 1978, 1981a,b; 1982). The laboratory evaluations demonstrated that selected organisms (clams, crabs, and benthic worms) were able to recover through as much as 1 m covering or were smothered by as little as a few centimetres covering of different types of dredged material. The organisms generally emerged from the deposits in a matter of hours, and minutes in some cases. These studies investigated combinations of sand dredged material deposited on mud and sand substrates and mud dredged material on mud and sand substrates. The most dramatic effect was noted when dissimilar dredged material was placed on either substrate. Impacts were highest when a sand dredged material was placed on a mud substrate, covering normally mud-dwelling organisms that were not suited for mobility through the sand. The same was true when sand-dwelling organisms were quickly smothered by a mud covering. Judicious selection of disposal sites where sand is placed on a sand bottom or mud bottom is imperative to minimize immediate or long-term physical impact at the site.

59. Site-specific field studies demonstrated that benthic organism recolonization of dredged material mounds formed during disposal was relatively rapid and the processes were attributed in some part to vertical migration (Oliver *et al.*, 1977). However, a significant number of organisms also may have been deposited with the dredged material and affected recolonization patterns.

(iii) *Metal uptake*

60. Metal availability and accumulation studies were conducted by Neff *et al.*, (1978) using the clam *Rangia cuneata*, the grass shrimps *Palaemonetes pugio* and *P. kadiakensis*, and the worms *Neanthes aracneodentata* and *Tubifex sp.*. Test sediment was taken from

Texas City and Corpus Christi, Texas, ship channels (15 and 30 per thousand salinity, respectively) and the Ashtabula River in Ohio (fresh water). Metals routinely measured were cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, and zinc.

61. For most metals studied by Neff *et al.* (1978), no uptake by organisms was evident. However, when uptake was shown to occur, the levels often varied from one sample period to another and were quantitatively marginal, usually being less than one order of magnitude greater than levels in the control organisms even after 1 month of exposure. Even though some sediment exhibited high levels of some metals, it is not valid to compare metal levels in organisms to total sediment chemical concentrations since only a variable and small amount of the sediment-associated metal is biologically available. In addition to not knowing the amount of metal available for biological uptake, animals in undisturbed environments may have naturally high and fluctuating metal levels. Therefore, comparisons to evaluate bioaccumulation should be made between control and experimental organisms at the same point in time.

62. Of a total of 168 animal/salinity combinations evaluated in the tests carried out by Neff *et al.* (1978), only 22 per cent showed significant accumulation due to sediment exposure. The largest uptake was of iron, a metal generally known for its low degree of toxicity in biological systems, but significant accumulations of lead were seen in a few of the short-term exposures, although these results could not be duplicated in long-term exposures. Relatively high uptake of lead occurred only in the polychaete *Neanthes* and was interpreted to be of potential ecological significance for this species. A literature search by Neff *et al.* (1978) showed that heavy metals in solution vary over several orders of magnitude in availability to benthic invertebrates. Although accumulations of heavy metals by organisms from the water has been documented, the literature shows no such clear evidence for accumulation of metals from sediments.

63. Neff *et al.* (1978) also investigated the depuration of heavy metals after the organisms were removed form the test sediment. In those 37 cases where there was uptake after an 8-day exposure, depuration during two or eight days in clean water was seen in seven instances, with the other 30 cases showing no decrease in metal concentration in the tissues.

64. The study by Neff *et al.* (1978) indicated that the chemical form of metals had important effects on their bioavailability. Elevated concentrations of heavy metals in tissues of benthic invertebrates were not always indicative of high levels of metals in the ambient medium or associated sediment. Although a few instances of uptake were seen to be of possible ecological significance, diversity of results among species, different metals, types of exposure, and salinity regimes strongly argued that bulk chemical analysis of sediment for metal content could not be used as a reliable index of metal availability and potential ecological impact of dredged material. Their work indicated that a biological assessment was necessary.

65. Neff *et al.* (1978) performed sequential and non-sequential chemical extractions on the sediment to evaluate the potential mobility of metals in different chemical forms and determine the total metal concentration in the sediment. For some species a correlation did exist, and for others a correlation did not exist, between any chemical or physical form studied and bioaccumulation of the metal. The authors stated:

> "At present, it does not appear that a simple extraction scheme can be developed that might indicate availability of sediment sorbed metals by benthic organisms. Additional data, based on a large number of different sediment types, may indicate, however, forms most likely to be accumulated by benthic organisms."

66. For some metals, there was apparent correlation between metal concentration in the sediment and in the associated infaunal and epifaunal macrobiota (Neff *et al.*, 1978). For other metals, no such correlation existed. These correlations often vary with sediment type. The correlation, when it occurs, may be due to direct or indirect transfer of metals from sediment to biota or it may represent the presence of a source of metals common to both the sediment and biota. Anderlini *et al.* (1976a) concluded that, if changes in metals in the water occurred as a result of dredging activities, the changes were either less than small natural fluctuations or were of short duration.

67. Both Neff *et al.* (1978) (short-term laboratory studies and literature reviews) and Anderlini *et al.* (1976a, b) (longer-term field work and back-up laboratory experiments) found the same heavy-metal phenomenon: the accumulation and release of certain heavy metals seem to vary with the metal, with the species, between sampling times, between sampling sites (dredged and not dredged), and within control sites or organisms or both. These variable results have not been directly correlated with dredging operations or sediment loading.

68. A field study that supports the laboratory results of Neff *et al.* (1978) was carried out by Simms and Presley (1977). These authors concluded that molluscs, crustaceans, and bony fishes from dredged areas of San Antonio Bay, Texas, were lower in almost every heavy metal than organisms from other areas where dredging was minimal. Molluscs were observed to concentrate metals more than any other organisms studied, but the levels observed were much lower than those thought to be lethal or toxic. Except for a few large fish, metal concentrations did not correlate significantly with size or growth stage. Vigorous shell dredging in the bay for 50 years apparently did not cause increases of heavy metals in the tissues of local organisms.

(iv) *Oil and grease*

69. The term "oil and grease" is used collectively in describing all petroleum hydrocarbon compounds in sediment. The petroleum hydrocarbon components are of natural and man-made origin and are primarily fat-soluble. Large amounts of contaminant oil and

grease find their way into the sediment of industrialized waterways either by spillage or as chronic inputs from municipal and industrial effluents, particularly evident near urban areas with major waste outfalls. A literature review demonstrated a broad variety of possible oil and grease components in sediment, the recovery of which was dependent on the type of solvent and methodology used to extract these residues (DiSalvo *et al.*, 1977).

70. The literature suggested long-term retention of oil and grease residues in sediment with minor biodegradation occurring. Trace contaminants, such as the PCBs and chlorinated hydrocarbons (DDT and derivatives), often occur in the oil and grease. Where oily residues of known toxicity become associated with sediments, the sediment retains toxic properties over periods of years, affecting local biota. Spilled oils are readily adsorbed to naturally occurring suspended particulates, and oily residues from municipal and industrial effluents are commonly found adsorbed to particles. These particulates are deposited in benthic sediment and are subject to resuspension during disposal (DiSalvo *et al.*, 1977).

71. Using the Elutriate Test, DiSalvo *et al.* (1977) observed that some soluble hydrocarbon residues were released to the water from sediments known to contain 2,000 to 6,000 ppm total hydrocarbons. Hydrocarbons concentrations in the elutriate (100 to 400 ppb) were from 11 to 400 times higher than background concentrations, yet all were well below acceptable effluent discharge standards. The amount of oil released during the Elutriate Test was less than 0.01 per cent of the sediment-associated hydrocarbons under worst-case conditions.

72. A test scheme was employed in which estuarine crabs (*Hemigrapsus oregonensis*), mussels (*Mytilus edulis*), and snails (*Acanthina spirata*), and a freshwater clam (*Corbicula sp.*) were exposed to contaminated sediment in order to determine magnitudes of uptake of hydrocarbons that were included in sedimentary oil and grease burdens. Overt mortality of test organisms was not directly attributable to exposure to contaminated sediment. Experimental evidence suggested slight uptake of hydrocarbons by salt-water test organisms incubated in the presence of Duwamish River sediment, which contained almost 500 $\mu g\ g^{-1}$ total hydrocarbons. Freshwater clams exposed for 30 days to Duwamish River sediment showed no well-defined uptake of hydrocarbons. Mussels and crabs exposed for four days to New York Harbor sediment containing 2,000 $\mu g\ g^{-1}$ total hydrocarbons showed average uptakes above background of about 50 to 70 $\mu g\ g^{-1}$ (2.5 and 3.5 per cent, respectively), of the sedimentary hydrocarbon concentration, (DiSalvo *et al.*, 1977).

73. The results from DiSalvo *et al.* (1977) indicate that selected estuarine and freshwater organisms experienced only minor mortality when exposed for periods of up to 30 days to dredged material that is contaminated with thousands of parts per million of oil and grease. Uptake of hydrocarbons from the heavily contaminated sediment appears minor when compared with the hydrocarbon content of the test sediment, and, when compared with results describing exposure of uncontaminated organisms under field conditions, total

hydrocarbon uptake ranged up to several hundred parts per million (DiSalvo *et al.*, 1975, 1977).

(v) *Pesticides*

74. Attempts have been made to trace pathways of uptake of sediment-associated DDT into the tissues of estuarine deposit-feeding benthic infauna (Nathans and Bechtel, 1977). The data obtained suggested the possibility of uptake of DDT under model laboratory conditions that may or may not be operative under field conditions.

75. Fulk *et al.* (1975) reviewed the literature on pesticides and PCBs in algae, suspended solids, bottom sediment, and water containing various chlorinated hydrocarbons. The studies that they reviewed reported results of tests conducted to determine the adsorbtion and desorption of chlorinated hydrocarbons on solids, and generally indicated that the materials were much more readily sorbed than desorbed.

76. Fulk *et al.* (1975) analyzed sediment from five locations for aldrin, dieldrin, endrin, lindane, 2,4-D esters, DDT analogues, toxaphene, and PCBs. They found that PCBs, dieldrin, and DDT analogues were the most prevalent. The desorption characteristics of the DDT analogues and dieldrin were studied. No release of DDT residues was observed. Some dieldrin release was observed in the parts per trillion range. On the basis of these laboratory studies, it appears that release of these water-insoluble pesticides will not occur to an appreciable extent during disposal.

77. During a disposal operation in San Francisco Bay, Anderlini *et al.* (1976b) monitored the release of PCBs and compounds of the DDT group from sediment, and the subsequent uptake by organisms. Some uptake if p,p-DDE was observed, but the levels of the other chlorinated hydrocarbons remained constant in *Mytilus edulis*.

C. FIELD INVESTIGATIONS

78. Short- and long-term chemical, physical, and biological impacts of open-water disposal have been determined by large-scale field investigations in numerous locations (Wright, 1978). Chemical water-column effects duplicated the laboratory results, in which only low levels of some nutrients and the metals iron and manganese were apparently released. Analyses of the dredged-material deposits at the sites showed initial elevated concentrations of chemical constituents in the IW, but with time the concentrations became similar to those in reference areas. Movement or release of chemical constituents out of the sediment of the disposal or reference sites was not apparent. Turbidity or suspended particulate in the water column following disposal was found in concentrations significantly lower (an order of mag-

nitude or more) than concentrations shown to have an impact on a broad range of aquatic organisms, and persisted for only a few hours. A significant impact noted in these studies was the mounding of dredged material on the bottom of the dump sites. Biological recolonization studies of these mounds showed that conditions returned to a pre-dump status; there was rapid biological recolonization of the fine-grained disposal areas, but sandy substrates exhibited slower recovery.

79. DMRP studies were conducted where organisms at selected sites were analyzed for metals or chlorinated hydrocarbon uptake (Wright *et al.*, 1977; Cobb *et al.*, 1977; Tatem and Johnson, 1977; Wright, 1978). Evidence from freshwater, estuarine and marine sites showed no increased uptake of numerous toxic and non-toxic metals and of chlorinated hydrocarbons by several organisms when compared with control or reference areas.

80. The Duwamish River enters Elliott Bay, a part of Puget Sound. An Elliott Bay deep-water estuarine site where highly contaminated material disposal was investigated. It is discussed here because the results have direct application to ocean disposal.

81. The entire river is tidal with horizontal and vertical variations in salinity levels. These levels depend upon tidal stage and river discharge. Low dissolved-oxygen concentrations (3 mg g^{-1}) occur near the bottom of the river. Although quite important as a waterway, the Duwamish is also a major migration route for salmon and trout.

82. Elliott Bay is a rather typical estuarine system with a surface layer of low-salinity water over a deeper layer of more saline water. During the summer, density stratification is present; in the winter, colder fresh water from the waterway entrains, and mixes with, warmer saline water in the bay. Hence, there is usually no stratification in the winter. Because it is an estuary, water-column chemical constituents tend to be rather variable. The waterway has created an underwater delta along the south side of the bay. The deltaic sediments consist mainly of silty sand mixed with wood and other organic debris. The dominant demersal fish in the bay during the winter are assorted soles, and the dominant benthic invertebrate is the pink shrimp. Worms and various molluscs are also important components of the bottom fauna (Tatem and Johnson, 1977).

83. In the past, the Duwamish was dredged with a hydraulic pipeline dredge and upland disposal was used. However, the increasing cost of land for upland disposal and a scarcity of sites required a shift to the use of mechanical dredging and open-water disposal from barges. In 1974, there was a spill of almost 1,000 lb of PCBs in the waterway that resulted in serious PCB contamination in the maintenance-dredged channel area of the river. The most highly contaminated sediment was hydraulically dredged and placed in an area impervious to contaminants (Blazevich *et al.*, 1977). All dredging operations and the upland containment of the highly contaminated sediment were carefully monitored by the U.S. EPA. The U.S. EPA found that there was a minimal release of metals, nutrients, and hydrocarbons in the dredging area (Blazevich *et al.*, 1977).

84. Less contaminated sediment was removed by a clamshell dredge, placed in barges, and transported to an experimental open-water disposal site. The material dredged from the waterway was an oily, black, fine-grained organic silt with a plastic texture (Tatem and Johnson, 1977). It was found to leave disposal barges in clumps or as a well-defined mass that fell to the bottom with velocities of up to 180 cm s^{-1}. Upon impact with the bottom, a dense surge of material flared outward at about 36 cm s^{-1} and could be detected more than 200 m from the point of impact. Suspended solids returned to ambient conditions within 10 min, but a slight reduction in light transmittance persisted for several hours (Tatem and Johnson, 1977).

85. The disposal of 114,000 m^3 of dredged material resulted in numerous mounds 2 to 3 m in height. The maximum radius of the deposit was approximately 200 m. Subsequent chemical analysis for PCBs at 6 and 9 months after disposal indicated that the composite mound was gradually spreading (Tatem and Johnson, 1977). This movement was probably brought about by currents gradually redistributing the dredged material. The spreading was not of sufficient magnitude to move contaminated sediment beyond the boundaries of the disposal site.

86. The majority of chemical changes in the water column during disposal were relatively minor (Tatem and Johnson, 1977). There were temporary increases in dissolved manganese, ammonia, phosphorus, and total PCBs. These changes occurred with increases in suspended particulate matter; when the particulate matter decreased, so did the concentrations of contaminants. The increase in particulate matter and associated chemical variables was of extremely short duration, usually less than 30 min. It is of interest that prior to disposal, the concentration of PCBs in the water column exceeded U.S. EPA criteria and that these concentrations increased after disposal.

87. As would be expected, the chemical changes observed in the deposited sediment were a reflection of the nature of the dredged material (Tatem and Johnson, 1977). Metals, nutrients, PCBs and oil and grease were present in the sediment of the disposal area in greater concentration after disposal than before disposal and reflected the concentrations found at the dredging site.

88. A number of biological variables were investigated during the Duwamish aquatic disposal field investigation, and a few of the variables showed major changes as a result of disposal. Density, biomass, number of species, and diversity of benthic invertebrates at the disposal site were depressed after disposal (when compared with pre-disposal values) (Tatem and Johnson, 1977). These effects were more apparent at the cental stations of the disposal site and least noticeable at the corner stations. Some decreases in these parameters were also noted at two reference stations. Nine months after disposal, the number of species present at the disposal site was comparable with the numbers present at the two reference sites, although the biomass values continued to be depressed for the central stations of the disposal

site. There was evidence that animals at the edges of the disposal site were stimulated by the dredged material.

89. There was little uptake of metals or PCBs by fish or by most invertebrates analyzed during and after the disposal operations. Pre- and post-disposal specimens were collected from the disposal site and from locations outside Elliott Bay (Tatem and Johnson, 1977). In addition, caged animals were held at the disposal site for up to three weeks. Mussels held in cages at the disposal site accumulated PCBs to levels slightly above background levels, but the increases were not statistically significant. It should be pointed out, however, that some of the animals collected from Elliott Bay prior to disposal contained substantial amounts of PCBs so that a slight uptake would not have been statistically significant.

90. Demersal fish and shellfish (shrimp) seemed to ignore disposal operations. There were fewer present during disposal at the disposal site than at the east reference site but they were present in about the same number as at the west reference site. After disposal, the number of fish decreased at the disposal site and at both reference sites; this decrease suggests a seasonal change in these organisms rather than an impact of disposal (Tatem and Johnson, 1977). The number of shrimp captured at the disposal site after disposal increased compared with those obtained prior to disposal. Shrimp at the reference sites either remained at the same level (east reference site) or increased erratically from month to month (west reference site). Overall, more shrimp were found at the disposal site after disposal than at either reference site, indicating that the shrimp were attracted to the disposal site. Two additional years of study at this estuarine site have verified initial findings, especially with respect to PCB concentrations.

D. PHYSICAL VERSUS CHEMICAL AND BIOLOGICAL EFFECTS

91. Specific research and monitoring findings with regard to effects of open-water disposal have shown that, with few exceptions, the logical and easily predicted physical effects are more important than chemical or biological effects. Physical effects include smothering of a shellfish bed, the disruption of a flow pattern, a change in salinity, or similar effects. These possible consequences of disposal operations can be persistent, often irreversible and cumulative. However, they can be infrequent and can be avoided with the judicious application of site-selection considerations (dispersal versus containment) and many other physical and biological factors.

92. The difficult problem of the effects of turbidity or suspended sediment particles on both water quality and aquatic organisms has been widely investigated. It was found that, except in unusually environmentally sensitive areas such as coral reefs, spawning areas, fish

migration routes, and public water supply, turbidity is primarily a matter of aesthetic impact rather than of biological impact. It is often advisable as well as logical to schedule dredging and disposal operations to avoid disrupting spawning activities and fish migrations. However, investigations showed that most adult and many juvenile organisms can tolerate turbidity levels and durations far in excess of what dredging and disposal operations produce.

93. With regard to benthic or bottom-dwelling organisms, their resiliency, once beyond the larval stage, has been demonstrated. Disposal sites can be, and are, rapidly recolonized by the establishment of new populations, by migration of organisms from adjacent unaffected areas, and by survival of some of the organisms buried by the dredged material deposit.

94. Colonization by opportunistic species can occur within weeks and by the original species within months. When the type of dredged material disposed of at a site has the same general grain-size distribution as the natural bottom (e.g. sand deposited on sand or silt on silt), survival of existing organisms is maximized. Conversely, a mismatch of sediment types can be quite detrimental. The conditions that could be most injurious to benthic organisms is when the disposal operations, primarily hydraulic pipeline operations, produce a fluid mud or fluff layer that is a difficult and alien environment for many organisms.

IV. CONSTRAINTS ON DISPOSAL OF CONTAMINATED DREDGED MATERIAL

95. A significant effort related to the management of contaminated dredged material is presented by Gambrell *et al.* (1978). Because of the usefulness of this reference, relevant portions of the summary are presented in the following text.

A. CONTAMINANT MOBILITY

96. The processes involved in the release or immobilization of most sediment-associated contaminants are regulated to a large extent by the physico-chemical environment and the related microbiological activity associated with the dredged material at the disposal site. Important physico-chemical parameters include pH, oxidation-reduction conditions, and salinity. Where the physico-chemical environment of a contaminated sediment is altered by disposal, chemical and biological processes important to mobilization or immobilization of potentially toxic materials may be affected. In some cases, substantial contaminant release does not always result in greater mobility of a given contaminant. Frequently, an altered physico-chemical environment that results in the release of contaminants from one chemical form will favour other immobilizing reactions. The influence of physico-chemical conditions associated with various disposal methods on contaminant release must be identified.

97. In addition to the chemical properties of the contaminant, the chemical and physical properties of the dredged material can influence the mobility of contaminants at disposal sites. There are a number of readily identified properties of dredged material that affect the mobility and biological availability of various contaminants. Some of these properties change when the sediment is moved from one type of disposal environment to another, while other properties are not affected by changes in water content, aeration, or salinity.

98. The major sediment properties that influence the reaction of dredged material with contaminants are amount and type of clay; organic matter content; amount and type of cations and anions associated with the sediment; amount of potentially reactive iron and manganese and the oxidation-reduction, pH, and salinity status of the sediment. Much of the sediment removed during harbour and channel maintenance dredging is fine-grained, high in organic matter and clay, and both biologically and chemically active. It is usually devoid of oxygen and may contain appreciable sulphide. These sediment conditions favour effective

immobilization of many contaminants, provided the dredging material is not subject to wave- or current-induced mixing, resuspension, and transport. Coarse-grained sediment low in organic content is much less effective in immobilizing metal and organic contaminants. These materials tend not to accumulate contaminants unless a contamination source is nearby. Should contamination of coarse-grained sediment occur, potentially toxic substances may be readily released upon mixing in a water column or by leaching and possibly plant uptake under intertidal or upland disposal conditions.

99. Many contaminated sediments are initially anoxic and nearly neutral in pH. Subaqueous disposal into quiescent waters will generally maintain these conditions and favour contaminant immobilization. Certain non-calcareous dredged material containing appreciable amounts of reactive iron and particularly of reduced sulphur compounds may become moderately to strongly acid upon gradual drainage and subsequent oxidation, as may occur under upland disposal conditions. Disposal in such an altered environment offers a high potential for mobilizing potentially toxic metals. In addition to the effects of pH changes, the mobility of most potentially toxic metals is influenced by oxidation-reduction conditions to some extent, and certain of the metals can be strongly affected by oxidation-reduction conditions. Thus, contaminated sediment that is coarse-grained and low in organic matter content poses the greatest potential for release of contaminants under all conditions of disposal. Sediment that tends to become strongly acid upon drainage and long-term oxidation also poses a high environmental risk under some disposal conditions.

100. For sediment recognized as representing a high environmental risk, disposal methods stressing containment of potentially toxic substances should be considered. Placement of dredged material heavily contaminated with potentially toxic substances in or adjacent to ecologically and economically important biological populations, or in areas where productive habitat development will occur, represents a high-risk disposal alternative. Likewise, disposal in high-energy aquatic environments may not be a desirable alternative because of the greater probability of long-term dispersion and subsequent transport of contaminants. Placing contaminated sediment in low-energy regimes to minimize resuspension and transport of contaminated solids will reduce the environmental risk of aquatic disposal. The most effective physico-chemical environment for immobilizing most potentially toxic metals, especially when sulphides are present, is near-neutral in pH, strongly reduced, and non-saline.

B. SUBAQUEOUS DISPOSAL

101. Subaqueous disposal of toxic dredged material within, or adjacent to, especially productive aquatic systems represents a high environmental risk. Unconfined disposal in any moderate- to high-energy hydraulic regime also increases the environmental risk because of

the likelihood of transport from the disposal site and chemical transformations of the contaminant to potentially more mobile and available forms as a result of altered physico-chemical conditions in the receiving aquatic environment.

102. Unconfined disposal (where stable mounding will not occur) in moderate- to high-energy subaqueous environments has the inherent disadvantage of greater surface area exposure to the water column. This may contribute to greater transport across the sediment/water interface because of resulting shorter diffusion distances for the small amount of some contaminants that may be mobilized within the reduced dredged material as soluble complexes. Also, greater spreading will increase the surface area, and thus the proportion of the total volume of the contaminated material that may become oxidized as a thin horizon at a sediment/water interface.

103. Certain subaqueous disposal alternatives offer the greatest potential for containment of potentially toxic substances associated with dredged material. Confined (stable mounding) subaqueous disposal of typically fine-grained reduced dredged material will result in little long-term transport from the disposal site. This type of disposal will maintain a strongly reduced physico-chemical environment and favour the stability of metal sulphide precipitates and insoluble complexes of metals with sediment organic matter of large molecular weight. Confinement by mounding in areas of low biological productivity in a low-energy hydraulic regime not subject to storm currents should pose a very low potential for adverse environmental effects. For most purposes, this implies ocean disposal or other deep-water placement where depths are about 30 m or greater. The optimum subaqueous disposal alternative, though not available for most projects, is confinement in a low-energy depression, where the contaminated dredged material can be covered with clean freshly-dredged material. The confined disposal and the capping alternatives result in the maintenance of reducing conditions favouring immobilization of most contaminants, minimum dispersion, and minimal surface area exposure of contaminated material to the water column and benthic organisms.

1. Short-term considerations.

104. Elutriate Test results and monitoring of water quality at disposal sites have generally shown that most contaminants are not released or are only released in negligible amounts during subaqueous disposal. Exceptions include manganese and ammonium-nitrogen for which short-term release in toxic concentrations may occur in the absence of mixing and dilution with receiving-site water (which should rarely be a problem). Where slightly elevated levels of potentially toxic substances have been found in receiving-site water, these levels usually decreased to pre-disposal levels quickly, often within minutes.

105. Little short-term adverse impact of subaqueous disposal would be expected for contaminated sediment transported out of a designated subaqueous disposal area because of the dilution that occurs at the disposal site, the association of released contaminants with the

solid phase, and the transient nature of any increase in the total levels of contaminants at disposal sites.

106. Little, if any, short-term adverse chemical impact on water quality or biological populations would be expected within the disposal site. An exception would be coarse-grained contaminated sediment that is low in organic matter and reactive iron contents from which substantial short-term contaminant releases may occur. As previously mentioned, this type of contaminated sediment occurs only under very localized conditions near a waste outfall because coarse-grained sediment is not an effective scavenger of most contaminants.

2. Long-term considerations.

107. The greatest potential for adverse environmental impacts associated with contaminated sediment disposed of subaqueously would be long-term gradual release and biological accumulation. Typical fine-grained reducing sediment placed in a low-energy subaqueous environment should result in mounding and immobilization of contaminants such that no long-term adverse effects will occur outside the designated disposal area and that surface water quality within the disposal area will not be affected. The potential for uptake by benthic organisms within the disposal area can be minimized by covering contaminated sediments with a layer of clean material.

C. INTERTIDAL DISPOSAL

108. Intertidal disposal of coarse-grained contaminated dredged material with low organic matter content may pose a high environmental risk because of the potential for contaminant leaching into adjacent ground or surface waters and the probable greater availability of many contaminants to plant life. Intertidal disposal near especially productive or sensitive aquatic habitats also represents a high risk, as does extensive habitat development on some contaminated dredged material deposited intertidally. Covering the contaminated material with a layer of clean dredged material or soil should reduce the potential for long-term impact of toxic material uptake by plants and animals. Physical confinement of the bulk of the dredged solids may be required to prevent erosion and subsequent dispersion of contaminated particulates into nearshore waters.

109. Intertidal disposal may present a higher environmental risk than subaqueous disposal because of the erosion and dispersion of the bulk of the solids that would result from the greater hydraulic energy conditions at some intertidal sites, the important and sensitive benthic and aquatic habitats usually associated with nearshore areas, and the demonstrated

potential of plants to take up certain contaminants and cycle them into wetland ecosystems.

1. Short-term considerations.

110. Unconfined intertidal disposal may result in elevated total contaminant levels in effluents associated with suspended solids during de-watering from initial consolidation and settling. Because of the proximity of most intertidal sites to important biological populations, such discharges should be minimized. It is anticipated that suspended solid levels in effluents from any intertidal sites will be more difficult to control than for upland containment facilities, where management to enhance suspended solids removal is more feasible.

2. Long-term considerations.

111. Potential long-term problems of intertidal disposal of contaminated sediment will be associated with gradual erosion and dispersion of contaminated dredged material in nearshore areas and uptake and possible cycling by organisms that become established on these sites. These risks can be minimized by covering the contaminated sediment with a layer of clean material and by taking precautions to prevent gradual long-term erosion of contaminated particulates. Leaching of most contaminants into groundwater or adjacent surface waters may be a long-term risk only if the contaminated sediment is coarse-grained with a relatively low organic matter content.

D. UPLAND DISPOSAL

112. Upland confinement of contaminated sediment for disposal purposes can be done in an environmentally safe manner, though in many cases it may offer little or no benefit over certain subaqueous disposal methods. Sediment heavily contaminated with potentially toxic materials should not be applied upland for the purposes of agricultural soil amendment or habitat development because of the potential for plant uptake, subsequent introduction into food chains, and possible human exposure from crop plants. In some cases, however, sediment slightly to moderately contaminated with certain potentially toxic substances may be used for many upland purposes.

113. Ponded upland containment (i.e., de-watered only by evaporation) where leaching and biological colonization can be controlled on a long-term basis can be an effective disposal method for highly contaminated sediment because the maintenance of strongly reducing conditions favours immobilization of most potentially toxic substances. However, the long-term management problems and the relatively low capacity implied by ponded containment makes

this alternative feasible only for certain low-volume projects. Leaching control and the maintenance of long-term flooded conditions will favour immobilization of most metals as sulphide precipitates in sediment containing appreciable amounts of sulphur. Confined upland disposal with management to minimize retention of suspended solids in initial de-watering effluents should be about as effective as ponded containment in containing most potentially toxic substances. Unconfined upland disposal not specifically intended for habitat development represents a moderate environmental risk because of the natural colonization and the implied greater initial spreading and resulting greater exposed surface area of contaminated sediment than with confined disposal.

114. Except for coarse-grained sediment with low organic matter and reactive iron contents, leaching of most contaminants into groundwater or adjacent surface waters is not expected to be significant if the dredged material does not become strongly acid upon oxidation. Short-term leaching of iron and manganese and long-term leaching of some nitrogen forms may be exceptions. Upland disposal of toxic-metal-contaminated non-calcareous sediment containing large amounts of reactive iron (especially total or pyritic sulphide) represents a high potential for long-term leaching. The strongly acid conditions associated with sulphide and pyrite oxidation will almost certainly result in substantial long-term mobilization and leaching of potentially toxic metals.

115. The use of contaminated dredged material for fill and other engineering purposes will present a low environmental risk under the following conditions:

(1) if extensive surface colonization by natural or managed biological populations is not permitted or the fill is covered with a layer of clean material greater than the expected rooting and burrowing depths of organisms.

(2) if organic matter, reactive iron, and silt and clay contents are moderate to high.

(3) if development of excessive acidity will not occur upon dredged-material oxidation.

1. Short-term considerations.

116. Short-term problems with upland disposal could involve elevated levels of contaminants associated with suspended particulates in initial de-watering effluents from the confinement sites, if applicable criteria for receiving surface waters are exceeded. Management to maximize suspended solids removal would be effective in reducing potential short-term release.

2. Long-term considerations.

117. Potential long-term problems are associated with contaminant uptake and cycling by organisms and leaching into subsurface aquifers. Groundwater contamination may be a problem if strong acidity development is expected upon long-term oxidation of the dredged material. A dredged material containing appreciable sulphide or pyrite represents a high long-term risk for leaching and contaminating groundwater with potentially toxic metals.

118. Liming the entire depth of an upland confinement facility for pH control may not be feasible for economic reasons and because of the additional disposal area capacity required to contain large amounts of lime. Liming may be an effective and feasible management tool for certain contaminated sediment materials applied as thin lifts for some land reclamation or soil improvement purposes.

V. ROLE OF CONTAMINANTS IN SELECTING DISPOSAL ALTERNATIVES

119. The sediment posing the most serious problems is that contaminated with more than one toxic material. Thus, the environmental evaluation associated with the various disposal alternatives will often have to consider more than one contaminant and the relative environmental threat of each. An assigned ranking of environmental risks to each of the various contaminants is not possible because of differences in toxicity, levels of contamination, and chemical behaviour under different methods of disposal. However, it is prudent to suggest that a very high level of concern should be given to sediment contaminated with cadmium, mercury, or certain chlorinated hydrocarbons. Sediment with high levels of iron, manganese, nitrogen, or phosphorus generally poses a very low environmental threat under most disposal conditions. Environmental problems associated with sediment contaminated with arsenic, chromium, copper, lead, nickel, zinc, and petroleum hydrocarbons can range from high to low depending on many factors. In many cases, potential environmental problems with these contaminants tend to be more manageable, and there is more flexibility in disposal alternatives than for sediment contaminated with cadmium, mercury, or certain chlorinated hydrocarbons.

A. MERCURY

120. Mercury is potentially one of the most hazardous of the toxic metals. Mercury losses from upland confinement sites may be especially associated with the fine-particulate phase in initial de-watering effluents.

121. Sediment highly contaminated with this element should be confined in such a way that mercury is isolated at the disposal site. Dredged material with considerable levels of naturally occurring organic matter and especially sulphide can effectively immobilize mercury. A reducing/near-neutral pH disposal condition favours the long-term stability of sulphide and organic complexes and will thus minimize mercury release. For sediment with low to moderate levels of mercury, oxidizing conditions generally enhance release to a small extent compared with reduced conditions. At high levels of contamination, a moderately acid oxidized disposal environment may result in substantial release.

122. Hydrous iron oxides can effectively scavenge traces of dissolved mercury in a water column. Increasing chloride levels may, however, reduce adsorption of mercury by hydrous oxides; increasing pH can overcome the chloride effect.

B. CADMIUM

123. Cadmium, like mercury, is potentially a very hazardous element in the environment. Cadmium can be readily taken up and concentrated by plants and subsequently enter food chains. The chemical mobility and plant availability of cadmium are strongly affected by oxidation-reduction conditions. Furthermore, oxidizing conditions can substantially increase soluble cadmium levels and plant availability. This effect is accentuated by decreasing pH. Thus, the potential for environmental contamination from cadmium-contaminated dredged material may be enhanced at many upland disposal sites as a consequence of expected pH and oxidation changes that favour increased solubility, plant uptake, and leaching. Maintenance of a strongly reducing disposal environment with near-neutral pH will be most effective in immobilizing cadmium.

C. LEAD

124. Lead is a potentially toxic metal often found in contaminated sediment in very high concentrations compared with mercury and cadmium. Fortunately, it is less toxic in equivalent concentrations in soil and sediment/water systems. Excessive levels of lead in fine-grained soils and dredged material can be effectively immobilized by sulphide and sediment organics under reducing conditions. Immobilization by organics and hydrous iron oxides is almost as effective under oxidized conditions. Moderately to strongly acid oxidizing conditions that may develop in certain dredged material placed in upland disposal facilities may result in a substantial long-term release of lead.

D. HYDROCARBONS (CHLORINATED AND NON-CHLORINATED)

125. Disposal methods that include long-term confinement of contaminated particulates should be effective in immobilizing chlorinated and petroleum hydrocarbon contaminants in contaminated sediment. There is no consistent effect of a given oxidation-reduction condition on the degradation rate of all chlorinated hydrocarbons, though the persistence of some do respond to altered physico-chemical conditions. For sediment contaminated with petroleum hydrocarbons of low toxicity, disposal methods that permit gradual dispersion in oxidized water columns and surface sediment should pose a low environmental risk while enhancing contaminant degradation.

E. SULPHUR

126. In dredged material containing potentially toxic metals, the presence of reduced sulphur as sulphide can contribute to effective immobilization of toxic metals if disposal methods are selected to maintain the initial reducing conditions of the dredged material. On the other hand, long-term oxidation of reduced sulphur compounds in some dredged material under upland conditions can result in development of strongly acidic conditions that can cause the release of substantial levels of toxic metals. This condition represents one of the greatest potentials for mobilization of toxic metals in dredged material.

VI. MANAGEMENT PRACTICES FOR AQUATIC DISPOSAL

A. DEEP-OCEAN DISPOSAL

127. Disposal at or beyond the outer edge of the Continental Shelf was thoroughly reviewed by Pequegnat *et al.* (1978). They evaluated the environmental aspects and impact of deep-ocean disposal, nearshore/offshore ecological trends and zonal analysis, and ecosystem dynamics and regional environments. Pequegnat *et al.* (1978) also considered factors such as those controlling spatial distribution and chemical fate, hydrobiological zones as disposal environments, and suitability of specific environmental areas for disposal of dredged material. Based on these considerations, they concluded deep-ocean disposal to be a generally viable alternative for highly contaminated material. Engineering and economic factors will dominate selection of their alternative.

B. CAPPING

1. Techniques.

128. Gambrell *et al.* (1978) have shown that depositing contaminated dredged material into subaqueous depressions, where available, is an effective method for immobilizing contaminants. The formation of mounds that are stable in quiescent waters will also result in good containment of most contaminants. For additional protection, covering highly contaminated dredged material with a layer of clean sediment will essentially seal the buried contaminants from overlying aquatic and benthic organisms. Covering will help in isolating the contaminated material from currents in low- to moderate-energy water columns and may minimize dispersion due to occasional storm currents. Also, the increased diffusion distance coupled with the immobilizing processes within the clean layer will effectively prevent passage and transport of trace amounts of contaminants from the depression of mounds by diffusion.

129. Frequently, a large dredging project will include sediment with a wide range of contamination levels. Dredging and mounding of the most contaminated material first, followed by dredging and covering with the cleaner material from the same project may be useful in confining and containing the most contaminated material (Gambrell *et al.*, 1978; Morton,

1980 a, b, c, d, 1981; Morton and Miller, 1980; and Morton and Korp, 1981).

2. Operational feasibility.

130. Morton (1980 b, c, d, 1981) and Morton and Korp (1981) demonstrated that, under certain conditions, the use of uncontaminated dredged material to cap contaminated dredged material is an operationally feasible, cost-effective, and environmentally sound method for disposal in the marine environment. These studies found that the additional management and operational controls required to conduct capping procedures were neither expensive nor complicated and were within the capabilities of today's dredging and disposal technology.

131. The operational feasibility of the capping technique was also demonstrated at a site in central Long Island Sound, and its application to slightly deeper waters on the Continental Shelf has been accomplished at the Mud Dump Site in the ocean in New York Bight (O'Connor and O'Connor, 1983). In their synthesis and evaluation of the New York Bight capping operation, O'Connor and O'Connor (1983) noted that the available data show that capping can be performed successfully. Furthermore, the thickness and stability of the cap can act to reduce losses of contaminants to the water column. They concluded that capping can be integrated with routine disposal operations to effectively cover and isolate contaminated dredged material.

132. Capping has also been used by Canada and Japan with reasonable success in nearshore disposal sites (depth less than 20 m) (International Maritime Organization, 1984; O'Connor and O'Connor, 1983) and was recognized as technically and scientifically feasible and to have application at low-energy sites up to 50 m in depths.

3. Additional concentrations.

133. Although indications concerning the environmental consequences of capping are very favourable, monitoring and assessment should be continued to determine whether long-term effects (such as sand/silt instabilities, bioturbation, and storm effects) reduce the effectiveness of the cap in isolating contaminants from the environment.

134. The cap should be thick enough to accomplish two goals: to inhibit release of contaminants of concern and to prevent bioturbation from exposing the capped material to aquatic organisms and to the overlying water column. Completed field research has shown that 1 m of cap material is sufficient to inhibit release of PCBs and pathogenic microorganisms from contaminated material. Laboratory research has shown about 0.33 m is sufficient to inhibit chemical-constituent release and that 1 m is sufficient to inhibit bioturbation from breaking the cap (Gunnison *et al.*, 1987; Brannon *et al.*, 1985, 1986).

135. Recent engineering and analytical guidance (Gunisson *et al.*, 1987; Truitt, 1987a, b) further demonstrates the viability of the capping techniques under appropriate water depth and hydrodynamic conditions.

4. Capping and the LDC.

136. The LDC prohibits the ocean dumping of material containing other than trace concentrations of cadmium, mercury, organohalogens, and petroleum hydrocarbons (Engler, 1980, 1981; U.S. EPA, 1977b). As a consequence of the reported capping investigations, it is felt that dredged material contaminated with these chemicals may be regarded as rapidly rendered harmless, and can be disposed of safely at sea when followed by capping with clean material. Moreover, the capping process can be conducted in complete conformity with Annexes I, II and III of the LDC through application of the guidelines for the Application of the Annexes to the Disposal of Dredged Material. Details of global management constraints and regulatory considerations are discussed in section VII.

C. DISPOSAL MANAGEMENT STRATEGY

137. The diversity of disposal alternatives and techniques for management of contaminated dredged material requires the development of an overall long-term management strategy for disposal. The selection of an appropriate strategy is dependent on the nature of the dredged material, nature and level of contamination, available dredging alternatives, project size, and site-specific physical and chemical conditions, all of which influence the potential for environmental impacts. Francingues *et al.* (1985) developed a procedure for contaminant testing and control in a management strategy that considers the nature and degree of contamination, potential environmental impacts, and related technical factors.

138. The management strategy is accomplished in the following sequence:

Step 1 Evaluate contamination potential.

Step 2 Consider potential disposal alternatives.

Step 3 Identify potential problems.

Step 4 Apply appropriate testing protocols.

Step 5 Assess the need for disposal restrictions.

Step 6 Select an implementation plan.

Step 7 Identify available control options.

Step 8 Evaluate design considerations.

Step 9 Evaluate appropriate control measures.

139. Most of the technologies identified in Francingues *et al.* (1985) are either commonly used by the U.S. Army Corps of Engineers in managing their dredging and disposal activities, or are being evaluated as part of ongoing research and development programmes.

VII. GLOBAL REGULATORY CONSIDERATIONS

A. APPLICATION OF THE LDC

140. The Signatory Countries take all practical steps to prevent any pollution of the sea that is liable to create hazards to human health, to harm living resources and marine life, to damage amenities, or to interfere with other legimate uses of the sea (Article 1 of the LDC).

141. The construction of the LDC is simple and straightforward. The articles that comprise the legal framework of the Convention develop the formal regulatory foundation. The annexes (i.e., Annex I Prohibited Materials; Annex II Materials Requiring Special Care; and Annex III Provisions for Developing Regulatory Criteria) develop the technical framework for the implementation of the articles. Interim technical guidelines for interpreting the annexes of articles comprise the day-to-day working level application of the LDC. The simplicity ends here.

142. Many legal and technical terms and constraints were ill-defined or even undefined by the founders, leaving to each of the signatory countries the very difficult job of implementation through their own domestic procedures. The LDC meets annually and conducts business on a consensus basis after debating the issues at hand. A vote is rarely taken and is regarded as the choice of last resort. Issues, questions, legal and technical positions, rule changes, modifications, or proposals can only be submitted to the annual consultative meetings by member nations and debated and acted upon by them. Technical and legal issues can also be referred to intersessional groups (e.g. Scientific Group on Dumping - SG) for further study or resolution. The LDC is an active and dynamic treaty organization that tries to incorporate the state-of-the-art in its deliberations while remaining responsive to many opposing views regarding ocean disposal. The most recent LDC activities significantly affecting dredged material disposal through promulgation of implementation guidelines were initiated at LDC VIII and are discussed below.

B. ACTIVITIES FROM LDC VIII TO LDC X

143. LDC VIII met in February 1984 and endorsed comprehensive Annex III Implementation Guidelines for all material proposed for ocean disposal. The structure of the Annex III guidelines includes the following sections:

Section A - Characteristics and Composition of the Matter.

Section B - Characteristics of Dumping Site and Method of Deposit.

Section C - General Considerations and Conditions.

144. LDC VIII concluded that all parts of the Annex III guidelines might not be applicable to dredged material, gave the LDC Scientific Group (SG VIII) the task to review this issue, and asked member nations to submit technical proposals to SG VIII.

145. SG VIII met in March 1985, and discussion led to the consensus that special guidance might be necessary. The group then recommended that an intersessional group of dredging experts meet to discuss the need for and the structure of guidelines for dredged material disposal (pending approval by LDC IX). LDC IX approved an intersessional Working Group to develop the guidelines.

146. The Working Group met in November 1985 and developed and approved, "Guidelines for the Application of the Annexes to the Disposal of Dredged Material." The structure of the dredged material guidelines is such that it replaces Section A of the Annex III guidelines and subsequently includes Sections B and C of the Annex III guidelines in their entirety. The sections of the dredged material guidelines are: Introduction (Background); Conditions under which Permits for Dumping of Dredged Material May Be Issued; Assessment of the Characteristics and Composition of Dredged Material; and Disposal Management Techniques.

147. The SG IX met in April 1986 and modified and approved the dredged material guidelines. The LDC X subsequently met in October 1986 and unanimously approved them. Their adoption is considered significant because the unique characteristics of dredged material are recognized and the regulation provides a holistic approach to the contrasting alternatives of ocean and land disposal.

148. The guidelines are founded on a comprehensive research base. These guidelines separate the regulation and assessment of dredged material from that of sewage sludge and industrial wastes and include exemptions and exclusions appropriate to dredged material. The guidelines present the availability of ocean disposal management techniques and define critical LDC terminology such as "rapidly rendered harmless" and "special care", in terms of disposal management strategy.

149. The guidelines also require a review of alternatives to ocean disposal through a comparative assessment of human health and environmental risks, hazards (safety) economics, and exclusion of future uses of disposal areas. Furthermore, the guidelines recognize that, for dredged material, "sea disposal is often an acceptable disposal option" and encourages productive/beneficial uses such as marsh creation, beach nourishment, land reclamation, or construction material.

VIII. CONCLUSIONS

150. There is obviously no simple procedure for selecting a method for the disposal of contaminated dredged material, but properly managed disposal in the aquatic environment appears to offer a logical and environmentally sound alternative to land-based disposal sites. The approach of carefully managing open-water sites should be considered a primary management solution to a perplexing problem. The same degree of waste management should also be strictly applied to land containment of contaminated dredged material.

151. With few exceptions, impacts of ocean disposal of dredged material are mainly associated with physical effects. These effects are persistent, often irreversible, and cumulative. Geochemically, contaminant releases are usually limited to nutrients with negligible releases of toxic metals and hydrocarbons. Biochemical interactions are infrequent with no clear trends, but uptake of toxic metals and hydrocarbons are usually negligible.

152. Land- and nearshore-based disposal alternatives appear to offer limited protection in relation to human impact as compared to ocean discharge and, moreover, are often excessively costly. Land-based alternatives often result in drastic change of the geochemistry of the dredged material with a subsequent enhanced release potential of chemical constituents. Land sites are usually located in or near highly productive nearshore areas or adjacent to, or in contact with groundwater aquifers.

153. Even highly contaminated dredged material can be disposed of in ocean locations if sufficient care is exercised in site selection to ensure that the material is isolated from the biotic zone of the marine system. This approach involves disposal site management using capping techniques or locating disposal in abiotic areas. Dredged material should be regarded as a highly manageable material for disposal in the marine environment.

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ANNEX IV

CONCENTRATION OF SELECTED CONTAMINANTS IN WATER, SEDIMENTS AND LIVING ORGANISMS

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INTRODUCTION

1. In this annex, an attempt has been made to review the concentrations of some of the most important inorganic and organic contaminants in water, sediments and organisms from various marine regions throughout the globe. In order to make the basic assessment more manageable, the contaminant categories have been limited to the major classes and to what has generally been regarded by GESAMP and other informed bodies as the most critical potential pollutants within each class. Thus, the review covers, in depth, principally levels of the heavy metals mercury, cadmium and lead, and residues of the organochlorine compounds PCBs and DDT. In the case of these specific contaminants, significant changes or improvements in sampling and analytical techniques over the last 10 - 15 years have rendered many of the earlier measurements totally inaccurate or at best questionable. Therefore, considerable emphasis has been placed on assembling the more recent data. Likewise, national and international intercomparison exercises have become fundamental means for ensuring quality control of data, and where possible, this requirement has also been considered in the selection of the data. The data have been mostly drawn from the open published literature on the basis of their recentness and quality. This has resulted in a listing of only what are considered to be the most reliable values for the various regions.

2. Every attempt has been made to ensure as broad a geographical coverage as possible but this has proved difficult for many areas of the globe, particularly those in the tropics and southern hemisphere. Gaps in information have been identified and in some cases unpublished reliable information from international regional seas programmes has been used to help fill the gaps. In the case where only questionable data exist for a given region, they have been excluded from the compilation.

3. For the sake of clarity in presentation, the data have been differentiated into coastal and open ocean; however, it should be noted that this distinction is often arbitrary and sometimes unnecessary, particularly in smaller, regional, seas where boundaries between the two realms are difficult to define. In either domain, the types of organisms that have been analyzed for contaminants are numerous, and an exhaustive cataloguing of all possible information is neither realistic nor within the scope of this review. Here emphasis has been placed mainly on certain well-recognized bioindicator organisms which are pandemic and can be used to make realistic interregional comparisons.

4. Finally, within the limits of available data and constraints of spatial and temporal coverage, an attempt has been made to delineate any evident spatial and temporal trends in existing contaminant concentrations. Because tissues of many marine organisms bioaccumulate trace contaminants and are usually much easier to sample and analyze than either sediments or sea

water, the contaminant data generated are less controversial. Hence, emphasis has been placed on marine organisms for trend analyses.

I. LEVELS IN WATER

A. OPEN OCEAN

1. Heavy metals

5. The most recent reports on mercury in open ocean waters stress the importance of ultraclean sampling techniques for obtaining reliable values (see Gill & Fitzgerald, 1985; 1987 for review). In this respect, reliable data are extremely limited and pertain mainly to the Pacific and Atlantic Oceans (Table 1). It is probably safe to say that inorganic mercury concentrations in the open ocean surface waters are often less than 2 ng l^{-1}. Furthermore, contrary to most trace metals, mercury concentrations generally display no distinctive gradient with depth, although sampling may not have been sufficiently refined to detect them. The most noteworthy finding to date suggests that concentrations in the northwest Atlantic are approximately double those measured in the northeast Pacific Ocean (Dalziel & Yeats, 1985; Gill & Fitzgerald, 1985- 1988). In the western Pacific there is some evidence that mercury concentrations in surface water decrease along a north-south gradient from roughly 6 ng l^{-1} at 40°N to 3 ng l^{-1} at 30°S (Nishimura *et al.* 1983). This gradient is thought to result from atmospheric transport from the continental areas and subsequent deposition via rain to surface waters. Furthermore, assuming the data in Table 1 are truly comparable, there is a tendency for higher concentrations in the northwestern Pacific region including the Bering and East and South China Seas. Two enclosed seas, the North Sea and the Baltic Sea, which are closer to high potential anthropogenic inputs of metals, show similar mercury concentrations to the North Atlantic. Too few reliable data are available for the Mediterranean. However, based on present information, the levels in open waters appear to be quite similar to those in the adjacent North Atlantic (see Aston & Fowler, 1985; Ferrara & Maserti, 1986).

6. The data in Table 1 indicate that cadmium concentration in surface waters can vary widely between approximately 0.2 and 60 ng l^{-1}, values which are within the range of surface and deep waters (0.1-125 ng l^{-1}) cited in recent reviews (Simpson, 1981; Bruland, 1983). The metal exhibits a nutrient-like distribution, i.e. depleted in the surface layers and increasing with depth. The strong correlation of cadmium profiles with those of phosphorus demonstrate the close coupling of cadmium with nutrient biogeochemical cycles. Open ocean concentrations, particularly those in the subtropical and central gyres, are generally the lowest, ranging from approximately 0.1 to 10 ng l^{-1}. Somewhat higher average concentrations are found in open waters in the enclosed seas (10-60 ng l^{-1}), particularly the Baltic and North Sea, which may result from enhanced river input (see Kremling, in press). In the Mediterranean, concentrations in surface waters from

the Alboran Sea in the extreme northwestern sector are higher than those in the nutrient-de pleted waters of the adjacent Northeast Atlantic (Boyle *et al.*, 1985). Furthermore, this surface enrichment (up to 13.8 ng l^{-1}) appears in a plume of water that originated outside the Strait of Gibraltar; however, it is not evident whether this enrichment is due to natural or anthropogenic sources. In the northwest Atlantic the finding of a significant onshore-offshore gradient from 22 to 0.22 ng l^{-1}suggests input in shelf waters from a continental source, presumably from river discharges (Bruland & Franks, 1983). In the North Pacific, similar increased cadmium concentrations in California shelf waters have been attributed to upwelling of deep, nutrient-rich waters (Bruland *et al.*, 1978). In general, there appear to be extremely few published data for the southern oceans; however, the relatively high values (17-54 ng l^{-1}) reported for the Weddell Sea are also probably a result of strong upwelling there (Mart *et al.*, 1982).

7. Lead is one element whose concentrations in open waters of the Atlantic and Pacific indicate anthropogenic input from chimney emissions and combustion of leaded petrol (Schaule & Patterson, 1981; 1983). It is also one of the most difficult elements to sample and measure without introducing external contamination; thus, reliable concentrations in marine waters are limited (Table 1). In the North Atlantic and North Pacific oceans, surface values range from approximately 5 to 50 ng l^{-1}. In both oceans, lead concentrations are significantly higher by a factor of seven to 10 in surface and thermocline waters compared with that from deeper layers, a feature which differs from the "nutrient-type" profiles for many other metals. This surface enrichment is attributed to atmospheric input emanating from the continents. Of particular significance are the approximately three times higher concentrations in North Atlantic surface waters compared to the North Pacific, which result in the greater industrial lead contamination along the eastern seaboard of North America (Schaule & Patterson, 1983).

8. Recent data also indicate that as of 1980, because of industrial lead emissions to the atmosphere, lead concentrations in surface waters of the North Pacific and North Atlantic have been conservatively estimated to be eight to 20 times as high as those in the South Pacific and are twice as high as natural concentrations (Flegal & Patterson, 1983). Furthermore, lead isotope measurements in the northeast Pacific have established that lead concentrations in surface waters 2,000 km off the North American coast originate from aeolian inputs of Asian lead, while those closer to shore are derived from North American industrial leads (Flegal *et al.*, 1986). However, along one transect from near the central California coast to Hawaii, lead concentrations increased from 5 to 15 ng l^{-1} (Schaule & Patterson, 1981), a trend consistent with oceanographic data but not consistent with expected or known anthropogenic inputs. Surface concentrations in the Mediterranean appear to be on the average somewhat higher than those in the North Atlantic and are likely to be due to the enclosed nature of this relatively small sea and its proximity to the highly industrialized regions of Europe. However, it should be kept in mind that lead concentrations in surface waters of the Sargasso Sea vary by as much 25 per cent throughout the year (Boyle *et al.*, 1986). Lead in the Baltic also appears to be only slightly higher than in the North Atlantic (Kremling, in press). There is some recent evidence that lead concentrations in open surface waters

of the the North Atlantic have begun to decrease in response to a general reduction in the use of leaded petrol in North America during this decade. For example, a nearly twofold decrease in total $Pb/^{210}Pb$ ratios in SargassoSea surface waters between 1979 and 1984 closely reflects the 2.6-fold decrease in U.S.A. leaded petrol consumption during the same five-year period (Boyle *et al.*, 1986). Similarly, lead concentrations in waters of the Gulf of Mexico have declined presumably due to the 40 per cent decrease in lead input via the Mississippi river during the last decade (Trefry *et al.*, 1985). On the other hand, an overall decrease is not yet evident in the North Pacific, due to a considerable contribution of atmospheric input from Asia (Flegal & Stukas, 1987; Flegal, pers. comm.).

2. Chlorinated hydrocarbons

9. Examination of organochlorine compounds in open-ocean water far from any point sources lead to a better estimation of global contamination (Tables 2, 3 & 4). Average PCB concentrations around 1 ng l^{-1} in oceanic water (Table 2) are generally lower than those reported for coastal waters (Table 8). Harding (1986), reviewing available information, could find no obvious temporal trend in DDT levels. Tanabe and Tatsukawa (1986) in another review suggest that PCB levels are higher in the North Atlantic than in other oceans, but this is not borne out by all available data (see Orlova, 1983). Nevertheless, there is clear evidence that PCB levels in surface waters of the northern hemisphere are higher than those in the southern ocean which typically have concentrations of 0.035-0.072 ng l^{-1} (Table 2). This difference most probably reflects the higher industrial production and use of PCBs in the northern hemisphere.

10. During a 1980-81 cruise in the western Pacific, eastern Indian and southern oceans, Tanabe *et al.* (1982) found generally similar DDT levels (approximately 0.005 - 0.06 ng l^{-1}) in open waters of the northern and southern hemispheres, except in areas near the Asian continent where elevated concentrations were found, in the range 0.05-0.12 ng l^{-1}. Other areas of elevated DDT concentrations identified during this and previous cruises were the Bay of Bengal, the Arabian Sea and the waters off Central America. Their data lend strong support to the hypothesis that contamination is a result of increased DDT use in tropical countries for agricultural and public health purposes; however, they point out that other factors such as particulate load in sea water can cause variation in DDT concentrations.

11. According to Harvey *et al.* (1974), PCBs averaged 20 - 40 ng l^{-1} in Atlantic surface waters in the early 70s but declined sharply in 1973 to about 1 ng l^{-1}. However, these former levels appear high in comparison to estimates of PCB releases during that period, and may have been an artefact due to analytical inaccuracies (see Risebrough *et al.*, 1976). More recent measurements give average values of approximately 0.04 - 0.5 ng l^{-1} for the Pacific and 0.06 - 0.25 ng l^{-1} for the Indian Ocean (Tanabe & Tatsukawa, 1986). Cyclodiene pesticides and HCH compounds are also present in open ocean waters (Harding, 1986). Between 1-10 ng l^{-1} lindane

(γ-HCH) have been found in open Pacific surface waters (Tanabe *et al.*, 1982). These values are higher than those (0.5-1.4 ng l^{-1}) reported for the Atlantic (Orlova, 1983) and may reflect enhanced inputs from pesticides use in Asia. Furthermore, total HCH residues in sea water were higher in the northern hemisphere of the Pacific region with α-HCH being more prevalent in northern hemisphere waters and γ-HCH dominating in the southern hemisphere (Tanabe *et al.*, 1982). In certain areas of the Northeast Pacific, there is some evidence that PCB concentrations were very low (less than 1 pg l^{-1}) in the mid-80s, and that α-HCH toxaphene dominated the chlorinated hydrocarbon profile in surface waters (R. W. Risebrough, pers. comm.). However, far more data are needed to verify these trends.

12. Chlorinated hydrocarbons have been found in water at all depths examined in the North Pacific, southern Indian, Mediterranean and Southern Ocean. No clear trends in chlorinated hydrocarbon concentration with depth have been determined, and depth distributions appear fairly uniform (Tanabe & Tatsukawa, 1986; Burns & Villeneuve, 1987). In one instance, a sub- surface PCB maximum of nearly 16 ng l^{-1} was observed in 1982 at 500 m in the northwestern Mediterranean; however, its origin is unclear and it was not present the following year (Burns & Villeneuve, 1987). In both open oceans and coastal areas, the highest chlorinated hydrocarbon concentrations are found in the surface microlayer which is naturally enriched in lipid compounds. Reviewing the literature, Harding (1986) found that concentration or enrichment factors for organochlorine compounds ranged from 10^4 to 10^7, with 10^5 to 10^6 in Narragansett Bay, U.S.A., the German Bight, western Baltic and the Sargasso Sea, and 10^4 to 10^5 in Blanca Bay, Argentina.

B. COASTAL WATERS

1. Heavy metals

13. A recent review of trace metal levels in nearshore and estuarine waters concluded that, while there was considerable variability, no major differences were apparent among different regions of the coastal oceans of the world (Segar & Davis, 1984). While no one area stood out as significantly more contaminated than any other, there was a definite tendency for the highest metal concentrations to be found near estuarine discharges, particularly those estuaries receiving industrial wastes. Almost all data used in the compilation were obtained before 1980 and presumably were subject to the sampling and analytical contamination problems inherent in open-ocean work. More recent data for coastal and nearshore waters are presented in Table 5. On the whole, the concentrations are much lower than those compiled by Segar and Davis (1984); nevertheless, similar general conclusions can be drawn from the newer data. As with information for the open ocean, the majority of the reliable coastal data come from the northern hemisphere.

14. Recent work has clearly shown that mercury levels are not necessarily elevated in

nearshore waters assumed to be contaminated. For example, levels in Puget Sound (0.2 - 1.0 ng l^{-1}) are similar to those measured in the open ocean (Bloom & Crecelius, 1983). Likewise, in the Baltic region the levels are not unlike those in the adjacent North Sea and North Atlantic, probably owing to the formation and rapid sedimentation of particulate forms of mercury (Kremling, in press). However, as expected in some areas near contaminated estuaries and bays, large increases in mercury are noted. One example is the Tagus Estuary where total dissolved concentrations at the outlet are as high as 80 ng l^{-1} (Figueras *et al.*, 1985). This can be compared with the 50-70 ng l^{-1} measured inside and just outside Minamata Bay in 1975 (Kumagai & Nishimura, 1978). The New York Bight also shows signs of large-scale mercury contamination with levels of dissolved mercury ranging from 10 to 90 ng l^{-1} (EPA, 1982 cited by Segar & Davis, 1984). Near estuaries, mercury levels vary considerably and the total dissolved fraction in adjacent sea water is thought to be related to both suspended load and organic material entering the sea (Figueras *et al.*, 1985; Cossa & Noël, 1987). There are only few reliable data for the Mediterranean Sea but, based on available data, it is not evident that Mediterranean coastal waters are higher than those of the northeast Atlantic excluding the Tagus Estuary (Ferrara & Maserti, 1986). The one data set from the Arabian Sea gives not indication of major contamination and suggests levels similar to those in other uncontaminated coastal waters.

15. Cadmium often shows distinct concentration gradients decreasing from nearshore to offshore waters. A typical example is evident at the outlet of the Elbe River in the Wadden Sea where concentrations drop from more than 500 ng l^{-1} in the estuarine mixing zone to less than 100 ng l^{-1} in the Wadden Sea (Duinker *et al.*, 1982). Cadmium concentrations within and outside estuaries are highly variable and are controlled by salinity, sediment load and nutrient chemistry. In smaller seas, river inputs of cadmium have a significant effect on concentrations in sea water. Evaluation of a large number of data from Baltic offshore and coastal waters suggests that concentrations in Baltic waters are less than two and five to 10 times higher than in the central North Sea and North Atlantic, respectively. Abrupt changes in cadmium concentrations can occur in North Atlantic coastal waters. For example, off the northwestern coast of Scotland, dramatic increases in cadmium up to 29 ng l^{-1} occur in a hydrographical front which forms in this region (Kremling, 1983). Elsewhere, outside the highly industrialized areas, coastal waters range between roughly 1 and 100 ng l^{-1}, and it is difficult to distinguish variations arising from anthropogenic inputs and those derived from water-mass movements or natural coastal run-off. Off Brazil, cadmium levels appear to be relatively low, typical of open ocean water; however, as in the case of other "anthropogenic" metals, more data on cadmium are needed from the southern hemisphere before any conclusions can be drawn.

16. As for the open ocean, the major source of lead to coastal waters is atmospheric input; therefore, outside the immediate influence of dense population centers, river input and industrial zones, lead concentrations in coastal waters are not unlike those in the open sea. However, in coastal zones adjacent to populated regions, where most measurements have been made, waters are highly contaminated with lead. An example are waters from the southern Californian Bight

where levels ranging from 25 to 150 ng l^{-1} are attributed to sewage and rainstorm run-off containing lead from petrol (Patterson *et al.*, 1976). Where lead has been measured carefully elsewhere, values are generally lower, in the range 5 to 100 ng l^{-1}, although possible contamination at some stage of sampling and analysis cannot be totally excluded.

2. *Chlorinated hydrocarbons*

17. Organochlorine compounds in sea water have been thoroughly reviewed by Harding (1986) and Tanabe and Tatsukawa (1986). Tables 6, 7, and 8 summarize the data for PCBs and other selected chlorinated hydrocarbons in coastal and nearshore surface waters. These measurements were made on bulk sea water samples which included particulate matter. Comparisons must be made with caution since the techniques used by the various investigators in most cases have not been intercalibrated, and with the very low concentrations reported (pptr), there is a possibility that some of the data are unreliable due to sample contamination or other analytical errors.

18. DDT residues are the most frequently reported compounds in coastal waters, with concentrations generally below 5 ng l^{-1} (Table 6). Notable exceptions are the high levels ($\sim$ 100 ng l^{-1}) reported near Marseille in 1972 and Blanca Bay, Argentina, in the early 1980s; however, the accuracy of these values, particularly the early ones, may be questionable. The alpha and gamma isomers of HCH insecticides are also frequently encountered, with concentrations ranging between 1 and 10 ng l^{-1} (Table 7). HCB has been consistently detected in waters off Norway but not quantified. In the northwestern Mediterranean, HCB concentrations were approximately 1.3 ng l^{-1} during the years 1980 and 1981. PCB concentrations typically lie in the range 1-10 ng l^{-1}(Table 8). The lowest levels are found off Antartica, reflecting the low level of contamination of this relatively remote environment. No temporal trend is detected from the global data between 1971 and 1980 in spite of greatly reduced use during the 1970s. However, in selected areas such as the northwestern Mediterranean where sea water analyses have been carried out by the same analyst for over eight years using the same methods, an approximate threefold decline in PCBs has been noted between the mid-1970s and the period 1978-82 (Burns *et al.*, 1985). This trend in decreasing PCB levels in near-shore waters has been confirmed by similar measurements made along the French coast in 1984 (Marchand *et al.*, 1988). Predictably, the highest levels of all chlorinated hydrocarbons are found in waters adjacent to highly industrialized and populated sectors of the coastal zone. For example, in the Seine estuary, at certain locations, PCB concentrations are as high as 370 ng l^{-1}, indicating a severe degree of contamination (Marchand, 1987).

II. LEVELS IN SEDIMENTS

A. OPEN OCEAN

1. Heavy metals

19. The available data on heavy metals in deep-sea sediments from open waters are few. Some typical values for mercury, cadmium and lead are given in Table 9. In general, published information indicates lower levels in deep-sea sediments than in those from nearshore, but this is not always the case since concentrations depend so much on sediment composition and grain size. For example, mercury levels in the deep north Atlantic range from 0.008 to 0.6 ppm. Much higher values might be expected near volcanically active sites such as hydrothermal vents or areas of active tectonic movements. Cadmium is generally less than 0.5 ppm (Segar & Pellenberg, 1973; Simpson, 1981); however, in upwelling areas near Antarctica, diatomaceous oozes contain as much as 3-60 ppm although concentations normally are less than 1 ppm. Lead is typically found in the range 8-60 ppm (Nriagu, 1978), with concentrations on the order of 47-80 ppm for clays and 13-17 ppm for oozes. Lead does reflect recent anthropogenic inputs to the deep sea since lead concentrations in the top few cm of two deep northeast Atlantic sediment cores are higher (15 and 21 ppm) than background levels (2.8 and 6.0 ppm) in the uppermost 10 cm. From these data it can be computed that approximately half of the contaminant lead inventory in the North Atlantic water column has accumulated in the deep sediments (Veron *et al.*, 1987).

2. Chlorinated hydrocarbons

20. The data base for chlorinated hydrocarbons in deep open ocean sediments is extremely limited. The most complete data set available was generated in a series of cruises taken throughout the Mediterranean in 1975-77 (Table 10). PCBs were detectable in the top few centimetres of all cores, some of which were sampled at water depths as great as 4,000 m. During a cruise in 1975 through the Ionian Sea and the western basin, PCBs were only detectable in the top 1 cm of the sediment. The analyses also showed a preponderance of hexachlorobiphenyl (quantified as Phenoclor DP 6) relative to pentachlorobiphenyl (DP 5) residues which suggested preferential adsorption and deposition of the more highly chlorinated biphenyl compounds. The lowest values (0.8 ppb), were found at the same depth on the sill slopes of the western basin and were thought to result from surface scouring either by accelerated currents or turbidity flows moving down the slopes. The highest concentrations occured in areas known to have a high sedimentation rate, viz.,

the relatively shallow Siculo-Tunisian sill and the station near the Algerian margin of the Algero-Provençal abyssal plain. During 1977 and 1978, PCBs in the top centimetre of cores ranged from 0.6 to 8.9 ppb dry. In most cores the highest levels were found in the top centimetre; however, in several of the sediments a subsurface maximum at about 3 cm was a regular feature. This may be due either to specific behaviour of these compounds in sediments and pore water or to active bioturbation, or both.

21. The only other published information pertains to an open North Atlantic sediment core raised from the Sargasso Sea in 1974 (Harvey & Steinhauer, 1976). The top 2 cm of sediments taken from a water depth between 5,500 and 5,800 m contained 0.5 and 0.3 ppb wet DDE and PCB, respectively. Assuming a 50 per cent water content of deep sea sediments, a PCB level of 0.6 ppb dry corresponds to the lowest PCB concentrations measured in corresponding deep Mediterranean sediments (Table 10).

22. Despite the paucity of data on chlorinated hydrocarbons in deep areas of the open ocean, recent evidence from sediment traps deployed at depth clearly shows that these compounds are still reaching the bottom. For example, between 1978 and 1980, Knap *et al.* (1986) measured average daily fluxes of PCBs, chlordane and dieldrin in particles (less than 125 µM) at 3,200 m in the Sargasso Sea of 4.4, 0.06 and 0.11 ng m^{-2} d^{-1}, respectively. The PCB concentration on the particles ranged from 50 - 350 ng g^{-1} dry, with a mean of approximately 150 ng g^{-1}. These levels are higher than those in deep surface sediments and suggest that some decomposition and compound recycling takes place post-deposition. The widely used pesticide lindane (γ-HCH), a common contaminant in coastal sediments, was not detected at 3,200 m, presumably because of its much greater water-solubility than PCBs or chlordane.

23. With only limited data from deep-sea sediments of the world ocean, virtually nothing can be deduced about spatial or temporal trends in the deep sea.

B. COASTAL

1. Heavy metals

24. Because of major differences in mineralogy, grain size, organic matter, water content and sources of anthropogenic inputs, sediments, perhaps more than any other substrate, show large variations in contaminant concentrations. This fact alone makes interpretation of levels in sediments extremely difficult without prior knowledge of sediment composition and history. This is underscored for nearshore or estuarine sediments in which concentrations of heavy metals such as mercury, cadmium and lead in sediments from a fairly restricted area or region may vary over two to three orders of magnitude (e.g. see Balls, 1985; Figueres *et al.*, 1985). Furthermore, far more information exists on contaminant levels in sediments because they integrate contaminant

loadings, and analytical problems are lessened owing to the relatively high concentrations of contaminants associated with sedimented materials. On the other hand, variability is enhanced by the different analytical methodologies used to remove contaminants loosely associated with, or strongly bound to, sediments. Therefore, metal levels reported for sediments must be viewed accordingly.

25. Recently reported heavy metal concentrations for a variety of nearshore sediments underscore this high degree of variation (Table 11). Mercury concentrations span over four orders of magnitude from less than 0.01 ppm in non-contaminated areas to 5-25 ppm in heavily-polluted embayments such as Raritan Bay and Minamata Bay. The range of cadmium concentrations is as great, reaching a maximum of 140 ppm near a major municipal sewage outfall off Los Angeles. The range for lead concentrations is much narrower, with most levels falling between 10-100 ppm. In the case of these three metals, there are abundant examples in the literature in which concentrations in sediment fall off with distance from the coast. In most of these cases there is clear evidence that the elevated levels in these estuarine and nearshore areas are derived from anthropogenic sources which enter via rivers, run-off and sewage outfalls.

26. Significant hot spots that are evident from Table 11 and the literature (Segar & Davis, 1984) are the New York and Southern California Bights, several estuaries on the west coast of the U.K., the Tokyo Bay, the Tagus estuary in Portugal, and several areas in the Mediterranean. These are all areas of high population density and industry development. Concentrations in sediments from these areas may not be directly related to industrial inputs, however, because circulation and discharge/dispersion mechanisms (point source vs. barge dumping over large areas) play an important role in determining absolute levels as well as the extent of contamination. For example, the Hyperion sewage sludge outfall off Southern California acts as a point source, but dispersion down canyon is excellent with elevated metal levels only in the immediate vicinity of the outfall. Greater loads of metals enter the New York Bight but, because of poor dispersion, the sewage sludge is dumped over a wide area, resulting in a more even distribution of lower concentrations. While these two nearshore zones in North America do show the effects of anthropogenic input of metals, the concentrations are not unusual or extreme compared to those found in fine-grained sediments of many other populated coastal regions.

27. When comparing the data in Table 11 with earlier compilations (Segar & Davis, 1984), no major differences are evident. In some cases, recent levels may be lower where efforts have been made to eliminate contamination by changes in release practices or dredging. In general, because sediments tend to trap metals entering the sea and integrate inputs, they are particularly good for pin-pointing locations of marine contamination.

2. Chlorinated hydrocarbons

28. There is a profuse literature concerning chlorinated hydrocarbons in nearshore surface sediments and much of it, particularly that for North America, published prior to 1980, has been reviewed by Segar and Davis (1984) and Ernst (1984). More recent sediment data for the most persistent chlorinated compounds, i.e. PCBs, and DDTs, are compiled in Table 12. Concentrations vary widely and are a function of both location and sediment grain size. In most studies the fine-grained fraction of the sediments was analyzed, thus enhancing the value of comparing results from different locations.

29. Most reliable data again originate from the more industrialized countries in the northern hemisphere. The highest values recorded in sediments are from New Bedford Harbour, Massachusetts, Escambia Bay, Florida and Palos Verdes, California, areas which all reflect input from a known point source. Other areas which have records of persistent inputs of PCBs and DDT residues are the Saronikos Gulf, New York Bight, Osaka Bay, Naples Bay and Marseille Bay. These are all sites containing major sewage discharges from highly populated and industrialized cities.

30. In the case of the Los Angeles city outfall at Palos Verdes, decreased DDT discharges between 1971-1977 led to a slight reduction in sediment concentrations in the mid-1970s (Bascom, 1982). However, during the years 1977-1981, when the input of DDT and PCB was relatively low and essentially constant, concentrations of these chlorinated residues dropped by an order of magnitude (Young *et al.*, in press). Nevertheless, even five to seven years after control measures were taken on release, high DDT and PCB concentrations have been found in fish from the area, a fact which indicates that sediments are acting for these persistent compounds as a reservoir from which they may entirely be re-mobilized and absorbed, directly or through water, by living organisms.

31. Three additional areas - the North Sea, the Baltic Sea and the Mediterranean - have been systematically studied for chlorinated hydrocarbon contamination under various regional monitoring programmes. As one example, coastal sediments from various countries around the Mediterranean have been analyzed for PCBs and DDT since the early 1970s by many laboratories which have intercalibrated their analytical methods (Fowler, 1987). From these studies, localized inputs on "hot spots" are readily identified. For example, the Athens sewage outfall in the Saronikos Gulf, with PCB sediment concentrations approaching 800 ppb dry, was pinpointed as a major source of PCBs in the Aegean Sea.

32. A similar survey in the vicinity of Naples found extremely high levels (3,200 ppb) in sediments from the Bay of Naples, presumably originating from local industries. Outside the bay, levels were in the general range of 10 to 30 ppb. Perhaps the most striking example of persistent PCB input from a point source comes from studies undertaken around the Marseille outfall at Cortiou, France. In the vicinity of the outfall, PCB concentrations in sediments from 20 to 50 m depth ranged between 3 to 16 ppm dry. Outside the immediate zone of contamination, concen-

trations in sediments dropped to background levels (10 to 30 ppb). Two other PCB "hot spots" in the Mediterranean are Augusta, Sicily (130 to 457 ppb dry sediments), and at 100 m depth just offshore the Casino at Nice, France (1,165 ppb dry). At the latter location, concentrations in nearby sediments from the same depth showed much lower concentrations (29 and 86 ppb, respectively), indicating a highly localized contamination.

33. Surveys of coastal sediments in the Adriatic Sea during the early 1970s recorded high concentrations of PCBs, primarily in the northern sector. The results indicated that the majority of these compounds originated from freshwater discharge in the north (e.g., Po River) and were for the most part transported in a southerly direction along the western shore. Later studies verified the relatively low concentrations of PCBs (less than 1.0 to 28 ppb dry) in sediments from both the northern and southern regions of Yugoslavia (Table 12).

34. Data on chlorinated hydrocarbons in sediments from the eastern Mediterranean are sparse; however, a survey has shown that of seven sediments sampled along the eastern coastline near Mersin, Turkey, only three contained detectable amounts of PCBs (2 to 4 ppb dry). Another study reported levels ranging from 0.7 to 0.9 ppb in sediments from nearby Antalya.

35. Similarly, the only published data on PCBs in sediments from the North African coast are those from spot samples taken on the circum-Mediterranean CALYPSO cruise in 1977. Concentrations in sediments from Tunisia and Egypt were in the range 0.5 to 2.0 ppb; significantly higher concentrations were noted off Algeria at Annaba (8 to 14 ppb) and Oran (325 ppb). Except for the PCB concentration at Oran, these values were generally lower than those measured on the European side of the Mediterranean and probably reflect the lower number of industrial and urban sources of PCBs along the north African shoreline (Fowler, 1987). These observations suggest a decreasing spatial trend in PCB concentration from west to east in the Mediterranean basin but most of the above-mentioned studies were not carried out over sufficiently long period of time to establish temporal trends. However, in one area of the Golfe du Lion near Montpellier, sediments were sampled at the same site over a three-year interval from 1972 to 1975. Although contamination of sediments was evident, no temporal trends were noted during the study. PCB concentrations averaged 340 ppb in 1972 and 390 ppb in 1975, merely confirming that chlorinated hydrocarbons are highly persistent in surface sediments.

III. LEVELS IN LIVING ORGANISMS

A. OPEN OCEAN

1. Heavy metals

36. Compared to work in the coastal zone, relatively few measurements of contaminant levels in organisms from the truly open-ocean waters have been reported, possibly because there is generally no perceived pollution in the open sea, and these more isolated areas of the ocean are not always of direct economic interest to the countries carrying out the survey. Such studies are very expensive and those that have been performed in the open seas are often ancillary to other studies or undertaken to establish a base line to draw conclusions on the spatial extent of contamination originating from land. Therefore, information from open-water organisms compared to that from similar species found in the coastal zone is often crucial.

37. Sampling of truly open-ocean species has primarily been focused on the plankton, a community of mixed pelagic organisms representing many phyla. Because these organisms have different physiologies, life-spans and bioaccumulative abilities, comparisons of concentrations in mixed plankton samples are often difficult to interpret. For that reason, temporal and spatial comparisons of contaminant levels in individual pelagic species should be given preference.

38. Because of their physiology and small size, mixed zooplankton are rapid accumulators of many contaminant classes. For heavy metals, one of the most recent surveys was carried out in the Mediterranean (Fowler, 1986). Cadmium and mercury levels were roughly 2 and 0.1 ppb dry, respectively, in mixed zooplankton. For krill (euphausiids) throughout the Mediterranean (Table 13), average mercury levels were the same as in mixed zooplankton. However, cadmium concentrations of about 0.3 ppb dry were much less. In general, these concentrations were similar to those reported for these types of plankton from the open North Pacific and North Atlantic. In addition, no onshore offshore trends in cadmium concentrations were evident in either mixed zooplankton or krill collected along a transect extending 90 km off Monaco.

39. In the deep open ocean available information for heavy metals comes from a few cadmium, lead and mercury analyses. As in coastal waters, these metals derive from both natural and anthropogenic sources and it is difficult to identify which source is the major contributor to the organism's body burden. Recent analyses of benthopelagic rat-tail fish from the deep north Pacific and north Atlantic indicate very similar average concentrations of cadmium (0.025 - 0.027 ppm dry) and lead (0.012 - 0.016 ppm dry) from the two oceanic areas (Windom *et al.*, 1987).

Furthermore, mercury concentrations in museum specimens of blue hake collected at depths of 2,000 to 3,000 m in the Northwestern Atlantic in the 1880s were similar to those measured in recently collected individuals, a finding which suggests that no significant increase from anthropogenic sources has taken place in this species during the last century (Barber *et al.*, 1984). From such data and other published reviews, there is no evidence of elevation of these metals in deep-sea species as a result of man's activities (Grassle *et al.*, 1986).

2. Chlorinated hydrocarbons

40. Likewise, sparse data for chlorinated hydrocarbons in zooplankton render it difficult to examine regional and temporal trends. In the case of PCBs, there are marked variations in reported concentrations (Table 14). PCB concentrations in Atlantic mixed plankton appear to be one to two orders of magnitude higher than those measured in Pacific and Mediterranean samples, but the Atlantic measurements were made prior to 1972 and could reflect a period of higher PCB input into open waters. More recent data for the Atlantic are lacking, but are estimated to be in the few ppb wet range (Tanabe & Tatsukawa, 1986). Concentrations of DDT and its degradation products of the same order of magnitude have also been measured in plankton samples from the Mediterranean and elsewhere (Fowler & Elder, 1980; Harding, 1986). The ratio of DDT to its degradation products (e.g. DDE) in zooplankton is useful in identifying hot spots of fresh DDT input such as that seen in 1977 in the eastern Mediterranean off Israel and the Nile delta (Fowler & Elder, 1980). A thorough review of DDT and other chlorinated pesticides in zooplankton has been made recently by Harding (1986). The review confirms the large range in concentrations which is a function of variable inputs, different species composition, and the prior history of the animals' movements as they migrate vertically through the water column.

41. Data on body burdens of organochlorine compounds in deep-sea organisms from the open ocean are very limited and pertain mainly to organisms obtained from the continental slope of eastern North America. Reviews of the literature indicate, however, that PCBs and DDT have penetrated the deep ocean and are accumulated to levels similar to those measured in coastal species (Grassle *et al.*, 1986). For example, recent analysis of livers from rat-tail fish (*Coryphaenoides armatus*) from greater than 3,000 m show concentrations of p,p-DDE and total PCBs ranging from 1.6 to 2.5 and 1.9 to 6.1 ppm wet weight, respectively. Furthermore, analyses carried out using high resolution glass capillary GC and verified by GCMS have identified the presence of hexachlorobenzene, toxaphene and chlordane in these bottom fish. The available data, which are limited to the northern hemisphere, demonstrate the occurrence of food web transfer; however, although feeding strategies differ in the deep sea, no evidence for food web magnification has been found. Nevertheless, some deep-water fish far removed from land-based sources of contaminants have indicated a high degree of local contamination by chlorinated hydrocarbons. For example, sable fish from a depth of greater than 1,000 m near the Farallon Islands off California contained total DDT and PCB concentrations in their livers of 9 and 7 ppm wet weight, respectively (Melzian *et al.*, 1987).

B. COASTAL

42. Of more relevance to assessing pollution impact are measurements of contaminants in nearshore organisms which are directly exposed to contamination from land-based sources. Two strategies have evolved regarding which organisms to study. One concerns monitoring edible species of economic importance with the principal aim of protecting the health of man. The other focuses on the use of comparable bioindicator species which are distributed widely throughout the world and can thus be used to make spatial and temporal impact assessments. Irrespective of the strategy and the types of tissues examined, contaminant concentrations are highly variable, often as a result of such intrinsic biological factors as growth, sexual stage, fat content, food regime, as well as of such environmental factors as temperature and salinity changes (Phillips, 1980). Very often these factors are not recorded and thus it is difficult, if not impossible, to derive temporal and spatial variation in contaminant levels from these "bioindicators". Nevertheless, living organisms have been extremely useful in identifying contamination hot spots as well as temporal trends over the long term. One organism which has received much attention is the mussel *Mytilus* because this and related sessile bivalve species are found throughout the globe and are edible. Over the last decade, numerous baseline contaminant data have been gathered for these and other bivalves which will prove useful for future comparison.

1. Heavy metals

43. Table 15 lists average values and ranges for heavy metals in mussels from several regions of the globe. These values have been summarized from a large number of published data, the original references for which can be found in the citations given in the table. This table is not comprehensive but is meant to give an indication of regional averages and ranges in areas where a sufficient number of data have been obtained from in-depth studies. It appears that mean values vary, in general, between approximately 1.0 and 5.0 ppm dry weight for cadmium and 0.1 and 0.4 ppm for mercury, although overall the ranges indicate a high degree of variability in both cases. Average lead values vary somewhat more, falling in a range of roughly 1-16 ppm dry. The high average cadmium concentrations in mussels from a pristine area in southern Oman in the Arabian Gulf presumably arise from the intense upwelling which brings deep phosphorus-rich (and presumably cadmium-rich) waters onto the coast.

44. Temporal and spatial trends are extremely difficult to establish, given the fact that many intrinsic and external factors can affect measured metal concentrations in space and time. Nevertheless, some of the more comprehensive and long-term bivalve studies carried out in the northern hemisphere have proved useful in locating local sources of contamination.

45. Shellfish monitoring around the U.K. in the last 20 years has shown somewhat higher metal levels along the Irish Sea coast (particularly Liverpool) than in the English channel and along the North Sea coast, although the concentrations have not given rise to concern for human health (Franklin, 1987). In the Mediterranean, average mercury concentrations in mussels generally ranged from 0.04 to 0.87 ppm wet weight but values as high as 7.0 ppm indicated local sources of contamination in the Adriatic (UNEP, 1987a). When compared to similar data generated from the ICES monitoring programme (Jensen, 1987), average mercury concentrations are much more variable and tend to be somewhat higher in the Mediterranean region. For the Mediterranean there is a general lack of reliable data from the southern coast. Present results therefore pertain to the more industrialized northern sector. Nevertheless, it appears that beside molluscs, some fish, crustaceans, marine mammals and birds from the northwestern sector of the Mediterranean also have higher average mercury concentrations than similar taxonomic groups from the Atlantic. This is thought to be due to higher natural environmental levels of mercury from geological origin. However, the pathways and mechanisms of transfer to top predators in the region remain obscure (UNEP, 1987a).

46. One of the more intensive and best co-ordinated studies on a national scale has been the U.S.A. Mussel Watch Programme which has been underway since the mid 1970s (Goldberg *et al.*, 1978; NOAA, 1987). Both mussels and oysters (Gulf coast) have proved useful in detecting hot spots of metal contamination. For example, bivalves from the Hudson Raritan Bay, New York, Boston Harbour, Massachusettes and San Pedro Bay, California reflect high ambient lead levels; those from Hudson Bay, Tampa Bay, Florida and Matagorda Bay, Texas are high in mercury and samples from Copano Bay, Texas, and the Delaware, Chesapeake and Hudson Bays are relatively high in cadmium (NOAA, 1987). Temporal trends are more difficult to establish because either the sampling has not been maintained for sufficiently long periods or the same species has not been sampled on different occasions. However, many of the heavily contaminated sites identified in earlier studies during the last decade (Goldberg *et al.*, 1978) continue to be recognized as hot spots (NOAA, 1987). On a local scale, the same technique but with higher resolution spatial sampling has been extremely useful in pin-pointing the origin of contamination sources such as the recent discovery of mussels containing 1,826 ppm dry lead adjacent to a lead slag site in Monterey Bay, California (Flegal *et al.*, 1987).

47. Sedentary fish have also proved to be useful indicators of trends in heavy-metal pollution. One dramatic example comes from a recent Dutch study in which a single species of teleost (*Zoarces viviparus*) from the Wadden Sea and the heavily contaminated Ems estuary has been analyzed for mercury since 1974 (Essink, 1988). Results clearly indicate a continued decrease in mercury content of fish since that time, commensurate with decreases of the major mercury discharges into the Rhine and Ems rivers. Between 1981 and 1987 the mercury concentration in fish decreased by a factor of two, whereas the estimated mercury discharge of the Rhine dropped by approximately a factor of three, suggesting the continued presence of other sources of mercury in this region of the North Sea.

48. In contrast to northern latitude studies, there is a paucity of comparable data from the southern hemisphere (Table 15), and in many instances earlier surveys carried out during the last decade have not been followed up, making it difficult at best to assess possible temporal trends in these regions.

2. Chlorinated hydrocarbons

49. Marine organisms have also been important in pin-pointing land-based sources of organochlorine pesticides and PCBs, particularly in the northern hemisphere. As with metals, the most commonly used species in past and present monitoring programmes are bivalves and edible fish. In many instances bivalves are preferable because of their greater bioaccumulation potential through filter feeding and the fact that the low organochlorine concentrations often encountered in fish flesh are difficult to quantify accurately. Recent published data for PCBs and total DDT residues in mussels and oysters from various regions of the world are set out in Table 16. It is immediately evident that the variability in any one region is large, often up to three orders of magnitude. This fact alone severely limits intra- and inter-regional comparisons. Furthermore, the use of different species, size differences, seasonality effects, different methods of quantification, etc., all contribute to making comparisons difficult. In general, the highest values are found in areas where there are known industrial and agricultural inputs (e.g. PCBs in Buzzard's Bay, U.S.A. and Osaka Harbour, Japan), and the lowest levels are found in organims from the more remote areas such as the Arabian Sea and the central coast of Brazil (Table 16). Much of the data in Table 16 have been generated during the 1980s and perhaps the most striking feature is the widespread occurrence of DDT residues and PCBs in marine organisms long after restrictions were placed on their use over a decade before. Also, it is noteworthy that the overall lack of reliable data from the tropical regions and the southern hemisphere makes it difficult to assess possible recent shift in the use of DDT from the northern hemisphere to these regions.

50. Insight can be gained by examining the results from national or regional monitoring efforts where there is a high degree of resolution in temporal and spatial sampling. Fish and shellfish monitoring, which has continued for nearly two decades in the Baltic Sea region has demonstrated a significant decrease in the concentration of DDT residues in living organisms during the 1970s and 1980 (HELCOM, 1986). For example, recently published data from the Finnish sea areas of the Baltic show that in the period 1979-1986, total DDT in herring muscle dropped from 0.7 - 2.2 ppm to 0.3 - 1.0 ppm (lipid weight), and corresponding PCB levels from 2.7 - 3.7 ppm to 0.3 - 1.1 ppm (Haahti & Perttila, 1988). Other pesticides persist, however, with present lindane concentrations in these fish varying from 0.010 to 0.017 ppm and HCB from 0.013 to 0.019 ppm (lipid weight). However, the Baltic studies did detect a slight increase in DDT in some areas after 1983 but the origins of these episodic inputs have yet to be explained (HELCOM, 1986).

51. Monitoring of fish and shellfish around the British Isles between 1977 and 1984 has shown that, while the levels of some organochlorine contaminants still remain relatively high in

some areas along the North Sea coast (e.g. Liverpool Bay), concentrations are generally lower than those previously reported (Franklin, 1987). Similar studies along the French Mediterranean coast have shown that PCB concentrations in mussels dropped by nearly an order of magnitude between 1972 and 1975 in response to regulations of PCB dispersion in the environment which took effect at that time (Marchand, 1987). However, subsequent measurements made in 1980-81 indicated a significant increase in PCB contamination, the origin of which is not known. Over the same period, total DDT in mussels decreased substantially and did not show a concomitant increase between 1975 and 1980-81. Persistent PCB hot spots were found in mussels from the Seine estuary (5,000 ppb dry) and at Toulon and Marseille near the Rhone outflow (2,000 ppb). Higher concentrations have been noted in other areas of the Mediterranean but the data are not yet sufficient for temporal trend analysis (UNEP, 1986, 1987b).

52. The NOAA National Status and Trends Mussel Watch Programme has examined chlorinated hydrocarbon residues in bivalves around the U.S.A. during 1984-86. Although too early to assess temporal trends, this study has been crucial in identifying persistent hot spots such as Hudson Bay, N.Y., San Pedro Harbour and Palos Verdes, California for DDTs,and Buzzards Bay, Massachusettes, Hudson Bay and New York Bight as a source of PCBs (NOAA, 1987). Interestingly, corresponding fish liver analyses lead to the identification of some different hot spots not always noted by analyses of mussels or oysters. The programme highlights one of the largest problems in using tissue data to identify contaminated sites, i.e. the impracticability of using a single species over a wide area. At many of the locations identified as hot spots in earlier surveys (see Goldberg *et al.*, 1978; Segar & Davis, 1984), PCBs and DDTs are still present in high concentrations in local bivalves - for example 6,808 ppb dry PCB in Buzzards Bay and 1,109 ppb dry total DDT in Hudson Bay.

53. A recent analysis of U.S. monitoring data, which includes information from different programmes undertaken throughout the 1970s and 1980s, gives some indication of the overall temporal trends (Stout, 1986). In the Southern California Bight, PCB levels in Dover sole (*Solea solea*) dropped by more than an order of magnitude ($\sim$1 ppm to 0.03 ppm wet weight) between 1972 and 1981. Likewise, levels in mussels from near the Los Angeles County sewer outfall decreased tenfold (2.5 ppm to 0.24 ppm) between 1971 and 1978; however, concentrations later increased to 0.56 ppm in 1979. The reasons for the increase are obscure but could be related to many factors such as remobilization of contaminated sediments, renewed dumping or change in analytical methods within the monitoring programme (ibid.). Another important observation has been the consistently higher PCB (360 ppb dry weight) and DDE (507 ppb dry weight) concentrations in mussels at Año Nuevo Island, a supposedly "pristine" area off central California (Martin & Castle, 1984). Similar observations have been made at San Miguel Island, California (Stout, 1986). The islands contain extensive rookeries for seals and sea lions, and these normally contain high concentrations of organochlorine residues in their tissues. In the absence of any known organochlorine inputs to these sites, it is hypothesized that these persistent residues are biologically recycled through the mammals and retained in the ecosystem.

54. On the east coast of the U.S.A. following the control of intentional PCB dumping into the Hudson river, PCB concentrations in striped bass dropped from 18 ppm wet weight in 1978 to 7 ppm in 1979 and to 4.8 ppm in 1981; however, the decline was greater between 1978-1979 than between 1979-1981, indicating a slowing down in the environmental depuration process (Stout, 1986). This, however, may be a unique case where total suppression of a major input has lead to a significant decrease of PCBs in animals living in the ecosystem. Over a much broader scale, chlorinated hydrocarbon residues are still widespread on both the east and the west coasts of the U.S.A., as evidenced by the levels in the years 1984-86 (NOAA, 1987), similar to those measured a decade earlier (Goldberg *et al.*, 1978; Farrington *et al.*, 1983).

55. Similar monitoring programmes in areas of low population density with limited agriculture and industrial activities give useful information on temporal changes in global background or baseline levels of organochlorine compounds. Bivalves and fish have been monitored at several locations in the Gulf and Arabian Sea areas since 1980 (Burns *et al.*, 1982; Fowler, 1988, unpublished data). PCB and total DDT concentrations in rock oysters from the coast of Oman are given in Table 17. The same oyster populations have been continually sampled and the same sampling and analytical methodologies have been applied by a single laboratory during the six-year period. It is evident that the concentrations are very low when compared with other regions and have varied little over the years. Lindane, aldrin, dieldrin and HCB are nearly always present but at levels normally less than 1 ppb dry. Concentrations such as these probably reflect atmospheric transport from sources outside the region and can be considered as baseline levels for future reference.

56. Convincing evidence of decline in organochlorine compounds between the 1970s and 1980s comes from studies of Arctic seals (Addison *et al.*, 1984; 1986). In seals from the Canadian east coast, DDT residue levels declined approximately 3 - 5 fold between the early or mid-1970s and 1982, whereas PCB concentrations fell by about half over the same interval. In Arctic ringed seals from the west coast of Canada, PCB levels declined by about the same factor. DDT levels, however, decreased little over the same period, suggesting a continuing supply of the pesticide to the western Arctic. While declining PCB concentrations closely reflect the ban on the manufacture of these compounds in the early 1970s, DDT is likely to have been transported atmospherically to the Arctic from the Far East where it had been used extensively until at least the late 1970s.

C. SUMMARY

57. It appears from the existing data on marine organisms that the highest contaminant concentrations occur, as would perhaps be expected, in regions adjacent to highly populated and industrialized zones (often located near major river/estuary systems) or in areas used intensively for agricultural purposes. Although this assessment relies heavily on monitoring data using sessile bivalves, similar conclusions can be drawn from the existing data for a wide variety of fish species.

58. Nearly all the supporting information comes from studies in the northern hemisphere at latitudes above 30°N. Therefore, although there is a small but growing data base from tropical zones and the southern hemisphere, it is far too early to draw any conclusions about a southward shift in chlorinated pesticide and PCB use (Goldberg, 1983) in these regions. Nevertheless, the few reliable southern hemisphere data that do exist demonstrate the widespread presence of various pesticides in coastal organisms. For example, in 1985, mussels along the Chilean coast were found to contain 3 - 43 ppb wet and 4 - 10 ppb wet lindane and aldrin, respectively (Ober *et al.*, 1987a), concentrations which are equivalent to, or surpass, those recently reported for mussels in the Mediterranean (UNEP, 1986; Pastor *et al.*, 1988). Further insight comes from recent analyses of fur seal blubber from Australia which reveal relatively high (average 4 ppm wet) total DDT concentrations (Smillie & Waid, 1987). When these data are compared with similar analyses of pinnipeds from other areas of the world (Table 18), it is apparent that total DDT concentrations in Australian seals are at least as high as some which have been reported for the northern hemisphere. In contrast, PCB concentrations in the same seals are low by world-wide standards.

59. Documented cases of declining organochlorine residue levels in marine organisms pertain mainly to the most contaminated areas such as the Hudson-Raritan Estuary and the Los Angeles Bight. In other areas, PCB and DDT concentrations have either fluctuated, stayed the same or actually increased over the years (see Stout, 1986). One example of their persistence in the environment comes from the California State Mussel Watch Programme which, as recently as 1986, reported high DDT (2.9 ppm wet) and PCB (2.2 ppm wet) levels in mussels from San Francisco Bay and San Diego Bay, respectively (Anonymous, 1987). Furthermore, despite restrictions or prohibitions on their use, other pesticides including dieldrin and chlordane were also found at elevated concentrations in mussels from various lagoons and bays in central and southern California (ibid.). These observations underscore the difficulties many developed and developing countries will face in the future in reducing levels of organochlorine pesticides and other compounds in their coastal ecosystems.

IV. GENERAL SUMMARY AND CONCLUSIONS

60. This review of recent concentration data underscores the widespread occurrence of some potentially critical metal contaminants (Hg, Cd and Pb) and organochlorine compounds (PCBs and DDT) in marine waters, sediments, and organisms of the global ocean. The highest concentrtions are generally found in the most highly-populated and industrialized regions which often are centred near or on major river estuaries. For the most part existing concentrations, particularly those in edible marine organisms, do not give rise to alarm; however, in some cases national and international concentration limits have been exceeded and this has caused some concern for human health. In some of these "hot spots" (e.g. Minimata Bay, Hudson River, Los Angeles Bight, Ems estuary) where measures have been taken to eliminate the source of acute or chronic contamination, a significant reduction in contaminant concentrations has resulted, but temporal data are either two sparse or have not been collected for a sufficiently long period to make accurate predictions on the environmental half-life of the contaminants. Nevetheless, the ubiquity of some persistant anthropogenic contaminants (e.g. organochlorine residues) in areas far removed from known input sources suggests that their residence time in the ecosystem will be long. Spatial data on a global scale are also limited; therefore, it is not yet possible to draw any firm conclusions about the long-term consequences of a shift in use of DDT and other chlorinated pesticides towards the tropics and southern hemisphere. However, the recently emerging information base from these regions indicates levels of some organic compounds in marine matrices that are as high or higher than those that have been measured in the northern hemisphere. More data from a much wider area are needed in order to elucidate this trend.

TABLE 1. CONCENTRATIONS OF Hg, Cd AND Pb IN SURFACE WATERS OF THE OPEN OCEAN

Location	Hg (ng l^{-1})	Cd (ng l^{-1})	Pb (ng l^{-1})
Northwest Atlantic	*0.7 ± 0.2[a], 0.5[e], 0.8[hh]	2.5 ± 0.9[h], 0.2[w]	33[m], 46[h], 26 ± 4[y], 8.1[dd]
Northeast Atlantic	1.2[c], 0.6[e]	11.2[c], 4.8 ± 1.4[h] 4.5 ± 2.9[f], 2[g], 1.1[x]	33[g], 33[c]
Northeast Pacific	*0.37 ± 0.13[a]	4.5[k], 2.2 – 9[l], 0.16[w]	14[m], 5 – 15[q]
Northwest Pacific	**5.0 ± 0.5[b]	14[z]	
Southwest Atlantic			
Southeast Atlantic			
Southwest Pacific	0.42[hh]	15[z]	4.6 ± 1.0[p], 3.5[dd]
Southeast Pacific	2.2[hh]	3.8 ± 2.4[h]	16[h]
Arctic	*2.3[c]	8.1 ± 1.4[i], 14.5[j]	14.8 ± 3.5[i]
Antarctic		17 – 54[aa], 13.7 ± 5.9[ff]	
Mediterranean	0.5 – 2.5[d]	8[q], 4.0 – 13.8[x], 14[bb]	30 – 150[g, n]
Baltic Sea	3.0[r]	30 ± 2.7[o], 34[r]	16 ± 4.5[o], 41 – 83[r], 50[gg]
North Sea	1.8[r], 3.3[s]	23[r], 59[s], 21 ± 6[f], 16[u]	52[r], 40[s], 31[u], 62[cc]
Indian	**4.4 ± 1.6[v]	15[t], 0.25[bb]	30[t]
Arabian Sea		3.4[ee]	
Bering Sea	*7.0 ± 3.3[v]		
Japan Sea	*5.9 ± 1.8[v]		
East & South China Seas	*5.7 ± 2.3[v]		

* Reactive Hg ** Total Hg

FOOTNOTE:

a. Gill & Fitzgerald (1985)
b. Matsunaga *et al.* (1975)
c. Olafsson (1983)
d. Copin-Montegut *et al.* (1986a)
e. Dalziel & Yeats (1985)
f. Kremling (1985)
g. Copin-Montegut *et al.* (1986b)
h. Nurnberg *et al.* (1983)
i. Mart *et al.* (1984)
j. Danielsson & Westerlund (1983)
k. Bruland (1983)
l. Boyle & Huested (1983)
m. Schaule & Patterson (1983)
n. Laumond *et al.* (1984)
o. Danielsson & Westerlund (1984)
p. Flegal & Patterson (1983)
q. Schaule & Patterson (1981)
r. Kremling (1987)
s. Brugmann (1986)
t. Danielsson (1980)
u. Balls (1985b)
v. Nishimura *et al.* (1983)
w. Bruland & Francks (1983)
x. Boyle *et al.* (1985)
y. Boyle *et al.* (1986)
z. Boyle *et al.* (1976)
aa. Mart *et al.* (1982)
bb. Mart & Nurnberg (1986)
cc. Brugmann *et al.* (1985)
dd. Flegal (1986)
ee. De Baar *et al.* (1987)
ff. Bordin *et al.* (1987)
gg. Brugmann (1988)
hh. Gill & Fitzgerald (1988)

TABLE 2. PCBs IN OPEN OCEAN SURFACE WATERS

Region	Year	N	(ng l^{-1})	Quantified as	Source
Mediterranean	1975	37	2.9 ± 3.6 (0.2–19.0)	Phenochlor DP–5	Harding (1986)
Mediterranean (west and east)	1975		2.9(0.2–19)		Elder & Villeneuve (1977)
" "	1977-79		0.7(0.1–2.5)		Villeneuve *et al.* (1980)
Northwestern Med.	1982		1.5–5.1		Burns & Villeneuve (1987)
North Atlantic Ocean					
Iceland to Nova Scotia	1971	8	25 ± ?	Aroclor 1260	Harding (1986)
Azores to Barbados	1973	8	2 ± ?	Aroclor 1260	"
Sargasso to New York Bight	1973	9	0.8 ± ?	Aroclor 1260	"
Northeast Atlantic 34°– 63°N	1972	19	39 ± 36 (1–150)	Aroclor 1260	"
Northwest Atlantic	1972	15	27 ± 24	Aroclor 1260	"
Gulf Stream, Sargasso			(1–88)	Aroclor 1260	"
Sargasso Sea	1973	9	1.4 ± 0.9) (< 0.09 – 3.6)	Aroclor 1254	"
North Pacific (N.E. Pacific Gyre)	1972	2	2.6 ± 1.4* (1.6 – 3.6)	Aroclor 1254	"
Northeast Pacific (off Mexico)	1975	23	(< 10 – < 300)**	Aroclor 1254	"
Northwest Pacific	1975	13	0.41 ± 0.10 (0.25–0.56)	Kanechlor 300, 400 and 500	"
" "	1976	8	0.54 ± 0.29 (0.29–1.11)	Kanechlor 300, 400 and 500	"
" "	1978	6	0.35 ± 0.13 (0.32–0.59)	Kanechlor 300, 400 and 500	"
" "	1979	5	0.33 ± 0.05 (0.22–0.38)	Kanechlor 300, 400 and 500	"
Indo-Pacific	1980–81	18	0.12 ± 0.06 (0.04–0.25)	Kanechlor 300, 400 and 500	"
Antarctic (below Australia)	1980–81	9	0.06 ± 0.01 (0.04–0.08)	Kanechlor 300, 400 and 500	"

* Mean ± sd (range)
** Pentachlorobiphenyls

TABLE 3. CONCENTRATIONS (ng l^{-1}) OF DDT AND METABOLITES IN OPEN OCEAN SURFACE WATERS

Region	Year	N	pp'DDT	op'DDT	pp'DDE	pp'DDD	Σ DDT	Source
Mediterranean	1972		(2–24)		(0.5–2.0)	(0.7–2.0)		Harding (1986)
"	1977	3	1.2		0.2 = Σ DDE + DDD		1.4	"
"	1978	16	0.4		0.2 = Σ DDE + DDD		0.6	"
"	1979	32	0.7		0.3 = Σ DDE + DDD		1.0	"
"	1980	35	0.5		0.5 = Σ DDE + DDD		1.0	"
Subarctic Atlantic	1977	10	0.4		0.5 = Σ DDE + DDD		0.9	"
" "	1978	73	0.2		0.4 = Σ DDE + DDD		0.6	"
" "	1979	54	0.4		0.2 = Σ DDE + DDD		0.6	"
" "	1980	49	0.2		0.1 = Σ DDE + DDD		0.3	"
Subtropical Atlantic	1977	15	0.7		0.1 = Σ DDE + DDD		0.8	"
" "	1978	30	0.3		0.2 = Σ DDE + DDD		0.5	"
" "	1979	49	0.7		0.1 = Σ DDE + DDD		0.8	"
" "	1980	39	0.5		0.2 = Σ DDE + DDD		0 7	"
Tropical Atlantic	1979	11	0.3		0.4 = Σ DDE + DDD		0.7	"
" "	1980	34	0.4		0.3 = Σ DDE + DDD		0.7	"
Northwest Atlantic								
Gulf Stream-	1972	6	8 nd++		2.9 ± 2.7	0.8 nd		"
Sargasso Sea					(0.35–7.5)			"
" "	1973	9	0.2 ± 0.1* (< 0.15 –0.5)	< 0.05				"
Northeast Atlantic								
Norway to Iceland	1973		+**		+	+	500 ± 90	"
" "	1975		+		+	+	500 ± 90	"

./.

TABLE 3 CONTINUED:

Region	Year	N	pp′DDT	op′DDT	pp′DDE	pp′DDD	Σ DDT	Source
North Pacific (N.C. Pacific Gyre)	1972	2	< 0.002+		< 0.01			Harding (1986)
Bering Sea	1978	7	+		+		0.02 ± 0.01 (0.01 – 0.04)	″
″	1981	2	0.0016,0.0007		0.0011,0.0007			Kawano *et al.* (1988)
Northwest Pacific	1975	13	+		+	+	0.11 ± 0.05	Harding (1986)
″ ″	1976	8	+		+	+	0.90 ± 0.26	″
″ ″	1978	6	+		+	+	0.25 ± 0.03	″
″ ″	1979	8	+		+	+	0.38 – 0.41	″
Northeast Pacific								
Mexico	1975	23	(< 1 – 20)		(< 1 – < 10)	(< 1 – < 80)		″
Indo-Pacific	1980-81	15	0.03 ± 0.03 (0.005 – 0.091)	0.01 ± 0.01 (0.001 – 0.018)	0.004 ± 0.003 (0.001 – 0.013)		0.04 ± 0.04 (0.007 – 0.13)	″
Indian Ocean								
Arabian Sea & Bay of Bengal	1976	6	+		+	+	0.10 ± 0.04 (0.06 – 0.16)	″
Antarctic (below Australia)		12	0.01 ± 0.01 (0.003 – 0.053)	0.003 ± 0.002 (0.001 – 0.006)	0.002 ± 0.001 (0.001 – 0.005)		0.02 ± 0.01 (0.005 – 0.058)	″
China Sea	1977	3	+		+	+	0.08 ± 0.02	″

\+ Mean
\+ + Not detected
* Mean ± sd
** Present

TABLE 4. CONCENTRATIONS (ng l^{-1}) OF HCH ISOMERS (BHC) IN OCEANIC SURFACE WATERS

Region	Year	N	α –HCH	β –HCH	γ –HCH*	Σ HCH	Source
Mediterranean	1972				(2 – 9.5)		Harding (1986)
"	1977	3			1.3		"
"	1978	16			0.7		"
"	1979	32			1.3		"
"	1980	35			1.5		"
Subarctic Atlantic	1977	10			0.5		"
" "	1978	73			0.8		"
" "	1979	54			0.4		"
" "	1980	49			0.3		"
Subtropical Atlantic	1977	15			1.3		"
" "	1978	30			0.6		"
" "	1979	49			0.5		"
" "	1980	39			0.8		"
Tropical Atlantic	1979	11			0		"
Northwest Atlantic	1972	6			0.04 ± 0.07 (0.10–0.15)**		"
North Pacific							
Bering Sea	1979	7				3.9 ± 0.4 (3.2–4.4)	"
"	1981	2	2.7,2.8	0.13,0.20	0.68,0.61		Kawano *et al.* (1988)
Northwest Pacific	1976	8				10.6 ± 5.4	Harding (1986)
" "	1978	6				1.3 ± 0.4	"
" "	1979	8				7.3 ± 3.6	"
Indo-Pacific	1980-81	15	0.9 ± 1.0 (0.1 – 3.4)	0.2 ± 0.3 (0.02 –1.0)	1.4 ± 1.3 (0.16 – 3.7)		"
Indian Ocean							
Arabian Sea & Bay of Bengal	1976	6				1.1 ± 0.6 (0.2 – 1.9)	"
Antarctic	1980-81	12	0.09 ± 0.07 (0.02–0.10)	0.03 ± 0.02 (0.008–0.66)	0.5 ± 0.3 0.19–0.94		"
China Sea	1977	3			(1.3–3.4)	2.2 ± 1.1	"

* Lindane (gamma isomer of 1,2,3,4,5,6,–hexachlorocyclohexane)

** Mean ± sd (range)

TABLE 5. CONCENTRATIONS (ng l^{-1}) OF Hg, Cd AND Pb IN COASTAL SURFACE WATERS

Location		Hg	Cd	Pb
Northwest	Atlantic			20 – 125[b]
	Cape Cod			21[e]
	Long Island Sound	*6 ± 1[j]		
	Gulf of St. Lawrence		22[s]	
	South Atlantic Bight			25 ± 10[t]
	Canadian Arctic		59 ± 11[v]	
	New York Bight	10 – 90[cc]	3 – 270[cc]	
Northeast	Atlantic			
	Greenland	*0.9[c]		
	Irish Sea		25[ii]	
	Bay of Biscaye	**0.7 – 5.5[n], **3[z]	13 – 20[q]	
	English Channel		20 – 25[o], 11–16[ii]	
	Tagus estuary outlet	**80[z]		
	Scottish Coast		9 – 19[ii]	
Northeast	Pacific			
	Monterey Bay		9 – 23[r]	7.6[a]
	Juan de Fuca Strait			6.5[g]
	Puget Sound	**0.2 – 1.0[k]	70 – 100[hh]	20 – 110[hh]
	Southern California Bight			25[u]
	Near Los Angeles outfall			150[u]
Northwest	Pacific			
	Tokyo Bay	3 – 5[bb]		
Southwest	Atlantic			
	Brazil		1.2 – 12[h]	
Southwest	Pacific			
	Australia		< 10 – 60[ff]	
	New Zealand (Taranaki harbour)		2 – 30[ii]	
Southeast	Pacific	0.1[oo]		4 – 11[ee]
Mediterranean		0.5 – 20[l]		
	Spain		7.6 – 9[f]	
	Rhône Delta		11 ± 6[x], 4[x], 20[x]	77[w]
	Italy	**6.3[y], *2.0[y]		
Black Sea			7.8[ll]	12[ll]
Baltic Sea		**3.3 (0.9 – 11[aa])	59[dd]	40[dd]
	Bothnian Bay		34 ± 9[p],39 ± 6[nn]	246 ± 66[nn]
	Bothnian Sea		36 ± 8[nn]	120 ± 29[nn]
	Northern Baltic Proper		39 ± 13[nn]	189 ± 24[nn]
	Gulf of Finland		36 ± 5[nn]	324 ± 24[nn]
	North German Coast		29 ± 6[p]	

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TABLE 5 CONTINUED:

Location		Hg	Cd	Pb
North Sea			46 ± 18[h]	
	Framvaren fjord, Norway	*0.3 – 25[kk]	56[ll]	73[ll]
	Kattegat/Skagerrak	5 ± 2 – 12 ± 3[gg]	22.5[d]	50[d]
	UK		**10 – 60[i]	**30 – 265[i]
	German Bight		71[jj]	
Indian				
	The Gulf	**9 – 25[m]	16 – 30[m]	60 – 120[m]
	Oman (Arabian Sea)	**4 – 18[m]	5 – 100[m]	50 – 120[m]
	Red Sea		5[bb]	
	India (Arabian Sea)	**30–130[mm]		
	India (Bay of Bengal)	** < 10–100[mm]		

* Reactive ** Total dissolved

FOOTNOTE:

a. Schaule & Patterson (1981)
b. Cited in Flegal & Patterson (1983)
c. Olafsson (1983)
d. Magnusson & Westerlund (1983)
e. Boyle & Huested (1983)
f. Boyle *et al.* (1985)
g. Flegal *et al.* (1986)
h. Kremling (1985)
i. Balls (1985a)
j. Gill & Fitzgerald (1987)
k. Bloom & Crecelius (1983)
l. Copin-Montegut *et al.* (1986a)
m. Fowler *et al.* (1984)
n. Cossa & Noel (1987)
o. Danielsson *et al.* (1985)
p. Kremling & Petersen (1984)
q. Boutier & Chiffoleau (1986)
r. Bruland *et al.* (1978)
s. Cossa (1987)
t. Windom *et al.* (1985)
u. Patterson *et al.* (1976)
v. Campbell & Yeats (1982)
w. Huynh Ngoc *et al.* (1988b)
x. Huynh Ngoc *et al.* (1988a)
y. Ferrara *et al.* (1986)
z. Figueres *et al* (1985)
aa. Brugmann (1985)
bb. Kumagai & Nishimura (1978)
cc. Segar & Davis (1984)
dd. Brugmann (1986)
ee. Flegal (1986)
ff. Denton & Burdon-Jones (1986)
gg. Gustavsson & Edin (1985)
hh. Paulson & Feely (1985)
ii. Smith (1986)
jj. Kremling & Hydes (1988)
kk. Iverfeldt (1988)
ll. Haraldsson & Westerlund (1988)
mm. Sanzgiryet *al.* (1988)
nn. Bordin *et al.* (1988)
oo. Gill & Fitzgerald (1988)

TABLE 6. CONCENTRATIONS (ng l^{-1}) OF DDT AND METABOLITES IN COASTAL AND NEAR-SHORE SURFACE WATERS

Region	Year	N	pp'DDT	pp'DDE	pp'DDD	Σ DDT	Source[X]
Baltic Sea							
Western Baltic Sea	1974	21	0.2 ± 0.1		2.4 ± 0.6		Harding (1986)
" " "	1975	18	0.2 ± 0.1		0.1 ± 0.1		"
Mediterranean							
Marseille	1971		(70–180)	(30–120)			"
Monaco	1974	3				0.04 ± 0.01 (0.04–0.05)	"
Northeast Atlantic							
British Isles	1974	31	+	+	+	0.05 ± 0.07 (0.01–0.25)	"
Brest	1977-78					1.5 ± 2.8	Marchand *et al.* (1983)
Northwest Atlantic							
North Carolina	1972	4	nd***(< 8.0)	7.9 ± 6.9 (3.5–18.1)	nd (< 0.8)		Harding (1986)
Corpus Christi Bay (Texas)	1980	8				1.32 ± 1.04 (0.2–3.1)	"
Southwest Atlantic							
Santos Estuary (Brazil)	1974-75	64				(nd–40)	"
Blanca Bay (Argentina) +	1980-81	15	37.9 ± 87.2 (nd–346.0)		nd		"
North Pacific							
Hawaii Canal	1971	6	9.0	0.3	3.6		"
			(0.4–41.0)	(0.1–0.6)	(0.1–10.0)		"

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TABLE 6 CONTINUED:

Region	Year	N	pp′DDT	pp′DDE	pp′DDD	Σ DDT	Source[x]
Northeast Pacific							
San Francisco Bay	1969	24	5 ± 5*	2 ± 1	2 ± 2	8 ± 5	〃
			(1–18)	(1–4)	(1–7)	(3–23)	〃
Monterey Bay	1970	4	+**	+	+	2.4 ± 0.2	〃
to San Diego						(2.4–2.7)	〃
California Coast	1973					(0.9–17.5)	〃
〃 〃	1974	7	0.5 ± 0.6	1.0 ± 1.6			〃
			(< 0.1–1.9)	(< 0.1–4.6)			
Southern California	1973	14	0.6 ± 0.3	0.10 ± 0.03		0.7 ± 0.3	〃
Bight			(0.3–1.3)	(0.06–0.13)		(0.34–1.43)	〃
〃 〃	1975	20	(< 0.05–0.8)	(< 0.05–5.9)	(< 0.03–2.2)		〃
North Sea							
German Bight	1974	24	0.3 ± 0.2		1.5 ± 0.6		〃
			(0.1–0.6)		(0.5–2.6)		
〃 〃	1975	17	0.33 ± 0.15	0.11 ± 0.06	0.20 ± 0.15		
			(0.11–0.63)	(nd–0.21)	(nd–0.57)		
Hanö Bight	1975	8	0.04 ± 0.08	1.7 ± 1.6			〃
			(nd–0.21)	(0.8–5.2)			〃
Antarctic++							
Syowa Station	1981	6	+	+		0.008 ± 0.008	〃
						(0.001–0.021)	

x Original references for data cited in Harding, 1986.
* Mean ± sd (range)
** Present
*** Not detected
+ o,p′DDT, 13.4 + 24.5, (nd – 94.5)
+ + o,p′DDT, +

TABLE 7. CONCENTRATIONS (ng l^{-1}) OF HCH ISOMERS (BHC) IN COASTAL AND NEAR-SHORE SURFACE WATERS

Region	Year	N	α –HCH	β –HCH	γ –HCH	δ –HCH	Σ –HCH	Source
Baltic Sea								
Western Baltic	1975	18	6.8 ± 1.3 (4.7–8.7)		5.0 ± 1.0 (3.4–7.0)			Harding (1986) ″
″ ″	1976	14	9.8 ± 4.1 (5.5–21.8)	0.8 ± 0.2 (0.3–1.3)	3.4 ± 0.8 (1.9–5.2)			″
″ ″	1978	13	7.5 ± 1.8 (4.1–11.0)	0.4 ± 0.2 (0.2–0.7)	2.9 ± 0.8 (1.8–4.5)			″
Mediterranean								
Marseille	1971				70 (40–90)			″
Monaco	1981-82	4			(nd–1.29)			″
French coastline	1984	12	< 1 –4		< 1 – 8			Marchand *et al.* (1988)
Northeast Atlantic								
German Bight	1975	17	6.5 ± 5.5* (2.0–20.0)		5.6 ± 4.7 (1.9–16.0)			Harding (1986) ″
Northwest Atlantic								
Corpus Christi	1980	8	0.27 ± 0.15		0.07 ± 0.05			″
Bay (Texas)			(< 0.05–0.49)		(< 0.01–0.11)			″
Southwest Atlantic								
Santos estuary, Brazil	1974-75	77					(nd–1,020)	″
Blanca Bay,	1980-81	15	22.4 ± 16.4		27.8 ± 22.3	6.6 ± 5.4		″
Argentina			(5.1–50.7)		(5.4–65.1)	(0.5–17.5)		″
North Pacific								
Hawaii Canal,	1971	6			0.9			″
(USA)					(0.3–2.0)			″
Antarctic								
Syowa Station	1981	6					0.48 ± 0.27 (0.29–0.39)	

nd: Not detected

* Mean ± SE

TABLE 8. CONCENTRATIONS (ng l^{-1}) OF PCBs IN COASTAL AND NEAR-SHORE SURFACE WATERS

Region	Year	N	Concentration	Quantified as	Source
Baltic Sea					
Western Baltic Sea	1974	21	2.9 ± 1.2 (1.1–5.9)	Clophen A 30	Harding (1986)
" " "	1975	18	1.1 ± 0.8 (nd–3.9)	Clophen A 50	"
" " "	1976	14	7.2 ± 4.1 (1.1–15.6)	Clophen A 60	"
" " "	1978	12	5.7 ± 2.5 (3.5–11.9)	Clophen A 60	"
Hanö Bight	1975	8	0.9 ± 0.9	Clophen A 50	"
Mediterranean					
Marseille	1971		(100–210)**		"
French coastline	1975	11	13.1 ± 12.3 (1.5–38.0)	Phenoclor DP-5	"
" "	1984	14	< 2–11		Marchand *et al.* (1988)
Monaco	1981-82	4	(0.2–1.2)	Aroclor 1254	Harding (1986)
Northeast Atlantic					
Liverpool Bay (British Isles)	1974	31	0.4 ± 0.3 (0.15–1.5)	Aroclors 1254 and 1260	Harding (1986)
German Bight	1974	22	3.1 ± 0.9	Clophen A 30	"
" "	1975	17	2.1 ± 0.9 (0.8–3.6)	Clophen A 50	"
Holland (Dutch coast)	1976		0.7–8.1		Duinker & Hillebrand (1979)
Oslofjord and Frierfjord, Norway	1976-77	3	5 ± 1 (4 – 10)	Aroclor 1254 and Clophen A 60	Harding(1986)
Brest	1977-78	96	4.3 ± 2.8		Marchand *et al.* (1983)
Gironde estuary	1985		10 ± 6 (3–25)		Marchand (1987)
Seine Bay (Channel)	1985		(3–6)		"
Seine estuary	1985		29 ± 12 (40–370)		"
Northwest Atlantic					
Narragansett Bay (USA)	1971		150 ± 40*	Aroclor 1254	Harding (1986)
New England	1974	6	0.8	Aroclor 1260	"
Corpus Christi Bay (Texas)	1980	8	4.8 ± 10.7 (0.1–31.0)	Aroclor 1260	"
Northwest Pacific					
Japan	1972		(0.3–13.9)		"
"	1973		(0–6.4)		"

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TABLE 8 CONTINUED :

Region	Year	N	Concentration	Quantified as	Source
Northeast Pacific					
California	1973		(5.4–16.3)		Harding (1986)
"	1974	7	12.7 ± 10.6 (3.0–35.6)	Aroclor 1254	"
San Francisco Bay	1978	1	0.66		De Lappe *et al.* (1983)
Farallon Islands	1978	2	(0.018–0.028)		"
Golden Gate Bridge	1978	2	(0.145–0.160)		"
Southern California Bight	1973	7	0.4 ± 0.10 (0.3–0.5)	Aroclor 1254	Harding (1986)
" "	1975	20	(0.04–2.0) +	Aroclor 1254	"
Antarctica					
Syowa Station	1981-82	6	0.05 ± 0.01 (0.03–0.07)	Kanechlor 300, 400 or 500	Harding (1986)

* Mean ± SE (range)

** Range

\+ Pentachlorobiphenyls

TABLE 9. CONCENTRATIONS ($\mu g\ g^{-1}$ dry) OF Hg, Cd AND Pb IN SURFACE SEDIMENTS FROM THE OPEN OCEAN.

Single values are mean or median; values in parentheses are ranges.

Region	Hg	Cd	Pb
North Atlantic Ridge	0.32 (0.008 – 0.6)[c]	0.23[d] 0.65[d]	15 – 21[e]
Mediterranean	(0.01 – 0.97)[a,b,f]		

a. Selli *et al.* (1972)
b. Kosta *et al.* (1978)
c. Aston *et al.* (1972a)
d. Aston *et al.* (1972b)
e. Veron *et al.* (1987)
f. Bargagli *et al.* (1988)

TABLE 10. PCB CONCENTRATIONS (ng g^{-1} dry) IN SURFACE SEDIMENTS FROM THE OPEN MEDITERRANEAN SEA (from Fowler, 1987)

Year	Region	Sample No.	Range	$\bar{X}$
1975	Algero-Provençal basin	5	0.8 – 9.0	4.0
	Gibraltar sill & Siculo-Tunisian sill	2	0.8	0.8
	Algerian margin	1	0.8	0.8
	Ionian Sea	3	0.8 – 5.1	2.8
1976[a]	Ligurian Sea	14	1.5 – 33	11
1977[b]	Tyrrhenian Sea	2	0.8 – 1.3	1.1
	Ionian Sea	3	1.2 – 1.6	1.4
	Aegean Sea	2	0.6	0.6
	Levantine & Central Basin	5	0.6 – 8.9	3.8

a Grab samples
b Undisturbed sediment cores (top 1 cm)

TABLE 11. CONCENTRATIONS (ppm dry) OF Hg, Cd AND Pb IN SURFACE LAYERS OF NEAR-SHORE SEDIMENTS. Single values are mean or median.

Location		Hg	Cd	Pb
Baltic Sea		0.01 – 1.0[a]	0.08 – 6.8[a]	7 – 150[a]
		0.23(0.08–0.80)[xx]	3.2(1.1–7.2)[xx]	54(35–94)[xx]
Northeast	Atlantic			
	Irish Sea	0.07 – 3.3[c]		
	Tagus Estuary, Portugal	0.02 – 9.4[u]		
	France	0.08 – 0.12[ii]	0.15 – 0.2[ii]	31 – 45[ii]
	Spain		1.7 ± 0.42[mm]	58 ± 22[mm]
	W. Africa	0.002 – 1.4[kk]	0.1 – 2.8[kk]	2 – 87[kk]
	Ivory Coast Lagoon	0.004 – 2.3[vv]		7 – 250[vv]
Northwest	Atlantic			
	Canada	0.04[pp], 0.61 ± 0.33[ccc]	0.16[pp], 0.10–0.47[ccc]	24[pp]
	Bermuda		< 0.25 – 0.99[p]	6.4 – 230[p]
	New York Bight	0.12 – 4.9[q]	< 0.47 – 9.6[q]	5 – 270[r]
	Mexico		0.1 – 2.4[bb]	10 – 91[bb]
	Trinidad		0.05 – 4.5[gg]	6.7 – 29[gg]
	South Carolina		0.01 – 0.46[nn]	0.3 – 30[nn]
Northeast	Pacific			
	Puget Sound	0.276[yy]	0.367[yy]	43.8[yy]
	Southern California Bight	0.13 – 4.4[e]	1.1–6.0[e], 0.4–140[f]	32–130[e], 10–540[f]
			0.3 – 1.3[cc]	4.5 – 12.5[cc]
	Costa Rica	0.022 + 0.029[hh]	0.12 ± 0.08[hh]	5.3 ± 3.0[hh]
Northwest	Pacific			
	Korea			25 – 120[zz]
	Seto Inland Sea		0.14 – 0.88[g]	14 – 43[g]
	Minimita Bay	25[v]		
	Malaysia		N.D. – 1.25[ll]	6.5 – 32[ll]
	Thailand		0.1 – 0.4[qq]	
			0.005 – 0.11[ll]	11–18[qq], 6.5 ± 1.5[ll]
	Phillipines			36 – 40[ll]
	China			21.6 ± 3.8[aaa]
Southwest	Atlantic			
	Argentina		1.9 – 3.1[n]	
	Brazil	0.2 – 1.4[rr]	0.3 – 1.3[rr]	15 – 70[rr]
Southeast	Pacific			
	Chile	0.11 – 0.49[aa]	1.05 – 9.16[aa]	8.6 – 74[aa]

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TABLE 11 CONTINUED :

Location	Hg	Cd	Pb
Southwest Pacific			
Port Phillip Bay, Australia		0.15 – 9.9[i]	4.6 – 180[i]
Manukau (Auckland) Harbour, New Zealand			98 – 247[uu]
New Zealand estuaries and harbours			43 + 19[ww]
New Zealand fjords and sounds			35 + 10[ww]
Mediterranean	0.01 – 37[w]	0.2 – 49[w]	4.5 – 280[w]
Adriatic	< 0.1 – 16.9[oo]	< 0.05 – 5.6[oo]	5.3 – 96[oo]
Ligurian Sea		0.3 – 7.0[j]	36 – 180[j]
Sicily	0.03 – 2.0[z]	2.5 – 4.6[s]	4.5–17[s], 7.5–20[z]
Bay of Naples	0.1 – 1.75[tt]	5 – 200[m]	
Saronikos Gulf, Greece	0.3 – 10[l]		
Israel	0.2 – 0.5[k]		15 – 28[k]
Spain	0.06 – 16.5[dd]	0.03 – 4.0[dd]	4.8 – 550[dd]
Thermaikos, Greece		0.2 – 5.1[ss]	18 – 246[ss]
Gulf			
Kuwait		0.75 – 3.0[o]	10 – 40[o]
Iraq		0.14 – 0.23[ee]	5.6 – 25.6[ee]
Oman	0.012 – 0.023[ff]	2.5 – 4.7[ff]	49 – 63[ff]
Indian Ocean			
Albany, Australia		0.26 – 7.6[h]	13 – 180[h]
Bombay, India	2.5(0.018–8)[t], 0.038–0.08[bbb]	10 ± 2[jj]	48 ± 7[jj]
Karwar, India	0.05 – 1.32[bbb]		
Bay of Bengal, India	0.95 – 5.3[y]		
Red Sea, Jordan		2 – 18[x]	83 – 225[x]
North Sea		0.29[b]	21.1[b]
Plymouth estuary	0.02 – 2.6[d]		

FOOTNOTE:

a. Brugmann (1987)
b. Nicholson & Moore (1981)
c. Rae & Aston (1981)
d. Millward & Herbert (1981)
e. Hershelman *et al.* (1981)
f. Katz & Kaplan (1981)
g. Aoyama *et al.* (1982)
h. Talbot (1983)
i. Talbot *et al.* (1976)
j. Cosma *et al.* (1982)
k. Amiel & Navrot (1978)
l. Griggs *et al.* (1978)
m. Griggs & Johnson (1978)
n. Sericano & Pucci (1982)
o. Anderlini *et al.* (1982)
p. Lyons *et al.* (1983)
q. Timoney *et al.* (1978)
r. Carmody *et al.* (1973)
s. Castagna *et al.* (1982)
t. Zingde & Desai (1981)
u. Figueres *et al.* (1985)
v. Kumagai & Nishimura (1978)
w. UNEP (1986)
x. Abu-Hilal (1987)
y. Sasamal *et al.* (1987)
z. Castagna *et al.* (1987)
aa. Salamanca *et al.* (1986)
bb. Paez-Osuna (1986)
cc. Thompson *et al.* (1986)
dd. Modamio (1986)
ee. Abaychi & Douabul (1986)
ff. Burns *et al.* (1982)
gg. Hall & Chang-Yen (1986)
hh. Dean *et al.* (1986)
ii. Carruesco & La Paquellerie(1985)
jj. Patel *et al.* (1985)
kk. Portman (1987)
ll. Gomez (1986)
mm. Gomez Parra *et al.* (1984)
nn. Sanders (1984)
oo. Donazzolo *et al.* (1984)
pp. Ray & Macknight (1984)
qq. Hungspreugs & Yuangthong (1983)
rr. De Luca Rebello *et al.* (1986)
ss. Voutsinou-Taliadouri & Satsmadjis (1983)
tt. Baldi *et al.* (1983)
uu. Aggett & Simpson (1987)
vv. Kouadio & Trefry (1987)
ww. Stoffers *et al.* (1983)
xx. Brugmann (1988)
yy. Bloom & Crecelius (1987)
zz. Lee *et al.* (1988)
aaa. Zhang *et al.* (1988)
bbb. Sanzgiry *et al.* (1988)
ccc. Pelletier & Canuel (1988)

TABLE 12. CHLORINATED HYDROCARBON CONCENTRATIONS (ng g^{-1} dry) IN SURFACE SEDIMENTS FROM COASTAL AND NEAR-SHORE AREAS

Location		Σ PCB	Σ DDT
Northwest	Atlantic		
	Gulf of Maine	< 100[e], 40 – 340[f] Trace–130[h]	
	New Bedford Bay, Mass.	8400[j]	
	Escambia Bay, Florida	< 30 – 480,000[j,k]	
	New York Bight	0.5 – 2,200[j,k]	
	Gulf of Mexico, USA	0.2 – 35[j,k]	< 0.03[u]
	Gulf of Mexico Carribean (Mexico)		0.3 – 2.27[aa]
	Chesapeake Bay	4 – 400[k]	
	Hudson-Raritan Estuary	286 – 1,950[l]	116 – 739[l]
Northeast	Atlantic		
	Brittany	< 0.5[c]	
	Brest, France (subtidal)	0.4 – 185[x]	< 0.1 – 17.6[x]
	(intertidal)	3.3 – 2,100[x]	8.0 – 52.2[x]
	Seine Estuary	15 ± 8[c]	< 0.1 – 0.4[x]
	Loire Estuary	51 ± 34[c]	< 0.5
	Irish Sea, UK	< 2 – 2,890[j]	
	Ivory Coast	2 – 213[w]	2 – 997[w]
Mediteranean			
	Venice	15[a]	1.4[a]
	Adriatic	1 – 17[b]	1.3[p], < 1 – 8[t]
	Northwestern	0.3–1,200[b], 11–61[r], < 2–250[cc]	3.6[r], 0.7–44[cc]
	Tyrrhenian	0.6 – 3,200[b]	4[d], < 20[d]
	Ionian	0.8 – 457[b]	
	Aegean	1.3 – 775[b]	7.1 – 1,893[s]
	Eastern	1.9 – 4.0[b]	
	Central & Western	0.5 – 323[b]	< 1 – 28[q]
	Marseille Bay	157 ± 12[c]	10 – 50[x]
Northeast	Pacific		
	San Pedro Basin, Calif.	1 – 13[i]	5 – 30[i]
	Santa Monica Basin, Calif.	0 – 9[i]	30 – 160[i]
	Palos Verdes, Calif.	80 – 7,420[j,k]	1,600 – 100,000[n,v]
	Puget Sound, Wash.	80 – 640[k]	
	San Francisco Bay	30 – 50[k]	
Northwest	Pacific		
	Gulf of Thailand	ND[g]	22 – 56[g]
	Osaka Bay, Japan	40 – 2,000[j]	
	Harimanada Bay	50 – 400[j]	
Soutwest	Pacific		
	Queensland estuaries, Aust.	6 – 350[m]	
	Port Phillip Bay, Aust.	< 10 – 390[m]	
	Bass Strait, Aust.	< 10[m]	
	Manukau Harbour, New Zealand	0.5 – 14.2[bb]	1.2 – 2.3[bb]

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TABLE 12 CONTINUED:

Location		Σ PCB	Σ DDT
Southwest Atlantic			
	Brazil	(N.D. – 84)[z]	(N.D. – 25)[z]
North Sea		11.5 – 40.5[j]	
	Dunkerque	134 + 134[c]	< 0.1[x]
	Norway	14 – 28[s]	0.16 – 0.36[s]
Baltic Sea			
	Sweden	40 – 160[j]	
	Finland	10(< 10–20)[j]	
	Kiel Bay	8.4 – 10.8[j]	2.0 – 2.4[x]
	Eckernforde Bight	134 – 212[s]	28 – 46[s]
Indian Ocean			
	East India		N.D. – 780[o]
	West India (Arabian Sea)		*43(14–358)[y]

* Factors of 2 used to convert from wet to dry weight concentrations.

FOOTNOTE:

a. Pavoni *et al.* (1987)
b. Cited in Fowler (1987)
c. Marchand (1987)
d. Baldi *et al.* (1983)
e. Larsen *et al.* (1984)
f. Larsen *et al.* (1984b)
g. Menasveta & Cheevaparanapiwat (1981)
h. Larsen *et al.* (1985)
i. Thompson *et al.* (1986)
j. Cited in Marchand (1987)
k. Cited in Segar & Davis (1984)
l. Young *et al.* (in preparation)
m. Richardson *et al.* (1987)
n. Young *et al.* (in press)
o. Sen Gupta (1986)
p. Vililic *et al.* (1979)
q. Amico *et al.* (1982)
r. Burns & Villeneuve (1983)
s. Cited in Ernst (1984)
t. Picer & Picer (1979)
u. Murray *et al.* (1981)
v. Smokler *et al.* (1979)
w. Marchand & Martin (1985)
x. Marchand *et al.* (1983)
y. Sakar & Gupta (1987)
z. Montone (1987)

aa. Rosales-Hoz & Alverez-Leon (1979)
bb. Fox *et al.* (1988)
cc. Marchand *et al.* (1988)

TABLE 13. MEAN TRACE ELEMENT CONCENTRATIONS ($\mu g\ g^{-1}$) IN EUPHAUSIIDS FROM DIFFERENT OCEANIC REGIONS. Means and ranges (in parentheses) are from single independent surveys (From Fowler, 1986).

Region	Cd	Hg
Mediterranean		
1975	0.9 (0.4–2.0)	0.233
1977	0.34 (0.11–0.66)	0.138 (0.028–0.239)
N.E. Atlantic	0.3	0.260
N.E. Pacific	2.8 (0.8–5.5)	0.090 (0.050–0.150)
" "	1.4 (0.9–2.2)	0.137 (0.050–0.440)
" "		0.223 (0.026–0.497)
N.W. Pacific	1.2 (0.4–2.2)	

TABLE 14. PCB CONCENTRATIONS (ng g^{-1} wet weight) IN OPEN OCEAN PLANKTON
(From Tanabe & Tatsukawa, 1986; Fowler & Elder, 1980)

Location	Year	N	Range	Mean
North Atlantic	1970		300 – 450	380
Northeast Atlantic	before 1972	22	10 – 110	
North and South Atlantic	1970-1972	53		200
South Atlantic	1971	4	18 – 640	200
Mediterranean	1977	7	1.7 – 25	7.1
Western North Pacific	1981	1		1.1
Bering Sea	1982	3	1.0 – 1.6	1.3
Western South Pacific	1981	3	1.2 – 2.3	1.7
Antarctic (Ross Sea)	1972	1		< 3
Antarctic (50–65°S, 124°–126°E)	1981	3	0.2–1.0	0.5

TABLE 15. AVERAGE CONCENTRATIONS (μg g^{-1} dry) AND RANGES OF HEAVY METALS IN THE MUSSEL *MYTILUS* AND CLOSELY-RELATED SPECIES. Statistics applied after exclusion of assumed outlying data or those from obviously contaminated areas.

Region	Cd	Hg	Pb
Baltic Sea & Kattegat	2.6 ± 2.6[a,m] (0.4–12.9)	(0.04–0.57)[a,m]	10 ± 2[i,m] (1.4–20)
North Sea	1.4 ± 0.8[a] (0.2–11)	(0.08–1.00)[a]	(0.9–12)[j]
Irish Sea	2.1 ± 1.2[a] (0.2–5.4)	0.36 ± 0.23[a] (< 0.09–0.96)	15.9(5.2–30.7)[l]
English Channel	1.4 ± 0.8[a] (0.5–4.7)	0.24 ± 0.12[a] (0.09–1.30)	5.3(3.2–7.2)[l]
Gulf of Gascony (Northeast Atlantic)	1.0 ± 0.8[a] (0.3–2.6)	(0.11–1.23)[a]	
Mediterranean	1.3 ± 0.7[a] (0.4–3.0)	(0.04–1.52)[a]	0.8 ± 0.8[e]
Adriatic	(0.73–1.6)[d]	(0.10–0.20)[d]	(1.18–2.13)[d]
Northwest Pacific	(0.4–5.2)[a]	(0.04–2.1)[a]	0.73 ± 0.51[h]
Southwest Pacific	(0.4–18.5)[a]	(0.10–2.7)[a]	0.67(0.1–2.0)[f]
Northeast Pacific	4.8 ± 2.6[a] (0.5–16.2)	0.24 ± 0.08[a] (0.09–0.49)	2.47 ± 4.0[c] (0.42–23.3)
Southeast Pacific	3.3[m]		(13.6–18.4)[n]
Northwest Atlantic	1.9 ± 0.7[a] (0.9–4.2)	0.16 ± 0.10[c] (0.05–0.47)	4.6 ± 3.2[c] (1.4–14)
Gulf of Saint Lawrence	2.0 ± 0.6[a] (1.0–4.2)	0.16 ± 0.08[a] (0.05–0.40)	
Southwest Atlantic	(3.6–6.0)[g]		(0.8–1.9)[g]
Northwest Indian Ocean			
Arabian Gulf (Oman)	16.4 ± 8.6[b] (8.3–25.5) 7.6[o]	0.082 ± 0.006[b] (0.076–0.088)	1.32 ± 0.99[b] (0.4–2.33)
(Pakistan)	(0.24–0.61)[q]	(0.064–0.070)[q]	(2.11–2.57)[q]
(GOA, India)	7.7[p]	0.50[p,r]	7.3[p]

If weight data are not reported, concentrations given in the original reference as wet weight have been multiplied
by 5.6 to obtain dry weight values (Cossa, 1987).

FOOTNOTE:

a. Cossa (1987)
b. Fowler (in press)
c. NOAA (1987)
d. Martincic *et al.* (1987)
e. UNEP (1986, 1987b)
f. Smith (1986)
g. Gil *et al.* (1988)
h. Hungspreugs *et al.* (1984)
i. Szefer & Szefer (1985)
j. Meeus-Verdinne *et al.* (1983)
k. Orren *et al.* (1980)
l. Franklin (1987)
m. Jensen (in press)
n. Ober *et al* . (1987a)
o. Burns *et al.* (1982)
p. Qasim & Sen Gupta (1988)
q. IAEA (1987)
r. Sanzgiry *et al.* (1988)

TABLE 16. RECENT AVERAGE CONCENTRATIONS (μg kg^{-1} dry) AND RANGES OF PCBs AND TOTAL DDT IN THE MUSSEL *MYTILUS* AND CLOSELY RELATED SPECIES COLLECTED IN RECENT YEARS

Region	PCBs	ΣDDT
Baltic Sea	(179 – 778)[q]	(62 – 739)[q]
North Sea	(106 – 362)[e]	(15 – 143)[e]
Irish Sea	(57 – 1,070)[e]	(92 – 590)[e]
English Channel	(380 – 480)[e]	(35 – 112)[e]
Mediterranean		
Adriatic	(17.9 – 68)[a]	(11.8 – 102)[a]
Spain	(10.8 – 1,264)[b]	(60 – 288)[b]
France	(83 – 1,825)[c]	
Northwest Atlantic		
USA	(10 – 6,808)[d]	(2.8 – 1,109)[d]
Canada	(11 – 258)[i]	
Northeast Atlantic		
France	(96 – 1,345)[c]	
Southwest Atlantic		
Brazil	4.1(2.5 – 10.5)[l]	2.3(1.23 – 5.61)[l]
Northeast Pacific		
USA	(607 – 2,052)[d]	(5.4 – 1,077)[d]
Northwest Pacific		
Hong Kong	(20 – 3,136)[f]	
Thailand	(11 – 241)[g]	(179 – 235)[g]
Japan	(3.1 – 364)[k]	
	864(17 – 2,682)[o]	
The Gulf	*(1.7 – 110)[h]	*(0.9 – 29.1)[h]
Northwest Indian Ocean		
Arabian Gulf		
Oman	4.9[n]	3.6[n]
	*(0.1 – 69)[h]	*(0.5 – 7.7)[h]
Pakistan	7.8[p]	213[p]
Southeast Pacific		
Chile		33.7[j]
Southwest Pacific		
Australia, Port Phillip Bay	(14 – 879)[m]	

If weight data are not reported, a wet weight/dry weight factor of 5.6 has ben applied (Cossa, 1987).

* Rock oysters

FOOTNOTE:

a. Najdek & Bazulic (1988)
b. Pastor *et al.* (1988)
c. Marchand (1987)
d. NOAA (1987)
e. Franklin (1987)
f. Tanabe *et al.* (1987)
g. Menasveta & Cheevaparanapiwat (1981)
h. Fowler (1988)
i. Delval *et al.* (1986)
j. Ober *et al.* (1987b)
k. Miyata *et al.* (1987)
l. Montone (1987)
m. Richardson *et al.* (1987)
n. Burns *et al.* (1982)
o. Watanabe *et al.* (1987)
p. IAEA (1987)
q. HELCOM (1981)

TABLE 17. AVERAGE CONCENTRATIONS ($\mu g\ kg^{-1}$ dry) AND RANGES OF PCBs AND TOTAL DDT IN ROCK OYSTERS (*SACCOSTREA CUCULLATA*) FROM SEVERAL LOCATIONS ALONG THE ARABIAN SEA COAST OF OMAN (from Burns *et al.*, 1982; Fowler, in press, unpublished results).

Collection Date	PCBs	ΣDDT
*September 1980	17.4 (7.8 – 39.5)	2.7 (1.03 – 4.4)
January 1983	8.5 (6.0 – 12.0)	
October 1983	23.7 (0.3 – 68.7)	4.9 (3.0 – 7.7)
April 1984	2.0 (0.1 – 4.0)	0.43 (0.14 – 0.86)
September 1985	8.4 (6.4 – 11.0)	2.9 (2.2 – 3.3)
April 1986	9.1 (6.4 – 13.0)	3.3 (1.1 – 8.4)
September 1986	3.4 (1.0 – 6.0)	2.1 (1.3 – 2.9)

* Data from Burns *et al.* (1982) converted using dry/wet weight ratio of 0.23.

TABLE 18. A COMPARISON OF PCB AND TOTAL DDT AVERAGE CONCENTRATIONS ($\mu g\ g^{-1}$ wet weight) AND RANGES IN PINNIPED BLUBBER (from Smillie & Waid, 1987)

Latitude		N	PCBs	ΣDDT	Sample
Arctic	76°N	28	$\bar{x}$ = 0.23 (0.06–1.10)	$\bar{x}$ = 0.06 (0.01–0.40)	Atlantic Walrus
Temperate N	53°N	7	$\bar{x}$ = 189.42 (22.00–576.00)	$\bar{x}$ = 10.85 (0.51–25.40)	Harbor Seal
	48°N	22	$\bar{x}$ = 4.00 (1.00–11.00)	$\bar{x}$ = 1.70 (0.60–3.10)	Harp Seal
	48°N	8	$\bar{x}$ = 15.70 (± 5.80)	$\bar{x}$ = 3.50 (± 1.00)	Gray Seal
	33°N	4	$\bar{x}$ = 17.10 (12.00–25.00)	$\bar{x}$ = 103.20 (51.00–203.00)	California Sea Lion
Tropical					
Temperate S	38°S	11	$\bar{x}$ = 0.69 (0.05–3.87)	$\bar{x}$ = 4.03 (0.03–12.05)	Australian Fur Seal
	69°S	1	0.04	0.17	Weddell Seal
Antarctic	70°S	20	$\bar{x}$ = 0.09 (0.01–0.76)	$\bar{x}$ = 0.07 (nd–0.15)*	Ross Seal

* nd = not detected

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ANNEX V

SELECTED CONTAMINANTS: TRIBUTYLTIN AND CHLORINATED HYDROCARBON BIOCIDES

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TABLE OF CONTENTS

I. TRIBUTYLTIN

1. Tributyltin (TBT), like its predecessor DDT, will become a transient in environmental science. Both are very powerful biocides and were extremely effective in their intended uses: TBT as an anti-fouling agent in marine paints, and DDT as an agricultural pesticide and as an anti-malarial agent through its decimation of carrier mosquitoes. But both impact upon non-target organisms in unacceptable ways. DDT may cause morbidities and mortalities among members of higher trophic levels such as marine birds and fish; TBT impairs the physiological activities of mid-trophic-level organisms such as molluscs, crustacea, and tunicates as well as algae. Initially, both were economically and socially attractive; but later, unexpected ecological consequences came about. DDT was initially banned by the United States in 1972. Afterwards, many countries in the northern hemisphere and some in the southern hemisphere followed suit. TBT has had its applications as an anti-fouling paint regulated or banned, following an action in France in 1982, and later in other European countries and some states of the United States. DDT is still used extensively in the tropics. But it is inevitable that as the awareness of their non-target effects develops with time, both of these substances will be phased out throughout the world. Since the DDT story has been told often, herein only the TBT environmental concerns will be developed. An up-to-date review of tributyltin in the environment has been prepared by Maguire (1987).

2. TBT is a member of the family of organotin compounds in which one to four carbon atoms are bound covalently to the tin atom. In this position the tin atom can accommodate up to three additional attached chemical groups. About 25 organotin compounds are commercially produced. Organotin compounds were first prepared in 1849 through interaction of methyliodide with tin metal. Dutch scientists recognized the biocidal properties of these substances in the early 1950s. Subsequently, they were used as fungicides, bactericides, and preservatives for woods, textiles, paper and electrical components.

3. TBT was first introduced into marine waters as an anti-fouling agent in marine paints in the mid-1960s. This compound was not only the most toxic substance ever deliberately put into the oceans, it is also the most effective anti-fouling agent so far devised for inclusion into marine paints to protect surfaces of ships and other structures from fouling organisms.

4. TBT's toxic impacts upon non-target organisms were first noted in the late 1970s in the Arcachon Bay, France, which supplies about 10 per cent of the oysters consumed in France. The Bay is also host to a large number of recreational vessels, berthed in marinas adjacent to the oyster farms. The Pacific oyster (*Crassostrea gigas*) had been flourishing there since its introduction in 1968. In early 1977, shells of the oyster were observed to be malformed, a condition characterised, by a hypersecretion of a gel that later became enclosed by a fine calcium layer. In addition, there

was little or no natural spatfall. Healthy oysters transplanted to these port areas suffered 50 per cent mortalities within 30 days. On the other hand, oysters taken from the Bay to areas free of recreational vessels rapidly assumed normal growth patterns. French scientists then hypothesized that there might be a relationship between the number of vessels and oyster morbidities as a consequence of the leakage of tributyltin from the boats to the organisms. Laboratory studies subsequently confirmed this induction. Tributyltin produced shell anomalies when it was introduced to the waters in which the organisms were placed. Further investigations showed that exposure of the organisms to TBT did not allow the settlement of spat.

5. Subsequently, concern developed in the U.K. about the effects of TBT paints on a faltering oyster industry. Over 90 per cent of the country's small yachts used TBT-containing antifouling paints. The Ministry of Agriculture, Fisheries and Food (MAFF) initiated analyses of TBT in coastal waters in 1982 and noted that the levels found corresponded to those that did not allow the settlement of oyster spat in the French laboratory experiments. Water concentrations in estuaries reached levels of 2,200 ng l^{-1}. When *C. gigas* was transported from estuaries relatively free of TBT to the Burnham-on-Crouch Estuary, where in 1982 the TBT concentrations ranged from 80 to 430 ng l^{-1}, the oysters rapidly developed malformed shells.

6. In 1982, the French government banned the use of TBT-containing paints on all pleasure craft less than 25 metres in length except on those with aluminum hulls. At first, the regulation applied only to vessels harboured on the Atlantic Ocean but was later extended to cover the entire French coast.

7. Subsequently the British scientists proposed 20 ng l^{-1} as a target level of TBT in natural waters to maintain their quality. This is based on the disruption of growth of *Ostrea edulis* at 250 ng l^{-1} and of *C. gigas* at 160 ng l^{-1}, or an average of 200 ng l^{-1}. A safety factor of 10, which takes into account the extrapolation of these acute data to the chronic situation, reduced the value to 20 ng l^{-1}.

8. However, the United Kingdom enacted more restrictive legislation following some observations on the effects of TBT levels in coastal waters on the populations of the common dogwhelk (*Nucella lapillus*), a snail. For example, upon exposure to extremely low levels of TBT, a few ng l^{-1}, dogwhelk underwent a phenomenon known as "imposex" in which females developed male characteristics including a penis and sperm ducts. The dogwhelks accumulate TBT from their foods, algae and other small organisms, which enrich themselves in TBT taking up coatings from the micro-layer at the air/sea interface. The lipophilic TBT is enriched in these surface films. The females' productivity falls down as the oviducts of the dogwhelks become stuffed with eggs which are blocked from release by newly formed male organs.

9. Organotins were found in Scottish salmon grown in sea cages coated with tributyltins to reduce fouling. The levels were reported in local newspapers and caused concern among the population. Although no public health hazards have been established, still the citizenry did voice concern about the ingestion of these toxic chemicals.

10. In 1986, the United Kingdom banned the production of TBT copolymer paints with more than 7.5 per cent TBT (measured as tin in the dry paint film) and of other paints with copper or accompanying anti-fouling agents with more than 2 per cent TBT. The latter restriction effectively prohibited the use of free-association paints.

11. In February 1987, the United Kingdom, recognizing that existing levels of TBT in marinas endanger the survival of some aquatic organisms, banned the retail sale of anti-fouling paints containing TBT. Further, a new environmental quality target level of 2 ng l^{-1} is being proposed (Duff, 1987).

12. Up to the present time the United States has taken no action at the federal level on the use of TBT containing paints. However, a number of states including Virginia, California, Washington, Alaska and Oregon, following the lead of France and the U.K., have banned or regulated the use of TBT on sea-going vessels and on fish culture and capture nets. The U.S. Environmental Protection Agency is now conducting a special review of the paints, as required by the Federal Insecticide, Fungicide and Rodenticide Act.

13. A prime participant in the United States activities is the United States Navy which argues that the use of TBT-containing anti-fouling paints, if applied to their entire fleet, would result in savings of hundreds of millions of dollars per year.

A. ANTI-FOULING PAINTS

14. Tributyltin-containing anti-fouling paints replaced the previously used biocidal formulations containing cuprous oxides. The latter suffered from an inability to kill plants and from the formation of insoluble salts on the paint surface. Beginning in the mid-1960s they were replaced to some extent by the tributyltin formulations (the following review is taken to a large extent from Anderson and Dalley, 1986).

15. There are two general formulations of paints involving tributyltin salts. The first, the so-called *free-association* paints, contain TBT physically dispersed in a hard matrix such as chlorinated rubber. The sea water penetrates the paint, taking into solution the TBT salt which then diffuses to the paint surface. The pores created by the dissolution of the TBT salt allow further penetration of the sea water into the paint. With time these pores become clogged with insoluble materials. The paints have an operational period of one to two years. An improvement in lasting power for such paints is obtained by employing an ablative paint, i.e., one in which the matrix is somewhat soluble. Initially, the free-association paints have a high release rate.

16. A second type, the copolymer paints, utilize a chemical bonding between the TBT and the paint matrix, a methacrylate or methylmethacrylate resin. Such paints depend upon the release

of TBT through sea water interacting with the hydrophobic copolymer through saponification. The portion of the film, previously insoluble with TBT, now becomes soluble upon the release of TBT. Such paints have an expected effective period of five to seven years. In addition, the release rate can be controlled through differences in the composition of the copolymer.

B. THE DOGWHELK AS A SENTINEL FOR TBT POLLUTION

17. Very sensitive organisms to TBT in environmental waters are female snails which through exposure develop male characteristics, the phenomenon of imposex. This was first observed in the female American mud snail (*Nassarius obsoletus*) (Smith, 1981) which upon exposure grew a penis and a vas deferens. Later, Bryan *et al.* (1986) found a similar situation with the dogwhelk (*Nucella lapillus*) in the waters of southwest England. These authors related the levels of TBT in the waters to the intensity of imposex. A "degree of imposex" was proposed to be the ratio of (female penis length)^3 /(male penis length)^{3}x 100. It should be pointed out that the biology and toxicology of imposex is not yet well developed.

18. The index was applied to Scottish sea lochs where the dogwhelks exposed to waters with significant small boat activity displayed imposex (Davies *et al.*, 1987a). Similarly, the same phenomenon developed in the whelks which lived near fish farms.

C. EXPOSURE LEVELS

19. The widespread concern about TBT in marinas and in fish farms has prompted a large number of surveys and monitoring programs. Based upon the toxic activity against snails, measurable levels (about 1 to 2 ng l^{-1}) are clearly unacceptable. Perhaps there are other organisms as sensitive to TBT. But enough relevant information is at hand to severely regulate TBT use.

20. Several recent monitoring programmes illustrate the problem of TBT contamination. Waldock *et al.* (1987) monitored 40 stations around the United Kingdom in 1986 with emphasis placed upon nine of them. Of the latter, one was an enclosed bay, one an open coastal site and seven were estuarine. The estuaries were involved with shell fisheries and six sites were involved with yachting activities. The samples were collected at slack or low tide and were usually taken monthly. The harbours and marinas had the highest concentrations which appeared to reflect the degree of flushing. Over half of the 250 samples had TBT concentrations exceeding 20 ng l^{-1}. In open estuarine areas, TBT again exceeded 20 ng l^{-1} in 67 per cent of the cases studied. Higher values of TBT were found at the Burnham-on-Crouch station during summer months, as compared

to winter months. In the spring, with the launching of yachts, the values rise. Finally, Waldock *et al.* (1987) confirmed the population decrease in the common dogwhelk in their stations, which they attribute to lowered reproductive capacity rather than to increased mortalities. Overall, their recorded concentrations ranged from under 1 ng l^{-1} in the winter to 1,500 ng l^{-1} in marinas sampled during the summer.

21. In the summer of 1986, Stallard *et al.* surveyed TBT at eight sites, primarily marinas of the California coast. TBT values in marina waters ranged from 20 to 600 ng l^{-1}, while lower values were usually found in harbours and on coasts. In marinas where the concentrations of TBT exceed 100 ng l^{-1}, there is usually a conspicuous absence of native organisms, especially molluscs. The north-coast marinas, which primarily berthed fishing vessels, also had occasionally high values of TBT in their waters, indicating that the fishing fleet is utilizing TBT-containing paints.

22. Water samples at 25 locations on the Canadian rivers Detroit and St. Clair had TBT in over 90 per cent of the water samples (Maguire *et al.*, 1985). The highest concentration was in a tributary of the Detroit River with a sub-surface value of 150 ng l^{-1} recorded. Similar values were found in the surface microlayer of the St. Clair River. Because the persistence of the microlayer varies from site to site, such measurements are at best indicative of the presence in sub-surface waters.

23. These measurements highlight the occurrence of TBT at levels at which they can do ecological damage in waters of the United Kingdom, Canada and the United States. Although the monitoring programmes, often taking place during a single season of the year and in restricted locations, are imperfect, they do emphasize the widespread occurrence of this biocide.

D. TBT IN PENNED SALMON

24. Fish-farming nets are often painted with TBT-containing anti-fouling paints so that the nets are not weighted down by fouling algae and other organisms and so that adjacent waters can have free access to the pens. Two recent investigations indicate that the farmed salmon accumulate TBT in their muscle tissues (Short and Thrower, 1986; Davies and McKie, 1987b). Both commercially raised and laboratory fish achieved levels in the parts per million range on a weight basis. Davies and McKie, working in Scotland experimentally, exposed their fish to waters containing 100 to 1,000 ng l^{-1} of TBT with levels in the fish reaching up to 1.5 ppm for grilse-sized fish. Similar values were found by Short and Thrower (1986). Cooking the fish proved to be ineffective in destroying TBT (Short and Thrower, 1986).

E. TOXICOLOGY

25. There have been extensive laboratory studies on the acute toxicity of TBT to marine organisms. Most of the studies have underestimated the effects of this biocide because TBT was not measured continuously in the system under study. Instead, the amounts introduced at the start were assumed to persist throughout without any microbial or photochemical decomposition or any absorption of the TBT upon the solid components of the system. These processes are well known to occur with TBT and have the effect of removing the compound from solution.

26. Acute exposures have been usually designed to study effects that occur within four days. But TBT is known to be slow-acting, and short-term laboratory experiments may lead to an underestimation of its biological activity.

27. Waldock *et al.* (1987) have summarized literature data on TBT toxicity. Fish, tunicata, echinodermata, bryozoa, crustaceans, molluscs, annelids, coelenterates, rotifers and algae provide examples of morbidities and mortalities at TBT levels that may be found in marine or estuarine environments. Representatives of these groups of organisms suffer sub-lethal effects at concentrations under 200 ng l^{-1}. On the basis of chronic exposures, molluscs and algae are more sensitive than other organisms. This was evident to Stallard *et al.* (1987) in their survey of California marinas, where the absence of macrophytes and mussels indicated TBT levels of about 100 or more ng l^{-1}. Perhaps the most sensitive organism so far identified is the common dogwhelk, where exposures as low as 2.5 ng l^{-1} produced imposex.

28. According to the compilation of Waldock *et al.* (1987) oysters are especially susceptible to the development of gross abnormalities, including shell thickening and chambering as a result of exposure. Fifty ng l^{-1} inhibited larval growth in *Crassostrea gigas*. Growth of newly metamorphosed European flat oysters (*Ostrea edulis*) was affected severely at 60 ng l^{-1}, and marginally so at 20 ng l^{-1}.

F. ENVIRONMENTAL PERSISTENCE

29. The degradation of TBT in natural waters takes the path of dibutyltin to monobutyltin to inorganic tin. The resultant products are far less toxic than the parent. Seligman *et al.* (1986) have found the environmental half-lives in sea waters to be in the range of seven to 15 days. Removal of TBT from the water column can take place through photolysis, microbial degradation and sorption on settling particulate phases. Biological degradation appears to be the most important factor. Photochemical decomposition appears to be slow with a half-life greater than 89 days (Maguire *et al.*, 1983).

30. Seligman *et al.* (1986) found that the TBT half-lives observed in waters exposed to light are slightly shorter than for those not exposed. They suggest that photosynthetic bacteria or algae may hasten the degradation process.

II. CHLORINATED HYDROCARBON PESTICIDES AND POLYCHLORINATED BIPHENYLS

31. Two groups of chlorinated organic compounds, the pesticides and the industrially used polychlorinated biphenyls (PCBs), have attracted the attention of marine scientists concerned with environmental quality. First of all, these compounds have been implicated in the mortality and morbidity of marine organisms, primarily at high trophic levels. Secondly, recent analytical advances, especially in gas chromatography and mass spectroscopy, have allowed both definitive and accurate analyses in organisms, waters, and sediments.

32. To understand the amounts and distributions of these substances in marine systems, detailed source-term data are crucial. Although figures on the production and use of chlorinated hydrocarbon pesticides and PCBs are lacking from most countries of the world, there is a variety of evidence that there are extensive leakages of these substances to the environment, especially in the developing countries of the tropics. On a world-wide basis, the value of pesticide imports has increased by 159 per cent between 1972 and 1984 (Postel, 1987, p.11). Although the use of DDT and BHC (benzene hexachloride) has been banned in most northern hemispheric countries, these compounds account for 75 per cent of the total pesticide utilization in India (Postel, 1987, p.16), while many of the countries of the tropics and southern hemisphere are increasing their imports as well as their capacities to produce biocides. Where several hundred thousand tons of DDT were produced annually in the 1960s throughout the world, present production of hard chlorinated pesticides is probably around several million tons. Can potent biocides or their degradation products affect marine ecosystems through decimation of non-target organisms?

33. The above statistics are complemented by unusually high levels of biocides in the atmosphere and in the oceans. C. B. Giam (personal communication) has indicated that hexachlorobenzene and hexachlorocyclohexane are usually the dominant hydrocarbon pesticides in the air samples that he has examined in the early 1980s. The concentrations in the islands in the northern hemisphere (Enewetak) and in the southern hemisphere (Samoa) are of the same order of magnitude, although somewhat less in Samoa. The DDTs and HCHs, as well as the PCBs, have been found in air, water, ice and snow in Antarctica (Tanabe *et al.*, 1983a). Although the concentrations decrease during their transport for lower latitudes, the isomers and degradation products of these three groups of substances did not change much during their movements.

34. The source-term information for the industrially used polychlorinated biphenyls has not been systematically obtained. They are of both economic and environmental interest to many sovereign nations. Although the United States and Japan essentially discontinued production in the early 1970s (Goldberg, 1976), production in France, Italy and Spain continued through the'

1970s (Fowler, 1986).

35. In addition, there is a substantial build-up of PCBs in the waters and air of the tropics and southern hemisphere (Tanabe, 1988b). Although production of these chemicals has markedly decreased in the industrialized nations over the last decade or so, materials in old products are still used. Much of this may eventually enter the environment, and levelling off of environmental levels may take a decade.

36. Recent leakages and spills on the African continent, coupled with their potential movements to the coastal ocean, justify the concerns of environmental scientists.. These industrial chemicals have been linked to the declining marine mammal populations in the Baltic and North Seas and in California coastal waters. Are the concentrations of these toxic chemicals in the ocean system attaining values that might jeopardize marine organisms and public health?

A. CHLORINATED HYDROCARBON BIOCIDES

37. Tanabe and his associates have carried out the most extensive recent surveys of chlorinated biocides in marine air and water. Tanabe and Tatsukawa (1981) indicated that there are most probably high uses of DDT and HCH (1, 2, 3, 4, 5, 6 hexachlorocyclohexane), including the α, β, δ and ε isomers, in developing countries, especially those in tropical zones. Their distribution in the mid-latitude atmosphere of the Pacific (Figures 1 and 2) cannot be explained by Asia being the only source. For example, Japan banned the use of both pesticides several years before the measurements were made and the concentrations in the coastal marine environment reflected this action through lower values in the associated waters. On the other hand, the open ocean waters surrounding Japan showed uniformly high values after the ban. These authors suggest that atmospheric transport is moving these pesticides from their sources about the tropical and mid-latitudes of the northern hemisphere.

38. ΣDDT and HCH isomers as well as the polychlorinated biphenyls were measured in air and water in the Antarctic, western Pacific and eastern Indian oceans between November 1980 and March 1981 and were found at all locations (Tanabe *et al.*, 1982 a and b) (Figures 1 and 2). Hexachlorocyclohexane (sum of α, β, and γ isomers) were higher in the northern than in the southern hemisphere in both air and water (Figure 1) whereas ΣDDT had about the same concentration in both hemispheres and was higher in the tropics (Figure 2). The drift of DDT use from the northern to the southern hemisphere is clear from these results.

B. POLYCHLORINATED BIPHENYLS

39. The industrially used PCBs were first found in the environment in 1966 (Jensen, 1966) through analyses of fish and wildlife. Transport to the marine environment to a large extent takes place through the atmosphere, a phenomenon which has lead to a world-wide dispersion, similar to the situation for some of the chlorinated pesticides. For the open ocean waters both the PCBs and the chlorinated hydrocarbon biocides are more concentrated in the Northern than in the Southern Hemisphere (Figure 3; Tanabe *et al.*, 1982 a and b).

40. Two groups of compounds, the polychlorinated dibenzo-p-dioxins and the polychlorinated benzofuranes, are found as by-product impurities during PCB production. They are highly poisonous and perhaps much of the toxic activities attributed to the PCBs are really due to these two sets of substances (Tanabe, 1988a). Much of the following is taken from a recent review by Tanabe and Tatsukawa (1986).

41. PCB concentrations in air and water (Figure 3) indicate higher concentrations in the coastal regions of the tropical and sub-tropical areas. The concentration differences between hemispheres were less than for ΣDDT. These levels in the developing world are attributed to high uncontrolled usage (Tanabe *et al.*, 1982 a and b).

42. The atmospheric concentrations over the ocean are usually below 1 ng m^{-3}. In general, values in the mid-latitudes of the northern hemisphere are higher than those in other parts of the atmosphere, reflecting the greater use of these chemicals on adjacent lands (Table 1). Tanabe *et al.* (1983a) observed an increase going from Antarctica (69°S; 40°E) to Mauritius in the western Indian Ocean (21°S; 58°E), suggesting that the low-latitude countries may act as a source of PCBs.

43. The Mediterranean waters in general have higher values near estuaries than in the open ocean, emphasizing the effects of river-borne loads. A maximum value of 38 ng l^{-1} was measured 0.5 km from the mouth of the Rhone (Fowler, 1986). River effects in general are dominant over the more ubiquitous atmospheric entries.

44. The open-ocean surface waters contain PCBs in concentrations of the orders of a few ng l^{-1} or less (Table 2). A few values higher by one or two order of magnitude were reported for 1971 and 1972; the reliability of these values is questionable. The mid-latitudinal waters of the North Atlantic ocean have the highest values, reflecting to a large extent atmospheric inputs of the industrialized world.

45. PCBs have been found at oceanic depths of many kilometres. The vertical transport is established by measurable amounts in deep living fish and sediments, as well as in the water column. The less soluble PCBs are effectively transported through sorbtion on sinking particles, especially biogenic solids (Elder and Fowler, 1977).

46. In the marine biosphere, highest levels are found in the higher trophic levels. Where open ocean plankton have PCB concentrations of the order of a few ppb, fish have concentrations one order of magnitude and marine mammals are an additional one to three order of magnitude higher. Mussel Watch data from throughout the world suggest that there are relatively uniform concentrations of PCBs in coastal waters with occasional hotspots (Fowler, 1986). Higher levels in fish are found near river mouths and near centres of industrial activity.

47. There is evidence for the metabolism of PCBs in organisms. For example, where sea water contains primarily the less chlorinated biphenyls (two or three chlorine atoms per molecule), more highly chlorinated substances were found in plankton and fish samples. For example, Dall's porpoise primarily contains biphenyls having four or more chlorine atoms per molecule.

48. The loading of various compartments of the marine environment with PCBs has been estimated by Tanabe and Tatsukawa (1986) (Table 3). The total amount of these compounds is about 230,000 tons with the major fraction in the water itself. Not unexpectedly, the North Atlantic provides the largest contribution. These investigators indicate that perhaps the integrated world production achieved a value of 1.2 million tons. If so, about 20 per cent of the cumulative world production is now in the open ocean environment.

49. There is mounting evidence of the effects of PCBs and DDT (and its metabolites) upon Baltic marine fauna (Olsson, 1986). It is at present difficult to separate the effects of these two groups of substances; perhaps they are additive. Olsson summarizes conventional wisdom and indicates that PCBs are the responsible substances, especially in regard to low reproduction rates of seals and sea lions.

50. Marine mammals have PCB, DDT and HCH concentrations that correspond with the levels in open-ocean surface water (Figure 4; Tanabe *et al.*, 1983). The organisms were taken from the Bering Sea, the western Pacific and the Southern Ocean. This relationship suggests that the blubber of marine mammals may provide an adequate integration of regional pollution by these halogenated hydrocarbons.

C. THE INTERNATIONAL MUSSEL WATCH

51. A worldwide monitoring of chlorinated hydrocarbon biocides and PCBs in mussels and oysters has been proposed to ascertain whether or not there may be unacceptable levels in these organisms, especially those from the southern hemisphere and tropics, (Goldberg, 1983). A comparison will made with the levels of the 1960s and 1970s in bivalves from the northern hemisphere where impacts upon higher trophic level organisms were evident. Although such a comparison may well be semi-quantitative at best, it will provide an indication of potential environmental degradation.

TABLE 1. PCB CONCENTRATIONS IN THE OPEN OCEAN ATMOSPHERE
(Tanabe and Tasukawa, 1986)

location	Year	PCB conc. (ng m^{-3})		
		N	Range	Mean
North Atlantic				
Bermuda	1973	4	0.15 - 0.50	0.30
Bermuda	1973	8	0.21 - 0.65	0.51
Bermuda, U.S.	1973	4	0.72 - 1.6	0.99
Grand Banks (45°N, 52°W)	1973	5	0.05 - 0.16	0.086
Newfoundland	1977	6	0.042 - 0.15	0.12
Gulf of Mexico	1977	10	0.17 - 0.79	0.35
Barbados	1977-1978	17	<0.005 - 0.37	0.057
North Pacific				
Enewetak Atoll (12°N, 162°E)	1979	14	0.35 - 1.0	0.54
Western Pacific (3-35°N, 105-151°E)	1980-1981	7	0.089 - 0.74	0.25
Western Pacific (43-53°N, 154-172°E)	1981	2	0.041 - 0.061	0.051
Western Pacific (41-46°N, 144-174°E)	1982	5	0.022 - 0.095	0.043
Bering Sea	1981	3	0.026 - 0.059	0.041
South Pacific				
Western Pacific (1-46°S, 151-157°E)	1981	5	0.083 - 0.50*	0.27
Indian				
Eastern Indian (1-44°S, 104-125°E)	1980	5	0.066 - 0.33*	0.15
Western Indian (20-54°S, 48-57°E)	1982	4	0.060 - 0.24	0.16
Antarctic				
53-65°S, 125-161°E	1980-1981	5	0.056 - 0.18	0.091
54-68°S, 38-58°E	1982	4	0.076 - 0.11	0.091
Syowa Station (69°00'S, 39°35'E)	1981-1982	11	0.017 - 0.17	0.061

* Excluding the coastal regions.

TABLE 2. PCB CONCENTRATIONS IN OPEN OCEAN SURFACE WATERS
(Tanabe and Tasukawa, 1986)

Location	Year	PCB conc. (ng l^{-1})		
		N	Range	Mean
North Atlantic				
Sargasso Sea	1973	8	< 0.9 - 3.6	1.0
Sargasso Sea, New York	1973	9		0.8
9-55°N, 9-73°W	1973-1975	39	0.4 - 8.0	2.9
North Sea and Scottish coast	1974	5	< 0.15 - 0.52	0.23
South Atlantic				
11-36°S, 2-33°W	1975	8	0.3 - 3.7	1.0
North Pacific				
Western Pacific (22-35°N, 141-154°E)	1975	13	0.25 - 0.56	0.41
Western Pacific (32-42°N, 133-143°E)	1976	8	0.29 - 1.1	0.54
Western Pacific (12-33°N, 129-138°E)	1978	6	0.23 - 0.59	0.35
Western Pacific (29-34°N, 137-146°E)	1979	5	0.27 - 0.38	0.33
Western Pacific (5-31°N, 107-152°E)	1980-1981	9	0.039 - 0.15	0.089
Bering Sea	1981	3	0.073 - 0.13	0.10
South Pacific				
Western Pacific (2-41°S, 152-156°E)	1981	5	0.081 - 0.21	0.12
Indian				
Eastern Indian (4-45°S, 104-123°E)	1980	6	0.057-0.25	0.14
Antarctic				
48-65°S, 124-163°E	1980-1981	7	0.042 - 0.072	0.058
Syowa Station (69°00'S, 39°35'E)	1981-1982	3	0.035 - 0.069	0.053

TABLE 3. ESTIMATED CONCENTRATION AND LOAD OF PCBs IN THE OPEN OCEAN ENVIRONMENT

Compartment mass PCB conc., load	North Pacific	South Pacific	North Atlantic	South Atlantic	Indian	Antarctic	Total load
Compartment mass							
Air (10^{16} m^3)	70	76	38	35	58	46	
Water (10^{19} l)	30	33	15	14	23	19	
Sediment (dry, 10^{15} g)	70	76	38	35	58	46	
Plankton (wet, 10^{15} g)			48 (whole ocean)				
Fish (fresh, 10^{14} g)			26 (whole ocean)				
Mammals (fresh, 10^{12}g)			85 (whole ocean)				
PCB conc.							
Air (ng m^{-3})	0.3	0.2	0.5	0.2	0.2	0.1	
Water (ng l^{-1})	0.2	0.1	0.6	0.1	0.1	0.05	
Sediment (dry, ng g^{-1})	0.4	0.2	1.0	0.2	0.2	0.1	
Plankton (wet, ng g^{-1})	2.0	1.0	5.0	1.0	1.0	0.5	
Fish (fresh whole, ng g^{-1})	10	2.0	30	2.0	2.0	0.2	
Mammals (fresh whole, μg g^{-1})	2.0	0.5	5.0	0.5	0.5	0.05	
PCB load							
Air	210	150	190	70	120	50	790
Water	60,000	33,000	90,000	14,000	23,000	10,000	230,000
Sediment	28	15	38	7	12	5	105
Plankton			20-240 (whole ocean)				130
Fish			1-80 (whole ocean)				40
Mammals			2-200 (whole ocean)				100
Total							231,165

(Tanabe and Tasukawe, 1986)

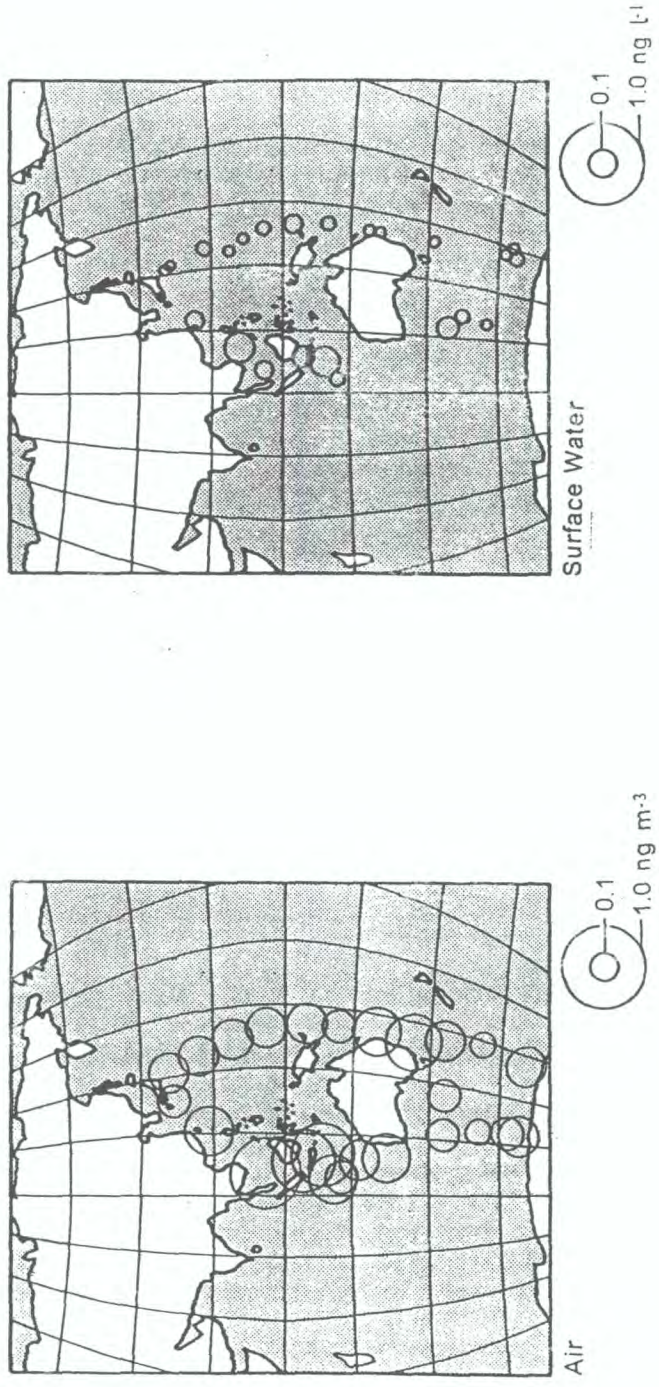

FIGURE 1. TOTAL DDT DISTRIBUTIONS IN OPEN OCEAN ATMOSPHERE AND WATERS
(Redrawn from Tanabe *et al.*, 1982a)

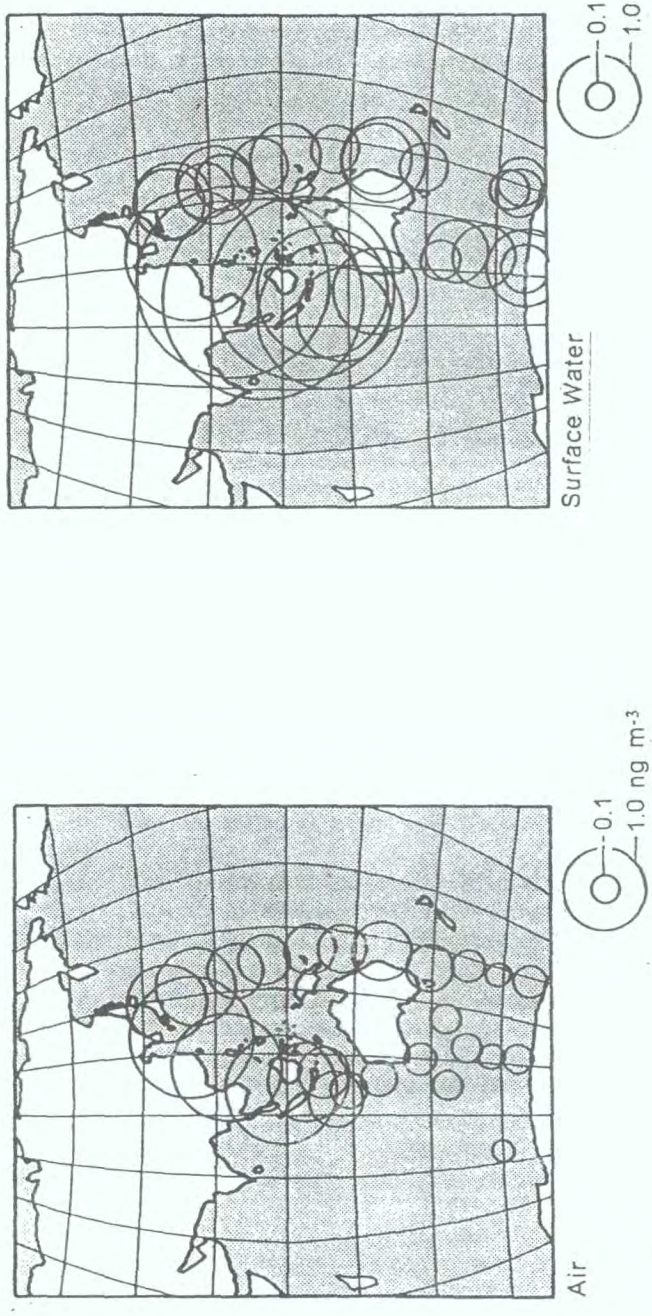

FIGURE 2. TOTAL HCH DISTRIBUTIONS IN OPEN OCEAN ATMOSPHERE AND WATERS
(Redrawn from Tanabe *et al.*, 1982a)

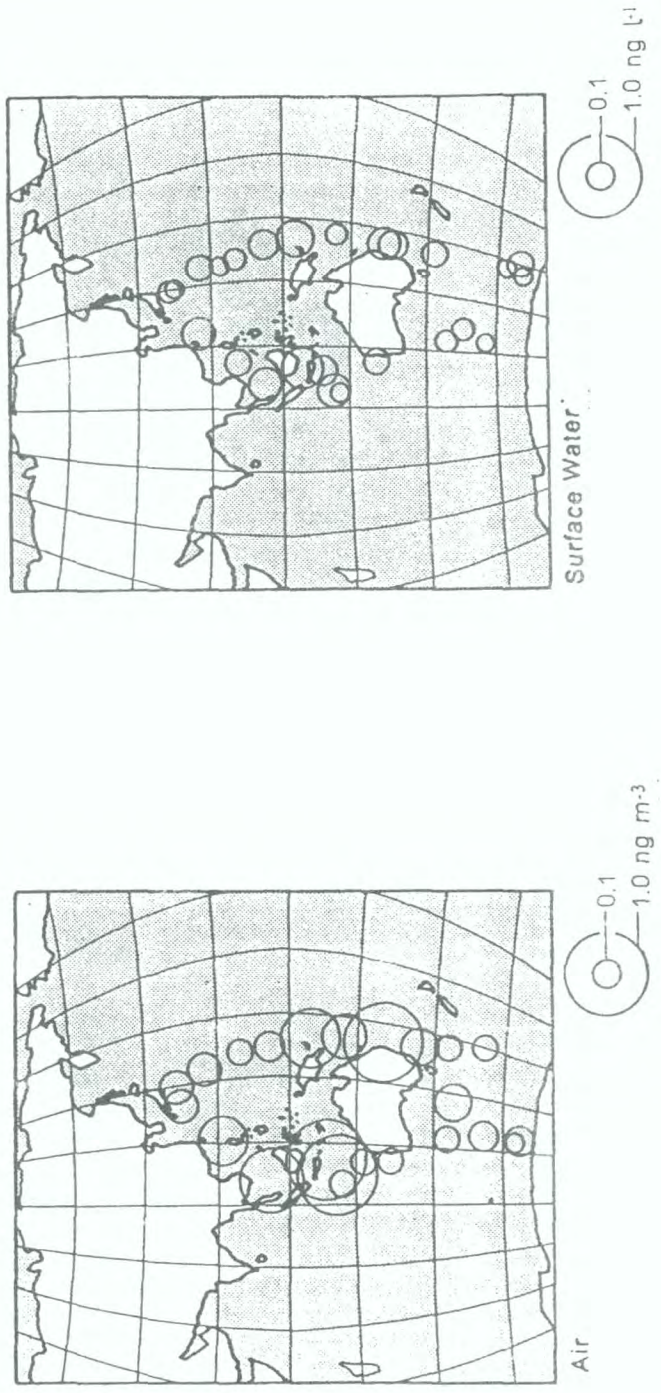

FIGURE 3. PCB DISTRIBUTIONS IN OPEN OCEAN ATMOSPHERE AND WATERS
(Redrawn from Tanabe *et al.*, 1982a)

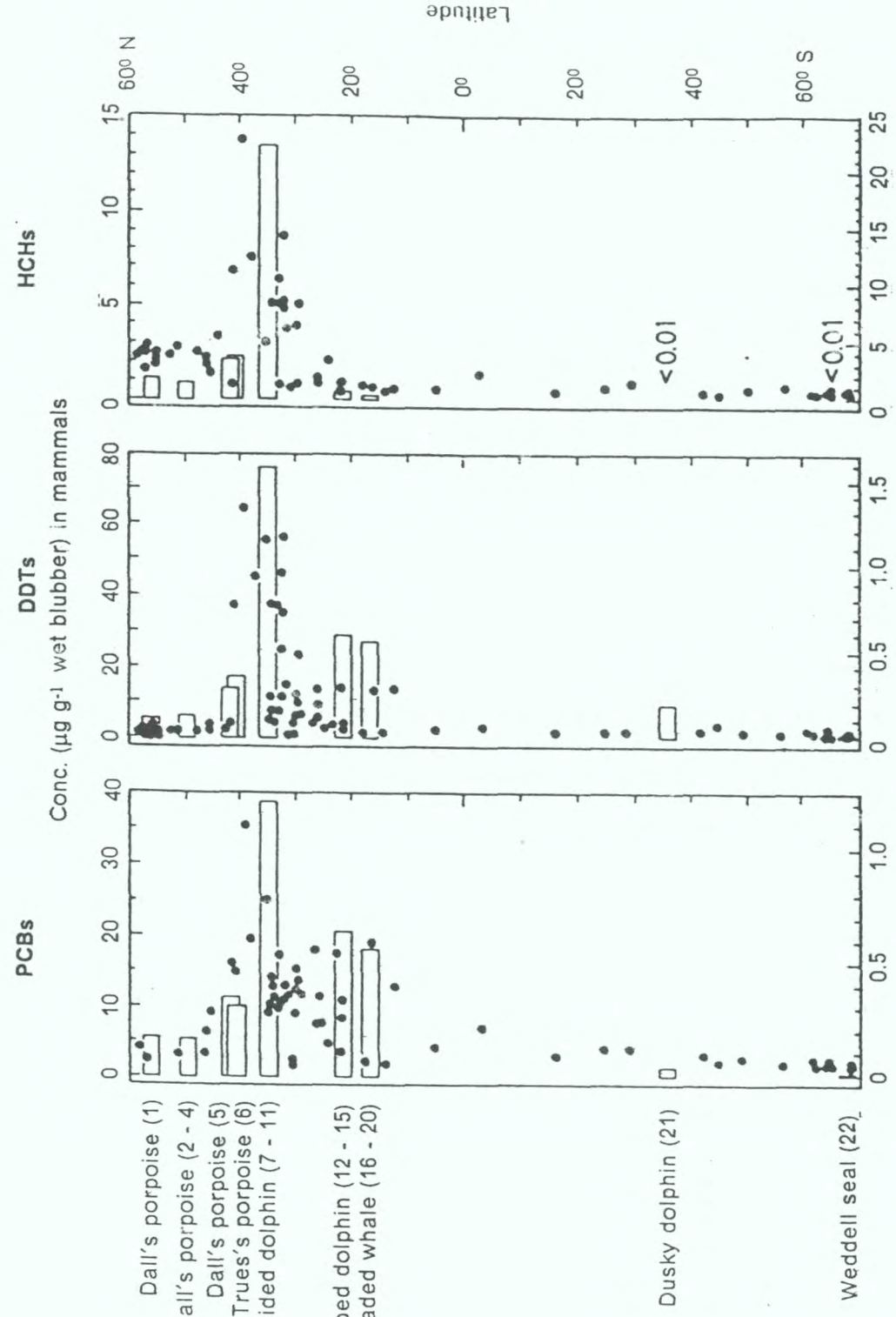

FIGURE 4. LATITUDINAL VARIATIONS IN PCB, DDT AND HCH CONCENTRATIONS IN MARINE MAMMALS.

Closed circles indicate concentrations in open ocean surface waters.
Bars represent the mean or single values of the compounds in marine mammals.
The parenthesized number correspond to the number of samples.
(Redrawn from Tanabe *et al.*, 1983b)

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ANNEX VI

MANIPULATIONS OF HYDROLOGICAL CYCLES

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INTRODUCTION

1. Although man-induced alterations of river flow date back to ancient civilizations, the era of major dam-building activity did not begin until the early twentieth century. With the development of engineering technology, particularly concrete technology, the building of great multipurpose dams became accelerated since the 1950s (Figure 1, Tables 1-3). This worldwide trend was motivated by the pressing need for more water resources for industry and urban developments, and by the energy crisis and the resulting need for alternative energy sources, particularly hydroelectric power. Successive years of severe drought in some developing countries, catastrophic floods in others, in addition to the vital need for promoting agriculture and land reclamation, induced these countries to dam their rivers for water storage and regulation, irrespective of downstream impacts. Almost all the main rivers of Western and Central Africa have been dammed (Table 4). At least 20 dams built on the Congo-Zaire and its tributaries are used as reservoirs and for power generation. The Nile waters are almost entirely withdrawn for various inland purposes. The Indus river is on its way to becoming entirely diverted for farm-land irrigation. Rivers in developed countries are even more thoroughly harnessed. The Danube, the Rhine and the Colorado are typical examples.

2. As a result of the increasing trend in river impoundment and artificial flood-water storage, the proportion of stable run-off has been augmented on every continent. Petts (1984) estimates that in Africa and North America about 20 per cent of the stable run-off is contributed by impoundments. In Europe and Asia the figures are 15 per cent and 14 per cent, respectively, while rivers of South America (4.1 per cent) and Australasia (6.1 per cent) are less affected. Croom *et al.* (in Petts, 1984) suggest that by the year 2000 about 66 per cent of the world's total stream flow will be controlled by dams.

3. Deforestation and various land uses have accelerated soil erosion. The relation on a global scale between geological erosion and man-induced soil erosion is not well known. According to some estimates (UNESCO), the present rate of erosion would be two and a half times the rate before man started to affect the landscape on a large scale. Although this figure is obviously hypothetical, the increased rate of land erosion increases the sediment load of natural rivers and its subsequent deposition in river beds and deltas. The result is a reduced river gradient, increasing the danger of bank breaches and flooding, as happens with the rivers of China.

4. In the first part of this chapter, the river inputs to the ocean are reviewed. This is followed by an overview of the upstream and downstream impacts of the man-made alterations of the natural hydrological cycles, both direct (e.g. impoundments) and indirect (changes in the vegetation cover). Five case studies from different continents are then examined.

I. RIVER INPUTS TO OCEAN SYSTEMS

5. River inputs to Ocean Systems (RIOS) were reviewed at a SCOR workshop held in Rome in March 1979 (UNEP/UNESCO, 1981).

A. DISSOLVED INPUTS

6. The mean discharges of the major rivers of the world, which drain about 40 per cent of the global land surface, are listed in table 5. In terms of their water discharge, the 10 biggest rivers account for 73 per cent of the total discharge.

7. The dissolved contents of rivers are highly variable. The major dissolved elements in surface waters are controlled by numerous parameters: chemical (solubility products of minerals), or geological (occurrence of mineral species, rock porosity, history of surface rocks exposed to weathering), climatic (temperature, rainfall, evaporation), geomorphologic (relief intensity) and biological (vegetation cover, biological uptake or release of elements) (Meybeck, 1981).

8. The chemical types of the world surface waters (Table 6) are obtained from a set of rivers. The inputs of dissolved silica and total ionic content to each ocean (Table 7) have been computed on the basis of 64 per cent of the river water drained to the oceans and of the typology of dissolved transport for the remaining 36 per cent. Owing to its smaller size and despite its poorly weathered watershed, the Glacial Arctic Ocean has high rates of dissolved inputs, whereas the Pacific Ocean has low ones (Table 7). The dissolved silica content of rivers is found to be directly dependent on the average temperature of the river basin (Meybeck, 1981). 73 per cent of the dissolved silica input to the oceans originates from the tropical zone. Highlands in temperate and tropical humid zones (12.5 per cent of the exoric part of the continents) produce 45 per cent of the dissolved silica and 41 per cent of the ionic inputs (Meybeck, 1981).

9. The transports of the major elements in dissolved and particulate phases show a positive correlation which reflects the general correlation between the total dissolved transport T_d and the particulate transport T_s. However, important geographic variations in the ratio of dissolved to total transport for a given element, the Dissolved Transport Index (DTI) are caused by the variability of T_d / T_s and by lithic factors, the DTI increasing greatly in sedimentary environments (Meybeck, 1981). The dissolved elemental transports T_d in major climatic zones, as defined by surface run-off q and average air temperature, are represented in Figure 2. (Meybeck 1979 in Meybeck, 1981).

B. RIVER SEDIMENT INPUT TO THE OCEANS

10. Several estimates of the amount of river sediment input to the oceans have been proposed. The estimates cover a rather wide range. Two approaches have been followed: one estimates the mass being carried oceanward by rivers, while the other estimates denudation of the continents. Sediment loads based on the latter method are considerably higher than those based on the former. Lopatin (1950) and Holeman (1968) estimate an input of 12.7 and 18.3 Mt y^{-1}, respectively, while Fournier (1960) reached an estimate of 64 Mt y^{-1}. The global budget of sediment input to the oceans has recently been updated and critically reviewed by Milliman and Meade (1983), who give an estimate of 13.5 Mt y^{-1}.

11. Such global estimates, however, are faced with at least two major problems: the questionable validity of a large part of the data and the near absence of documentation on the bed-load of rivers. On the other hand, the sediment transport of rivers and their discharge to the oceans are not in a steady state (Meade, 1980). The case of the Yellow River in China is a dramatic example (Figure 3). Only 24 per cent of the sediment that flows into the lower Yellow River actually reaches the ocean. The remaining 76 per cent are deposited on an alluvial plain and in the river delta.

12. Rivers draining humid mountains, such as the Ganges-Brahmaputra, and desert areas, such as the Colorado and the Orange, tend to have very high sediment concentrations. Concentrations are low in rivers draining low-lying areas, whatever the latitude. The lowest concentrations are in those rivers crossing natural or man-made lakes (Milliman, 1980). It is probable that the sediment load of smaller rivers is underestimated in worldwide budgets, since the smaller is the drainage basin the less able it is to store sediments. Milliman and Meade (1983) assume that for every order-of-magnitude increase in drainage-basin areas, the sediment yield decreases about 7-fold (Figure 4).

13. The following brief overview of river sediment delivery is summarized from Holeman (1968) and Milliman and Meade (1983) (Table 8; Figure 5).

14. The Yellow River in China carries the largest amount of sediment of any river in the world. The Ganges in India is the second largest sediment carrier. The major rivers between Korea and Pakistan contribute more than 6,400 Mt y^{-1}, nearly half of the total world input. The Chinese rivers and those draining the Himalayas and Southeastern Asia have an average yield of about 600 t km^{-2} y^{-1}. This area therefore contributes about 50 per cent of the world river sediment budget.

15. The large islands of the western Pacific Ocean are among the most important producers of river sediment. They include Japan, Taiwan, the Philippines, Indonesia, New Guinea and New Zealand. Because of their active tectonism, their heavy rainfall and steep slopes, these islands

contribute considerable quantities of river sediments to the Ocean. The heaviest contributor is Taiwan with an average sediment yield to the ocean of about 10,000 t km^{-2} y^{-1} and a calculated annual sediment load of about 300 Mt, only slightly smaller than that of the United States. A conservative sediment yield of 1,000 t km^{-2} y^{-1} from the 3 10^6 km^2 of the large western Pacific islands gives a total discharge of river sediment of 3,000 Mt y^{-1} from these islands.

16. The South American continent contains three of the world's largest rivers emptying into the Atlantic Ocean, the Amazon, Orinoco and Parana-Plata Rivers that are rated 1, 3 and 9 in terms of water discharge and 1, 18 and 5 in terms of drainage area. Very few accurate measurements of sediments, however, are available from the latter two rivers and the figure given for the Amazon is an approximate estimate. Though a small river, the Magdalena River, discharging to the Caribbean, appears to transport more sediments than either the Orinoco or the Parana-Plata Rivers.

17. Rivers draining the eastern part of North America have small sediment loads. Though the St Lawrence drains a large basin, most of its sediment load is trapped in the Great Lakes. The Mississipi is the largest river discharging to the Gulf of Mexico but, because of reservoir construction, bank stabilization and improved soil conservation, its sediment load has considerably decreased.

18. Sediment loads in rivers of western North America have also been decreased by damming. The sediment load of the Colorado river has dropped from 135 Mt y^{-1} to less than 0.1 Mt into the Gulf of California. The discharge of the Columbia river has also been reduced. A large annual load is carried by the Fraser River which drains British Columbia, 20 Mt y^{-1}, mainly fine sand. The MacKenzie River, draining into the Beaufort Sea, has the second largest draining area in North America. Its sediment load ranges from 57 to 199 Mt y^{-1}.

19. The rivers that drain Europe are small and carry little sediment. The Danube, by far the largest, has an estimated sediment discharge of 67 Mt y^{-1}. The Eurasian Arctic is drained by the Ob, Yenisei and Lena Rivers. Their total water discharge is of 385 to 560 km^3 y^{-1}, but their combined sediment discharge is small (63 Mt y^{-1}).

20. The Niger and the Zaire Rivers together drain about 5 10^6 km^2 of West Africa. The Zaire is the second largest river in the world in terms of both drainage basin and river flow. Sediment loads from both rivers are relatively small, however, because they drain low-lying land and, in the case of the Zaire, empty into lakes prior to reaching the ocean.

21. East Africa has several large rivers of which the largest is the Zambezi, followed by the Limpopo (Mozambique) and the Rufiji (Tanzania). The Zambezi sediment load appears to have dropped to less than half, as a result of damming.

22. The Nile sediment discharge to the Mediterranean has dropped from about 150 Mt y^{-1} to almost nil after damming in 1965.

II. IMPACTS OF HYDROLOGICAL ALTERATIONS

23. Among the most pervasive of man's effects on the hydrological regime and sediment load of rivers are the changes in vegetation cover and soil structure that have accompanied deforestation, intensive farming and overgrazing. No less drastic and complex, however, are the impacts of river control by dams and reservoirs.

A. ALTERATIONS RESULTING FROM CHANGES IN THE VEGETATION COVER

24. Compared to natural conditions, most agricultural practices imply a reduction of vegetation cover at least during part of the year, including loss of surface litter and of humus content. The humus content is related to temperature and humidity. Especially in tropical and subtropical regions, cultivation will tend to reduce the humus content and to increase erodibility.

25. Due to the protection by the leaf canopy and by the ground cover of litter and vegetation, forests and woodlands are normally characterized by low surface run-off, high infiltration rates and insignificant soil erosion. The forest soils often have a relatively porous structure which facilitates ground-water supply. when the forest cover is removed, the hydrological conditions change, soil erosion, nutrient levels, and run-off increase substantially.

26. Bormann *et al.* (1974) reported on the export of particulate matter, erodibility and the relative importance of dissolved substances for particulate matter in exported materials following deforestation at Hubbard Brook. The mature forested ecosystem is little affected by erosion. Deforestation, accompanied by repression of growth by herbicides, increased export 15 times. The increase in export was exponential, slow in the first two years after cutting, rising sharply in the 3rd year. The increases in particulate matter export are primarily due to increases in erodibility rather than in flow rates. The first response to deforestation is the mobilization of nutrients and leakage in stream water. While the particulate matter rises sharply as biotic control of erodibility weakens, dissolved substance export declines, probably because of diminution of readily available nutrients stored in the system. The average ratio of annual net export of dissolved substance to particulate matter shifts from 2.3 before, to more than 8.0 after deforestation during the first two years (Figure 6).

27. Losses of nitrogen due to the clearing of tropical forests are higher than those of

phosphorus and carbon (Van Bennekom and Salomons, 1981). The lumbering of temperate forests leads to very high (more than 4,000 μM) concentrations of inorganic nitrogen, nearly all in the form of nitrate, in stream water, corresponding to losses of 400 mmol N m^{-2} per year. This value, assumed to hold for two years after clearing, combined with annual deforestation of 0.5 to 1.6 per cent of the 48.5 10^6 km^2 forested areas (Woodwell *et al.*, 1978, in Van Benekom and Salomons, 1981), gives an annual contribution from this source of 0.4 Tmol of dissolved nitrogen.

28. Attempts have been made to estimate the percentage of the material eroded upland that is transported more or less directly to the outlet. Significant relationships were found between this "sediment delivery ratio" and some of the drainage basin characteristics, in particular, the drainage area (Figure 4) and the relief-length ratio (R/L, Figure 7). The sediment yield from a drainage basin, however, must be seen as reflecting the magnitude and relative importance of the various sources and of the processes, pathways and sinks involved in the movement of eroded material from the point of detachment to the outlet. The sediment budget approach to investigating sediment yields provides a means of integrating these processes (Hadley *et al.*, 1985; Figure 8).

B. ALTERATIONS RESULTING FROM RIVER IMPOUNDMENT

29. The river environment, and to some extent the delta, the estuarine and the coastal zones are affected by processes operating in the drainage basins. Dams interrupt the pattern of downstream transfers. The discharges, the sediment and organic loads and the water quality become then governed by the releases from the reservoirs.

1. Alterations of flow regime

30. Three types of flood regulation reservoirs are common (Petts, 1984):

- retarding reservoirs (A, Figure 9) which are simple flood-storage basins which release water through uncontrolled outlets. The flood peak will be reduced and displaced in time;

- detention reservoirs which have a storage volume larger than that of the maximum flood. The flood peak is completely absorbed and the outflow is fully controlled (B, Figure 9);

- small-capacity detention basins. An effective flow regulation is achieved by the prior evacuation of stored water. The flood peak is absorbed, the flow rate being slightly raised and spread over a longer duration (C, Figure 9);

31. Flood control and reduced flow rate downstream from reservoirs may combine with the release of oxygen-depleted water to create or aggravate severe pollution problems. The assim-

ilation capacity and the flushing capacity for domestic and industrial effluents are greatly reduced.

32. Reduced flow rates donwstream from reservoirs have direct effects on biological cycles. It is assumed (Rozengurt *et al.*, 1985) that natural fluctuations of mean freshwater supply to estuaries, within a given cycle of wetness, vary within 25 per cent of normal 50 - 60 years averages. The most useful measure of hydrological variability is the coefficient of variation, C_V (standard deviation divided by the mean). The median value of C_V for 126 rivers investigated by UNESCO (McMahon, 1982) was found to be 0.25, but tropical and cold-climate rivers are 25 per cent less variable than those in semi-arid and temperate regions (Figure 10). If diversions within a cycle do not exceed the natural deviations from the average flow, the estuarine ecosystem, with its natural resilience, would survive the regulated water supply fluctuations, since they are within the range of natural conditions. In many parts of the world, however, massive water diversions from estuaries have greatly reduced or eliminated major fisheries. The periodicity and timing of the flow and the water quality are also major factors beside the flow volume.

2. Quality alterations of reservoir releases

33. The storage of water in open reservoirs induces physical, chemical and biological changes within the stored water. In consequence, the water discharged from impoundments can be of different composition and can show a different seasonal pattern from that of the natural river, except for reservoirs of short retention time.

34. Of the many factors that influence the quality of reservoir discharges (Figure 11), stratification and the level, or elevation, of the release, are particularly important. Reservoirs act as thermal regulators and nutrient sinks.

35. In natural rivers, the relatively small volume of water, together with turbulent mixing and the large surface area in contact with the atmosphere, allow a rapid response of stream water temperature to the prevailing meteorological conditions. In reservoirs, however, a characteristic seasonal pattern of thermal behaviour develops due to temperature-density differences in the water. The density gradient is poorly developed in spring but becomes increasingly well-defined as summer progresses.

36. The elevation of the outlet will determine the quality of releases from a stratified reservoir. Once a steady rate of outflow has been achieved, a relatively narrow layer of approximately constant density will be withdrawn, so that the water quality of the outflow will vary considerably if releases are abstracted from the epilimnion or the hypolimnion.

37. During summer, surface release from a stratified reservoir will discharge well-oxygenated, warm and nutrient-depleted water, whilst low-level outlets will produce relatively cold, oxygen-depleted and nutrient-rich releases which may contain high concentrations of iron, manganese and, hydrogen sulphide. The release of iron and manganese may occur in sufficient concentrations to

produce a precipitate on the stream-bed.

38. The formation of organic deposits rich in iron and manganese on the bed of regulated rivers appears to be common below soft-water reservoirs. A black friable deposit was observed within the bed materials of the lower river Elan (Wales, U.K.) below the Craig Goch reservoirs (Truesdale and Taylor, 1978, in Petts, 1984).

39. Seasonal pulses are climatically induced, and significant water-quality changes below reservoirs can occur annually with the autumnal overturn. Sudden and extreme nutrient peaks have been reported from below the Kariba Dam on the Zambezi river (Hall *et al.*, 1977, in Petts, 1984) and the Ucha Reservoir, U.S.S.R. (Petts, 1984).

3. Alterations of sediment transport

40. Reservoirs act as major sediment traps within the drainage basin of impounded rivers, although their trapping efficiency is variable.

41. Early attempts to forecast a reservoir's trap efficiency (TE), i.e. the percentage of incoming sediment deposited within the reservoir, were based upon ratios between the reservoir storage capacity and the drainage area. The relationship (Brown, 1944) is represented as:

$$TE = 100\,[1 - 1/(1 + 0.1\,C/DA)]$$

Where C = reservoir capacity, DA = drainage area. The use of a capacity/inflow ratio (C/I) has been recognized as a more accurate index than the C/DA (Brune, 1953). The reservoir retention-time has been identified as an important parameter in determining the trap efficiency. It has been estimated that short retention times (less than one day) commonly show trap efficiencies of between 40 per cent and 90 per cent. For longer retention times, the trap efficiency usually exceeds 90 per cent (Table 9).

42. Because of the differential rates of settling within impoundments, the sediment loads released from dams will have a different particle-size composition. All of the coarser material will be retained, but the lighter particles, especially clays, may remain in suspension. Some clays, such as the montmorillonite group, may react with dissolved salts, producing early flocculation. The kaolin clays may remain in suspension for longer periods and their proportion be greater in the released sediment load (Petts, 1983).

43. Beside any sediment release from dams, the suspended sediment load for the river below a dam will also be derived from unimpounded tributaries, effluent outfalls and the erosion of fine material from the river banks.

44. Erosion of the river banks can form a major source of suspended sediments for down-

stream reaches, particularly where the river meanders across a broad flood-plain. The imposition of unnatural flow regimes, characterized by highly variable discharges, can increase the rate of bank erosion. The channel degradation and scour below dams also provide an important sediment supply for the river downstream. This process, however, will often lead to redistribution of the sediment, the material moved from narrow sections becoming deposited in wider sections of the channel, where the flow is quieter.

45. However, the more important sediment source for channel sediments below dams is the unregulated tributary. The regulation of floods will lower the effective base level for tributaries and, as a result, rejuvenation of the tributaries will be induced. Consequently, the sediment yield from a tributary may be increased for several years after closure of the dam, until the side-stream has adjusted to the lowered base level.

III. CASE STUDIES

A. THE YELLOW RIVER AND THE YANGTZE IN CHINA

46. River control has long been a major issue in Chinese history and will remain a long-term major issue in the future. As a result of the uneven distribution of rainfall and of the disproportion between irrigation needs and water resources in China, there is a trend towards the excessive use of land and water resources at the expense of forests, pastures, rivers and lakes. In return, flood and drought problems are aggravated, thus forming a vicious circle (Zhengying, 1983).

47. A tremendous amount of river engineering work has been carried out on numerous rivers, particularly, on the seven major rivers (Table 10). Numerous storage projects of various sizes were constructed: 86,800 reservoirs, 6.4 million storage ponds of a total storage capacity of 410 km^3. Dikes along 160,000 km of rivers have been built or renovated (Zhengying, 1983) (Table 11).

1. The Yellow river (The Huang Ho), a case of natural river displacement

48. In the past 4,000 years, several radical changes have occurred in the Yellow River's course. At different times, the river has entered the Yellow Sea at points varying by as much as 500 miles (Figure 12). From 2278 BC to 602 BC, it occupied its northernmost course, entering the Gulf of Bo Hai. From 602 BC to AD 70, both the river and its mouth shifted to the south of the Shantung Peninsula. From AD 70 to 1048, the Yellow River again shifted north, much along its present bed. In 1194, the river occupied its southernmost course. After two more shifts, it remained stable for more than 500 years until 1855, when it moved once more north of the Shantung Peninsula (Figure 13).

49. Of the world's major rivers, the Yellow River carries by far the greatest load of suspended sediments (Figure 14). Its annual run-off ranges from 20 km^3 (1960) to 86 km^3 (1964) (Zhengying, 1983). Flowing through a loess plateau, the Yellow River erodes and transports an average annual sediment load of 1.6 billion tonnes. Much of this is deposited in the river channel below the loess plateau and in the estuarine zone. Aggradation of the channel and progradation of the delta reduce the river gradient, thus increasing deposition in the river bed, resulting in periodical breaches and subsequent displacements of the channel in history.

50. During several centuries of relative stability before 1855, the Yellow River carried down 300 to 500 km^3 of sediment to the Yellow Sea. The Jiangsu coastline prograded seaward as much

as 70 km. As a result of the southward transport of sediment-laden Yellow River waters along the coast, the north shore of the Yangtze River delta accreted nearly 100 km (Figure 12, Milliman, 1984).

51. This trend has changed during the last 37 years, the shoreline eroding nearly everywhere (Figure 13), more than 60 km^2 of Jiangsu shoreline being eroded annually (Milliman, 1984). The last natural diversion of the Yellow River, therefore, resulted in no change in the shoreline, until about 100 years after the diversion. This possibly resulted from erosion of the submarine delta, until the foreshore gradient became steep enough and erosion proceeded onshore (Milliman, 1984).

2. The Yangtze

52. The long-term average annual run-off of the Yangtze is of 979 km^3. The main problems in this basin stem from the deterioration of soil and from inadequate water conservation as a consequence of deforestation in the upper reaches, and from the narrowing of the river channel in its middle and lower reaches as a result of the reclamation of the flood-plains with polders, intensifying the threat of floods. This threat is aggravated by the high peak and large flood volume. The record large floods of 1931 and 1954 reached peak discharges exceeding 60,000 $m^3\ s^{-1}$ and 3-day flood volumes of over 20 km^3, largely exceeding the discharge capacity of the river, and resulting in the breach of dikes. The archives record 19 breaches of the Jingjiang dikes by the river from 1644 to 1949 (Zhengying, 1983).

B. THE INDUS RIVER MANAGEMENT

53. The Indus River is one of the world's largest rivers in terms of drainage area, of river discharge and sediment load. The Indus delta is located at the head of the Arabian Sea.

54. Since the 1940s, human activities have greatly altered the discharge pattern of the Indus and, as a result, the transport of sediment. Four general types of engineering activities have occurred along the river (Milliman *et al.*, 1984):

- three major dams have been built for water storage and hydroelectric power, at Tarbela, Mangla and Bhakra (Figure 15);

- an extensive system of canals for the transfer of water to and from various river branches as well as for farmland irrigation;

- barrages which contribute to control the river and direct its flow to channels (Figure 15);

- embankments and dikes which prevent river overflow, restricting the flow to the main channels.

55. Water resource development plans appear to be oriented towards the ultimate total capture and diversion of the river water for irrigation. In some near future, the Indus will no longer make a freshwater input to the delta area and the Arabian Sea (Snedaker, 1984).

56. Such wide-scale alterations of the Indus already have a drastic impact both on the river and its delta.

57. Before the man-made changes upstream, the Indus delta prograded at the annual rate of about 30 m (Kazmi, 1984). Under prevailing conditions of extreme wave energy, this progradation was only possible because of the high amount of silt and sand that was deposited and retained by the delta (Wells and Coleman, 1984). The active delta has now shrunk from approximately 2,500 km^2 to a small 250 km^2 triangular zone.

58. Soon after completion of the dams, the sediment load of the river fell sharply. Though downdam erosion restored some of the load, it appears that the sediment input to the estuary has decreased from a previous 300 Mt annually (according to Kazmi, 1984) or 250 Mt annually (according to Milliman *et al.*, 1984) to less than 100 Mt in 1974 - 75 (Milliman *et al.*, 1984), but marked early fluctuations persist. Whereas the water discharge in the 1950s exceeded 100 $km^3\ y^{-1}$, it was frequently less than 60 $km^3\ y^{-1}$ in the 1960 - 1970s (Figure 16).

59. Embankments and levees, while preventing the flooding of the flood plain, resulted in an increased salinity in ground water (Beg, 1977).

60. Flooded soils and salt-water intrusion appear to be a problem in the lower Indus delta. Much of the lower delta plain receives salt-water flooding from sea level elevation associated with the southwest monsoon winds in summer (Wells and Coleman, 1984). Surface salt accumulation and the effects of hypersalinity are presumably associated with the partial deterioration of the rice and fishery industries (Snedaker, 1984).

61. Observations of the mangrove forests of the Indus delta made in 1977 and in 1982 showed a progressive deterioration, undoubtedly associated with the reduction of the freshwater river flow (Snedaker, 1984). The forested areas are now wholly restricted to the well-flushed banks of the tidal channels; the interiors of the deltaic islands appeared bare. This process of deterioration will eventually lead to the loss of the remnants of the mangrove ecosystems in the Indus delta. It should be noted, however, that the shrimp annual harvests remain high, in spite of the steady decrease in the area of mangrove forests (Snedaker, 1984).

C. EFFECTS OF RIVER MANAGEMENT ON THE BLACK SEA AND THE SEA OF AZOV

1. River input

62. The Black Sea has a drainage area of 1,864,000 km^2. Most drainage comes from the eastern Russian platform and only 15 per cent comes from the high mountain areas. Several large rivers empty into the Black Sea (Figure 17 & 18).

63. Using Soviet data, Shinkus and Trimonis (1974, in Ross, 1977) summarized the sediment input to the Black Sea and the Sea of Azov by rivers. There is a yearly discharge of 374 10^6 km^3 into the Black Sea and the Sea of Azov which brings in about 15 10^6 tonnes of solids in suspension, 55 per cent of which comes from the Danube. An additional 15 Mt y^{-1} is supplied by traction load mainly during the spring-time flood season. The total annual contribution of dissolved solids is of the order of 100 Mt. The largest concentrations of suspended material occur around the edges of the Black Sea and Sea of Azov while little material is present farther from shore (Ross, 1977).

2. Alterations in river inputs

64. The rivers draining into the north-western and north-eastern Black Sea – the Danube, the Dnepr, the Dnestr, the Don and the Kuban – have been extensively manipulated since 1950. Man-made changes were intended to serve several purposes: to generate hydropower, to provide inland water reserves for industrial municipal and agricultural needs, and to accomodate river shipping.

65. A series of hydropower stations were constructed, creating large storage lakes. The total area of the Dnepr and Dnestr storage lakes reached about 7,160 km^2. The Don and Kuban reservoirs covered 2,850 km^2 (Table 12). On the Danube, 28 hydroelectric plants have been built in Germany alone, but the "Iron-Gates" plant in Yougoslavia-Romania is five times greater than all of them together.

66. Large amounts of fresh water have been retained inland in order to allow for river transportation. The development of new inter-river routes in the USSR has increased the amount of fresh water retained inland, with the subsequent intrusion of higher-salinity water into estuaries. Irrigation, however, has become the largest source of water consumption. Irrigation canals to arid

areas have added to the irretrievable losses of fresh water (Table 13 and 14). Projects to irrigate about 2.6 million hectares east of the Danube will require the withdrawal of about 1/3 of the present river water flow.

67. Large storage lakes have encouraged the rapid increase in industrial water consumption. The increased water consumption in the Black Sea Basin resulted in the deterioration of water quality. About 50 per cent of the water consumed goes back to the river system without sufficient treatment (Tolmazin, 1985). Concentrated outfalls and scattered sources discharge such chemicals as cyanides, ammonia, phenols and heavy metals (Tolmazin, 1985). The most severe pollution problems, however, appear to be caused by agricultural run-off, bringing down fertilizers, pesticides and organic compounds. As a result, the major storage lakes have become eutrophic, and previosuly very productive river habitats are now depleted.

3. Impact on the ecosystems of the Black Sea

68. The reduction in freshwater flow has induced salinity and density changes in the coastal areas of the Black Sea. In addition, the overloading of these river systems with industrial and sewage wastes has seriously depleted oxygen supplies.

69. Inland water management projects have produced a salinity increase of 0.19 per thousand in the Black Sea, but the increase is more marked in the north-west sector. The size of the surface outflow to the Mediterranean Sea is decreased, and in the meantime the bottom inflow through the Bosphorus is accentuated. The slow increase in volume of the bottom anoxic layer in the Black Sea may, in time, have a drastic effect on this basin by gradually reducing the thickness of the productive and well-oxygenated surface layer.

70. In the immediate vicinities of the Dnepr and Dnestr estuaries, salinity has increased by 2.0 - 2.5 per thousand. The dredging of navigational canals and the decreased river flow have resulted in an increased penetration of salt water up the estuaries. A correlation was found between salinity anomaly (S^-) and anomalies in the river discharge (Q') (Figure 19).

4. Stratification and oxygen deficit

71. Oxygen deficit became a problem in the north-west Black Sea (NWBS) and the Sea of Azov. The oxygen depletion results from two factors: the increased stratification in estuaries and the presence of organic materials from irrigated lands. The large-scale schemes of river-flow control led to an increased vertical density gradient because of the intrusion of a salt wedge. Vertical stability became generally higher without significant seasonal variations. Oxygen depletion in the NWBS was a serious problem during 1973-1975, following an itensified irrigation programme in the Dnepr and Dnestr basins. As a result, mussel fields were destroyed (Salsky, 1977, in

Tolmazine, 1985) and bottom fish were driven toward the surface, where they eventually died washed up on the beaches. The coastal waters of Bulgaria and Romania were also affected. Similar cases of oxygen depletion and fish mortality have occured in the Sea of Azov.

5. Fish catches

72. The Black Sea has historically been one of the most biologically productive regions in the world. The Sea of Azov in particular, supported a large fish population. The productivity of this area, however, has been sharply curtailed by man-induced changes in the environment.

73. Major fishing industries have suffered a drastic decline. The catch of bottom fish in NWBS, such as flounder, dropped from 80,000 tonnes before 1949 to only 4,000 tonnes in 1971 - 1975. On the whole, the total catch decreased from 300,000 tonnes to about 100,000 tonnes in 1971 - 1975. The decline of major fish stocks is attributable to a reduction in the brackish water environments where anadromous species reproduce. (Tolmazin, 1985). There is no longer a regular commercial fishery along the Dnepr and Dnestr estuaries. The only remaining brackish water habitat that is relatively healthy is in the vicinity of the Danube river.

6. Impact of water withdrawal on the Sea of Azov

74. The Sea of Azov, one of the most productive low-salinity regions in the world, provides strong evidence in support of the view that freshwater inflow plays a major role in maintaining the biological productivity of the sea and its estuarine systems (Figure 20). Water withdrawals from the Sea of Azov have grown as high as 46 per cent (Figure 21). The total losses of freshwater supply between 1950 and 1975 account for almost 250 km^3, or about 11 $km^3\ y^{-1}$.

75. This sustained trend in declining water supply had a negative impact on physical properties and biological productivity (Rozengurt *et al.*, 1985, Tolmazin, 1983):

1. increased salt intrusion into the Sea of Azov, the Don and Kuban rivers-estuarine systems;

2. accumulation of salt throughout the system resulting in the rise in mean salinity from 9 ppt to 14 - 16 ppt (Figure 22);

3. a 60 per cent reduction in primary and secondary productivity and over 95 per cent reduction in catches of anadromous fish (Figure 23 and table 15). Russian scientists have determined (Rozengurt *et al.*, 1985) that the reduction of runoff of about 1 km^3 and the resulting increase in salinity of about 1 ppt in the spawning grounds of anadromous fish reduces the Sea of Azov stocks by about 25 - 35,000 tonnes per each generation;

4. following the rise of mean salinity, there was a massive invasion of billions of medusae into

the Sea of Azov and the Don River Delta (Figure 24), causing a serious threat to many indigenous species. The jellyfish created severe problems such as food competition with fish and public health problems.

D. MANAGEMENT OF THE NILE RIVER AND ITS EFFECTS ON THE EAST MEDITERRANEAN

76. Man's intervention in the flow of the Nile dates back to Pharaonic times. Modern intervention began with the construction of the "Delta Barrage" near Cairo in 1861. The barrage sluices were opened to let the flood waters flow, but made possible the beginning of perennial irrigation instead of basin flooding. The practice was developed with the low Aswan Dam built in 1902 and twice raised in height, in 1907 and 1929. The low Aswan Dam was also provided with sluices to let the flood waters flow with their sediment load. Total sediment trapping began with the completion of the High Aswan Dam.

1. The pre-damming Nile regime

77. The High Aswan Dam was completed in 1965 and, as a result, the summer of 1964 saw the last "natural" flood discharge to the East Mediterranean. Until then, the Nile flow followed a rather regular two-phase pattern, the flood wave with its heavy load of suspended solids reaching Southern Egypt in June-July, rising to a peak by the end of September (Figure 25), and dropping to its low winter level by December. During the low-level phase, until the onset of the following flood season, the two river outlets at Rosetta and Damietta remained tightly closed, to prevent salt-water intrusion, by means of a weir-dam on the former and an earthen dam on the latter. Both dams were then opened in mid-August to let the excess of flood-water flow to the Sea.

78. Large yearly fluctuations in the total freshwater discharge are recorded over the past centuries, following fluctuations in the rainfall over the Ethiopian highlands and the Ugandan plateau. The average discharge for the five years 1959-1963 amounted to 42.9 km^3. In 1964, it was higher, 52.89 km^3. The Nile stream then progressed rapidly eastward and then northward along the coasts of Egypt and Asia Minor, its northward extension depending upon the height of the flood wave. The fertilizing effects of this periodical inflow of nutrient-rich waters into an otherwise oligotrophic sea were analogous in many ways to those of the Nile river itself on the arid lands it runs through (Halim, 1960). Considerable amounts of nutrient salts were brought down to the South East Mediterranean with the flood waters and distributed over a wide area by the flood stream.

79. Assuming the outflow of 43 km^3 (the 1959-1963 average), and based on concentrations

measured in flood water sampled 3 km upstream from the outlet (Halim, 1960), the amount of silicate and phosphate brought down to the sea during the flood season would be of about 410,000 tonnes of silicate and 8,260 tonnes of phosphate. These figures, however, appear to be an underestimate, as has been shown by Halim and Morcos (1966), since gradual dilution of the spreading stream waters was accompanied by the release of silt-adsorbed nutrients. The fertilizing effect of the flood water, both on land and in the sea, was mostly due to its suspended material.

80. The sudden release of the nutrient-rich stream waters in early August immediately stimulated a thick and continuous diatom bloom in the surrounding sea waters, the flood itself remaining devoid of plankton owing to its opacity. The phytoplankton biomass rose from 40-125 10^3 cells l^{-1} in the first week of August to 2 to 12 million cells l^{-1} few days after flood release. Soon after the onset of the bloom along the Nile delta (Figures 26 and 27), large shoals of Sardinella (*S. eba*, *S. aurita* and *S. granigera*) were attracted to the coastal belt to feed on the dense diatom crop. The stomach contents of the fish in this season were entirely composed of diatoms (Halim, 1974). The fishing season coincided with the flood season, extending from September to November. The Sardinella catch accounted for 30 to 40 per cent of the total sea fish landings of the year. The shrimp fisheries were next in importance (Figure 28). Spawning for many benthic organisms such as echinoderms, polychaetes and lamellibranchs appears to have been induced by, or at least coincided with, the Nile bloom, their pelagic larvae contributing a large proportion to the zooplankton crop in this season.

2. The sediment load

81. The average annual suspended sediment load for the period 1904-1963 at Gaafra, 1,100 km upstream from the outlets, is estimated to have been about 160 Mt y^{-1}, ranging from 50 to 300 in 1913 and 1954, respectively. It was higher in the nineteenth century with an estimated average of 200 Mt y^{-1} from 1825 to 1902. These estimates were obtained by using an empirical model based on measurements taken during nine flood seasons (Quelennec, 1976). Almost all of the suspended sediment load (98 per cent) occurred during the flood season:

Month	July	August	September	October	November
Percentage	2	45	38	12	1.5

82. Numerous sites of deposition within the catchment area in the Nile valley and delta reduced the actual sediment input to the sea. Major deposition areas were the irrigated farm lands, the reservoirs, and the Nile river bed. Irrigated land comprises basin irrigation and perennially irrigated areas. Since 1886, the basin-irrigated surface area was gradually decreased from 8,000 km^2 to 2,500 km^2 in 1964, from a total cultivated area of 25,000 km^2. It is estimated that this change-over to perennial irrigation reduced the silt deposition per unit area from 1.85 kg m^{-2} to 0.76 kg m^{-2}. The low Aswan Dam is provided with low level sluices to allow for the easy flow of

the silt, the heaviest load being in the lowest water layer, and is usually open during the flood, so that silting up of the reservoir has not been of much significance.

83. The length of the Nile from Aswan to the Rosetta estuary is of 1,180 km with an average surface width of 900 m. The often suggested accretion rate of 1 mm y^{-1}, if applicable to the whole stretch, would only account for about one million m^3.

84. The total sediment load calculated for Gaafra is consistent with the measurements made at Lake Nasser in 1975 for the total sediment accumulation since the completion of the High Aswan Dam (Anonymous, 1978). It appears therefore that some 130 - 140 Mt of sediments did annually reach the Mediterranean through both Rosetta and Damietta outlets in the period 1903-1963. Before 1000 AD, the Nile had seven outlets which have been replaced by the present two outlets. Since then, two new and nearly independent sub-deltas became superimposed upon the ancient delta, the Rosetta and Damietta promontories (Figure 29), each with its own sediment supply and sediment distribution pattern in the Mediterranean (Anonymous, 1978).

3. The post-damming conditions

85. Gradual filling of the reservoir behind the dam, and full regulation of the river became complete in 1968-69. The intake of both Nile branches at the level of Cairo is shown in Figure 30 for the years 1956 to 1976, and the average monthly suspended load downstream of Aswan for pre- and post-impoundment years in Figure 31.

86. The new Lake Nasser behind the dam has a storage volume larger than that of the maximum flood and serves as a detention reservoir. The flood wave is completely absorbed and the outflow is now fully controlled. Some surplus fresh water, however, still reaches the Mediterranean, a volume estimated to be about 2.5 to 4 km^3, but the suspended sediment load in its totality is deposited on the lake bottom upstream from the dam.

87. Subsequent surveys of the continental shelf waters (Emara *et al.*, 1973, Mostafa, 1984), carried out during autumn, show conditions in sharp contrast with the past (Figure 32). A typical high-salinity levantine water mass extends in front of the delta. The phosphate content never exceeds the East Mediterranean range (0.04 - 0.10 µM). The major phytoplankton bloom now develops in winter, the autumn bloom being insignificant. Observations made off Alexandria show a drop in the standing crop in the autumn of 1965 to about 5 per cent of its corresponding value in 1964 (Guerguess, 1970).

88. The drop in primary productivity had dramatic consequences on the fishery resources. Since 1964, the total landings have been steadily decreasing so that, by 1969, the yield was less than 20 per cent of its value in 1962-63. The effect on the Sardinella fisheries is even more pronounced, since this fishery depended directly on the Nile bloom. In 1962, 18,000 tonnes were landed. In 1968 and 1969, the landings did not exceed 460 to 600 tonnes. The shrimp fisheries, next in im-

portance, have also been affected but less abruptly, owing to the more complex cycle of the Penaeids. About 8,000 tonnes were landed in 1962, but by 1969 the yield had dropped to 1,130 tonnes, stabilizing at this level. Figure 28 suggests a recovery from this trend after 1979, with a significant rise in Sardinella landings. However, this rise is only apparent and due to an increased fishing effort and to improve the fishing methods.

4. Shoreline changes

89. The position of the coastline of the Nile Delta has been determined by the higher rate of deposition of sediments during the annual floods of the river, versus the eroding action of waves and currents which transport the sediments. On a longer time scale, the Nile Delta cone was building up with steady uniformity. On a shorter time scale, it was oscillating around a state of unstable equilibrium, the oscillations caused by fluctuations in sediment discharges. Man's intervention changed the pattern into one of erosion.

90. Changes in the coastline of the Nile Delta can be followed from topographic surveys since early 1800 supported by recent drilling data (Anonymous, 1978; G. Sestini, unpubl.). Until about 1910, there was active progradation of the two sub-deltas and their offshore silty-mud cones. An overall 25 per cent decrease in suspended load since, due to the decrease in rainfall over Ethiopia, was followed by a general retreat. Erosion at Rosetta from 1915 to 1964 (Anonymous, 1978) was of the order of 30 m a year.

91. With the absence of Nile-sand supply, the coast is now subjected to a process of dynamic re-adjustment (Figure 29), the promontories fast retreating and the embayments advancing. Since 1964, the Rosetta promontory has eroded at a progressively accelerating rate, as shown by successive surveys, being 125 m y^{-1} and 211 m y^{-1}, respectively, in 1973 and 1982 (Anonymous, 1978) (Figure 33). The only source of sand at present is the shore itself and the shallow marine shelf. Large-scale offshore erosion occurs in areas where winter erosion is no longer compensated by summer deposition, and such changes cause offshore sediment movements from the nearshore regimen. The sand budget and the hydrodynamic processes have been monitored since 1972 by the Caostal Protection Institute with UNDP/UNESCO assistance (Anonymous, 1978).

5. Towards a balance sheet

92. Any balance sheet of the positive and negative after-effects of the High Aswan Dam will have to take into account at least three major benefits; the magnitude of the hydropower generated, the doubling of the farmland area and, no less crucial, the regulation of water supply to the country despite a decade of severe drought in the Sahel-Sudan-Ethiopian belt. Life in the Nile Valley and delta is entirely dependent on the river Nile as a water-artery, rainfall being insignificant. The deficit in the flood-water supply necessitated a continuous emergency withdrawal of

large volumes of water from the reservoir to protect the country from the catastrophic effects of drought. From 1978 - 79 to 1984 - 85, the total deficit compensated from the reservoir reached 73.5 km^3 (Figure 34).

E. IMPACTS OF WATER DIVERSIONS ON THE SAN FRANCISCO BAY-DELTA-ESTUARY ECOSYSTEM

93. Over the past 130 years, the San Francisco Bay and Delta (Figures 35 and 36), the largest inland estuary of the west coast of the U.S.A., has been altered greatly by land and water development. Upstream impoundments, diversions and pumped exports from the delta have reduced river flows by more than 50 per cent (Figure 37). The long-term effects of the upstream impoundments, diversions and removals are just beginning to be understood. (Davoren and Ayres, 1984; Rozengurt *et al.*, 1985) (Figures 36 and 37).

94. Less than 150 years ago, the 2,978 km^2 of the delta (the Sacramento-San Joaquin Rivers) were freshwater and tidal marsh. Through marsh reclamation, and damming and diverting of the rivers, the conditions in the delta today are largely under man's control. More than 80 per cent of the marshland was leveed and developed for agriculture in the early 1900s. Since then, about 100 levee failures have occurred. Wind erosion, compaction and consolidation of the delta's peat sand and silt soils have reduced the levels of the 57 islands used for agriculture as much as 7.6 m below sea level (Davoren and Ayres, 1984, Unesco, 1984).

95. Between the mid 1940s and 1978, two major water storage and transport projects were completed, the Central Valley Project (CVP) and the State Water Project (SWP). The project's pumps, located in the southern corner of the delta, send water via canals to the San Joaquin Valley (the southern half of the Great Central Valley) and the Tulare Lake Basin for irrigation, and to the Los Angeles, San Diego and San Jose areas for urban needs. Pumping these large quantities out of the southern delta causes reversal of currents in some delta channels and kills or removes young fish, eggs and larvae by entrainment. Since the mid-1940s, when operation of the CVP began, efforts to reduce mortalities at the pumps have failed. The SWP pumps began operating in 1967, further complicating the bay/delta environment for aquatic life (Davoren and Ayres, 1984).

96. Between 1950 and 1978, freshwater diversions amounted to a total of 286 km^3, 40 times the volume of the San Francisco Bay. For 28 years, therefore, an average of 10 km^3 y^{-1} was withdrawn from river inflow to the delta. This decline in water supply had a negative impact on both physical properties and biological productivity (Figures 38, 39 and 40):

1. an increase in salt intrusion into the upper part of the delta and bay, resulting in an increase of mean salinity in the bay from about 20 to 27;

2. massive reduction in the sediment load discharge to the delta-bay-coastal zone ecosystem by 60 - 75 per cent of the 8 10^6 tonnes discharged per year under natural run-off conditions;

3. correlation of records of commercial catches of salmon, striped bass, and shad with spring run-off to the bay for pre-project years (1915-1940) indicate that there were significant landings only when spring outflows were 3.4 - 6.2 km^3 for the preceding 2 - 4 years. In contrast, the current range of mean spring-water supply is 2 to 5 times less. In recent years, commercial fishing for salmon, shad and striped bass has been prohibited. Sport catches have declined to about 30 per cent of the levels of 20 years ago, despite an increase in sport fishing effort (Rozengurt *et al.*, 1985) (Figure 40);

4. Between 1967 and 1982, when reliable counts were made of winter salmon runs past Red Bluff Dam, the average annual volume of water diversion was approximately 12.2 km^3, and cumulative withdrawals from the Sacramento-San Joaquin River supply to the estuarine system reached about 190 km^3 (Figure 41) between 1967 and 1982. During the same period, the number of winter-run Chinook Salmon returning to spawn in the upper reaches of the river was reduced as much as 60 times, despite the release of millions of hatchery-reared juveniles into the western delta (Figure 41).

97. In the literature and numerous unpublished reports on the status of Chinook Salmon spawning populations in the Sacramento-San Joaquin watershed, four factors have been proposed to explain the population decline: dams, water diversions, pollutants, and the loss of 95 per cent of their habitat. While all of these factors may contribute to the reduction of the salmon population, the data of Rozengurt *et al.* (1985) strongly suggest that the gradual increase of cumulative losses of water and nutrients resulting from diversions will continue to be the principal factor governing migration, spawning success and recruitment in this stock for years to come.

APPENDIX

MAN-MADE CONNECTIONS BETWEEN OCEANIC BASINS:

THE SUEZ CANAL

98. Although the waters of the Suez Canal are insignificant in volume compared to those of the adjacent seas, this man-made connection has become the site of a large-scale experiment in biological migration between the Atlanto-Mediterranean and Indo-Pacific basins.

99. The present connection between the two basins is not the first. According to several authors (Gohar, 1954), the palaeogeography of the Northern Red Sea provides evidence of alternating upheavals and subsidences, connecting or disconnecting the two basins. The Mediterranean and the Red Sea organisms intermixed at least during two periods, the end of the Eocene and the early Pliocene. Towards the close of the Pliocene, the Isthmus of Suez became finally uncovered and the two basins completely isolated from each other.

100. During historical times, however, man-made indirect freshwater connections were dug out. The northern extension of the Gulf of Suez was joined to the Mediterranean via the Nile by a canal dug by the Pharaoh Sesostris in 2000 BC, and once more in the 5th and 3rd centuries BC (the ships of Hatshepsut could dock at Thebes on their return from "Punt"). Whether there are remnants of the Mediterranean and Red Sea faunas from the Eocene and early Pliocene connections in the respective basins is a question of particular interest which has, as yet, received no satisfactory answer. The only organisms apparently providing some evidence of the survival of the original Mediterranean population in the Red Sea are the molluscs (*Fossularca lactea, Acanthochiton discrepans*) (Fox, 1926). A converse example is provided by the occurrence of the Indo-Pacific alga *Acanthophora delilei* in the East Mediterranean before the opening of the Canal (Aleem, 1983).

101. Circulation in the canal is largely determined by two factors, the high salinity and high density water mass in the Bitter Lakes, and the monsoon winds over the Indian Ocean. From January to June, the mean sea level is 30 - 40 cm higher in Suez than in Port-Said, resulting in a northward transport of high-salinity water reaching to Port-Said. From July to September the sea-level gradient disappears and surface waters are driven southward by the North winds. A stagnation period follows in October-December, before the current direction is reversed. This period is accompanied by a rise in salinity at the northern end of the Bitter Lakes.

102. The anomalous high-salinity gradient from the Bitter Lakes towards the two canal ends'

has been steadily decreasing since its opening: 168 g l^{-1} in 1869, 68 in 1872, 52 in 1924, 43.25 in 1935, 43.55 in 1954, 44 - 46 in 1966. In the meantime, the depth of the Bitter Lakes was increasing as a result of the dissolution of the salt beds (El-Sharkawy and Sharaf-El-Din, 1983).

103. The role of the canal as a pathway for migrating organisms does not entirely depend on its suitability as a habitat. As a habitat, the canal remained inhospitable to the fauna and flora of both seas for a long time. Fox (1926) was struck by the great poverty of the canal fauna in 1924, "the banks and the bottom are devoid of life, except for the piers, and there are no sea weeds." The lakes, however, were rich. The high turbidity caused by continuous ship traffic, the lack of suitable substrates, the continuous digging, the comparatively higher temperature of the canal waters, and the hypersalinity of the Bitter Lakes were cumulative obstacles to the penetration and settlement of organisms from both ends. An increasingly rich and varied fauna and flora, however, appears to have gradually established itself in the canal in the following years (Gruvel, 1935; Aleem, 1983), evidencing the changes in the canal conditions and the lifting of the salinity barrier. The immigration process appears to be steadily intensifying.

104. More organisms have crossed the canal and established themselves in the new areas than in the canal itself. Some, however, have penetrated the canal and luxuriated in it but have not extended beyond it (the scyphomedusa *Cassiopea andromeda*, the ascidian *Ascidia nigra*). Steinitz (1970, in Oren, 1970) listed 140 erythrean migrants into the Mediterranean and 41 Atlanto-Mediterranean forms in the Red Sea. Subsequent records bring this figure to about 200 erythrean immigrants (Aleem, 1983; Halim, unpubl.). The northward trend is obviously predominant. The erythrean migrants include organisms of different habits: active migrants such as fish and macrocrustaceans, passive migrants, such as microplankton, sponges, microcrustaceans, bivalves, algae, eel-grasses.

105. The area of extension of the erythrean immigrants reaches up to Turkey, Greece and the Adriatic to the north and north-west (an Indo-Pacific pycnogonid was recently reported from the N. Adriatic), and along the North-African coast to Tunisia. Their extension is expected to be favoured by the control of the fresh-water Nile outflow (Halim *et al.*, 1967). A gradual and radical alteration of the faunal and floral composition of the East Mediterranean is taking place at an increasing rate.

106. Competition between the migrants and the corresponding indigenous species for their ecological niches led in several instances to the predominance of the former over the latter (the fish *Sphyraena obtusata*, more successful than the Mediterranean *S. sphyraena*).

107. The flux of the Indo-Pacific species has enriched the East Mediterranean with several economically valuable fish (*Mugil seheli, Sphyraena obtusata, Siganus siganus*) and crustaceans (the crab *Neptunus pelagicus*, the shrimp *Penaeus japonicus*, the bivalve *Meleagrina occa*). On the other hand, three Mediterranean fish became widespread in the Gulf of Suez (*Syngnathus algeriensis, Pseudoserranus cabrilla, Sciaena aquila*) (Gohar, 1954).

TABLE 1. MAJOR WORLD DAMS CONSTRUCTED BEFORE 1983
(Data from Mermel, 1982. Reproduced by permission of *Water, Power and Dam Construction*)

A. Large Reservoirs (threshold 65 10^6 m^3)

Name	Completion date	Location	Reservoir Capacity (10^6 m^3)
Owen Falls	1954	Lake Victoria/River Nile, Uganda	204.8
Bratsk	1964	River Angara, USSR	169.3
High Aswan	1970	River Nile, Egypt	164.0
Kariba	1959	River Zambezi, Zimbabwe	160.4
Akosombo	1965	River Volta, Ghana	148.0
Daniel Johnson	1968	River Maniconagan, Canada	141.9
Bennett W. A. C.	1967	River Peace, Canada	74.3
Krasnoyarsk	1972	River Yenisei, USSR	73.3
Zeya	1975	River Zeya, USSR	68.4

B. Highest Dams (threshold 225 m)

Name	Completion date	Location	Dam height (m)
Grand Dixence	1962	River Dixence, Switzerland	285
Vaiont	1961	River Vaiont, Italy	262
Guavio	1982	River Orinoco, Columbia	250
Mica	1973	River Columbia, Canada	245
Chicoasén	1981	River Grijalva, Mexico	245
Sayano-Shushenskaya	1980	River Yenisei, USSR	242
Mauvoisin	1957	Drange de Bagnes, Switzerland	237
Chivor	1975	River Bata, Columbia	237
Oroville	1968	River Feather, USA	235
Chirkei	1977	River Sulak, USSR	233
Bhakra	1963	River Sutlej, India	226

C. Largest Hydroelectric Power Dams (threshold 4,000 MW)

Name	Completion date	Location	Planned power capacity (MW)
Grand Coulee	1942	River Columbia, USA	10,830*
Tucurui	1982	River Tocantins, Brazil	6,480*
Sayano-Shushenskaya	1980	River Yenisei, USSR	6,400
Krasnoyarsk	1972	River Yenisei, USSR	6,000
La Grande 2	1982	River La Grande, Canada	5,328
Churchill Falls	1971	River Churchill, Canada	5,225
Bratsk	1964	River Angara, USSR	4,600
Ust-Ilim	1980	River Angara, USSR	4,500
Cabora Bassa	1974	River Zambezi, Mozambique	4,000

* The latest updatings have given corrected figures for the planned rated capacity of Grand Coulee Dam as 5,494 MW and of Tucurui as 8,000 MW (Mermel, 1983).

TABLE 2. MAJOR WORLD DAMS UNDER CONSTRUCTION AND DUE FOR COMPLETION BY 1990 (Data from Mermel, 1982. Reproduced by permission of *Water, Power and Dam Construction*)

A. Large Reservoirs (threshold 65 10^6 m^3)

Name	Completion date	Location	Reservoir Capacity (10^6 m^3)
Guri	1985	River Caroni, Venezuela	136

B. High Dams (threshold 225 m)*

Name	Completion date	Location	Dam height (m)
Rogun	1985	River Vakhsh, USSR	325
Nurek	1985	River Vakhsh, USSR	300
Inguri	1985	River Inguri, USSR	272
Tehri	1990	River Bhagirathi, India	261
Kishaw	1985	River Tons, India	253
El Cajon	1985	River Humuya, Honduras	226

C. Hydroelectric Power Dams (threshold 4,000 MW)

Name	Completion date	Year of initial operation	Location	Planned power capacity (MW)
Itaipu	1985	1983	River Paraná, Brazil/Paraguay	12,600
Guri	1985	1968	River Caroni, Venezuela	10,000
Corpus Posadas	1988	1990	River Paraná, Argentina/Paraguay	6,000
Yacreta-Apipe	1988	1986	River Paraná, Argentina/Paraguay	4,050

* In the latest update, Mermel (1983) recorded that the Borocua Dam, Costa Rica, will be the fifth-highest dam when completed, at 267 m.

TABLE 3. GLOBAL DISTRIBUTION OF MAJOR DAMS AND RESERVOIRS BY NUMBER
(Petts, 1984 after Mennel, 1981. Reproduced by permission of *Water, Power and Dam Construction*)

	Completed to 1981			Dams under construction (109)
	Large reservoirs (25)	High Dams (24)	Hydroelectric Power Dams (77)	
NORTH AMERICA	4	6	21	14
U.S.A.	2	4	10	5
Canada	2	2	11	9
CENTRAL & SOUTH AMERICA	1	2	11	25
AUSTRALIA & NEW ZEALAND	0	0	0	1
SOUTH EAST ASIA	1	0	6	10
Japan	0	0	4	4
China	1	0	2	6
SOUTH WEST ASIA	0	1	0	12
India	0	1	0	5
AFRICA	5	0	1	5
EUROPE	0	5	2	6
U.S.S.R.	9	3	9	8

TABLE 4. WEST AND CENTRAL AFRICAN DAMS
(Collins *et al.*, 1983, in UNEP, 1985)

COUNTRY	No.	NAME	YEAR	RIVER	REFERENCE
Angola	9	Cambambe	1962	Cuanze	A, Sarmento & Alfonso (1962)
		Gove	1970	Cunene*	B
		Matela	1969	Cunene*	B
		Queve	1968	Queve	B
Cameroon	4	Sangana	1965	Sangana*	B, Rubin (1968)
		Mbakaou	1971	Sangana	A
		Bamendjin	1971	Mbam	A
		Edea III	1971	Sangana	B
Congo	3	Inga I	1973	Congo	B
		Shongo	1970	Congo	B
			1983	La Fini R	B
Eq. Guinea	0?				
Gabon	0				A
Ghana	9	Mauronga	1978	Tano*	B
		Henang	1967	Pra*	B
		Akoscabo	1961	Volta*	Mackintosh (1965), Ly (1980)
		Kpong	1981	Volta	Quartey and Allen (1981)
Guinea	2	Grandes Chito	1982	Konkoure	Rubin (1968)
Ivory Coast	11	Kolotali		Nzi (Bandama)	Rai (1974)
		Bouake		Kan*	Rai (1974)
		Ayame I	1959	Bia*	A
		Ayame II	1964	Bia*	A
		Kosson	1972	Bandama*	B
		Tabbo	1975	Bandama*	B
		Buyo		Sassandra*	A
		Soubre		Sassandra*	A
		Bandama	1972	Bandama	Chan (1973)
		Cavally	1965 (?)	Cavally	Rubin (1968)
		Ticbisson		Kan*	Rai (1974)
Liberia	1	Mt. Coffee	1966	St. Paul	A

TABLE 4. (CONTINUATION)

COUNTRY	No.	NAME	YEAR	RIVER	REFERENCE
Mauritania	1	Foum el Geita	198(?)	Gorgol	B
Nigeria	18	Bakolori	1975	Niger*	A, B Sargent (1979)
		Ejigb		Aro	A
		Gusau		Sokoto	A
		Ikere		Oyan	A
		Igbaja		Oyin	A
		Kishi		Soro	A
		Kainji	1968	Niger*	McDowell *et al* (1983)
		Shiroro	1977	Kaluma	Baylis (1972) Runin (1968)
		Kangimi	1977	Kangimi	B
		Birnin Kuda	1970		B
		Baganda	1970		B
		Karage	1971		B
		Keffin Guna	1971		B
		Tiga	1974		B
		Ikega	1982	Oyan	B
		George	1982		B
		Jebba	198(?)	Niger	B
		Dadin Kowa	1982	Gangola	B
Senegal	4(?)	Bakel	1965	Senegal*	Rubin (1968)
		Manantili	1980	Senegal*	B, Sall (1982)
		Diama	1981	Senegal*	B
		Kekriti(?)	1975		B
Sierra Leone	4(?)	Mange	1961		A, Rubin (1968)
		Bumbana	1960		Rubin (1968)
		Guma	1966	Rokel	B
				Mano	B
Togo	2	Kpime	1968		A, Rubin 1968)
		Koza	1975	Koza	Sargent (1979)
Zaire**	15	Inga I	1973	Congo*	
		Inga II	1982	Congo*	
		Kolwezi		Labudi	
		Lubumbashi		Lufira	
		Kinshasha		Congo*	
		Kisangani		Congo*	
		Ulindi		Ulindi	
		Lowa		Lowa	
		Luvua		Luvua	

A = World Register of Dams (1977)
B = Water Power (1966-1983)
* = Rivers with more than one dam construction
** = All of the dams concerned were constructed on the Zaire (Congo) and its tributaries.

TABLE 5. MAJOR RIVERS OF THE WORLD: MEAN DISCHARGE (m^3 s^{-1})

		Mean discharge (m^3 s^{-1})	Cumulative discharge (10^3 km^3 s^{-1})	Cumulative percentage contribution to the total discharge from continents Continent
1	Amazon	175,000		
2	Congo	39,640		
3	Orinoco	33,950		
4	Yangtse Kiang	22,000		
5	Brahmaputra	19.200		
6	Mississippi	17,800		
7	Yenissei	17,800		
8	Lena	16,300		
9	Paraná	14,900		
10	Mekong	14,900	371.5	31.3
11	Saint Lawrence	14,160		
12	Irrawady	13,560		
13	Si Kiang	12,500		
14	Ob	12,200		
15	Ganges	11,600		
16	Tocantins	11,000		
17	Aur	10,300		
18	MacKenzie	9,710		
19	Columbia	7,960		
20	Indus	7,550	482	40.6
21	Magdalena	7,500		
22	Zambezi	7,080		
23	Danube	6,530		
24	Yukon	6,220		
25	Niger	6,090		
26	Uruguay	5,500		
27	Ogoove	4,710		
28	Huang Ho	3,900		
29	Sepik	3,800		
30	Frazer	3,540	536.9	45.3
31	Northern Dvina	3,380		
32	Pechora	3,360		
33	São Francisco	3,070		
34	Godavari	2,690		
35	Pyasina	2,550		
36	Koksoak	2,550		
37	Neva	2,540		
38	Rhine	2,535		
39	Fly	2,450		
40	Purari	2,450	564.4	47.6
50	Cagayan	1,740	584.8	49.3
60	Shatt el Arab	1,450	602.2	50.8
80	Cuenza	950	625	52.7
100	Murray	737	642	54.1
120	Aux Feuilles	589	656	55.3
150	Dong Nai	473	671	56.6
200	Grey	294	690	58.2

From the World Register of Rivers Discharging to the Oceans, Unesco Division of Water Sciences, Internal Report.

TABLE 6. MAJOR CHEMICAL TYPES OF SURFACE WATERS (ORDER OF DOMINANT IONS EXPRESSED IN meq l^{-1}; THE SAMPLE CORRESPONDS TO A WATER DISCHARGE OF 23,260 km^3 PER YEAR). (Meybeck, 1981)

Cations	Anions	Percentage (A)	(B)	(C)
$Ca^{2+} > Na^{+} > Mg^{2+} > K^{+}$	$HCO_3^- > > Cl^- > SO_4^{2-}$		33.1	
	$HCO_3^- > SO_4^{2-} > Cl^-$		2.5	
	$SO_4^{2-} > HCO_3^- > Cl^-$	0.5		0.5
$Ca^{2+} > Mg^{2+} > Na^{+} > K^{+}$	$HCO_3^- > SO_4^{2-} > Cl^-$		46.7	
	$HCO_3^- > Cl^- > SO_4^{2-}$		15.0	
$Na^{+} > Ca^{2+} > Mg^{2+} > K^{+}$	$HCO_3^- > Cl^- > SO_4^{2-}$	0.4	1.0	
	$SO_4^{2-} > Cl^- > HCO_3^-$			0.1
	$Cl^- > HCO_3^- > SO_4^{2-}$			0.1

(A) Rain-dominated type $0.5 < \Sigma i < 1.5$ meq l^{-1}

(B) Rock-dominated $0.75 < \Sigma i < 8$ meq l^{-1}

(C) Evaporitic type $\Sigma i > 9$ meq l^{-1}

TABLE 7. SPECIFIC LOADS DISCHARGED TO THE OCEANS (Meybeck, 1979, 1981)

		Atlantic	Glacial Arctic	Indian	Pacific
Silica	ms	1.9	1.6	0.8	0.77
	mv	0.6	1.0	0.2	0.2
Total ionic content	ms	15	36	7.8	6.6
	mv	4.5	24	2.1	1.7

ms: load per unit ocean area (t km^{-2} y^{-1})

mv: load per unit ocean volume (t km^{-3} y^{-1})

TABLE 8. COMPARISON OF PROPOSED WORLD BUDGET OF HOLEMAN (1968) AND MILLIMAN AND MEADE (1983).

Area	Drainage area (10^6 km^2)		Sediment yield (t km^{-2} y^{-1})		Sediment Discharge (10^6 t y^{-1})	
	Holeman	Milliman & Meade	Holeman	Milliman & Meade	Holeman	Milliman & Meade
N. & C. America	20.48	17.50	87	84	1,780	1,462
S. America	19.20	17.90	57	97	1,090	1,788
Europe	9.2	4.61	32	50	290	230
Eurasian Arctic		11.17		8		84
Asia	26.6	16.88	543	380	14,480	6,349
Africa	19.7	15.34	25	35	490	530
Australia	5.1	2.20	41	28	210	62
Large Pacific Islands		3.00		1,000		3,000
TOTALS	100	88.60	183	150	18,300	13,505

Note: Northern Africa, Saudi Arabian peninsula and western Australia are primarily desert, and assumed to have little annual discharge of river sediments.

TABLE 9. IMPOUNDMENT EFFECTS ON SUSPENDED SEDIMENT LOADINGS (Petts, 1984)

River/Reservoir	Country	Annual suspended sediments loads		Proportion of natural load (per cent)	Source
		Natural	Below dam		
Danube	Romania-Yugoslavia	23.8 10^6 t	3.5 10^6 t	15	Bruk *et al.* (1981)
S. Saskatchewan	Canada	1.81 10^6 t	0.7 10^6 t	37	Rasid (1979)
Ume Älv	Sweden	100 10^3 t	40-50 10^3 t	50	Nilsson (1976)
Angermanälven	Sweden	92 10^3 t	34 10^3 t	37	Nilsson (1976)
Indalsälven	Sweden	76 10^3 t	26 10^3 t	34	Arnborg (1967)*
Rheidol	U.K.	26.7 10^3 t	2.7 10^3 t	10	Grimshaw & Lewin (1980b)
	WINTER	6.9 10^3 t	1.2 10^3 t	17	
	SUMMER	19.8 10^3 t	1.5 10^3 t	8	
	SUMMER+	4.7 10^3 t	1.0 10^3 t	21	
Vaal	South Africa	0.49 10^3 ppm	0.17 10^3 ppm	35	Schwartz (1969)
Missouri	U.S.A.	1.3 < 3.2 10^3 ppm	0.47 < 0.8 10^3 ppm	25	Neel (1963)
Nile	Egypt	0.6 10^3 ppm	0.05 ppm	8	Abul-Atta (1978)
	WET SEASON	1.15 10^3 ppm	0.05 ppm		
	DRY SEASON	0.06 10^3 ppm	0.05 ppm		
Callahan	U.S.A.	2.9 10^3 mg l^{-1}	0.34 10^3 mg l^{-1}	12	Schreiber & Rausch (1979)
Vltava	Czechoslovakia	0.64 10^3 mg l^{-1}	0.3 10^3 mg l^{-1}	46	Brádka (1966)
			0.06 10^3 mg l^{-1} + +	9	

* Reported in Nilsson (1976).
\+ Excludes a severe overbank flood of August 1973, which produced total suspended-sediments loads for that month of 30,995 t on the natural Afon Ystwyth and 1,035 t on the impounded Afon Rheidol.
\+ + After construction of a second upstream impoundment.

TABLE 10. PRINCIPAL CHARACTERISTICS OF SEVEN MAJOR RIVERS IN CHINA

No.	River	Catch-ment area (10^3 km^2)	Annual average run-off (10^9 m^3)	Annual average sediment load (10^6 t)	Water and soil loss area (10^3 km^2)	Popu-lation (10^6)	Max. recorded annual runoff (10^9 m^3 y^{-1})	Min. recorded annual runoff (10^9 m^3 y^{-1})
1	Yangtze	1,800	979	478	265.5	345	1,360/1954	676/1978
2	Yellow	750	56	1,640	391.3	82	86.1/1964	20.1/1960
3	Huai	270	50	14	52.7	125	84.1/1954	6.3/1966
4	Hai-Luan	320	29.2	81	123.8	98	45.8/1963	5.0/1920
5	Pearl	450	341	69	35.6	76	529.2/1915	127.7/1969
6	Liao	230	15.7	41	85.6	29	30.2/1954	4.7/1979
7	Songhua	550	76		72.2	47	121.4/1960	45.1/1968
8	Country total	9,600	2,600		1,203.4	1,031		

TABLE 12. THE MORPHOMETRIC AND FLOW CHARACTERISTICS OF STORAGE LAKES AND PROJECTED ENERGY PRODUCTION OF THE RIVERS ENTERING THE BLACK AND AZOV SEAS (Tolmazin, 1985)

Storage Lake	River	Total volume km^3	Available volume km^3 (Y)	Annual discharge (Q) km^3	$\eta = \frac{Y}{Q} - \%$	Installed capacity MW	Mean annual output KWH	Construction		
								Start	First unit	Last unit
Krasnodar	Kuban	3.1	2.9	11.0	26.4	NA	NA	1966	1973	1975
Tsymliansk	Don	23.9	11.5	22.0	52.3	200	700	1948	1952	1954
Kiev	Dnepr	3.7	1.2	33.1	3.6	350	635	1960	1965	1968
Kanev	Dnepr	2.6	0.33	43.9	0.8	420	823	1963	1972	1975
Kremenchug	Dnepr	13.5	9.0	48.4	18.6	625	1,506	1954	1959	1960
Dneprodzerzhinsk	Dnepr	2.4	0.52	50.6	1.0	352	1,250	1956	1963	1964
Dneproges	Dnepr	3.3	0.46	51.6	0.9	650	3,640	1944	1947	1950
Dneproges (expansion)	Dnepr					828	500	1969	1974	1980
Kakhova	Dnepr	18.0	6.8	51.9	13.1	351	1,420	1951	1955	1956
Dubossary	Dnestr	1.7	0.9	8.0	11.2	50	300	1949	1954	1959
Mogilyov-Podolsky	Dnestr	2.4	1.2	6.0	20.0	702	830	1975	(1983)	

Sources: Didkovsky, 1961; Bronfman *et al*, 1979; Rozengurt, 1974; Hydroelectricity, 1970; Electrification, 1977; Vendrov, 1979.

TABLE 13. WATER CONSUMPTIONS ($km^3\ y^{-1}$) FROM THE DNEPR RIVER BY VARIOUS WATER USERS (Tolmazin, 1985)

	observed			expected		
	1970	1975	1980	1985	1990	2000
Water users						
1. Municipal	0.3	0.4	0.5	0.6	0.7	0.8
2. Industrial	1.8	2.9	4.5	5.6	6.8	7.3
3. Agricultural (without irrigation)	0.7	0.9	1.0	1.1	1.1	1.2
4. Diversion within the basin	0.8	0.9	3.4	4.0	4.4	5.1
5. Irrigation	1.2	3.0	6.0	9.2	12.4	17.5
6. Interbasin water transfer for irrigation	1.7	2.9	4.4	7.0	9.7	14.1
7. Evaporation	3.5	3.5	3.5	3.5	3.5	3.5
8. Seepage	1.0	1.0	1.0	1.0	1.0	1.0
Total	11.4	15.5	24.3	32.0	39.0	49.5

TABLE 14. IRRETRIEVABLE LOSSES OF FRESH WATER IN THE DNESTR, THE DON AND KUBAN RIVERS (km^3) (Tolmazin, 1985)

Years	1970	1975	1980	1985	2000
River					
Dnestr	0.8	1.8	2.5	4.0	N/A
Don	5.4	7.6	N/A	11.0	2-20
Kuban	2.6	4.8	N/A	8.1	9-15

Sources: Bronfman *et al.*, 1979.

TABLE 15. ANNUAL CATCHES IN HUNDRED TONNES AND EXTRACTION OF CAVIARE IN THE SEA OF AZOV (Tolmazin, 1985)

Years	Pike-perch	Bream	Roach	Sturgeon family	Vimba and Shamaya	Herring	Caviare
1921-1936	244.6	446.1	n/a	29.5	4.88	52.0	2.0
1948-1952	96.4	76.1	n/a	15.7	3.67	6.3	1.0
1954-1961	19.9	29.9	38.6 (in 1955)	4.3	0.90	3.8	0.1
1962-1963	13.1	19.5	n/a	4.4	0.45	0.49	n/a
1973-1975	n/a	3.2	n/a	1.5	n/a	occasional	n/a

Sources: Gusev, 1967; Krotov, 1976.

FIGURE 1. World dam construction. Since 1971 the annual rate of dam-building has not declined but, for the countries represented in the 1973 *Register*, has been maintained at about 400 per year; this figure is increased, however, to 700 per year if all countries are considered (Mermel, 1981). (Redrawn from Beaumont, 1978, quoted by Petts, 1984)

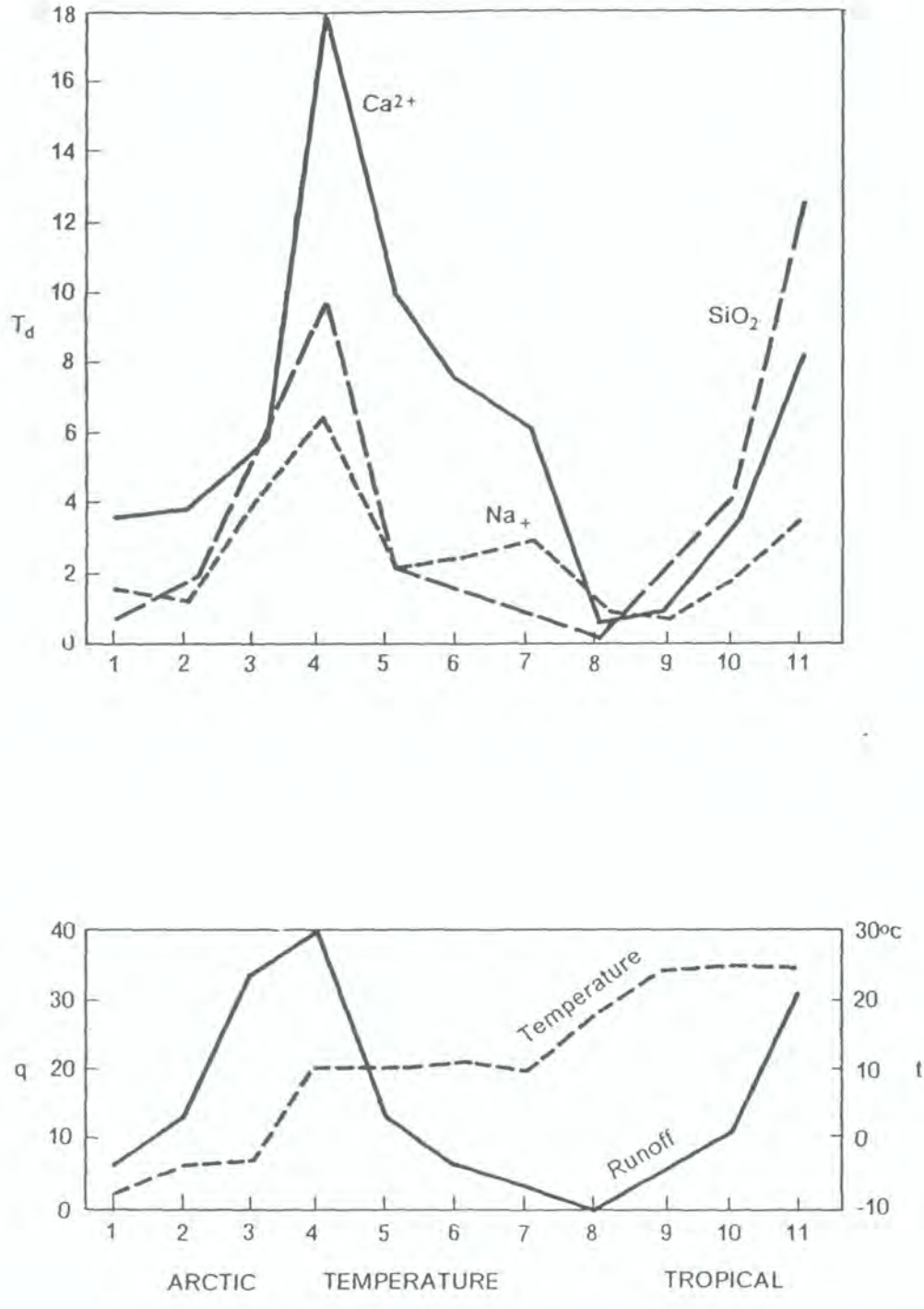

FIGURE 2. Dissolved elemental transports T_d (t km^{-2} per year) in eleven major climatic zones defined by surface run-off q (l s^{-1} km^{-2}) and average air temperature t (°C) (redrawn from Meybeck, 1979). The arid zone is No. 8.

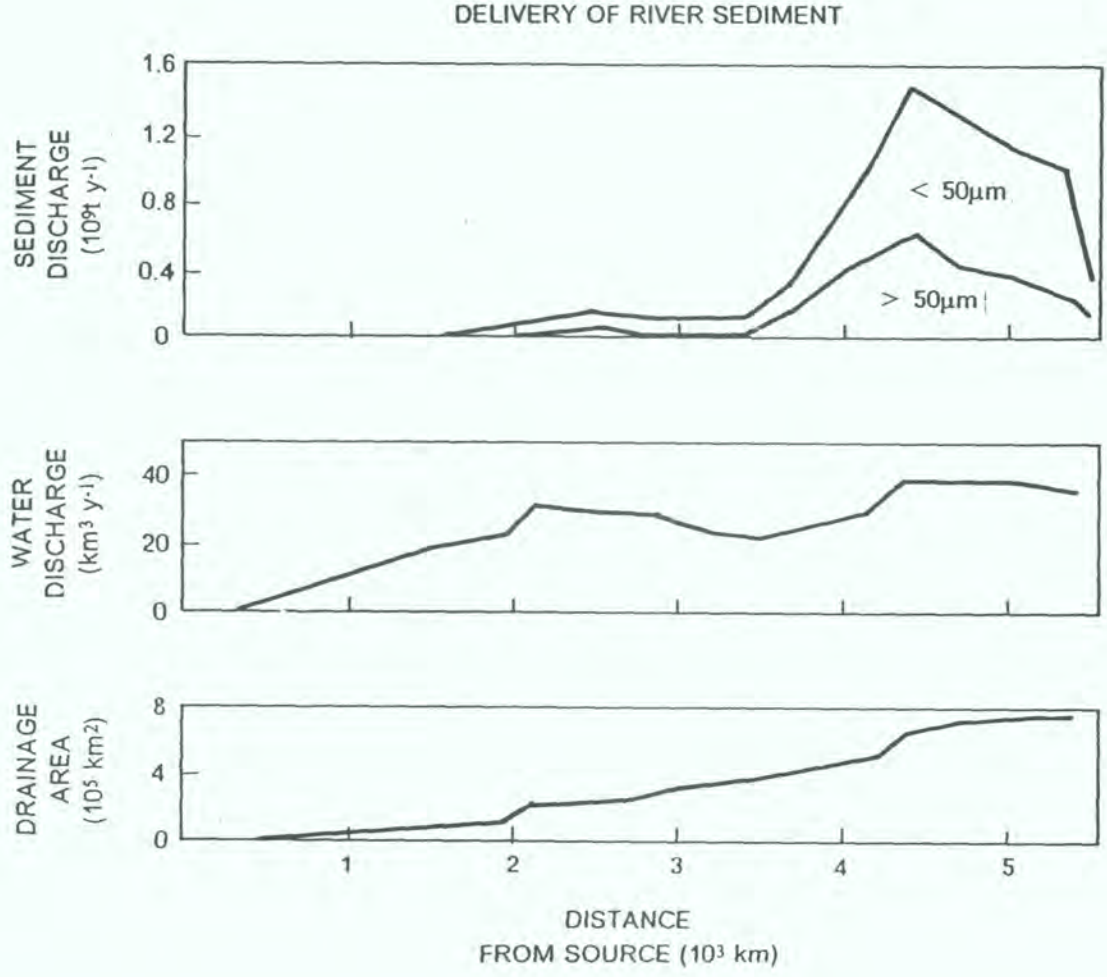

FIGURE 3. Graphs showing sediment discharge, water discharge, and drainage area along the Yellow River (Huangho) of China. Modified after Long and Xiong (1981) and based on continuous daily measurements at a number of gaging stations during 1965-1974. Sediment discharge (top graph) increases markedly as the river enters the loess region (about 3,500 km from source), decreases as the river flows across the alluvial plain (4,500 - 5,350 km), and decreases most markedly in the delta region (5,350 - 5,500 km). Last downstream gaging station is at Lijin, about 5,350 km from the source of the river. (Redrawn from Milliman and Meade, 1983).

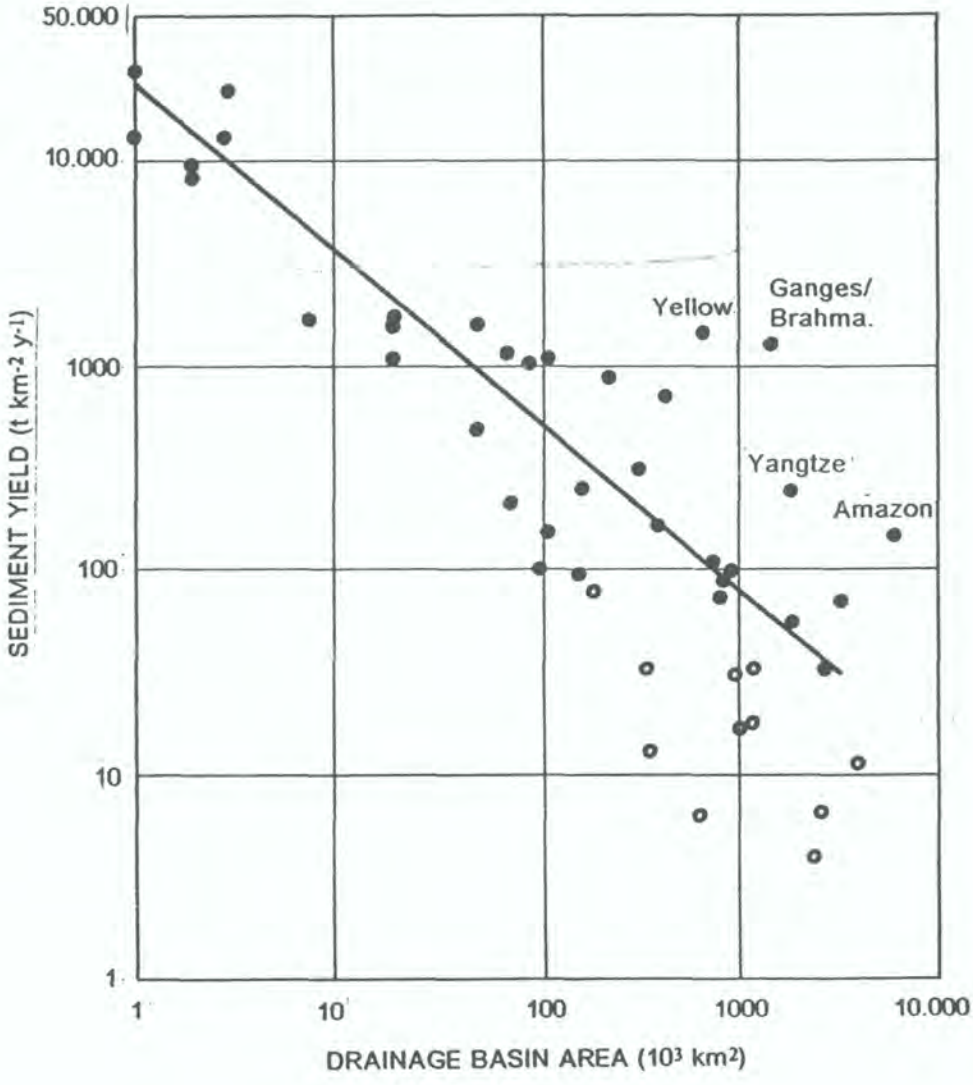

FIGURE 4. Comparison of sediments yields and drainage basin areas for all major sediment-discharging rivers (greater than 10 10^2 t y^{-1}. Open circles represent low-yield rivers draining Africa and the Eurasian arctic. Smaller basins have larger yields, although the largest rivers (Amazon, Yangtze, Ganges/Brahmaputra and Yellow) all have greater loads than their basin areas would predict. (Redrawn from Milliman and Meade, 1983).

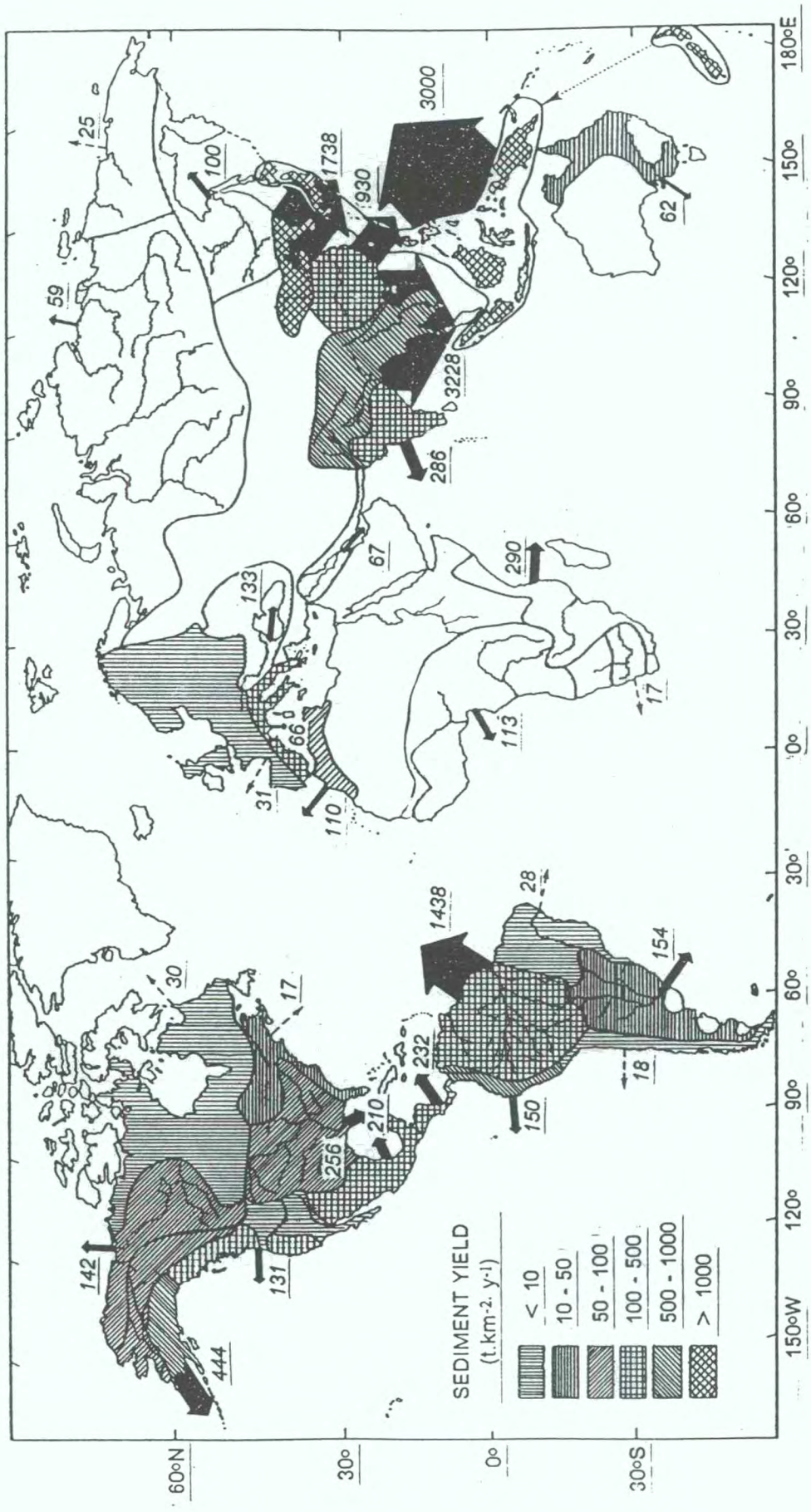

FIGURE 5. Annual discharge of suspended sediment from various drainage basins of the world: width of arrows corresponds to relative discharge. Numbers refer to average annual input in millions of tons. Direction of arrows does not indicate direction of sediment movement. The sediment yields and major rivers of the various basins also are shown; open patterns indicate essentially no discharge to the ocean. (Redrawn from Milliman and Meade, 1983).

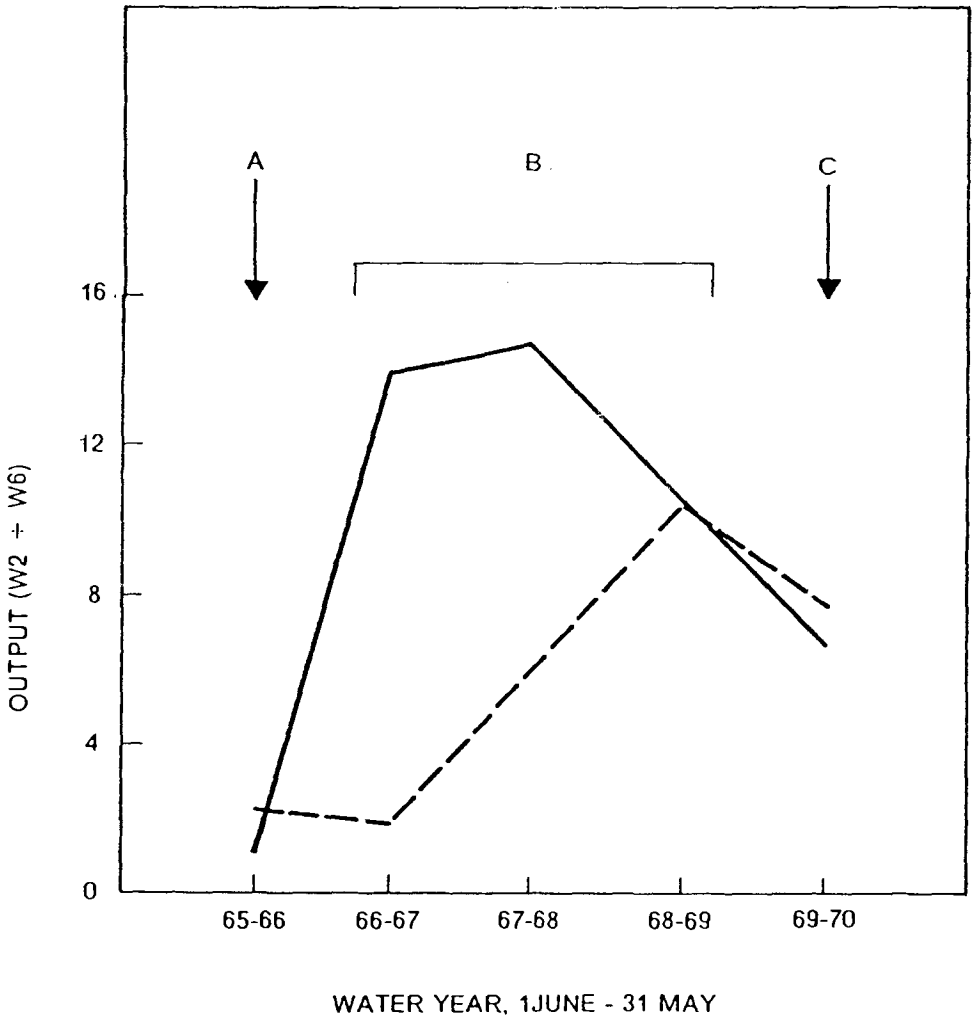

FIGURE 6. Ratio of dissolved substance output (NDS_{W2} / NDS_{W6}), solid line, and particulate matter (organic + inorganic) output (TPM_{W2} / TPM_{W6}), dashline, for several water years. A, forest cut Nov. 1965 - Jan. 1966; B, repression of plant growth by herbicides; C, regrowth begins. (Redrawn from Bormann *et al.*, 1974).

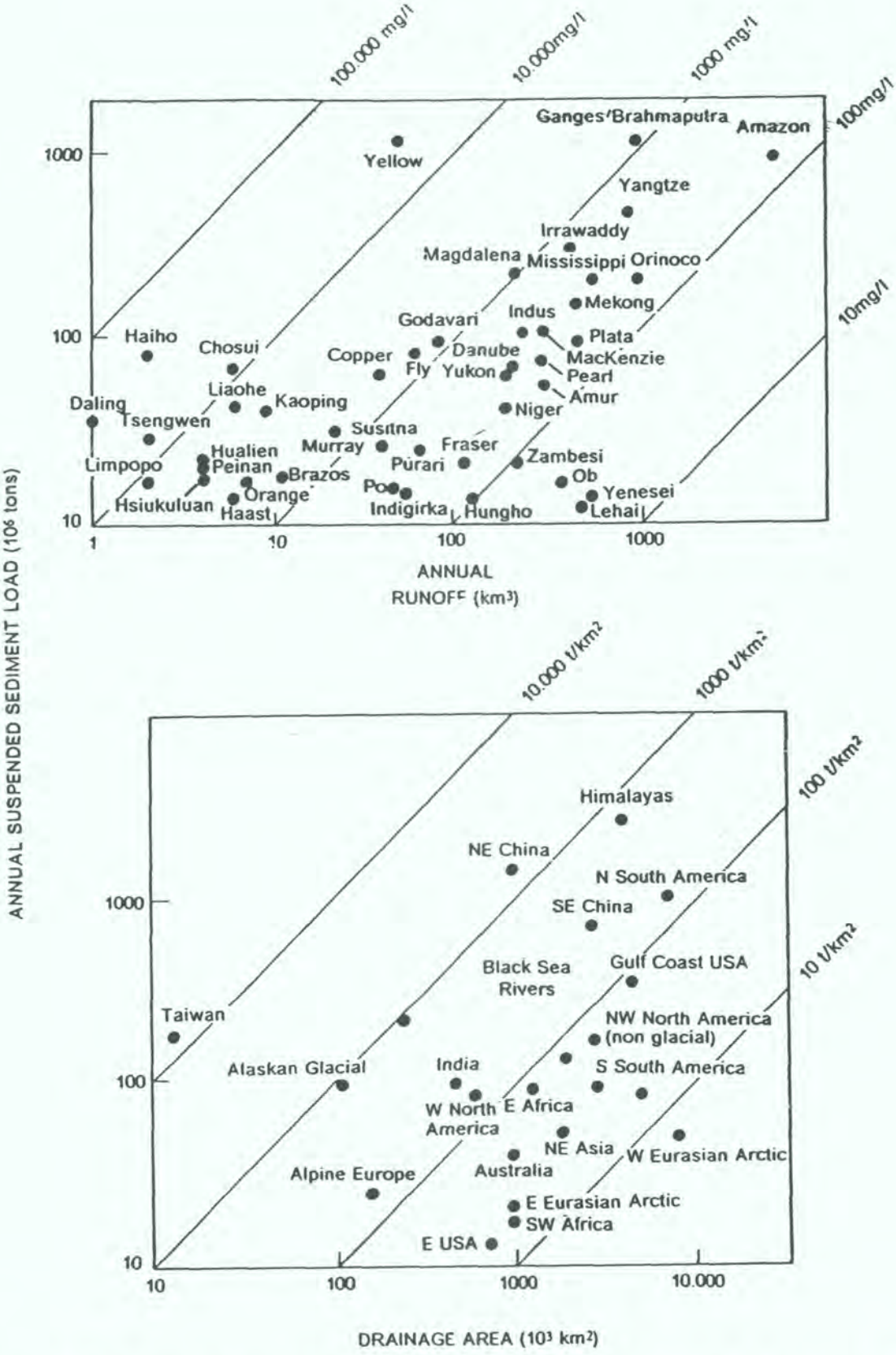

FIGURE 7. Variation of annual suspended sediment load with runoff (upper) and drainage area (lower). Note that the average concentration (upper) is highest in Asian and glacial rivers as well as those rivers draining arid areas (e.g., Orange, Brazos, and Murray). Yields for Asian and glacial rivers also are large, but desert rivers (Australia, SW Africa) have small yields. (Redrawn from Milliman and Meade, 1983).

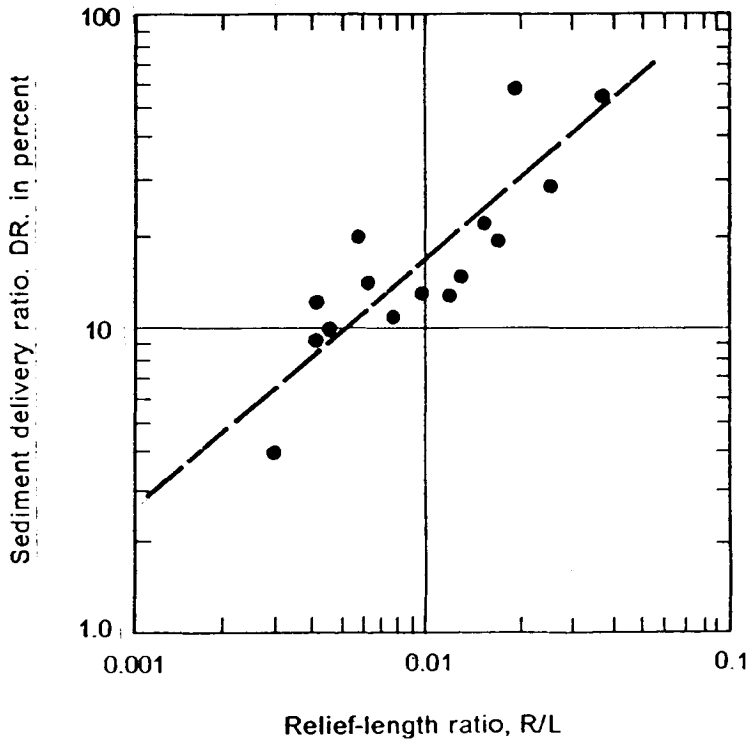

FIGURE 8. Relationship of sediment delivery ratio and relief-length ratio for 15 basins in southeastern U.S.A. (Redrawn from Roehl, 1962, quoted by Hadley *et al.*, 1985).

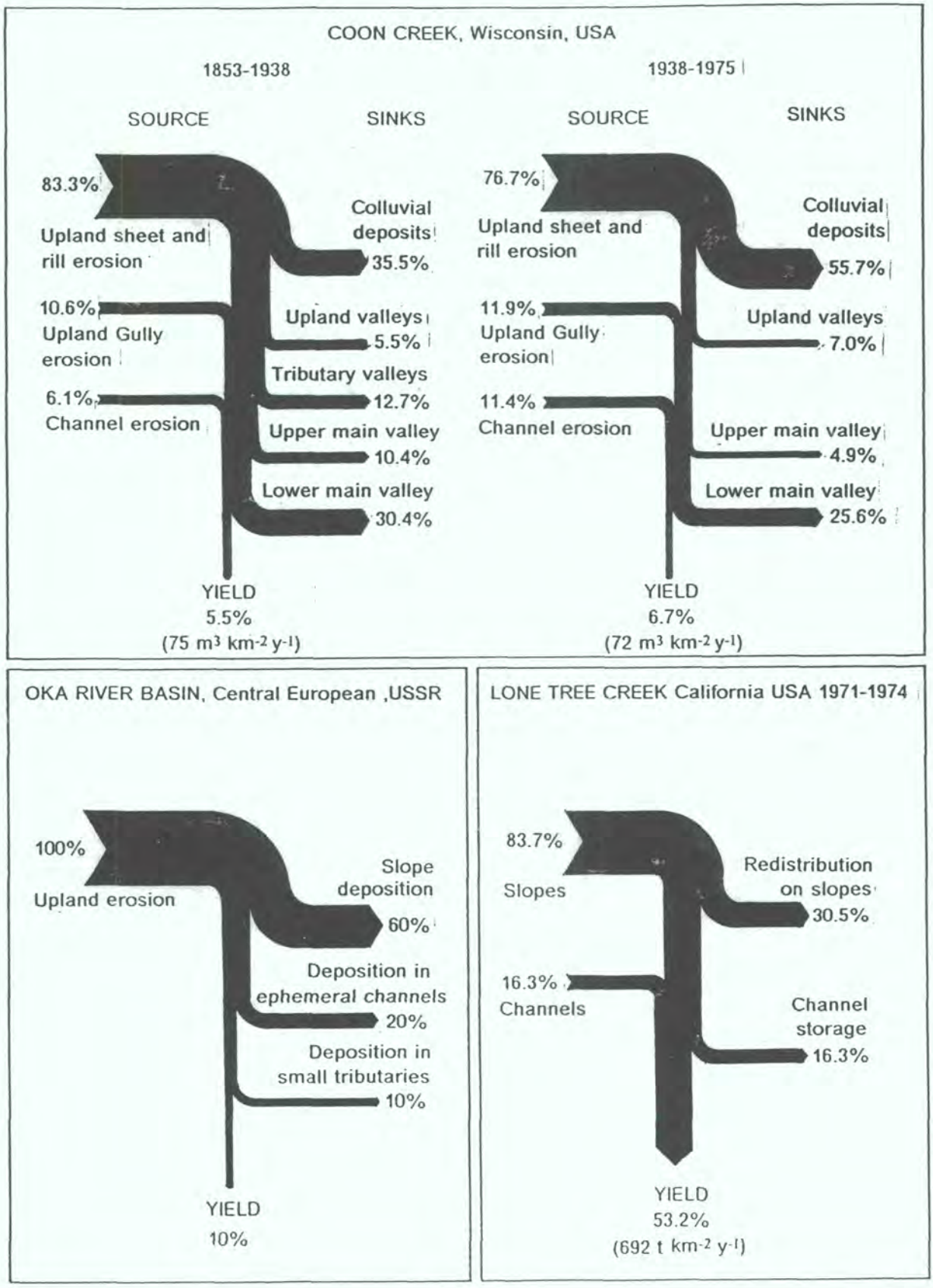

FIGURE 9. Tentative sediment budgets for Coon Creek, Wisconsin (360 km^2), 1853-1938 and 1938-1975; Lone Tree Creek, California (1.74 km^2); and the Oka River, U.S.S.R. Based on data presented by Trimble (1981), Lehre (1982), and Zavlavsky (1979). (Redrawn from Hadley *et al.*, 1985).

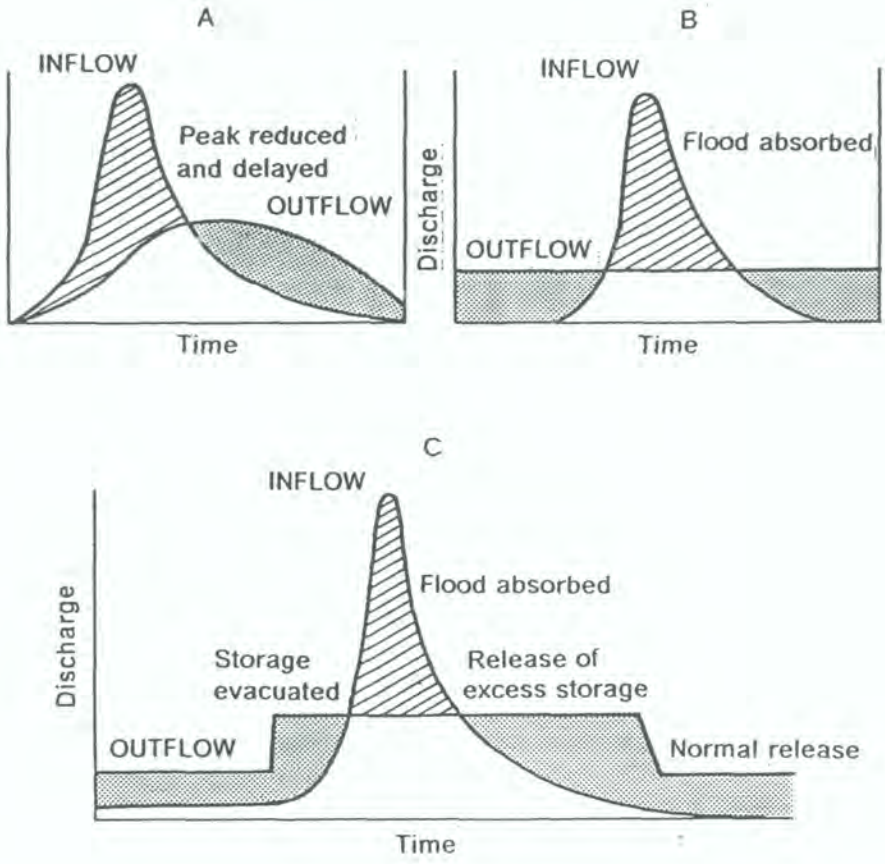

FIGURE 10. Primary types of flow regulation: reservoir attenuation (A), reservoir storage (B), and release manipulation (C). (Redrawn from Petts, 1984).

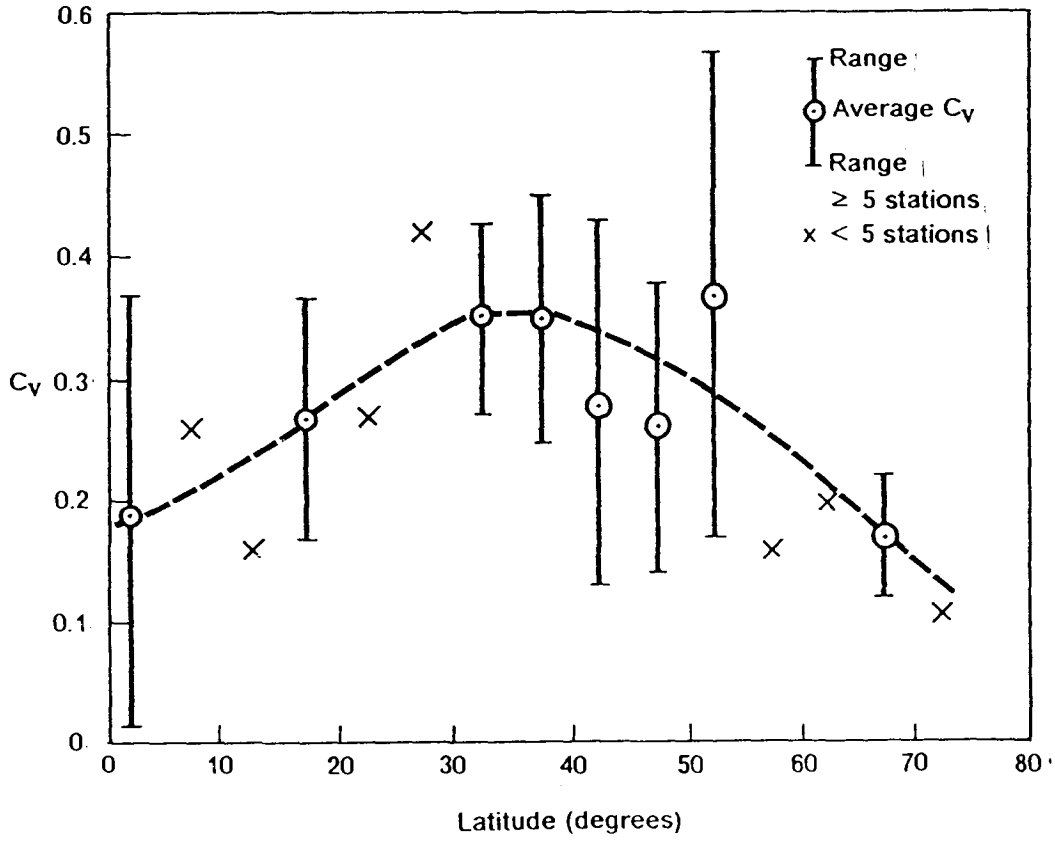

FIGURE 11. Relationship between Coefficient of Variation of Annual Flows and Latitude. (Redrawn from McMahon, 1982).

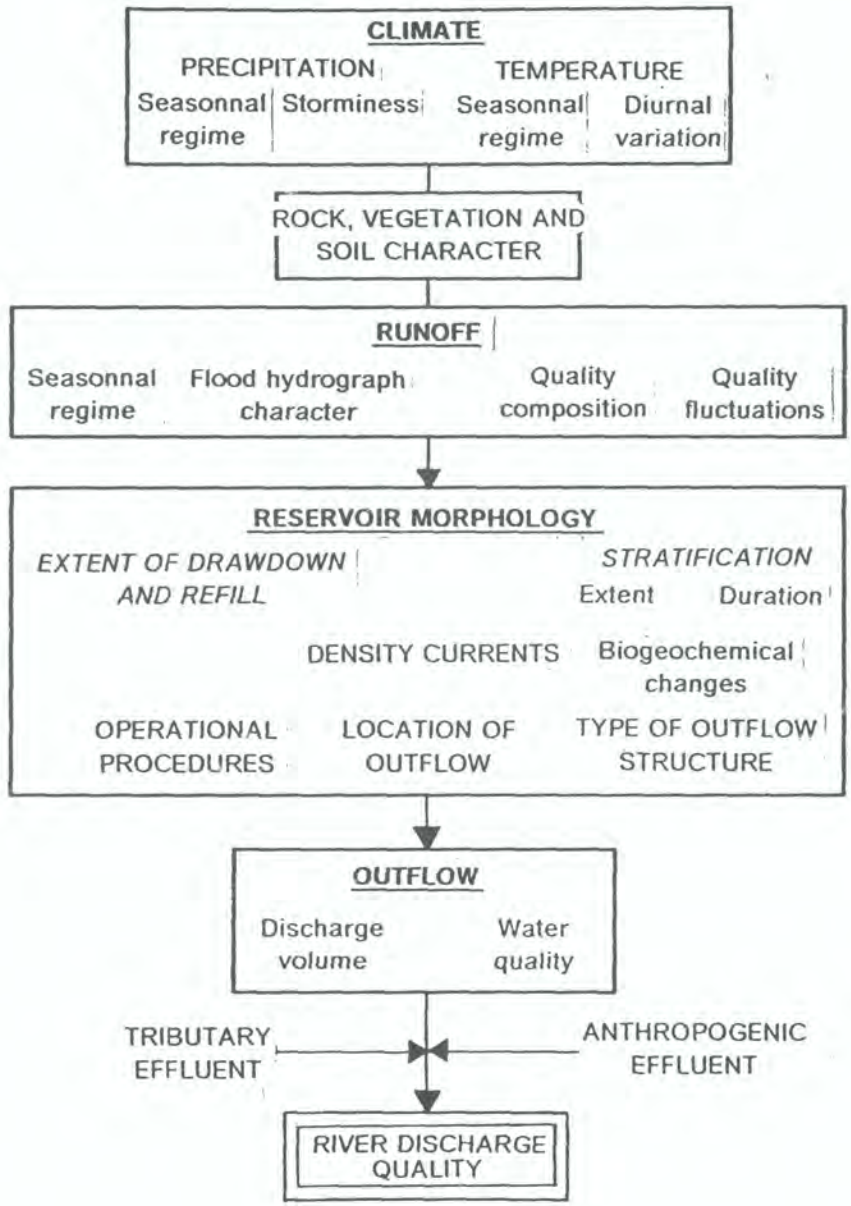

FIGURE 12. Factors affecting the water quality characteristics of dammed rivers. (Redrawn from Petts, 1984).

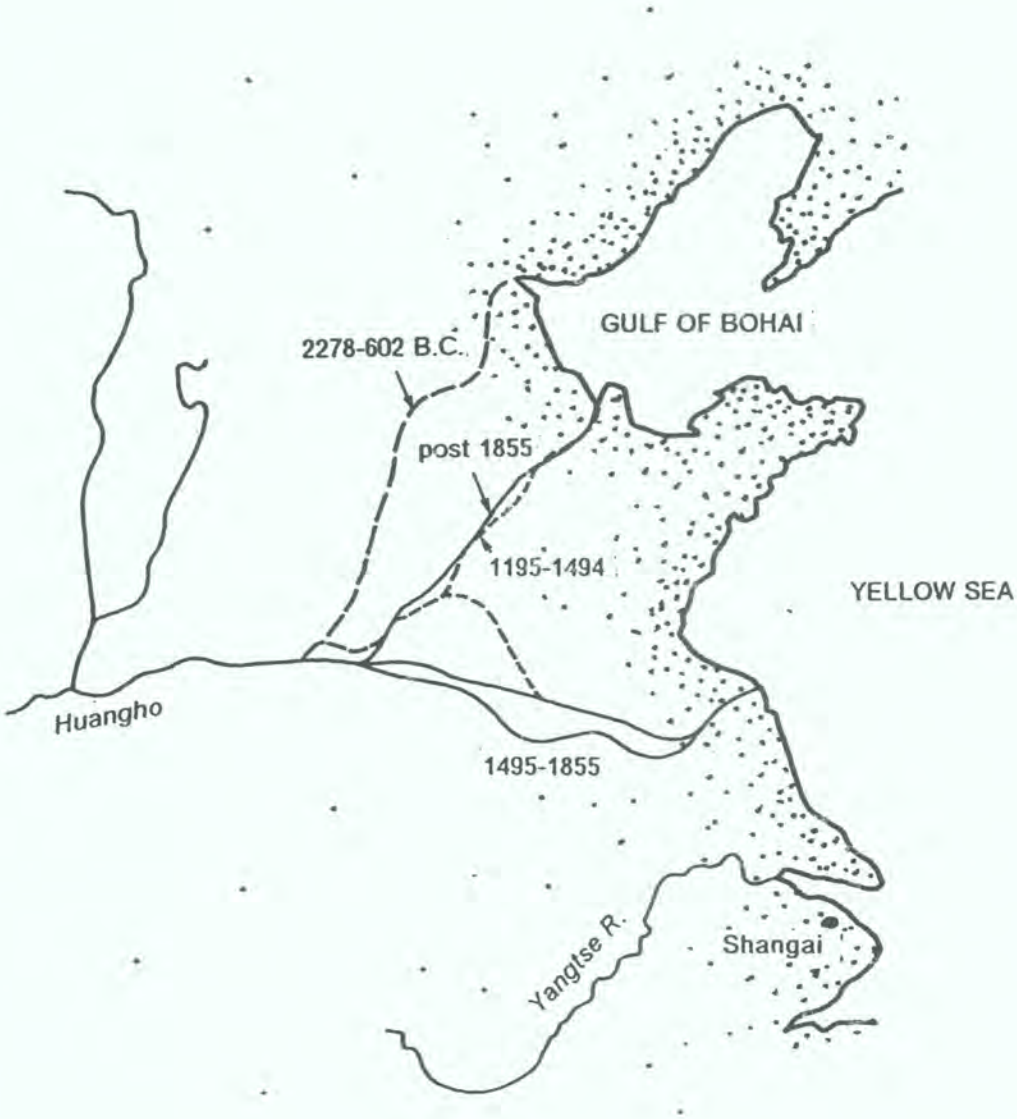

FIGURE 13. Displacements of the Yellow River course in the last 4,000 years. (Redrawn from Milliman, 1984).

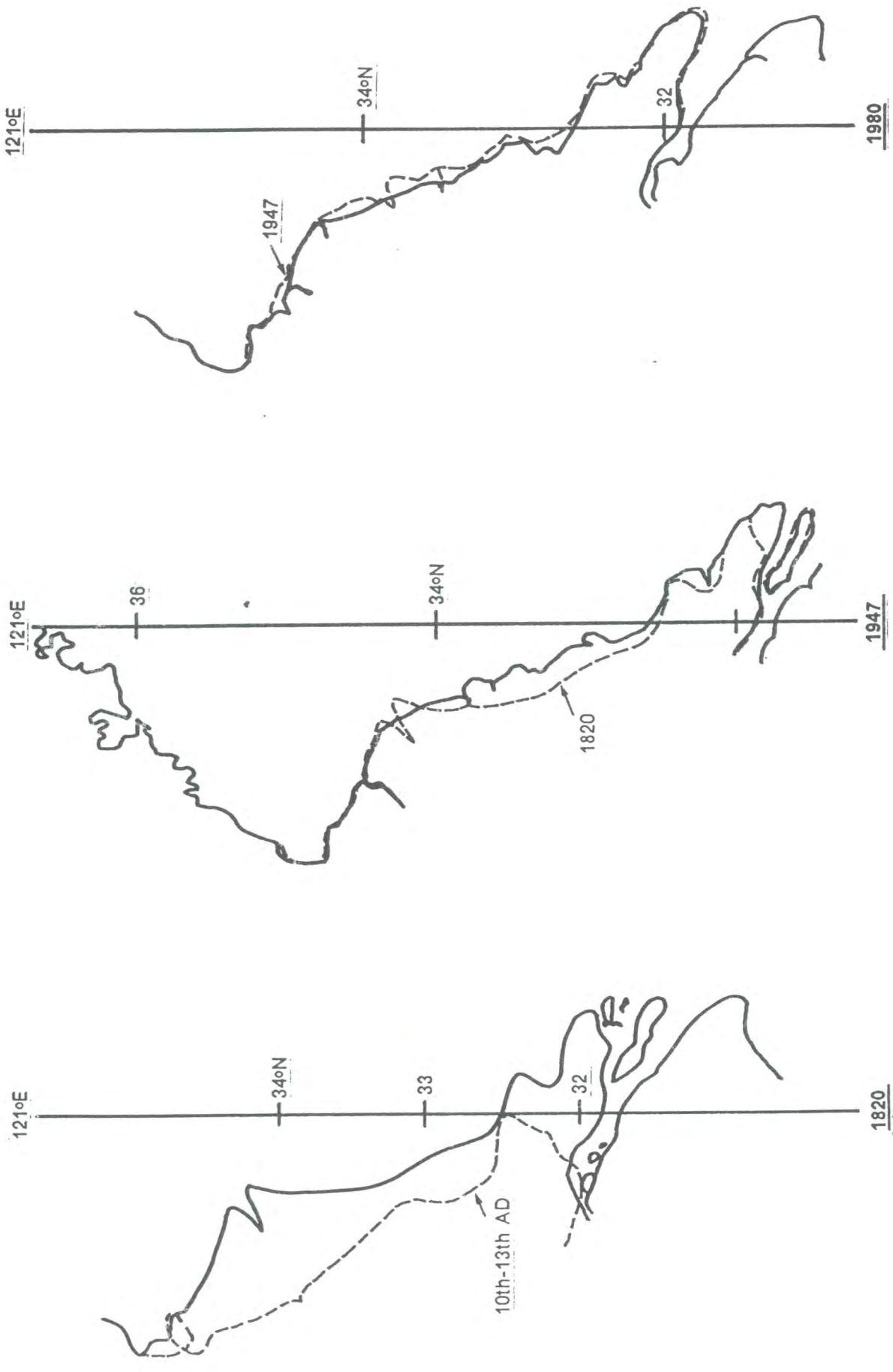

FIGURE 14. The Jiangsu coastline from 1820 to 1980. (Redrawn from Milliman, 1984).

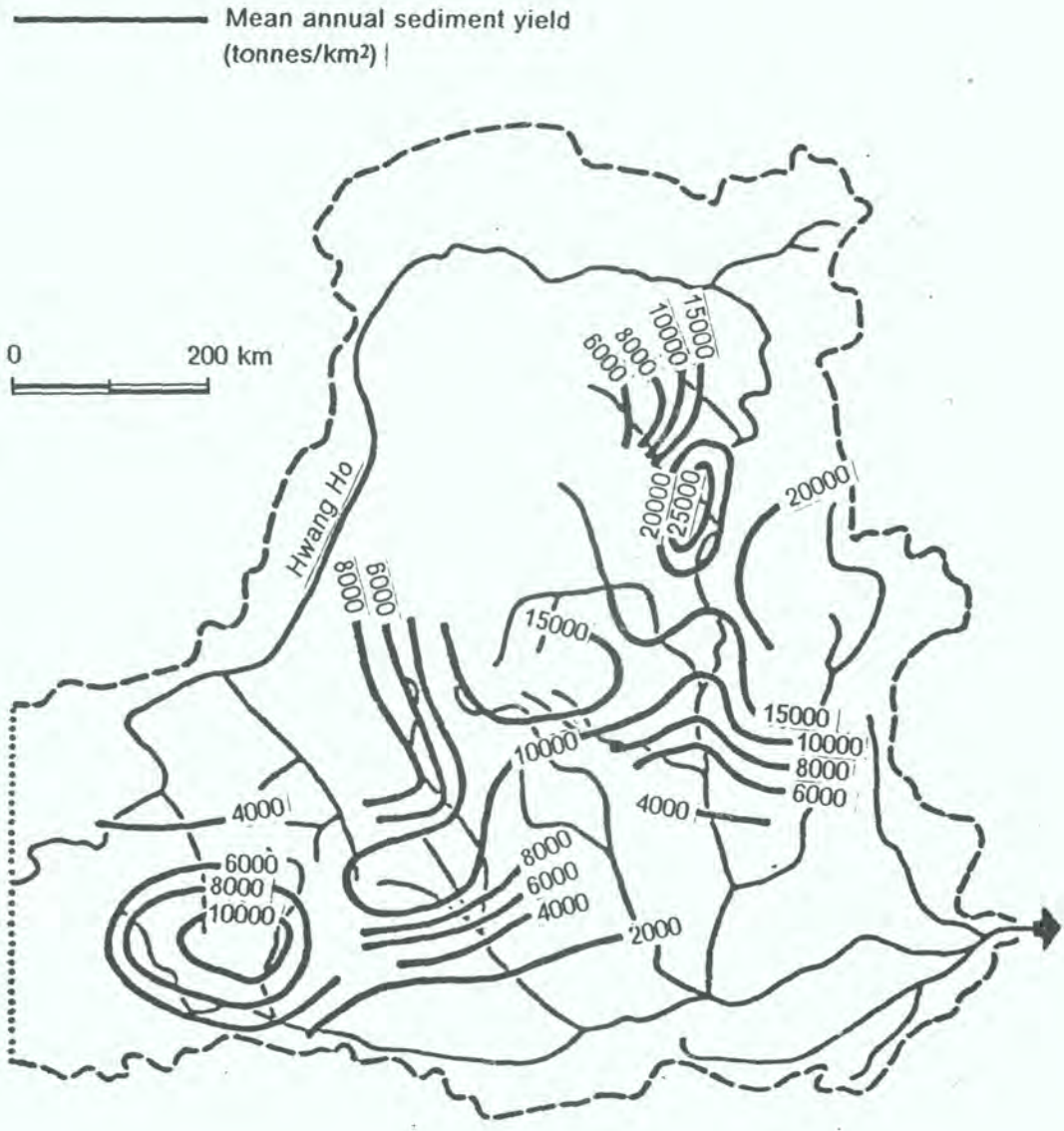

FIGURE 15. Mean annual suspended sediment yields in the loess region of the middle reaches of the Yellow River, People's Republic of China. (Redrawn from Hadley *et al.*, 1985).

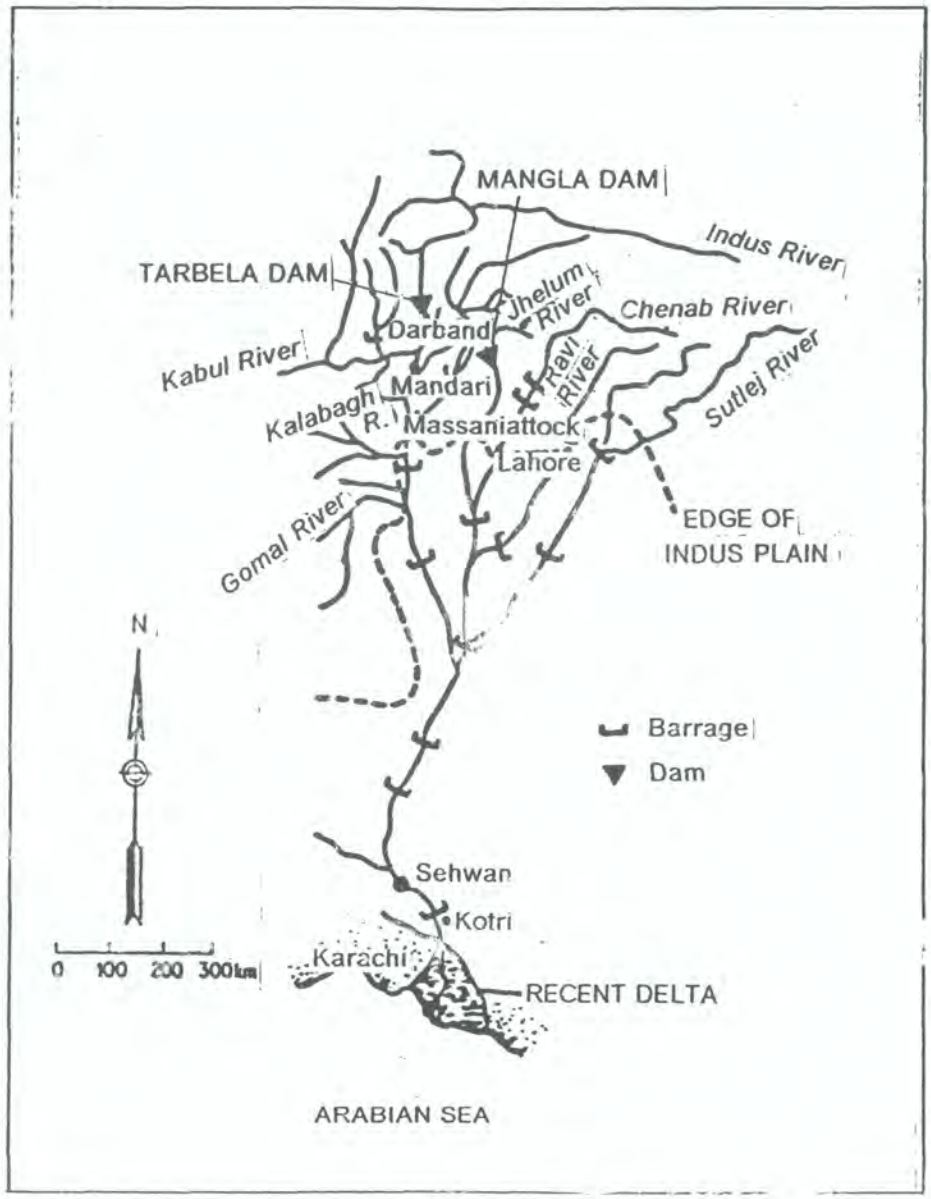

FIGURE 16. The Indus river basin (Redrawn from Milliman *et al.*, 1984).

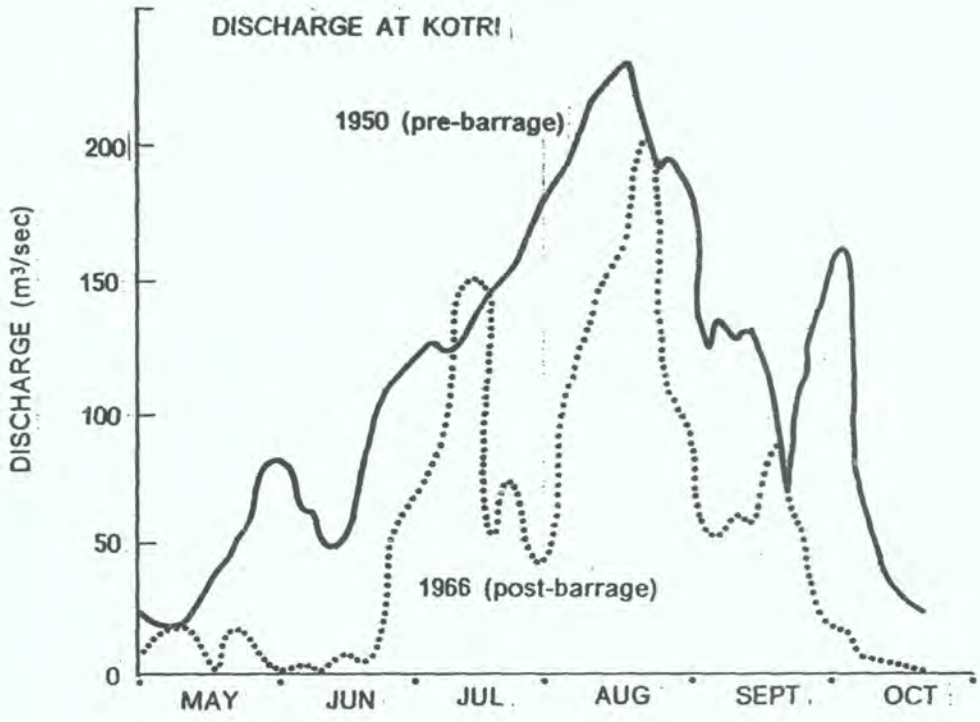

FIGURE 17. Comparison of discharge values at Kotri before and after barrage construction. (Redrawn from Milliman *et al.*, 1984).

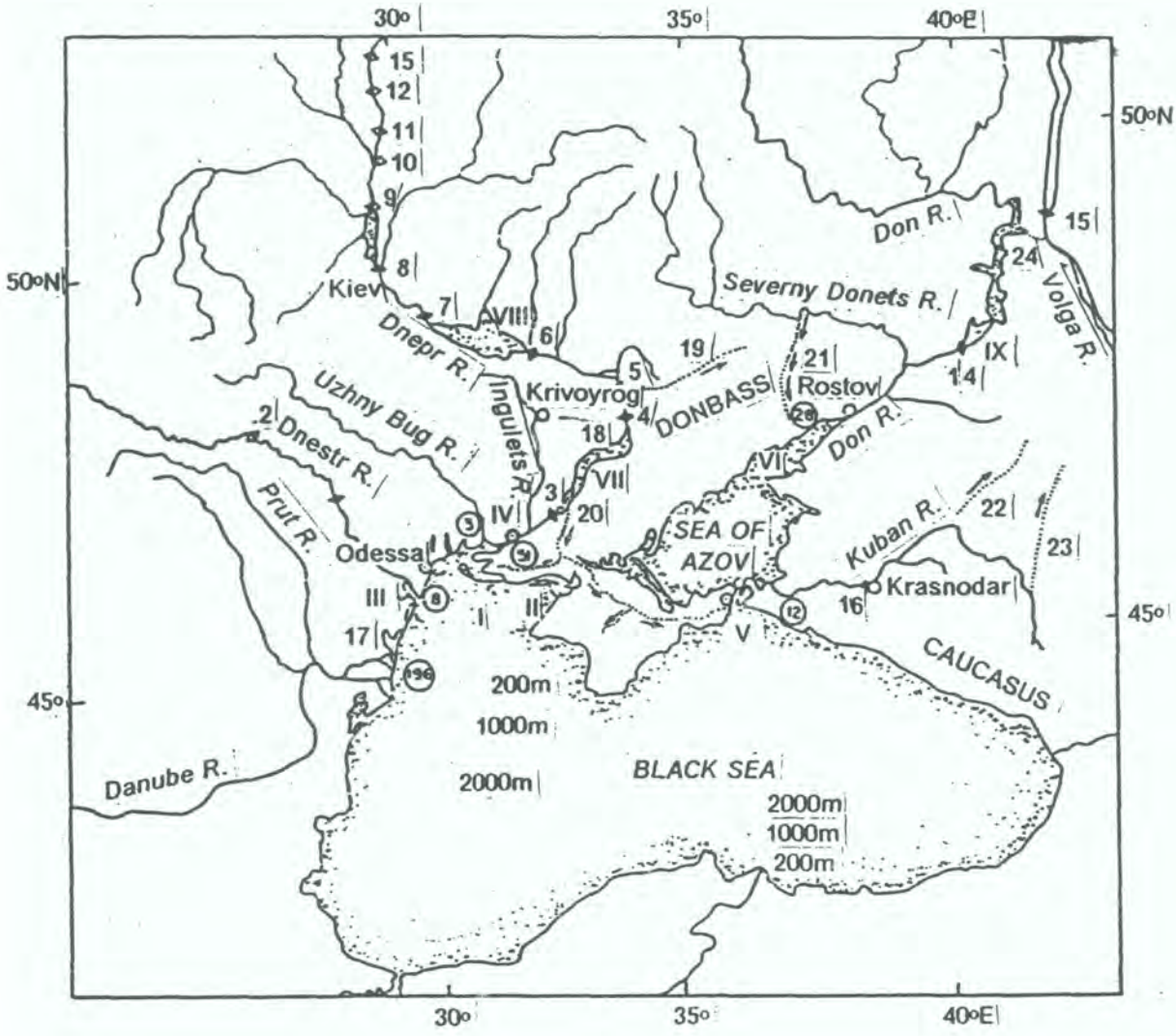

FIGURE 18. Major rivers, estuarine regions, and associated geographical settings of the Black Sea used in the study (depth in metres). (I-IX) *Water Bodies*: (I) Northwestern Black Sea, (II) Karkinitsky Zaliv (Bay), (III) Dnestr Estuary, (IV) Dnepr Estuary, (V) Kerch Strait, (VI) Taganrogsky Zaliv (Bay), (VII) Kakhovskoye Vdkhr (Vodokhranilishtche-Storage Lake), (VIII) Kremenchugskoye Vdkhr, (IX) Tsymlianskoye Vdkhr. (1-8) *Hydropower stations* (1) Dubossary, (2) Mogilev-Podol'sky, (3) Kakhova, (4) Dneproges, (5) Dnesprozderzhinsk, (6) Kremenchug, (7) Kanev, (8) Kiev, (9) Lubech, (10) Rechista, (11) Zhlobin, (12) Viliakhovka, (13) Mogliyow, (14) Tsymliansk, (15) Volgograd, (16) Krasnodar. (17-24) *Irrigation and water supply channels*: (17) Danube-Sasyk, (18) Dnepr-Krivoy Rog, (19) Dnepr-Donbass, (20) Dnepr-Crimea, (21) Severny Donets-Donbass, (22) Nevinnomyssky, (23) Kuban-Kalais, (24) Don-Volga. The arrows indicate the direction of the water transport. Encircled numbers indicate the annual river water discharge in $km^3\ y^{-1}$. (Redrawn from Tolmazin, 1985).

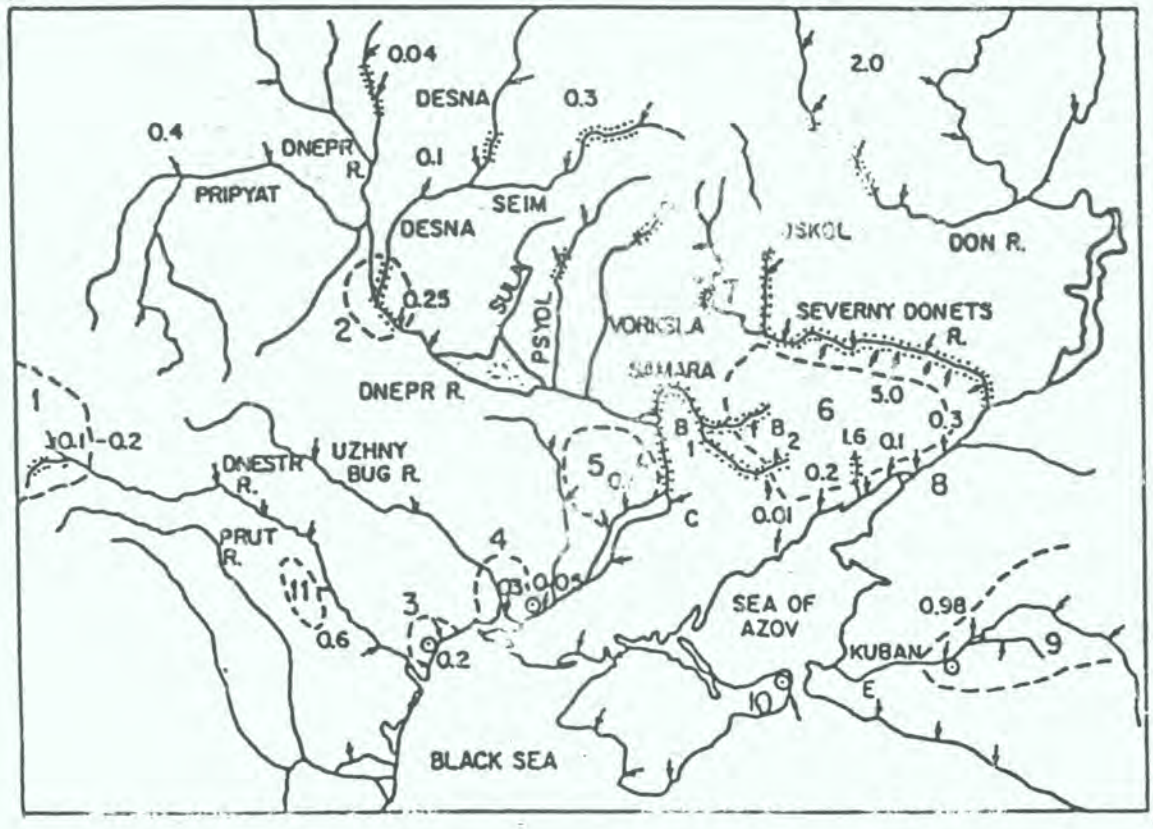

FIGURE 19 Industrial regions (numbers) and known untreated industrial effluents (letters) in the Black Sea and Azov Sea basins: (1) L'vov-Drogobych, (2) Kiev, (3) Odessa, (4) Nikolaev-Kherson, (5) Krivoy Rog, (6) Donbass, (7) Kharkov, (8) Rostovna-Donu, (9) Krasnodar-Maikop, (10) Kerch, (11) Kishinev; (A) Krivoy Rog ore mines, Western (B_1) and Central (B_2) Donbass coal mines, (C) ore mines and processing plant near Dneprorudnyl. Concentrated waste-water outfalls are indicated by the arrows, the double-dots mark the place of reported complete destructions of river habitats. (Redrawn from Tolmazin, 1985).

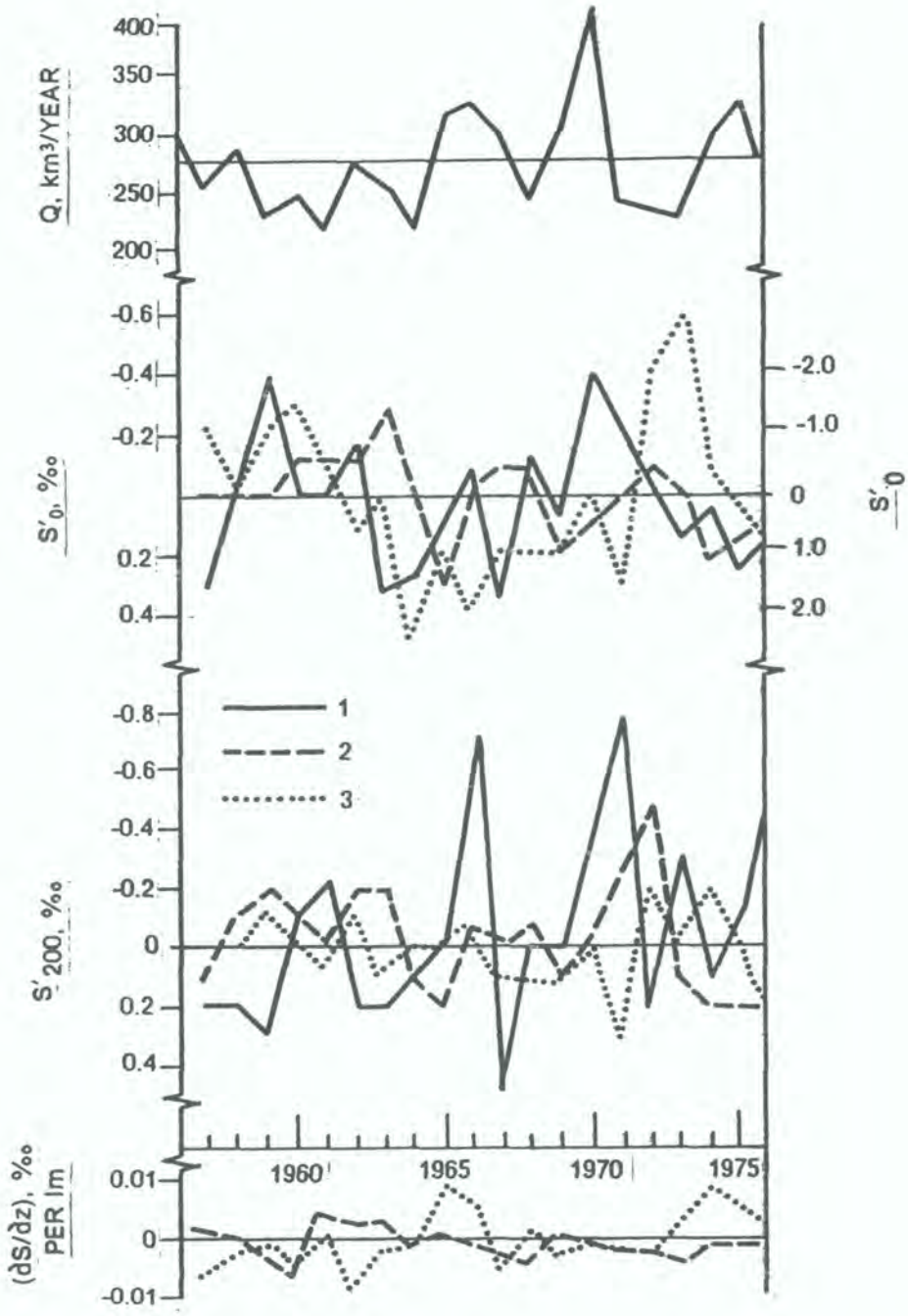

FIGURE 20. Year-to-year variability of river runoff into the NWBS (Q), salinity anomalies (deflects) at the surface (S_0) and depth of 200m (S_{200}) and anomaly of vertical salinity gradient (dS /dz), according to Blatow and others (1980): 1, NWBS; 2, western; and 3, eastern parts of the Black Sea. (Redrawn from Tolmazin, 1985).

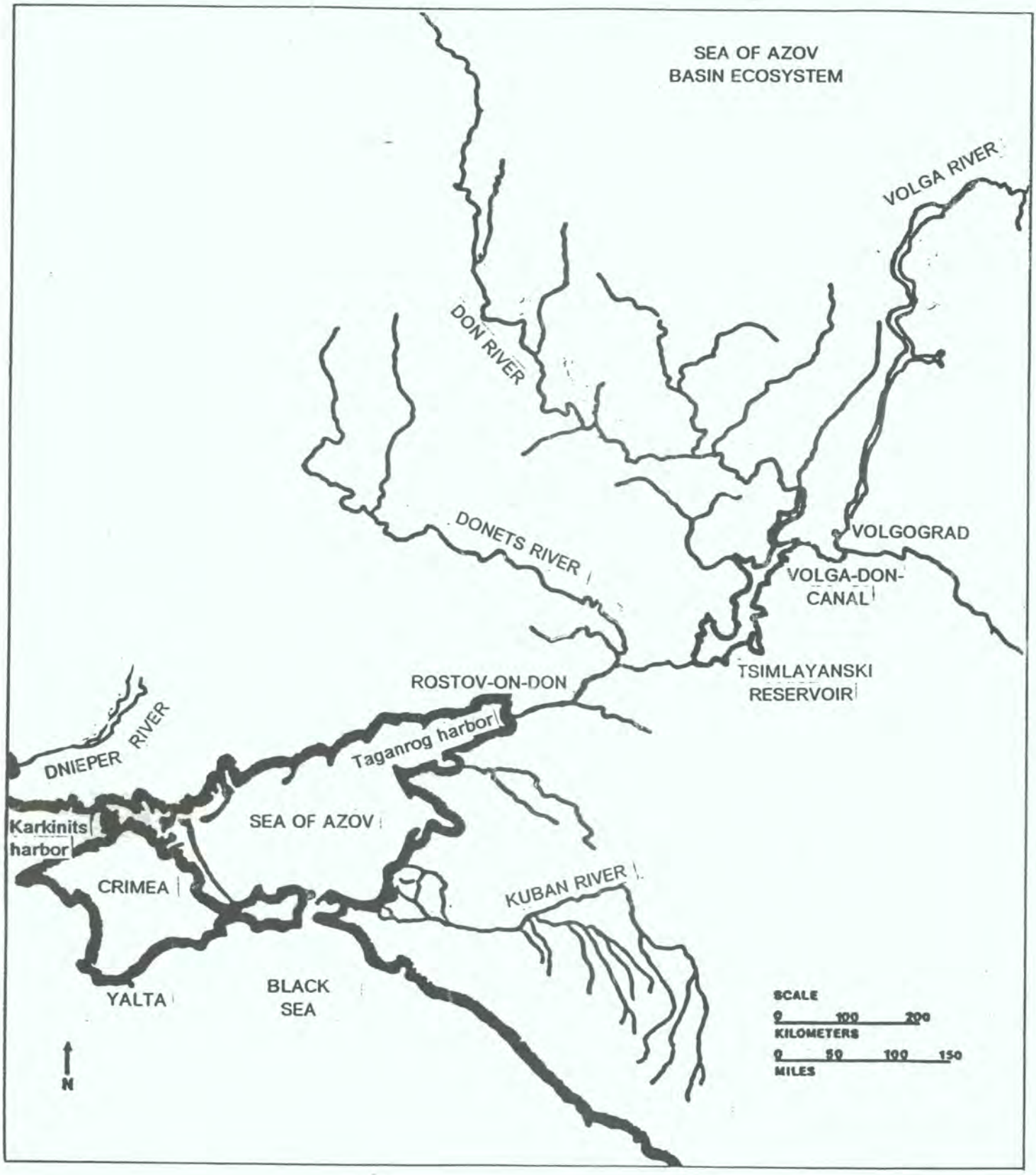

FIGURE 21. Geographical map of the Sea of Azov and its basin watershed in the southern U.S.S.R. (Rozengurt *et al.*, 1985).

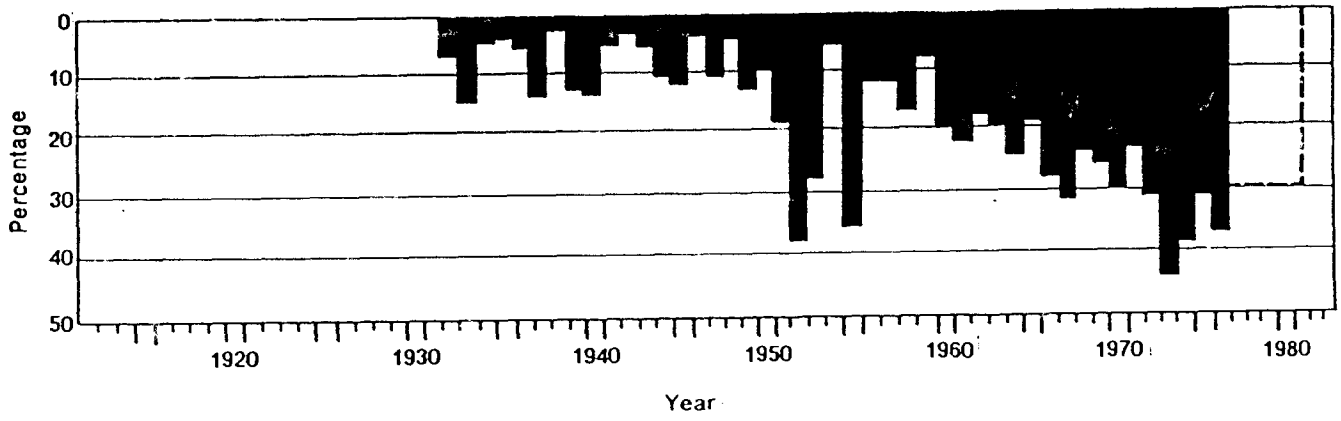

FIGURE 22. Freshwater diversions from the Don-Kuban river system expressed as the percentage of the annual natural river inflow to the Sea of Azov. (Redrawn from Rozengurt *et al.*, 1985).

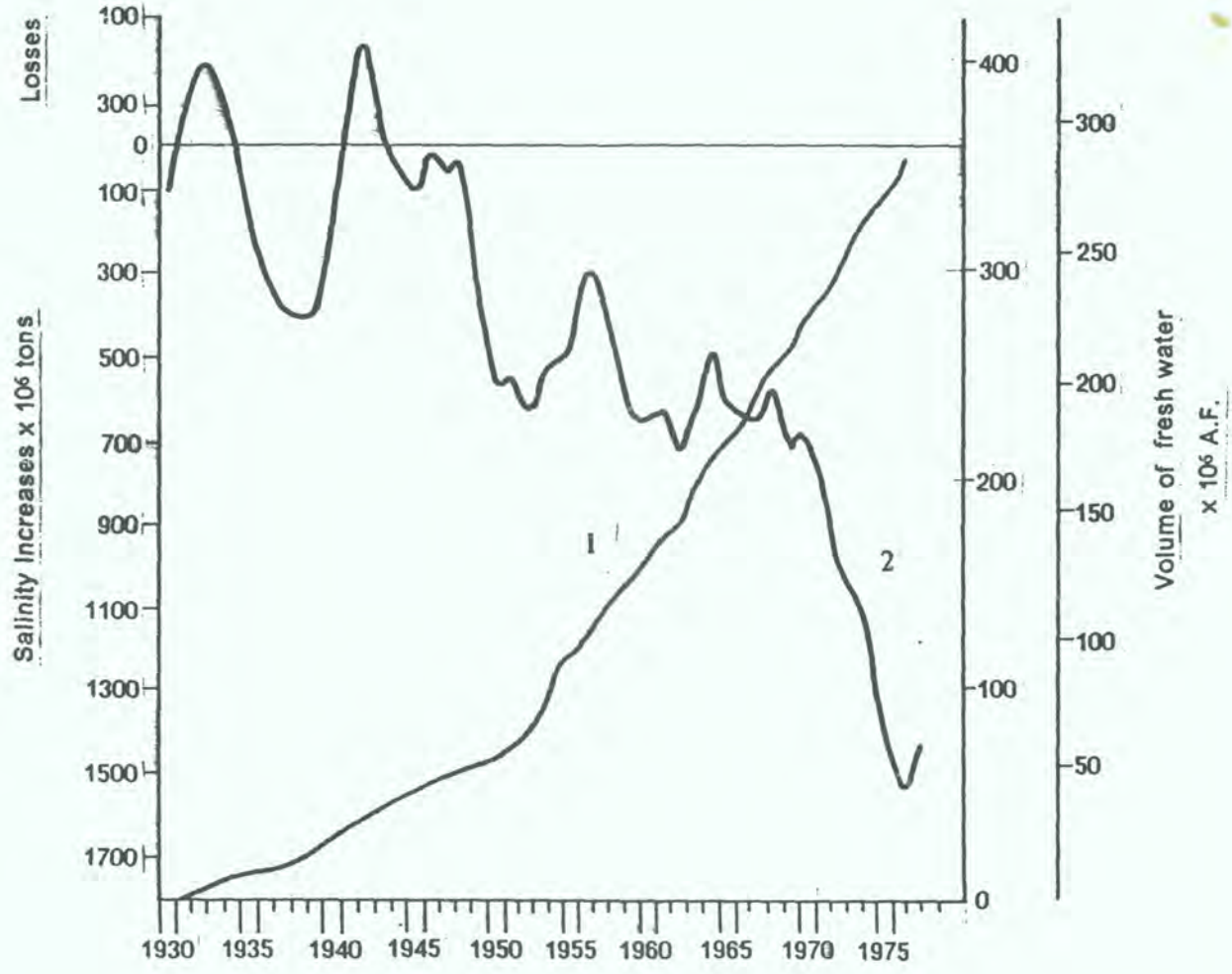

FIGURE 23. Cumulative curves: (1) freshwater losses and (2) accumulation of salinity in the Sea of Azov. (Redrawn from Rozengurt *et al.*, 1985).

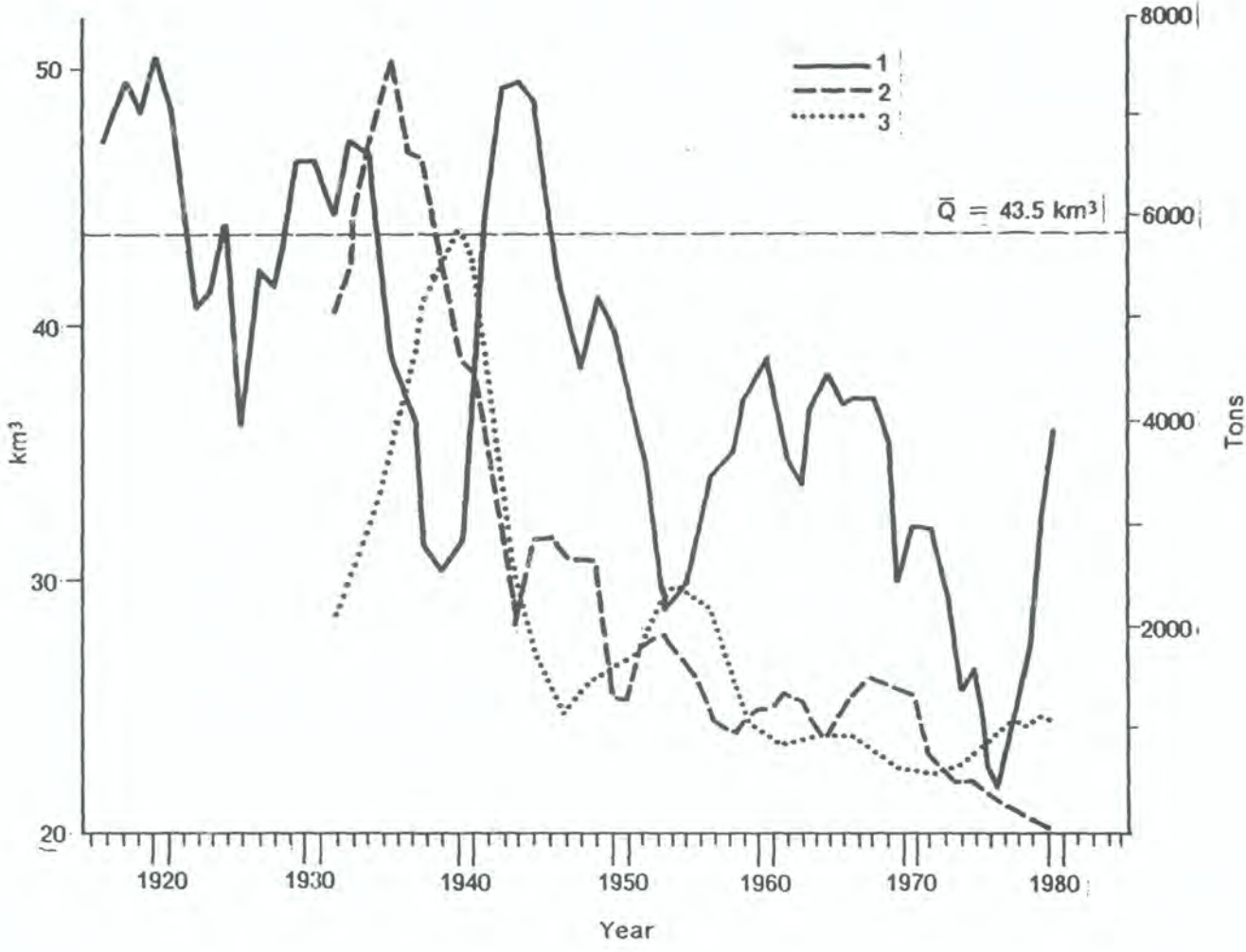

FIGURE 24. Fluctuations in the 5-year running average of (1) regulated combined river inflow to the Sea of Azov and commercial catch of anadromous fish (2) Russian sturgeon (*Acipenser guldenstadti*), beluga (*Huso huso* L.) and sevruga (*Acipenser stellatus* Palas), (3) Kerch (Black Sea) shad (*Alosa kessleri pontica*). (Redrawn from Rozengurt *et al.*, 1985).

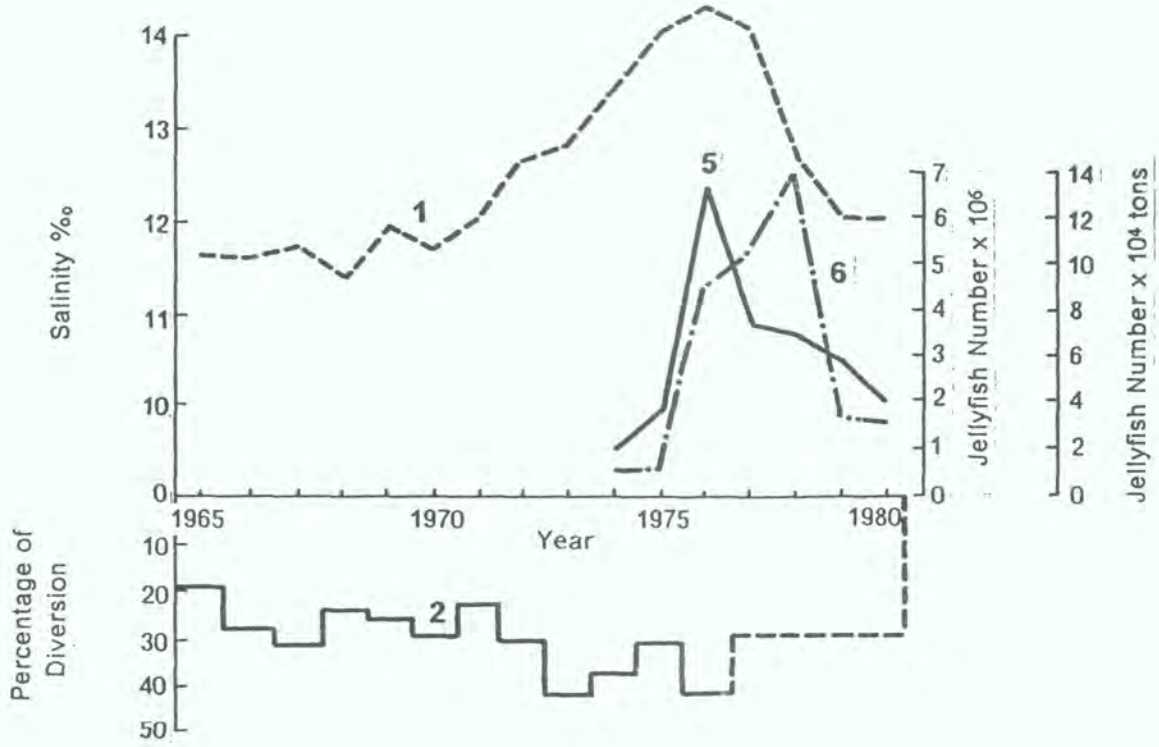

FIGURE 25. Population explosion of the marine jellyfish (*Aurelia*) inside the formerly brackish Sea of Azov as a result of increased freshwater diversions and the resulting rise in salinity concentrations. (1) Annual average salinity, (2) combined average annual freshwater diversions expressed as a percentage of the natural runoff to the Sea of Azov, (5) raw weight of jellyfish (*Aurelia aurita* and *Rhizostoma*) in millions of tons, (6) combined number of jellyfish in billions. (Redrawn from Rozengurt *et al.*, 1985).

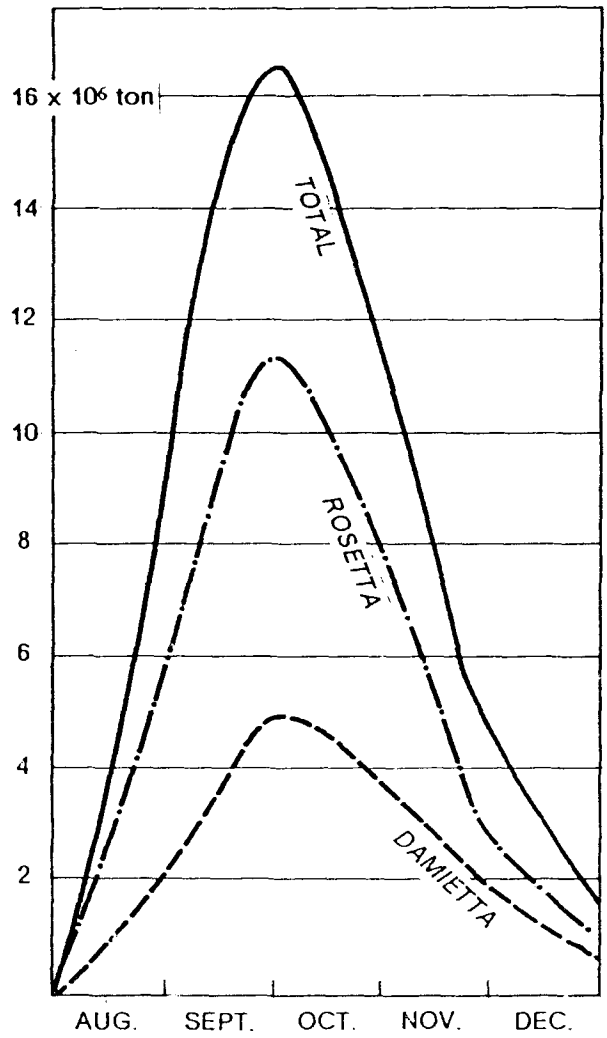

FIGURE 26. Average monthly discharge of Nile water for the five years 1959-1963. (Redrawn from Halim, 1967).

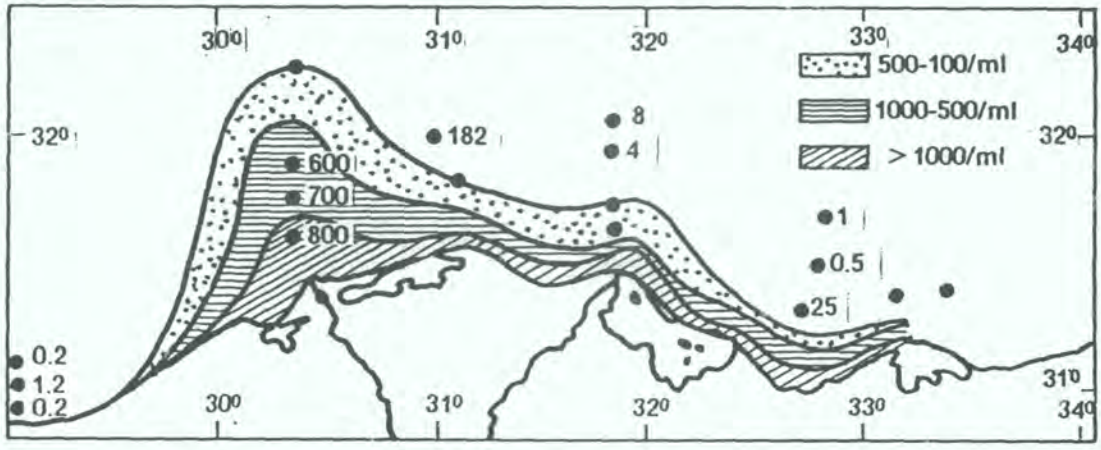

FIGURE 27. The "Nile bloom", following the flood in October 1964 (redrawn from Halim *et al.*, 1967): *Nitzschia seriata* 50-75 %, *Asterionella japonica, Chaetoceros affinis.*

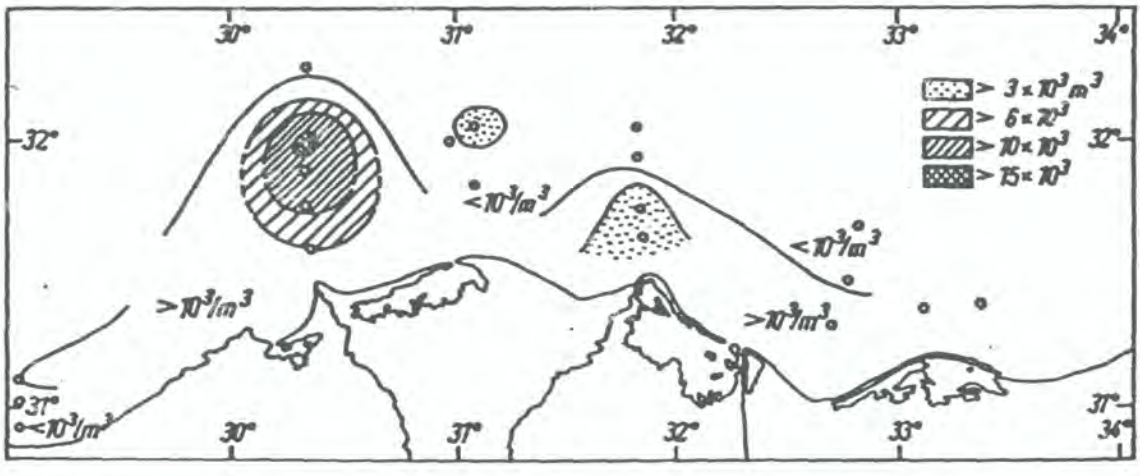

FIGURE 28. Distribution of zooplankton in the upper 10 m layer during the Nile flood season (October 1964). The dense patches fall in the intermediate zone of the bloom. (Samples from vertical 10 m phytoplankton net hauls). (Redrawn from Halim *et al.*, 1967).

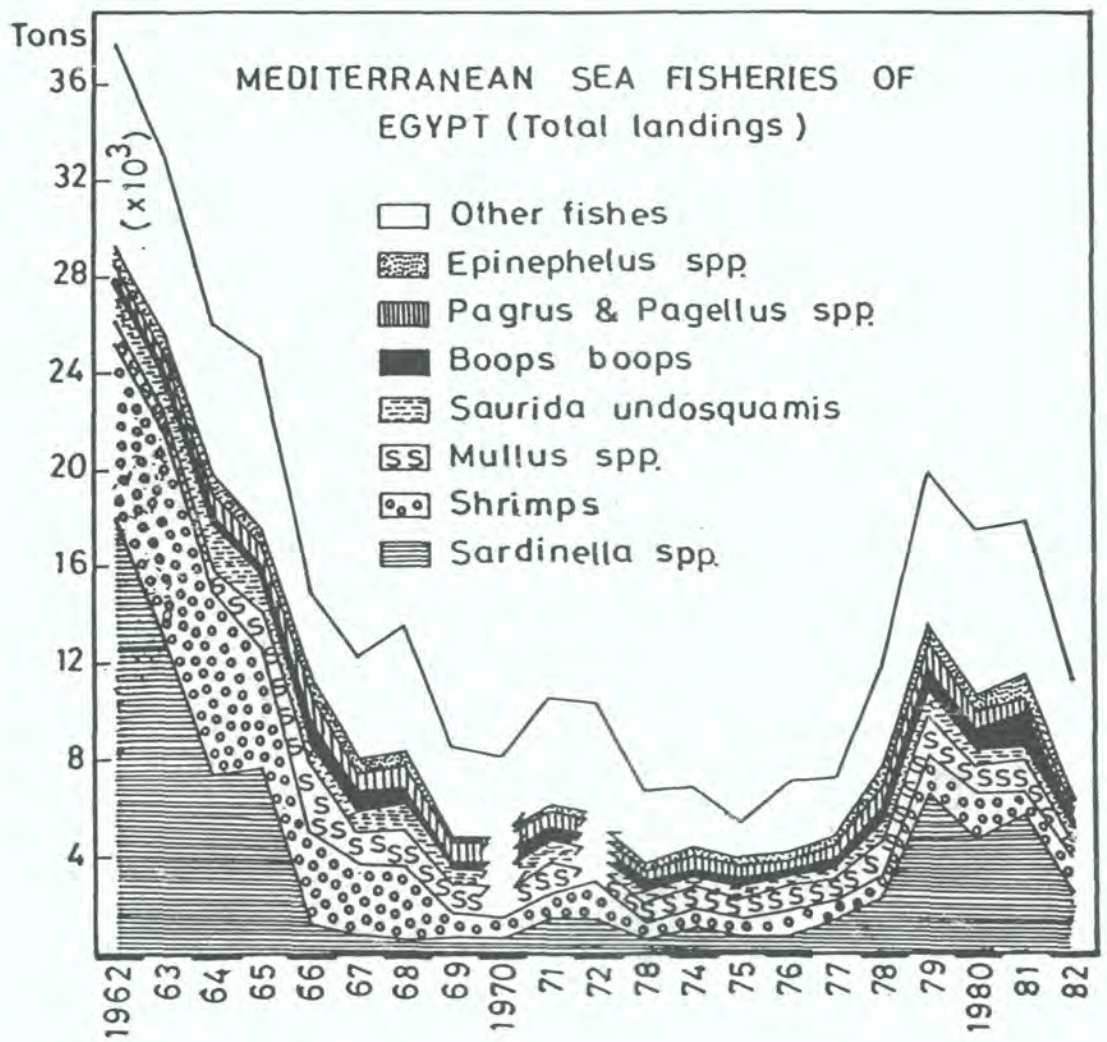

FIGURE 29. Total fish and shrimp landings from Egyptian Mediterranean waters from 1962 to 1982 (redrawn from Sh. K. Guerguess, pers. comm.).

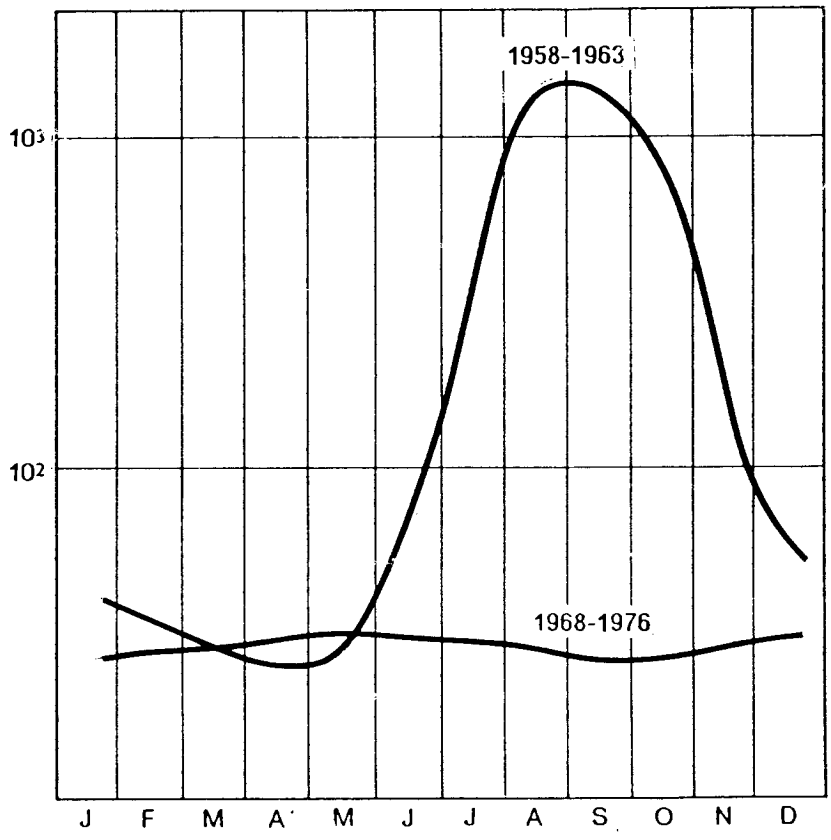

FIGURE 30. Average monthly suspended load of the Nile, downstream of Aswan, pre- (1958-63) and post-impoundment (1968-76), in ppm.

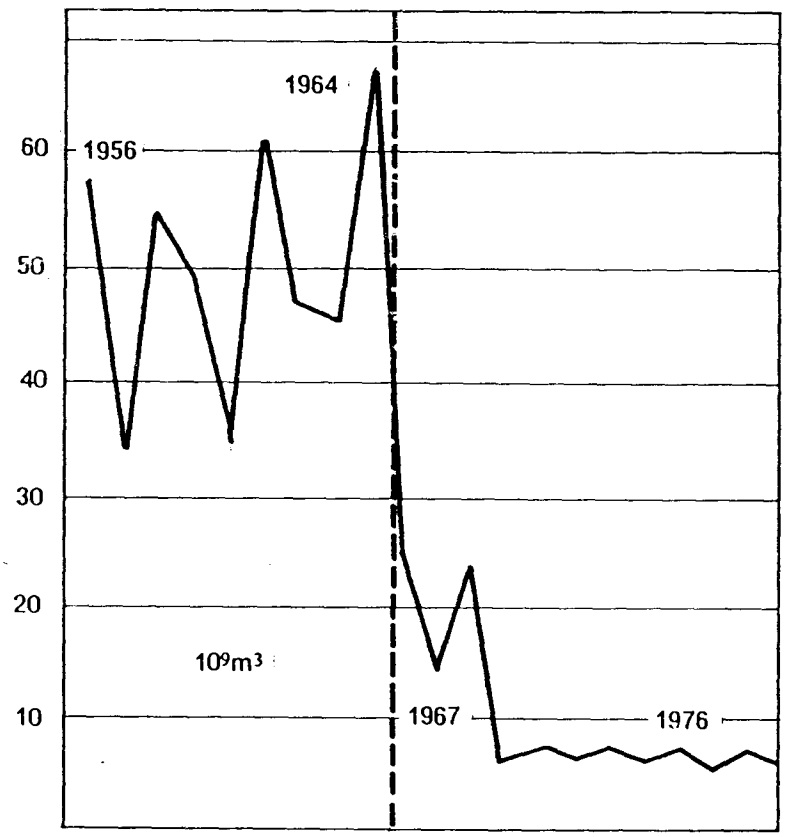

FIGURE 31. Total water intake of the two Nile branches, pre- (1956-65) and post-impoundment (1965-76).

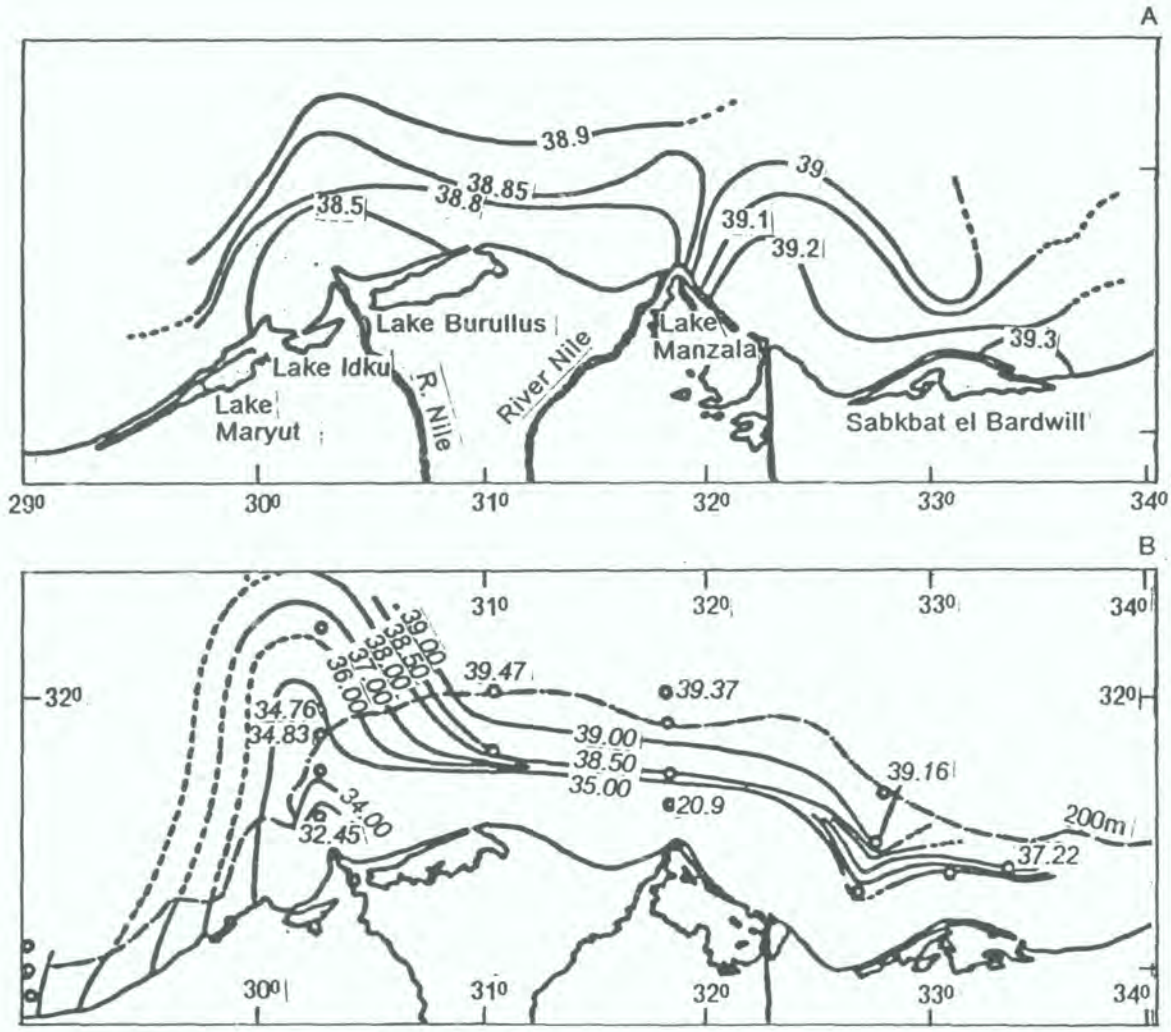

FIGURE 32. Surface isohalines off the Nile delta before and after the High Aswan Dam.

A. November 1982 (redrawn from Mostafa, 1985).

B. October 1964 (redrawn from Halim *et al.*, 1967).

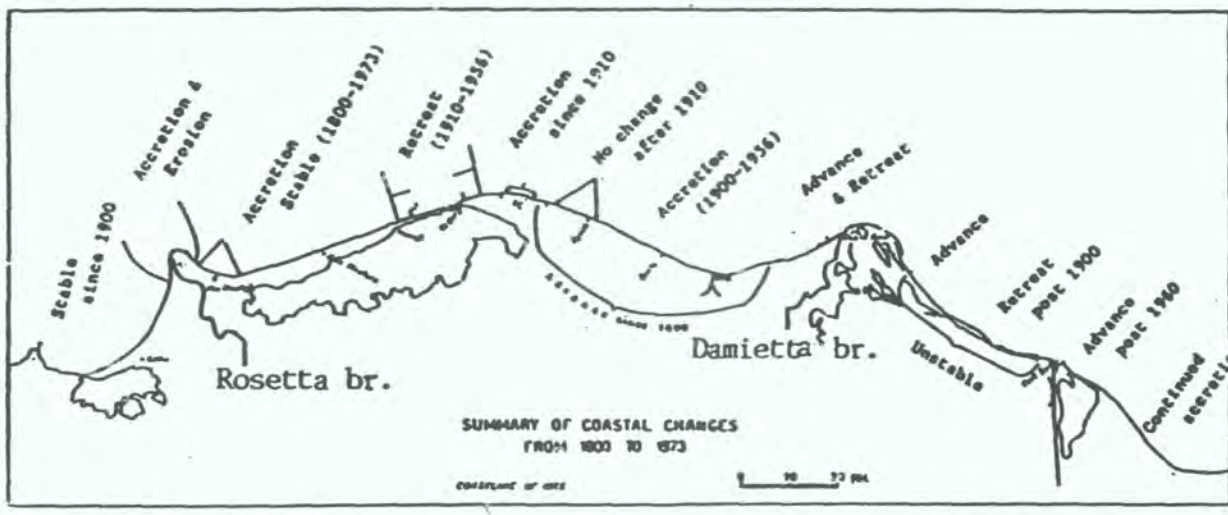

FIGURE 33. Nile Delta coastline and summary of coastal changes. (Redrawn from Anonymous, 1978).

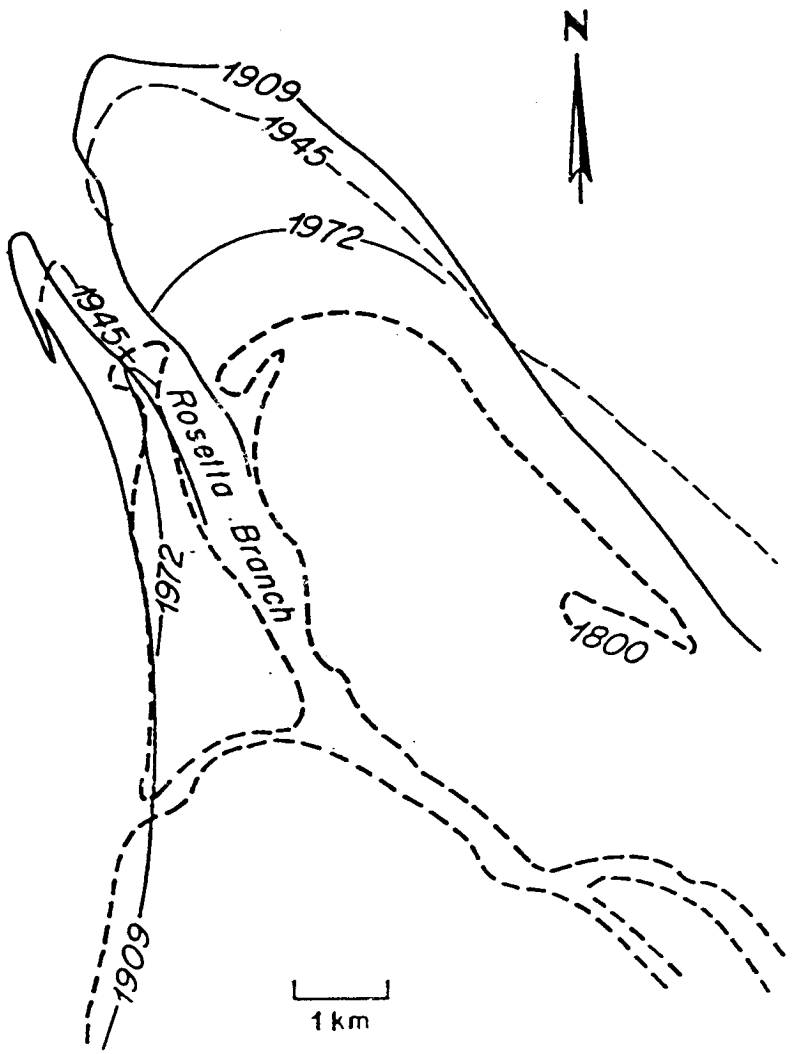

FIGURE 34. Accretions and retreats of the Rosetta promontory since 1800. (Redrawn from Anomymous, 1978).

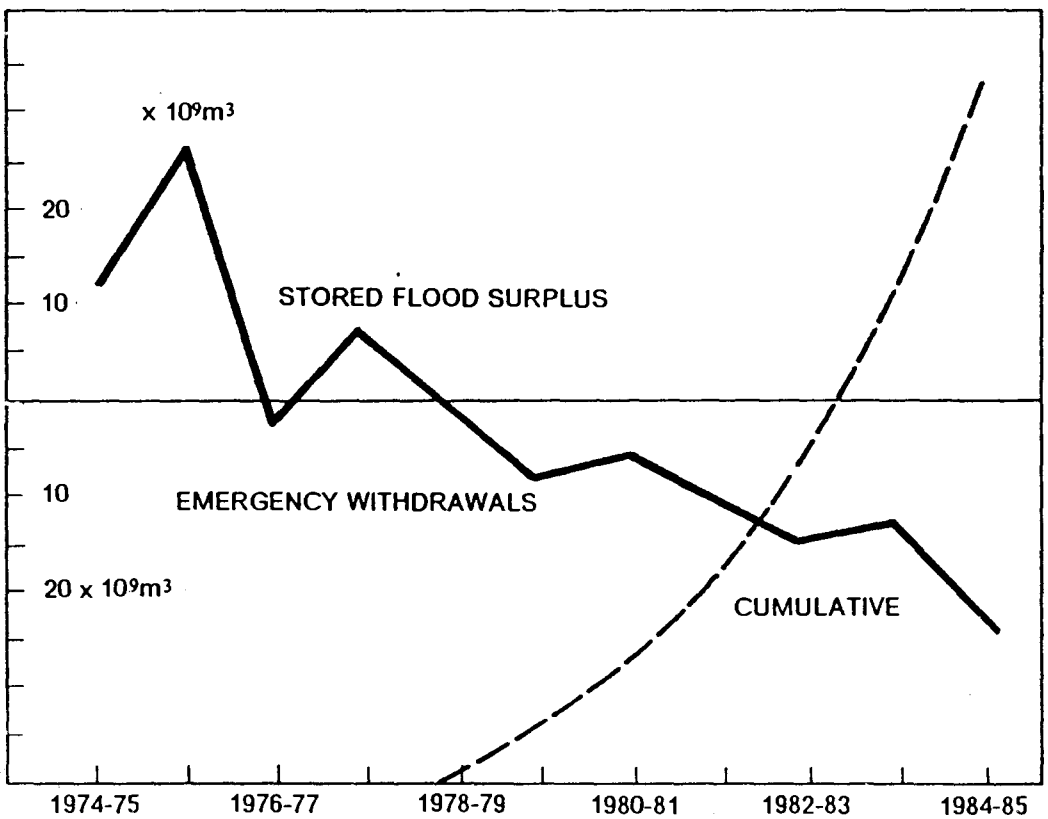

FIGURE 35. Emergency withdrawals from Lake Nasser reservoirs to compensate for low flood inputs, from 1976-77 to 1984-85. (Redrawn from Kenawy, 1985).

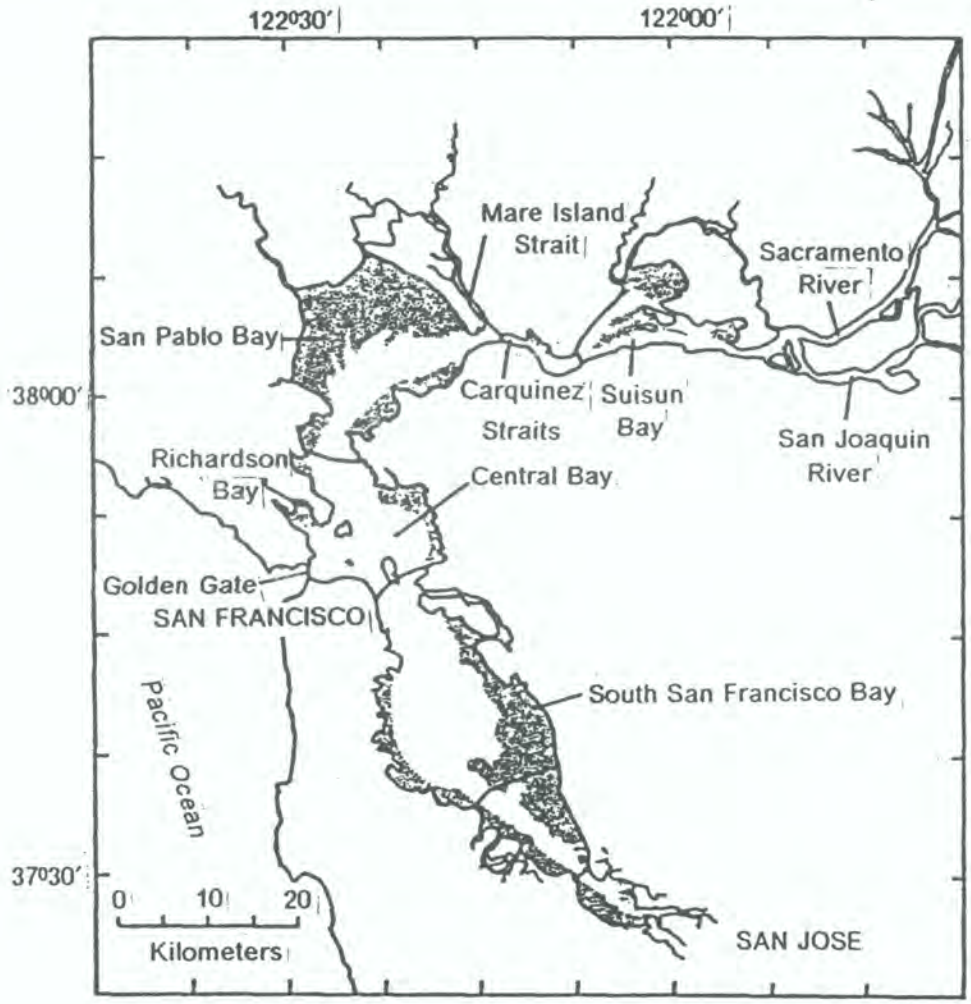

FIGURE 36. San Francisco Bay and western Delta. (Adapted from Luoma and Cloern, Pacific Division AAAS, San Francisco, 1982, quoted by Davoren and Ayres, 1984).

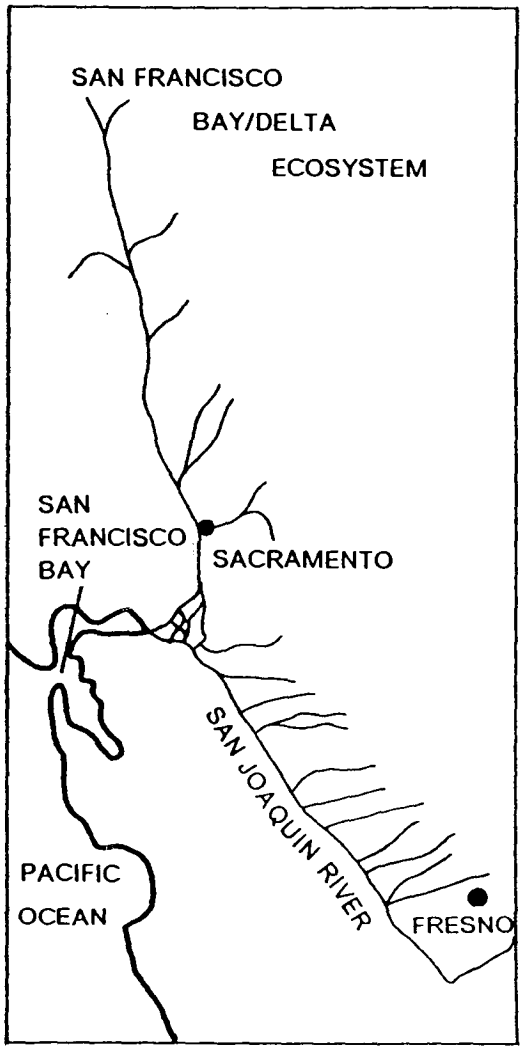

FIGURE 37. The Sacramento- San Joaquin river basin. (Redrawn from Rozengurt *et al.*, 1985).

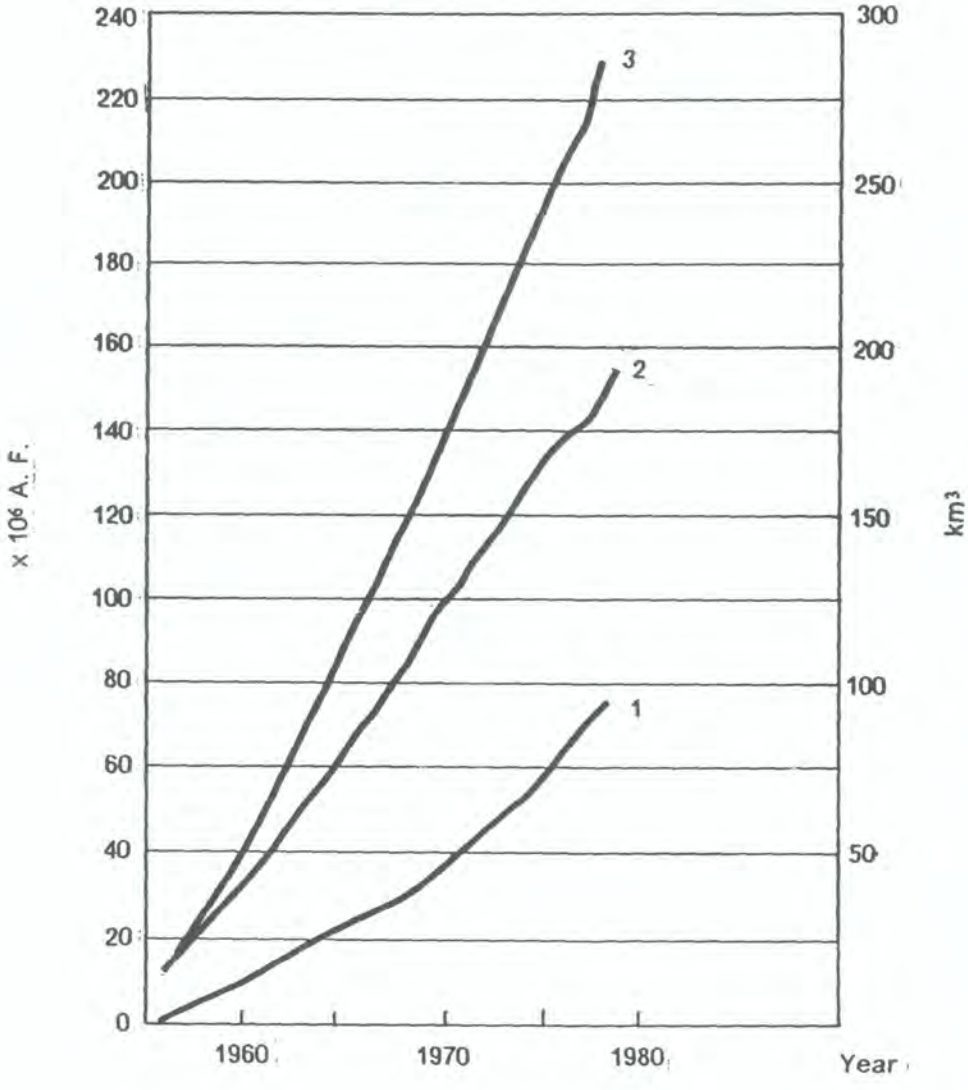

FIGURE 38. Cumulative quantity of freshwater diverted (and so lost to the Bay/Delta system) from withdrawals (1) within the Delta, (2) upstream of the Delta and (3) combined. (Redrawn from Rozengurt *et al.*, 1985).

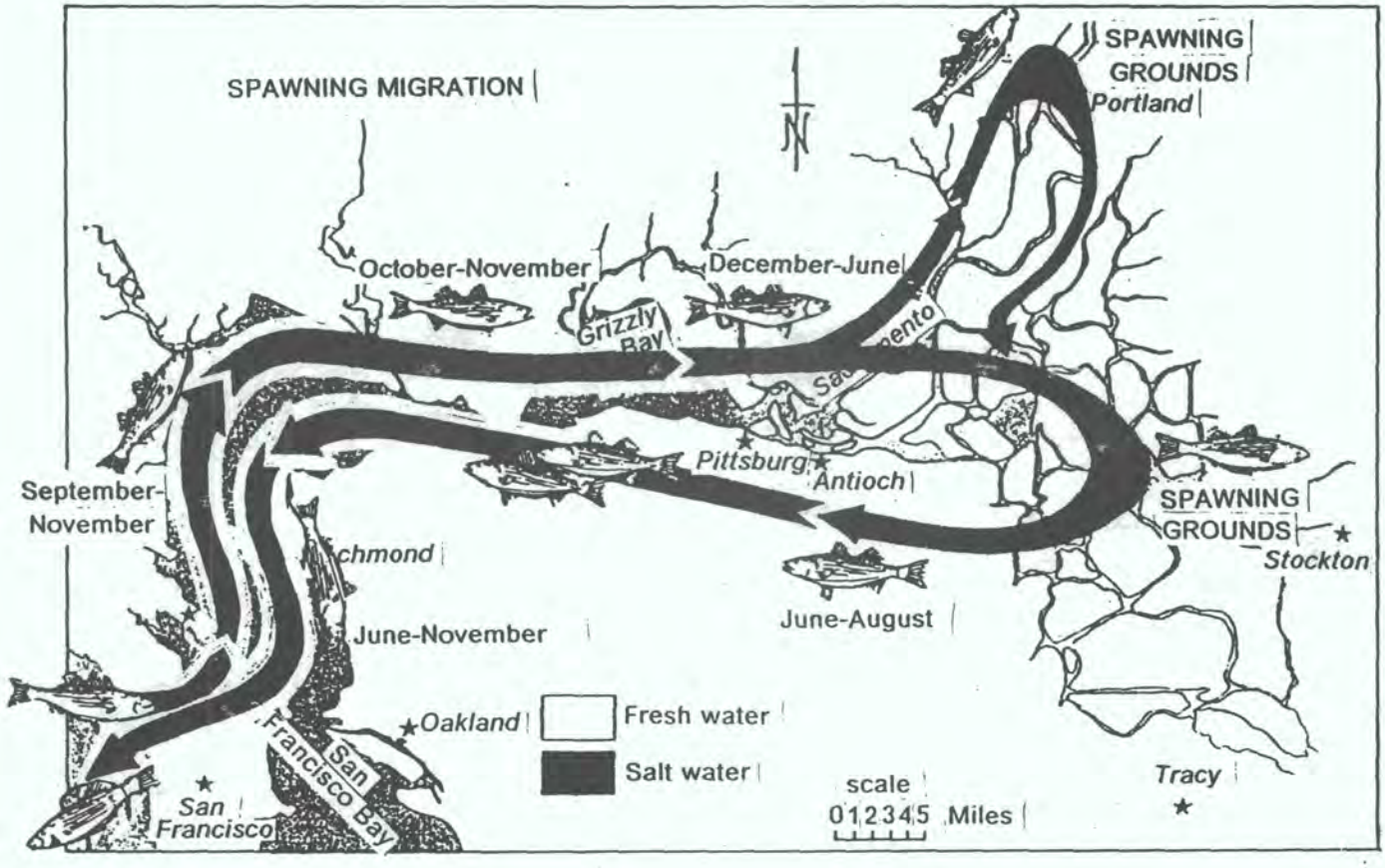

FIGURE 39. Spawning patch of striped bass (*Morone saxatalis*). (Redrawn from Davoren and Ayres, 1984).

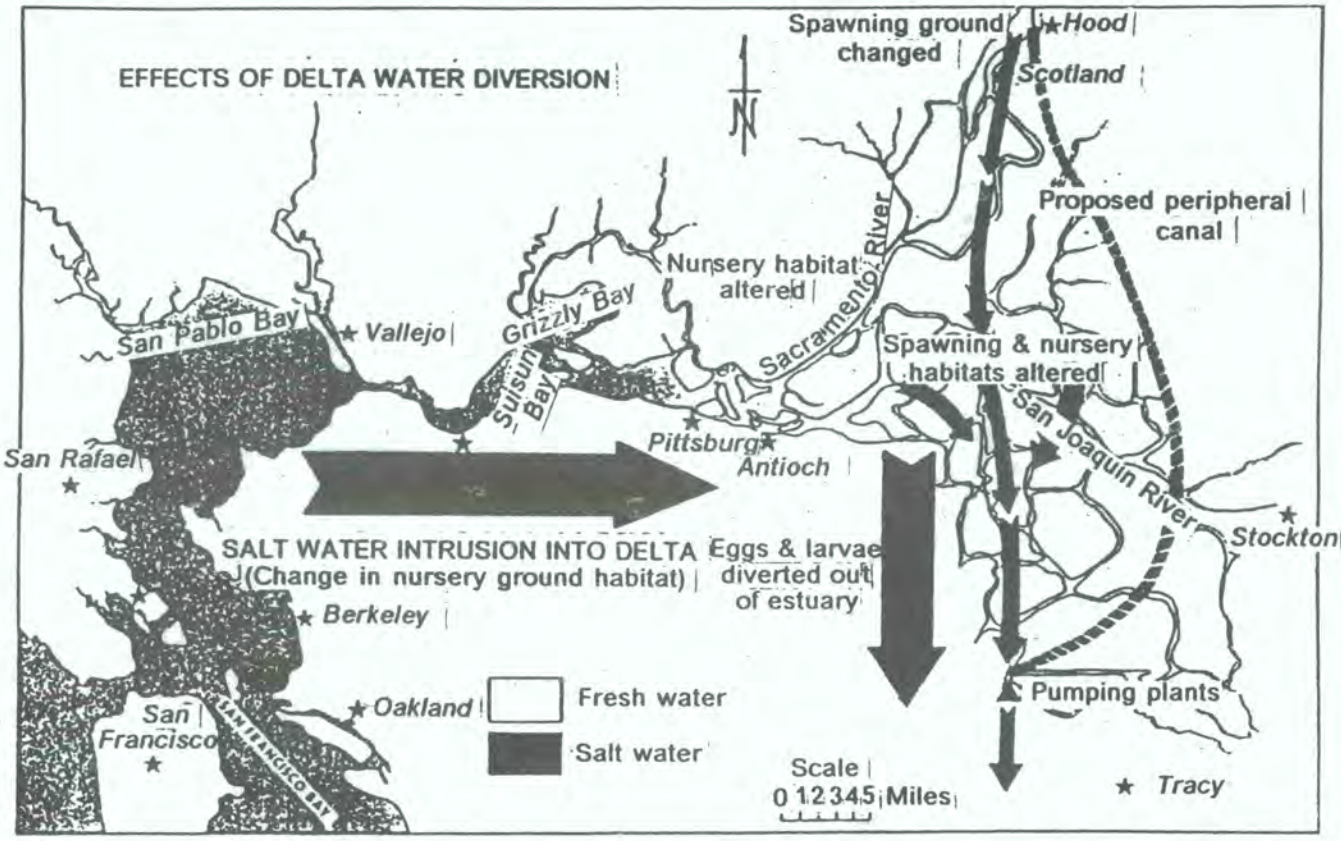

FIGURE 40. Pump diversions' effects on striped bass and other species. (Redrawn from Davoren and Ayres, 1984).

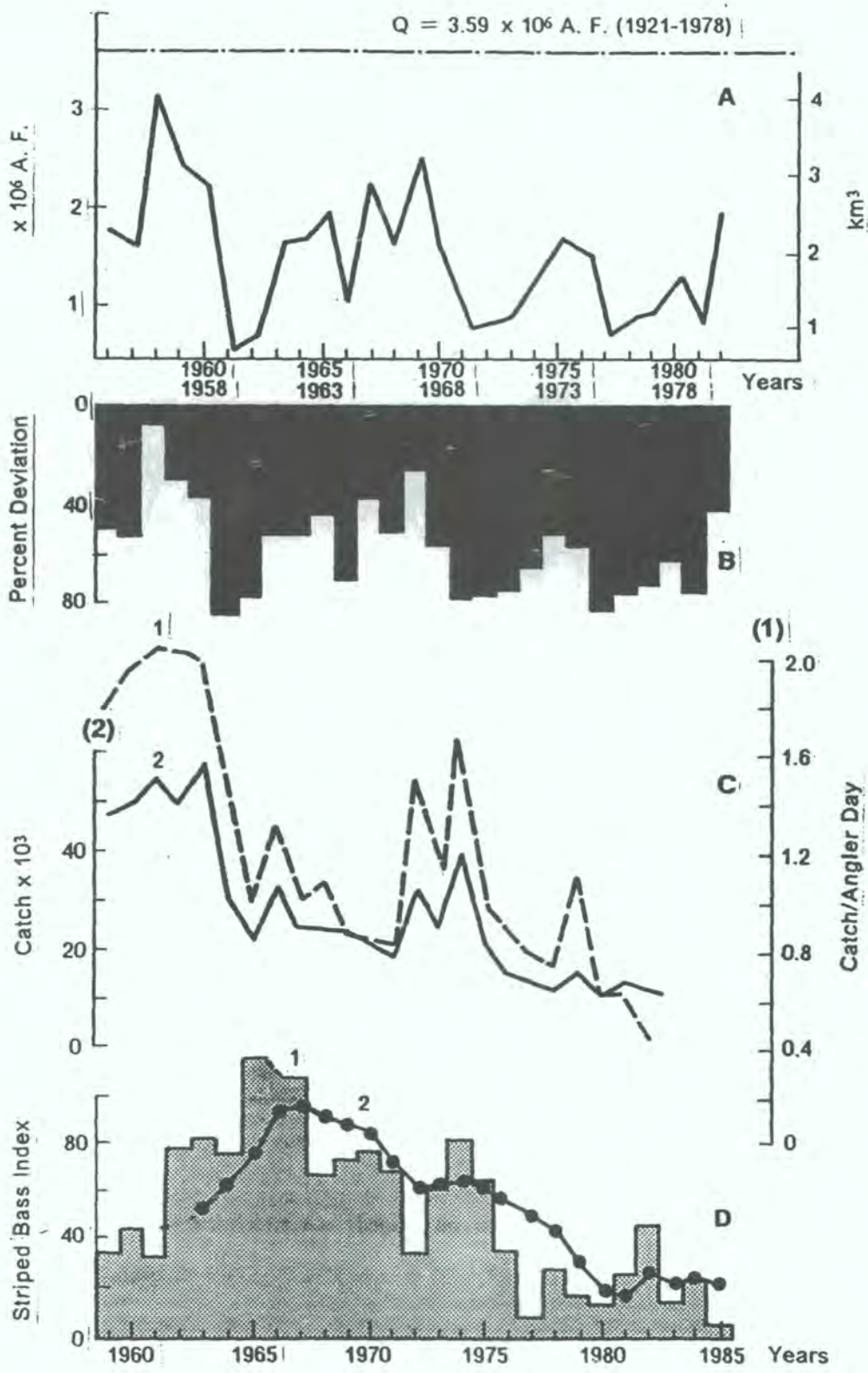

FIGURE 41. (A) Fluctuations of Delta regulated water supply to San Francisco Bay during spring (April-June). Data represent 3-year running means (e.g., 1958-1960). (B) Deviation in percentage of Delta regulated water supply to San Francisco Bay of mean spring natural runoff. (C-1) San Francisco Bay striped bass party boat catch/angler day (1959-1982). (C-2) Total striped bass party boat catch/season in number of fish (1959-1982). (D-1) Annual juvenile striped bass abundance index (1959-1985). (D-2) Five-year running means of striped bass abundance index (1959/63-1981/85). (Redrawn from Rozengurt *et al.*, 1985).

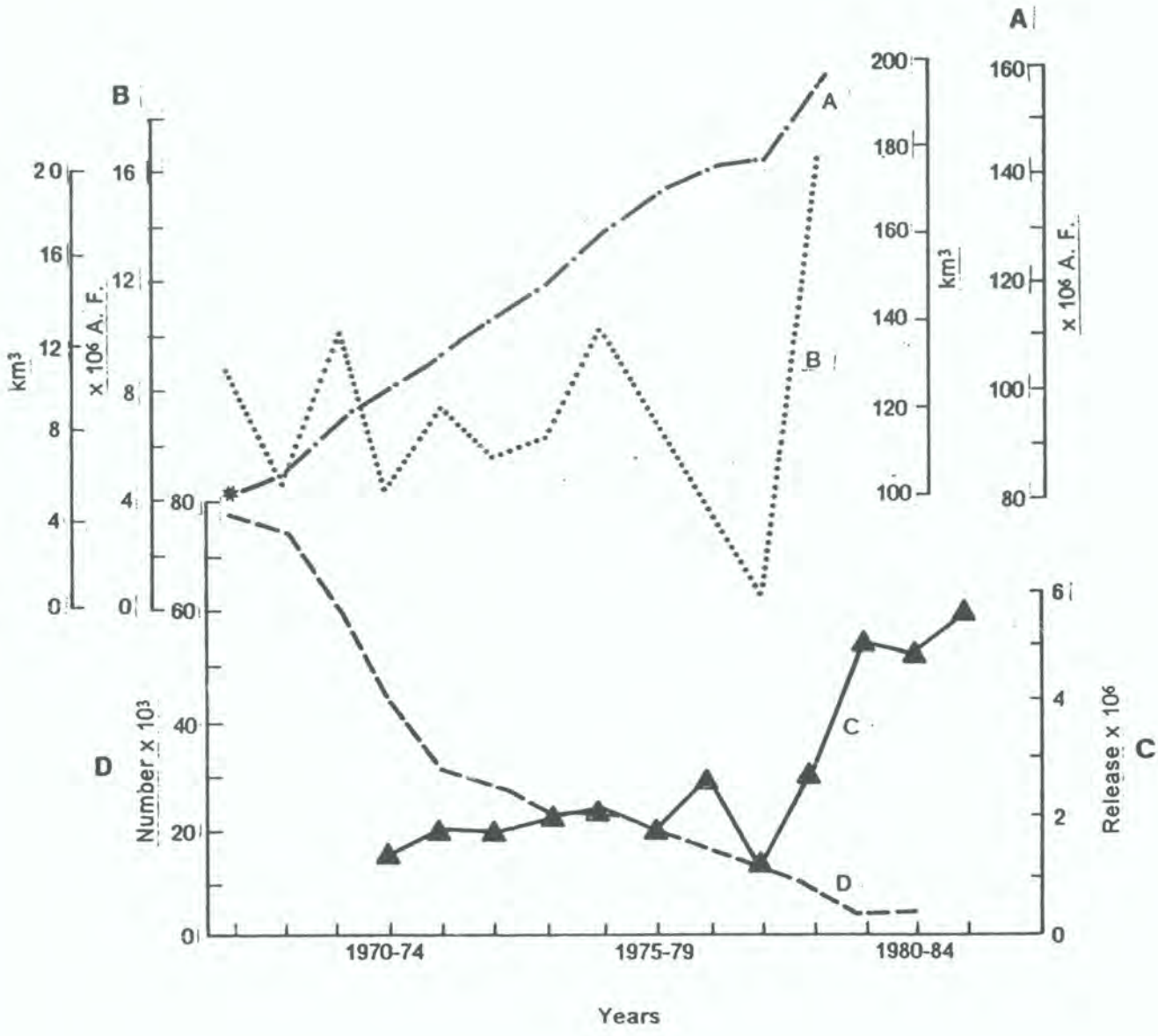

FIGURE 42. (A) Cumulative combined upstream diversions of the Sacramento-San Joaquin river systems (1967/69-1977/78). First data point is sum of diversions from 1955-1967. (B) Annual gross upstream diversions of the Sacramento-San Joaquin river systems (1967-1978). (C) Annual release of yearling chinook salmon juveniles from California State hatcheries (1970-1981). (D) Five-year running mean of winter-run spawning salmon past the Red Bluff diversion dam (1967/71-1980/84). (Redrawn from Rozengurt *et al.*, 1985).

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