NATURAL RECOVERY OF COLD WATER MARINE ENVIRONMENTS AFTER AN OIL SPILL

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J. M. Baker[1], R. B. Clark[2], P. F. Kingston[3], and R. H. Jenkins[3]

1 Consultant, Shrewsbury, England

2 Dept. of Biology, University of Newcastle upon Tyne, Newcastle upon Tyne, England

3 Institute of Offshore Engineering, Heriot-Watt University, Edinburgh, Scotland

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INTRODUCTION AND SUMMARY OF FINDINGS

This review of the published literature examines natural cleaning and natural recovery of ecosystems and biological communities following oil spills in cold water regions of the world. The scientific literature permits generalisations to be drawn; but oil spills in exceptional circumstances may produce exceptional effects. In a number of cases, long-term studies of recovery processes are incomplete, and recovery time scales suggested involve some extrapolation.

The following is a list of major findings from our review, grouped according to the primary subject areas of the paper.

BASIC CONCEPTS

- Consideration of the overall, long-term impact of a particular spill must take into account the relative proportions of oiled- and unoiled-habitats in the area of interest; and, in the case of oiled habitats, the relative proportions of different habitat types. Some areas may have a high percentage of habitats likely to recover relatively quickly, whereas other areas may have a high percentage of habitats likely to recover more slowly.

- For many cases, recovery of the availability of human services of a system impacted by oil is not closely related to biological recovery and is generally more rapid than biological recovery.

- Current evidence indicates that hydrocarbons are present in all environments. They originate from both natural and human sources and serve to define background levels of petroleum hydrocarbons.

- Clean, in the context of an oil spill, is defined here as the return to levels of petroleum hydrocarbons that have no significant detectable impact on the function of the ecosystem. The size of the ecosystem is obviously an important consideration. It is not microscopic, but is large enough to include the major plant and animal communities. Practical application of this definition requires taking into account the relative proportions of oiled- and unoiled-habitats. It does not necessarily require a return to some pre-existing background level or the complete removal of petroleum hydrocarbons from the environment.

- Recovery of an ecosystem damaged by petroleum hydrocarbons begins as soon as the toxicity or other adverse property of the oil has declined to a level that is tolerable to the most robust colonising organisms. Recovery processes can begin in the presence of residual oil.

- Recovery of an ecosystem is marked by the re-establishment of a healthy biological community in which the plants and animals of the community are functioning normally. It may not have the same composition or age structure as that which was present before the damage and will continue to show further change and development. It is difficult or impossible to say whether an ecosystem that has recovered from an oil spill is the same as, or different from, that which would have developed in the absence of the spill.

- Ecosystems are in a constant state of flux due to natural causes. These fluctuations can be as great as, or even greater than, those caused by the impact of an oil spill.

NATURAL CLEANING

- Most of the toxic components in a fresh oil spill on the surface of the sea rapidly evaporate. After evaporation, these toxic components disperse into the atmosphere and are rapidly diluted to background levels. Most of the remaining toxic components that did not evaporate, dissolve and disperse in the water column to low concentrations.

- Generally, the higher the aromatic content of an oil, the higher the toxicity. Weathered crude is less toxic than fresh crude because most of the toxic components have evaporated. Most of the toxic compounds are readily degradable.

- Oil concentrations in the water column below oil slicks are very low.

- The persistence of oil slicks on the sea surface is dependent upon the type of oil spilled and sea state conditions. Some slicks can be removed by natural processes within a few days. Other oils can form stable, highly viscous emulsions (mousses), which may persist for weeks or months in the open ocean. Eventually these slicks will form tarballs that are relatively harmless to biological systems.

- Microbial degradation is an important process in the eventual disappearance of oil from the marine environment. Degradation rate is controlled by oxygen and nutrient availability, temperature, chemical composition and surface area of the oil, and in some cases the activities of other organisms.

- Degradation is oxidative, and therefore the rate is reduced when oxygen concentrations are low, as is sometimes the case in fine sediments.

- Although some oxidation products resulting from biodegradation or photolysis are toxic, their rates of generation are slow at the surface of spilled oil because they are controlled by diffusion. If these products are leached from the surface of the oil and enter the water column, they are rapidly diluted to low concentrations. Therefore, they are not likely to have significant ecological impact.

- High-energy rocky shores usually do not accumulate oil, and if impacted are subjected to rapid cleaning by wave action.

- Oil does not penetrate easily into fine sediments in the intertidal zone, but can sink into shingle, gravel, and coarse sand. In some cases oil may penetrate to the water table, which forms a natural barrier to further penetration.

- On sheltered shores with high biological productivity, oil can penetrate down biological pathways, *e.g.*, worm burrows and plant root systems. Oil may persist for many years in the sediments, especially if oxygenating biological activity (*e.g.*, new burrow formation) is depressed.

- Large accumulations of oil or mousse may incorporate beach material and harden to form asphalt pavements; these are gradually eroded but may persist for many years on the upper shore and on sheltered beaches.

- Physical 'removal' of oil by natural processes alone does not eliminate oil from the environment; it redistributes it. This redistribution can be beneficial—*e.g.*, when wave action cleans the shoreline, it facilitates dispersion of the oil in the water column, and increases the surface area of the oil droplets, thereby encouraging other degradation processes. However, such redistribution may also involve sediment-bound oil being transported to the seabed.

- The toxicity of hydrocarbons on the seabed can vary widely, depending on the composition of the oil, the organism exposed to it, the transport pathways, and the extent to which the hydrocarbons can be degraded in the bottom sediments.

NATURAL RECOVERY

- Diving seabirds suffer heavy mortalities from oil. However, the mortalities arising from a single oil spill are not significantly different from natural mass mortalities experienced from time to time and are significantly less than annual mortalities from fishing activities.

- There is no evidence that seabird populations are declining as a result of oil spills. In fact, North Atlantic populations of most species have been increasing in recent years despite heavy annual losses from oil pollution.

- Kills of adult fish from exposure to oil are rare. The only important casualties from oil spills are rockfish and shellfish in near-shore waters, and fish in mariculture installations.

- Loss of pelagic eggs and fish larvae, when these are present at the time of an oil spill, has had no detectable impact on the fish stocks available to the fishing industry.

- Annual recruitment of fish stocks fluctuates naturally, and the size of the catchable stocks is determined more by the activities of the fishing industry (*e.g.*, over-fishing) and by climatic changes than by any other factor.

- Although the toxic components of petroleum hydrocarbons kill planktonic organisms, there is no evidence that these effects have any ecological significance in open waters. In closed waters, effects may persist for several months.

- To date, there are insufficient data in the scientific literature on marine mammal mortality and recovery to assess impact on breeding populations.

- Estimates of recovery times vary depending upon the environment. Past experience has shown that exposed, rocky shores usually recover in 2 to 3 years. Other shorelines show substantial recovery in 1 to 5 years with the exception of sheltered, highly productive shores (*e.g.*, salt marshes), which may take 10 years or more to recover.

- Subtidal sand and mud systems recover in recognisable successions. Usually, recovery times are 1 to 5 years, but they can be 10 years or longer in exceptional cases.

- Biological recovery of low-energy soft-substrate (sand, mud, *etc.*) ecosystems follows a generally well-defined course; but there are several alternative recovery routes for hard-substrate (rock, boulder, *etc.*) ecosystems, and the restored community may not be the same as that before the damage.

- Sublethal effects have been shown to occur after an oil spill and have been considered in the estimate of recovery times stated here. There is no evidence that sublethal effects are of any longer-term ecological significance.

- The early colonisers, once the physical and toxic effects of the oil ameliorate, play an active role in the breakdown of the remaining hydrocarbons.

- Removal of oil using drastic cleaning methods, beyond initial bulk oil removal, can actually delay recovery because the cleanup also removes living organisms and damages the habitat.

SCOPE AND OBJECTIVES

The objectives of this paper are, in the context of an oil spill, to provide:

- Practical working definitions of the terms 'clean' and 'recovery'
- A framework for understanding natural cleaning and recovery processes and their duration in cold water environments

The review draws only on published scientific information relating to oil spill incidents and relevant field trials and excludes the tropical environment.

This paper has been organised so that concepts, definitions, and other background information are presented in the earlier sections. The reader who wishes to skip this tutorial information should turn to the section entitled 'Natural Recovery,' which addresses recovery of biological systems (birds, fish, *etc.*).

An italicized summary statement is given at the beginning of major sections in the paper. The primary message of the paper can be obtained by reading only the italicized statements. A Glossary that gives definitions of selected technical words or phrases used in this report is also provided.

BASIC CONCEPTS

WHAT IS MEANT BY CLEAN?

Clean, in the context of an oil spill, is defined here as the return to a level of petroleum hydrocarbons that has no significant detectable impact on the function of an ecosystem. The size of the ecosystem is obviously an important consideration. It is not microscopic, but is large enough to include the major plant and animal communities. Practical application of this definition requires taking into account the relative proportions of oiled- and unoiled-habitats. This definition does not necessarily require a return to some pre-existing background level, or the complete removal of hydrocarbons from the environment.

Authors such as Myers and Gunnerson (1976) have concluded that both biogenic and petrogenic hydrocarbons are ubiquitous. Thus it is unrealistic to define clean as a complete absence of hydrocarbons or a complete absence of petrogenic hydrocarbons.

Examples of published information are given in Table 1 (water samples), Table 2 (sediment samples) and Table 3 (mussel samples). The first general observation that may be drawn from these examples is that all samples contain hydrocarbons.

It can be seen from Tables 1 to 3 that background levels vary considerably. General observations from the tables are:

- Higher background levels in water, sediment, and biota are found in nearshore waters, particularly in urban and industrialized bays and inlets, because of the proximity to land-based discharges and combustion products and the greater frequency of shipping.

- Hydrocarbons may accumulate in sediments or biota to higher background concentrations than are found in water.

In the absence of a spill, these may be called background hydrocarbons. Possible sources are

- Organisms, *e.g.*, leaf waxes and hydrocarbons synthesised by algae

- Natural seeps of petroleum hydrocarbons, such as occur, for example, at Scott Inlet and elsewhere along the northeast coast of Baffin Island (Levy, 1981), and in the Santa Barbara Channel (Spies and Davis, 1979)

- Airborne combustion products, either natural (*e.g.*, from forest fires) or man-made (*e.g.*, from the burning of fossil fuels)

- Normal operational discharges (*e.g.*, bilge water) from ships and boats, including tankers, cargo ships, fishing and cruise vessels, and private boats

- Normal land-based discharges, *e.g.*, rainwater run-off from roads and urban areas, sewage discharges

It is possible to distinguish petrogenic from biogenic hydrocarbons only by using highly specific and sensitive analytical methods, notably capillary gas chromatography (GC) and gas chromatography–mass spectrometry (GC-MS). Petrogenic hydrocarbons can be distinguished, for example, by their n-alkane distribution, pristane/phytane ratio, and the area of the unresolved complex mixture (UCM) 'hump' on the GC traces. Polynuclear aromatic hydrocarbons (PNAH or PAH) occur in crude oil but are also a significant constituent of combustion products (both natural and man-made).

Further information on this subject is available in the literature, *e.g.*, Broman *et al.* (1987), Foster and Wright (1988), Johnston *et al.* (1985), Knap *et al.* (1982), Law and Fileman (1985), Little *et al.* (1987), Mattsson and Lehtinen (1985), Mix and Schaffer (1983a, b).

The capacity of organisms such as bivalves to accumulate hydrocarbons (and other compounds) has led to their use as 'indicator' organisms to reflect spatial and temporal patterns in the compound of interest. As suggested by Table 3, mussels have been particularly widely used for this purpose. 'Musselwatch' sampling design and data interpretation have to take into account possible sources of variation; for example, the season (stage of reproductive cycle) or site (*e.g.*, tidal depth).

The interpretation of hydrocarbon data from sediments should consider the sediment properties, notably grain size and indigenous organic matter, which influence hydrocarbon adsorption and retention (Law and Fileman, 1985; Little *et al.*, 1987). The interpretation of hydrocarbon analytical data in general has to take into account the fact that different analytical methods (*e.g.*, infrared, ultraviolet fluorescence, GC) measure different things, so results from different methods are not directly comparable. Even if the same method is used, analytical variability also has to be taken into account (Farrington *et al.*, 1988; Howells *et al.*, 1989; Law *et al.*, 1987). Farrington *et al.* (1988), considering the results of a 22-laboratory intercomparison exercise, conclude that there is a need for much improvement for within- and between-laboratory precision.

WHAT IS MEANT BY RECOVERY?

Oil spill damage takes various forms: commercial, recreational, ecological, or aesthetic (though these may be interrelated). Recovery processes take many forms, which act on different time scales.

Recovery of the Use of Resources

> *The sea and coastlines are used in various ways for commercial and recreational purposes (human services) by the human population. Human uses of a spill-impacted area generally resume as soon as bulk oil is removed. In many cases, the availability of human services is not closely related to biological recovery and is usually more rapid than biological recovery.*

Nature of Resources. Human uses of the resources of the coastal regions of cold water marine ecosystems include fisheries (commercial and sport finfish; shellfish), tourism, nature viewing (including bird-watching), hunting, camping, boating/kayaking and, under rare circumstances, sunbathing/beach use.

The availability of a resource (except for shell fisheries) is usually restored as soon as bulk oil is removed from the water surface and from the most heavily impacted region, which is the intertidal zone. Failure of the actual use of a resource to recover as quickly as the resource becomes available is usually a consequence of public perceptions.

Fisheries. Commercial and sport fishermen are generally excluded from fishing grounds where oil is floating on the water because of the risk of fouling fishing gear. Often it is possible to fish in areas unaffected by oil, and commercial fishing can continue with very little interruption even after a major oil spill. This was the case for larger fishing vessels in Brittany, France, following the wreck of the *Amoco Cadiz* (Fairhall and Jordan, 1980).

Fish stocks are rarely directly affected by oil spills, and a fishery in an area that has been exposed to oil can be reopened as soon as the area is free of floating oil. Recovery usually takes place in a matter of days or weeks and is independent of the biological recovery of damaged ecosystems.

On the other hand, when the fishery resource itself is damaged (*e.g.*, clam beds, lobster fishery, mariculture installations), commercial damage will persist until exploitable stocks are restored. That may require deliberate restocking and a delay of 2 to 10 years, depending upon the age at which new stocks reach a commercial size.

Tourism. Many forms of tourism are unrelated to biological conditions, and therefore should be unaffected by oil spills, as long as the coastline and the water are clean. Minor oil contamination of bathing beaches by tarballs is regarded as a nuisance, and many coastal resorts regularly remove tarballs along with other beach litter in order to preserve local amenities. A detailed study made in the U.K. (PAU, 1973) could find no evidence that oil pollution had a detrimental effect on tourism, even at popular resorts in southwest England that had suffered from oil spills. This was not the case in Brittany following the *Amoco Cadiz* oil spill in 1978, where the tourist season that year was commercially the poorest on record; even coastal sites far away from those impacted by oil were affected (Fairhall and Jordan, 1980). Holidays on the south Brittany coast, far from the spill, were cancelled, with oil being given as the excuse. Recovery to pre-spill tourism levels occurred the following summer.

Tourist activities which are related to the biological environment, *e.g.*, bird-watching, resume as soon as bulk oil is removed from an impacted area.

Seaweed Harvesting. In some areas seaweeds and kelp are harvested for the manufacture of alginates used in shampoos, cosmetics, and bath liquids, food additives, and for processing into animal feeds. Since oil does not adhere to the slimy surface of seaweeds and kelp, these plants are little affected by an oil spill. The factory processes used in the manufacture of products from kelp eliminate gross hydrocarbons, so even residual contamination of the crop does not affect its commercial value. This is not true of seaweeds harvested from the shore. The start of the seaweed harvest on the Brittany coast had to be delayed one month until the oil had cleared sufficiently, following the wreck of the *Amoco Cadiz* on the French coast in 1978. The kelp harvest was normal, but the harvest of intertidal seaweeds was seriously affected, largely because these seaweeds had been removed in large quantities by clean-up teams, but also because of the high levels of oil remaining in the areas that had been impacted (Fairhall and Jordan, 1980).

As mentioned above, human services in a spill-impacted area are generally available as soon as bulk oil is removed from both the water surface and the shoreline. However, biological recovery involves processes that begin in the presence of oil and continue to operate well after bulk oil is removed. Consequently, recovery of the availability of human services is achieved, in most cases, before biological recovery is realized.

Biological Recovery Processes

Biological recovery of an ecosystem damaged by an oil spill begins as soon as the toxicity or other damaging properties of the oil have declined to a level that is tolerable to the most robust colonising organisms. Subsequent events follow a well-established course which differs with the nature of the ecosystem, and the early colonisers assist in the removal of the oil.

Soft Substrata. In soft substrata (mud, sand) the following sequence of events follows damage caused by the input of any organic material such as sewage sludge, wood pulp waste (Pearson and Rosenberg, 1978), natural algal decomposition (Spies *et al.*, 1988) or oil (Kingston, 1987).

The initial phase of recovery is characterised by a small number of species, but in enormous numbers. For example, 'opportunistic' species of polychaete worms breed all the year round and young stages are always available to recolonise; they can tolerate adverse conditions that exclude most other animals and they feed on organic material. In the absence of competition and with an abundant food supply, they multiply rapidly. As conditions in the area improve, other, less hardy species are able to establish themselves and, by competition, reduce the numbers of the first colonisers. This process proceeds as conditions continue to improve and more sensitive species are able to re-establish themselves. Eventually a diverse fauna, characteristic of the area, is restored.

Hard Substrata. On hard substrates (rock, boulder, *etc.*) the recovery process is less predictable because the initial colonisers are all seasonal breeders, and the first species to colonise a bare surface may inhibit colonisation by others. On hard surfaces, it is the interaction between plants and animals and competition for space, rather than the physical and chemical conditions, that determine the path taken by the recovery process.

Lewis (1982) has made a long study of this subject and gives an expert account. Bare rocks are first colonised by a film of diatoms and the young stages of seaweeds (algae). If other events do not supervene, these will develop into dense stands of kelp (subtidal) or seaweeds (intertidal) with a rich associated fauna of worms, crabs and other crustacea, winkles and sea snails, *etc.* However, these dense stands of algae cannot develop if herbivores colonise the area at an early stage. The most important herbivores are sea urchin and abalone (subtidal) and limpet (intertidal). They graze down the young seaweeds, but make little or no impact on established stands of algae. If, because of this grazing pressure, the rocks are kept free of algae, they are available for colonisation by other encrusting animals such as barnacles that live happily with the grazers. Mussels, if they become established first, exclude both seaweeds and limpets.

Thus, the assemblage of plants and animals that recolonise a denuded hard surface depends on the time of year when colonising forms are available, and the first to arrive may determine the subsequent course of events. Predators such as starfish or carnivorous sea snails (whelks, *etc.*), if in sufficient numbers, may eliminate the mussels or barnacles and result in further change. The various routes that may be taken in the recovery of damaged biological communities on hard substrata are no different from the changes that occur in these environments from natural causes.

Natural Fluctuations

> *Marine ecosystems are not stable, as is commonly assumed, but are in a natural state of flux and show erratic fluctuations from purely natural causes. This has now been established for all sectors of the marine environment for which adequate long-time series of observations have been made.*

Different species of North Atlantic plankton monitored between 1948 and 1969 show wide year-by-year abundance fluctuations (Figure 1, Glover *et al.*, 1972). Cod in the North Atlantic, as recorded in numbers of fish landed, show a similar volatility in abundance (Figure 2, Jones, 1982). Herring and sardine abundance in traditional Japanese and Scandinavian waters can be traced from historical records back to the start of the 15th century; they too show periods of abundance and dearth that can be related to climatic change (Figure 3, Jones, 1982). Even the subtidal benthos, normally regarded as a stable environment, shows short-term changes, presumably related to climatic fluctuations (Figure 4, Buchanan *et al.*, 1978).

Subtle changes occur in intertidal muddy substrates. In one closely studied mud flat in northeast England, one common species of polychaete worm failed to breed in two successive years, for unknown reasons, and was replaced by a different, but closely related species (Olive *et al.*, 1981).

Rocky shores show the effects of climate and the biological interactions described above (Figure 5, Lewis, 1972) and naturally undergo wide fluctuations in the extent of cover by seaweeds, barnacles, mussels, *etc*., in each case with corresponding changes in the fauna associated with them.

Natural Change and Impact Assessment. Sometimes damage attributed to oil spills can be caused by other factors. Limpets are dominant herbivores on rocky shores. If they are scarce, a cover of seaweeds follows; if they are abundant, seaweeds cannot become established and the rocks are colonised by other encrusting fauna. Young limpets settle on the rocks in early fall but are susceptible to air frost during the next few weeks. In years when there are early air frosts, most of the limpets are killed and the population is so small that seaweeds may develop (Figure 6, Bowman and Lewis, 1977).

Limpets are also killed by high summer temperatures. In one case on the north coast of Scotland, an accidental release of diesel fuel impacted a rocky shore. Afterwards, most of the limpets were found to be dead and the shore became densely covered with seaweed. This might have been regarded as a classical case of oil spill damage; but in fact, the beach had been regularly monitored, and it was known that the limpets had been killed by an exceptionally hot summer. These changes were already in train and the oil spill was irrelevant (Bowman, 1978).

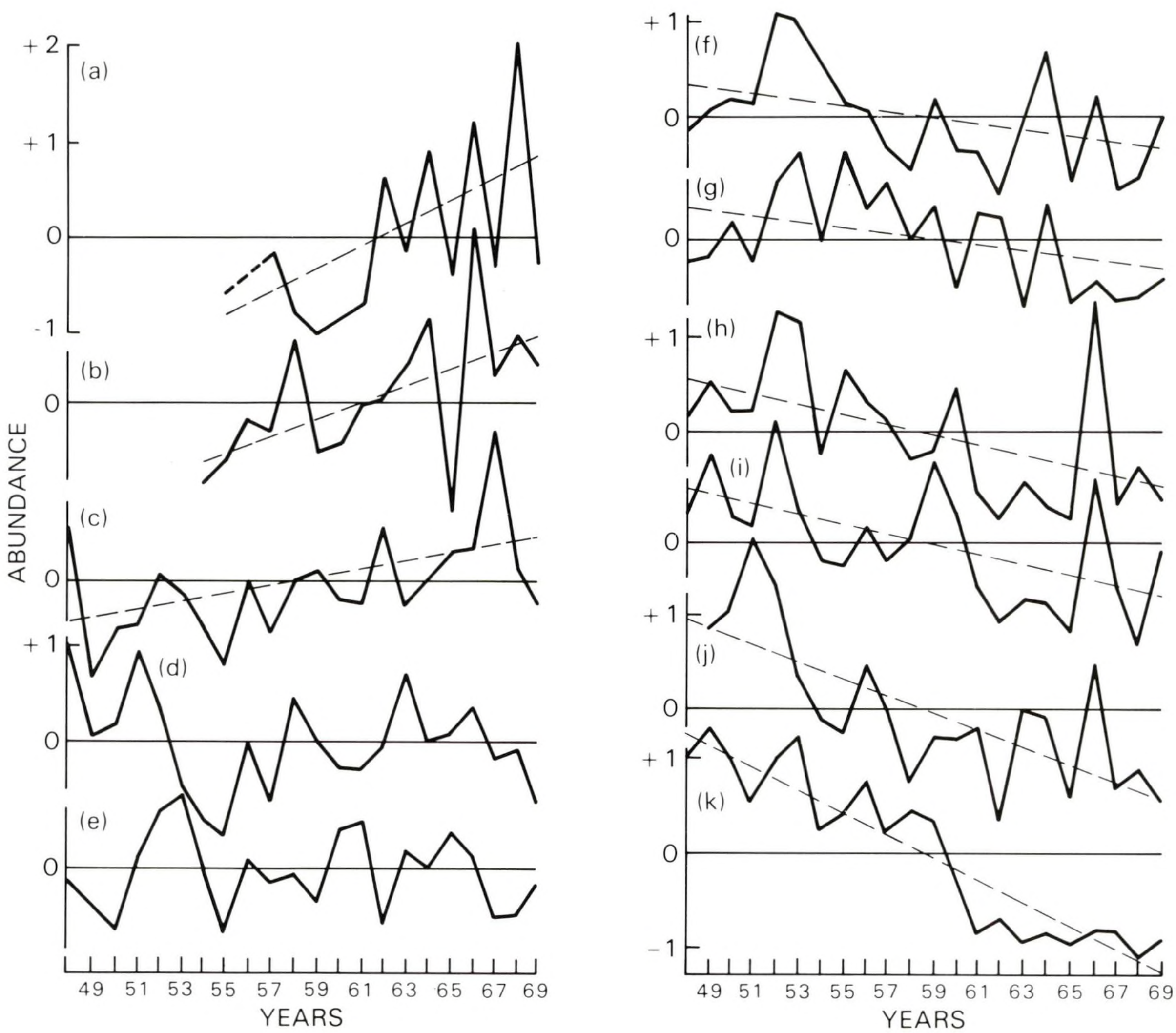
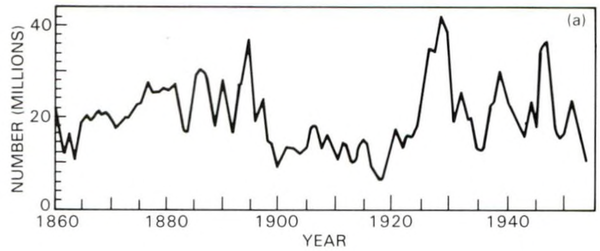


Figure 1: Changes in the abundance of zooplankton species in the North Atlantic. The organisms are: (a) *Pleuromamma borealis*; (b) *Euchaeta norvegica*; (c) *Acartia clausi*, (d) *Temora longicornis*; (e) *Clione lamacina*; (f) *Calanus helgolandicus* and *C finmarchicus* stages V and VI; (g) *Metridia lucens*; (h) *Candacia armata*; (i) *Centropages typicus*; (j) *Spiratella retroversa*; (k) *Pseudocalanus* and *Paracalanus*. (After Glover *et al.*, 1972.) The vertical axes give abundance variations expressed as standard deviations about a mean of zero. An annual abundance variation of one standard deviation less than the mean is –1. Standard deviations from the mean are calculated from annual abundance measurements for the 21-year period.



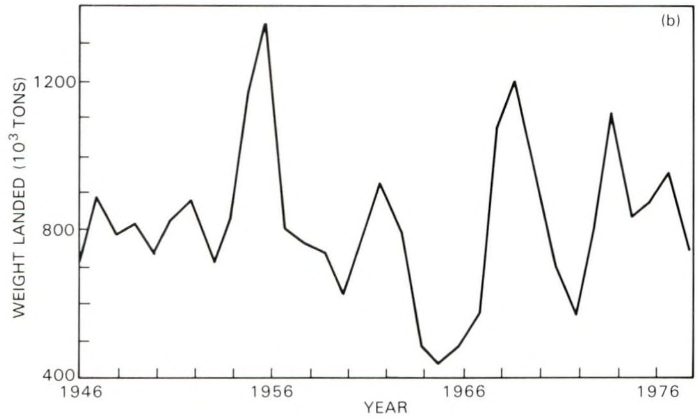


Figure 2: Landing of cod (a) in the Lofoten, Norway, fishery and (b) in the whole northeast Atlantic fishery (after Jones, 1982).

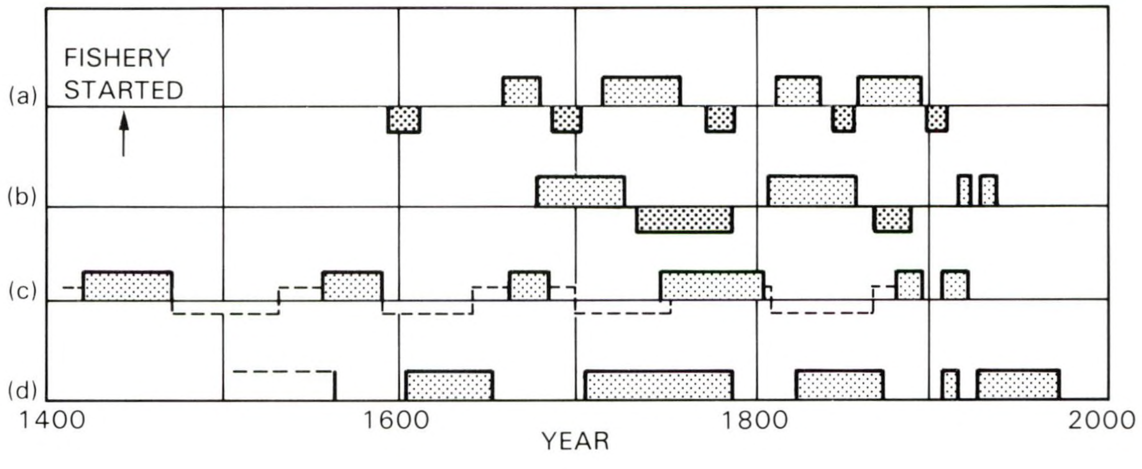
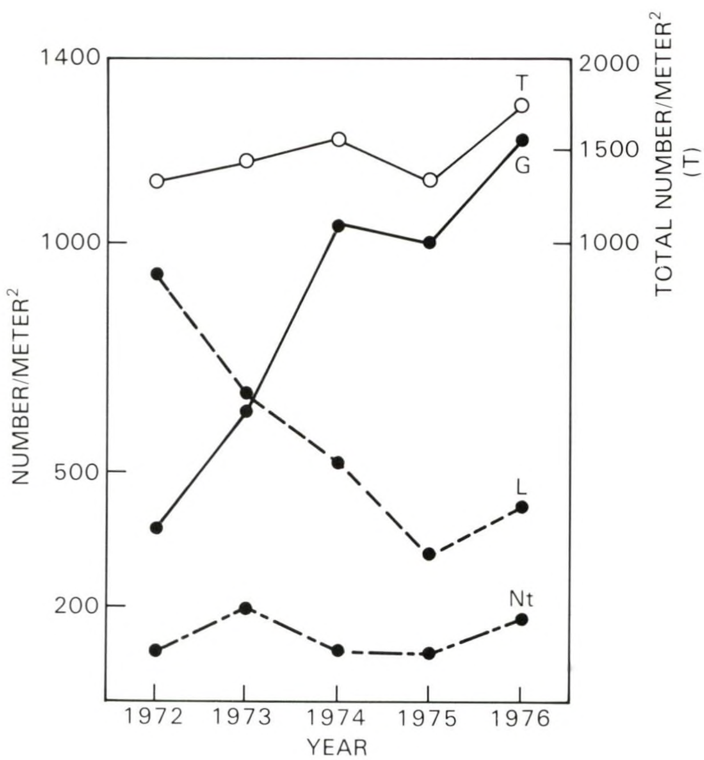


Figure 3: Periods of good and bad years (above and below line, respectively) for (a) Hokkaido herring; (b) Japanese sardine; (c) Bohuslan herring; (d) Atlantoscandian herring. (Dashed lines are speculative.) (After Jones, 1982.)



Gaining species 1972–6 (G)

Species	Gain/meter^2
Paraonis gracilis	238
Prionospio malmgreni	144
Myriochele oculata	129
Thyasira flexuosa	115
Owenia fusiformis	104
Magelona minuta	100
Tharyx multibranchiis	27

Losing species 1972–6 (L)

Species	Loss/meter^2
Spiophanes bombyx	219
Chaetozone setosa	129
Ampelisca tenuicornis	112
Amphiura filiformis	50
Nephtys hombergi	31
Scoloplos armiger	26
Rhodine gracilior	24
Ammotrypane aulogaster	17

Neutral species 1972–6 (Nt)

Species
Ampharete finmarchica
Glycera rouxi
Goniada maculata

Figure 4: Contribution of the 17 most abundant species to the total number of individuals at one subtidal station off the northeast coast of England (after Buchanan *et al.*, 1978).

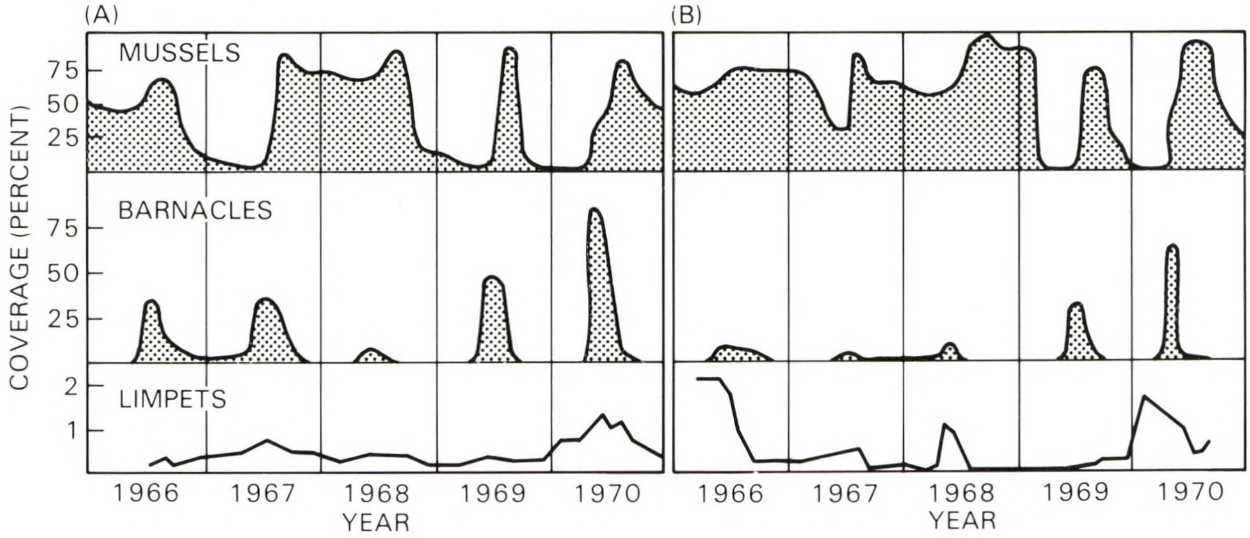


Figure 5: Fluctuations in the percentage cover on rocks by mussels, barnacles and limpets at two sites (A and B) on the north Yorkshire coast, 1966–1970 (after Lewis, 1972).

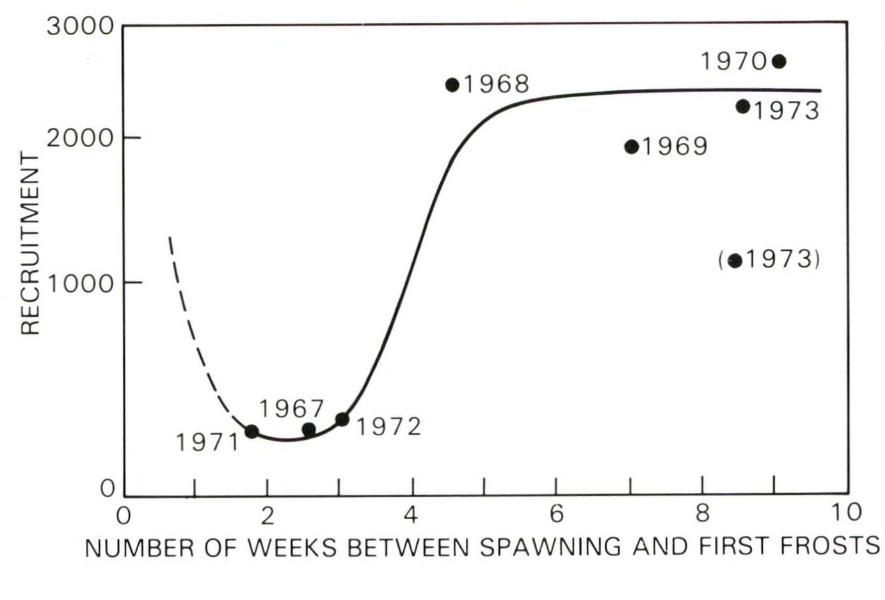


Figure 6: Relation between recruitment of limpets, *Patella vulgata*, and the duration of the frost-free period after spawning for the years 1967–1973. In 1973 there were two successional spawnings. (After Bowman and Lewis, 1977.)

In the same way, examination of two species of sea snails in Milford Haven in 1981 showed that there had been no substantial recruitment of young forms for 4 to 5 years (Figure 7). Milford Haven is the largest deep-water oil terminal in Europe, with five oil refineries around it; there are inevitable oil spills from tankers and these waters also receive waste water from refineries. It would be a natural assumption that the failure of recruitment of the sea snails while the oil terminal and refineries were in full operation was related to oil exposure. Not so. The two species are near the northern limit of their geographical range in Britain, and none of the fringe populations from south Wales to the north of Scotland had a successful recruitment during that time. The failure was due to climatic reasons, not oil (Lewis, 1982).

Given the above examples, it is clear that caution must be exercised in assessing the apparent damage as being due to oil spills.

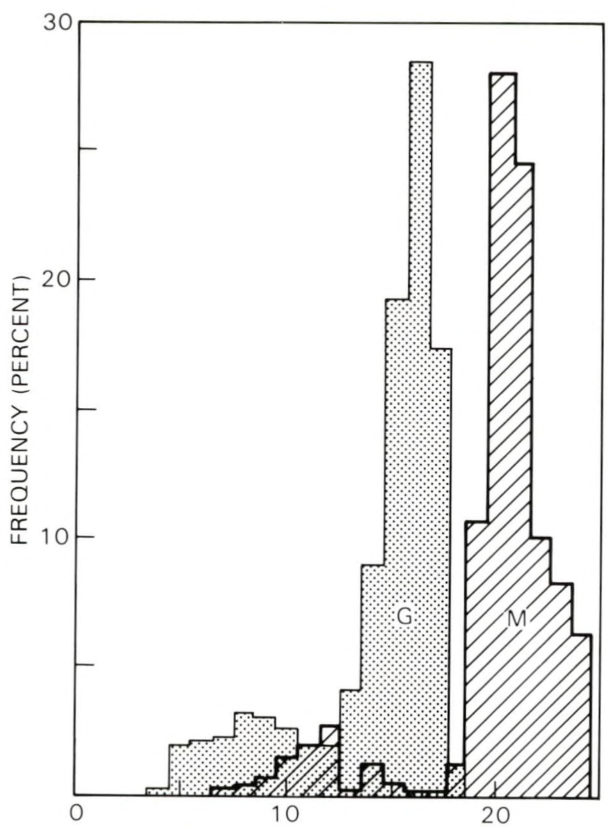


Figure 7: Size distribution of the intertidal gastropods *Gibbula* (G) and *Monodonta* (M) in a population on the Pembrokeshire coast, October 1981 (after Lewis, 1982).

A Working Definition of Ecological Recovery

Recovery is marked by the re-establishment of a healthy biological community in which the plants and animals characteristic of that community are present and functioning normally. It may not have the same composition or age structure as that which was present before the damage, and will continue to show further change and development. It is impossible to say whether an ecosystem that has recovered from an oil spill is the same as, or different from, that which would have persisted in the absence of the spill.

Ecological recovery is a contentious subject on which there is room for legitimate disagreement (Southward, 1982; Clark, 1982). A synthesis of Clark's analysis (Clark, 1989) follows.

The state to which an environment returns after damage is often unpredictable. It depends upon the time of year, the availability of recolonising forms, biological interactions, climatic factors, *etc*. In any case, the ecosystem, had it not been damaged, would have changed in subtle or major ways during the period required for recovery.

The species of plants and animals generally available in the locality, which are the main components of a community appropriate to a particular habitat, re-establish themselves in northern waters within 2 to 3 years (RCEP, 1981). The members of the community then function and interact in a normal way, and for practical purposes the ecosystem has 'recovered'.

The restored communities will, of course, continue to show fluctuation and change, depending on climatic factors, biological interactions, the aging of some members of the community, or the establishment of rarer, but ecologically unimportant, plants or animals. Controversy arises over the view taken of these post-recovery changes.

Particularly in northern waters, many species are at the fringe of their geographical range. They exist in small, scattered populations and have a precarious existence. If one such population is eliminated by environmental damage, its nearest neighbours from which recruitment could take place may be many miles away. Recolonisation of a damaged area by these rare species will therefore depend on exceptionally favourable circumstances and may be protracted. It may not take place at all if climatic factors make these sub-optimal environments even less suitable.

It is unreasonable to demand that recovery depends on the re-establishment of the same age structure in the population as existed before the environmental damage. Lobsters, for example, can live about 20 years, although there are very few such grandfathers in any natural population and probably none in a lobster fishery. The existence of a few geriatrics in the population therefore is not a realistic criterion for recovery of a damaged ecosystem. In any case, few natural populations show an actuarial age structure; they are often dominated by a single year-class with little recruitment in the years before or after (see Figure 7, Lewis, 1982).

COLD WATER ECOSYSTEMS

Marine cold water ecosystems include those habitats found in temperate, boreal/antiboreal, subpolar, and polar seas (see Figure 8). One of the most significant differences between tropical and cold water habitats is the annual temperature variation. In tropical surface waters this difference is no more than 2°C, whereas in cold water ecosystems it may reach 10°C (Ekman, 1967). Temperature changes are seasonal in cold water ecosystems and are thus important in controlling cyclical biological events such as reproduction. As a result of thermally induced mixing of water layers, seasonal temperature changes are an important factor in enhanced productivity in these regions.

Animal and plant communities from cold water ecosystems tend to be less stable than those from lower latitudes, owing to the harsher environmental conditions. As a consequence, there can be considerable natural variability in community species composition from year to year. Animals from polar and subpolar regions tend to adopt reproductive strategies that involve either viviparous (live-bearing) or oviparous (direct development from egg to miniature adult) development. Since such strategies are associated with greater parental care, but fewer offspring per reproductive cycle, these animal populations are less likely to recover from major environmental damage as rapidly as those more southerly species producing vast numbers of planktotrophic (plankton-feeding) larvae.

The reader is referred to Appendix A for further details.

PHYSICAL AND CHEMICAL BEHAVIOUR OF SPILLED OIL

When oil is spilled at sea, a series of complex interactions of physical, chemical, and biological processes is immediately set in train (Koons, 1987). These processes are collectively called 'weathering' and are described in Appendix B. Evaporation of the volatile components is especially dominant in the initial phase following a spill. The evaporated components include most of the toxic constituents in crude oil, which, after evaporation, are rapidly diluted to background levels in the atmosphere. The persistence of oil slicks on the sea surface is dependent upon the type of oil spilled and sea state conditions. Some slicks can be removed by natural processes within a few days; other crude oils can form stable, highly viscous emulsions (mousses) and may persist for weeks or months in the open ocean. Eventually these slicks will form tarballs that are relatively harmless.

Oil concentrations in the water column below oil slicks are low. Because of the churning actions of waves in shallow waters, oil may become incorporated into sediments or more concentrated in the water column. Beached oil may be washed off and may also become incorporated into subtidal sediments. Alternatively, the oil on the beach may be refloated and transported to impact other stretches of the coastline, or dispersed into the water column as fine droplets. However, oil transported from beaches is usually weathered and less toxic than fresh oil. Oiled sediments on beaches can be buried under fresh deposits of sand. High-energy rocky shores usually do not accumulate oil and, if impacted, are rapidly cleaned by wave action.

The reader is referred to Appendix B for further details.

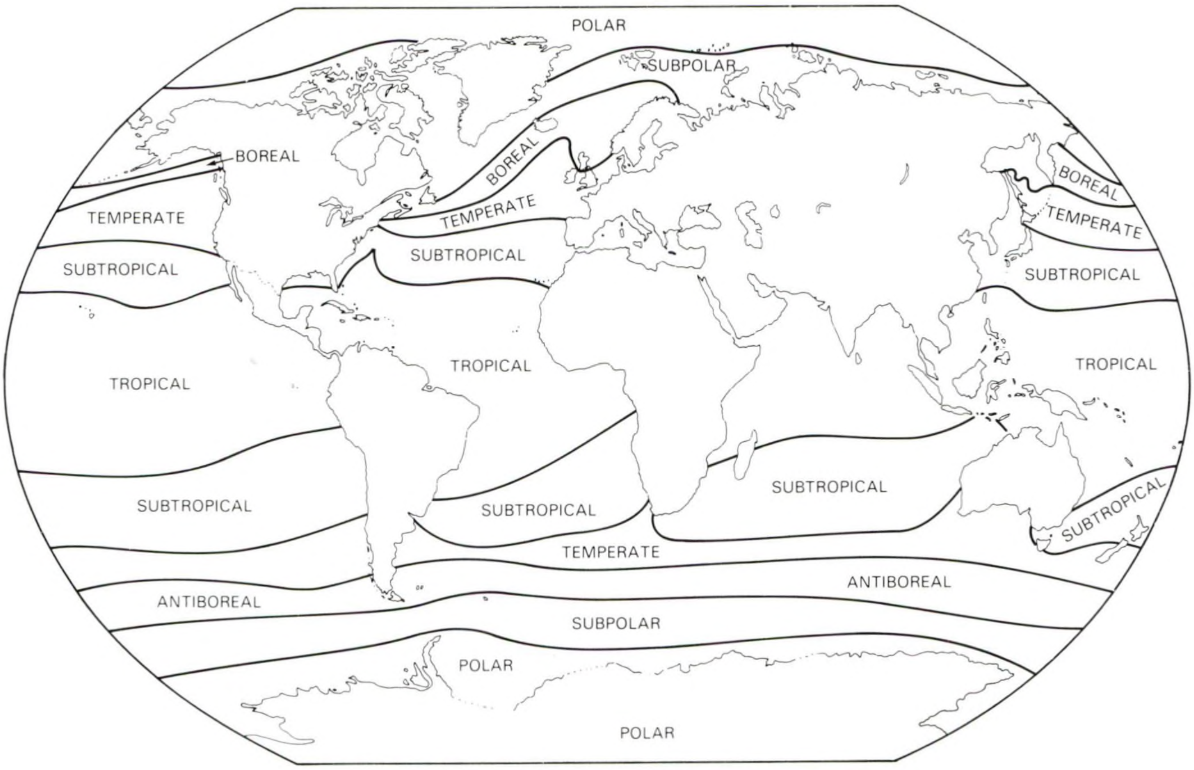


Figure 8: Map of the world showing the distribution of temperate to polar waters (Bogdanov, 1963).

NATURAL CLEANING

PHYSICAL REMOVAL BY NATURAL PROCESSES

The penetration of oil into shores, and the subsequent retention, redistribution, and escape of subsurface oil are influenced by shore physical characteristics, including exposure to wave action, sediment grain size, and position of the water table. Sediment transport pathways may move oil along shores or into subtidal areas, where grain size is again important in influencing oil retention.

Physical removal of oil from the shoreline by natural processes by itself does not eliminate oil from the environment; it redistributes it. Complete removal of oil from impacted shorelines, in terms of degradation of hydrocarbons to water and carbon dioxide, is effected partly through abiotic chemical reactions but mainly by biodegradation. Nevertheless, physical redistribution can be beneficial; for example, wave action may remove oil patches from a rocky shore and redistribute the oil in the form of relatively small drops in the water. This cleans the shore, and increases the surface area and aeration of the oil, thus facilitating biodegradation.

Physical removal from a particular area of a shore may also take place via sediment transport into an adjacent area ashore (McLaren, 1984) or into subtidal areas, where it is likely to continue to travel in the direction of sediment movement (McLaren and Little, 1987). The partitioning of oil into shoreline stranding, particle formation and transport, and sedimentation and biodegradation has been modelled by Gundlach *et al.* (1985) and Gundlach (1987). The estimation and quantification of oil on shorelines has been discussed by Owens (1984, 1987).

There is considerable information on physical removal/residence times of oil for different types of shore, resulting both from observations following spills and from experimental work. Such observations (a range of which are summarized in Table 4) form the basis for a number of discussions on factors determining the behaviour of oil on shores (*e.g.*, Owens, 1978, 1985; Tsouk *et al.*, 1985; Wolfe, 1987). In particular, they have been used to produce shore vulnerability or sensitivity indices which summarize the behaviour of oil on different types of shore and which can be used for predictive purposes and oil spill contingency planning. The best known index, used as the basis for many projects, is that of Gundlach and Hayes (1978), and this is reproduced in Table 5. An example of a regional application (the index applied to the Alaskan coast) is given by Gundlach and Hayes (1982). Vulnerability indices may be constructively used with removability indices which rate shore types in terms of how easily oil may be removed from them (Hann, 1988).

Factors affecting residence times include

- Overall exposure of the shore, from exposed rocky headlands to sheltered tidal flats and marshes (see Table 5). This in turn depends upon a number of variables that include fetch, speed, direction and frequency of winds, and open angle of the shore (Ballantine, 1961).

- Localized exposure/shelter—even on an exposed shore, cracks, crevices, and spaces under boulders can provide sheltered conditions where oil may persist.

- Steepness/shore profile—extensive, gently sloping shores dissipate wave energy.

- Substratum—oil does not penetrate easily into fine sediments, especially if they are waterlogged, but can penetrate into shingle, gravel and some sand beaches (see Hayes *et al.*, 1979, and Long *et al.*, 1981, in Table 4). In some cases oil may reach the water table, which forms a natural barrier to further penetration.

- Height of the stranded oil on the shore—oil spots taken into the supratidal zone by spray can persist for many years, where it weathers to produce tarry residues (see Mottershead, 1981 in Table 4). Conversely, oil on the mid- and lower-shore is more likely to be removed by water action. It is common to have stranded oil concentrated in the high tide area (see Keizer *et al.*, 1978, and Owens *et al.*, 1987a, b, in Table 4).

- Oil type—*e.g.*, viscosity will affect movement into and out of sediment shores.

- Volume of oil—*e.g.*, heavy loadings lead to greater retention times in sediments (see Harper *et al.*, 1985 in Table 4).

CHEMICAL AND BIOLOGICAL DEGRADATION

The main type of change that hydrocarbon molecules may undergo on entering the marine environment is oxidation. This occurs either through abiotic chemical reactions, or through enzyme-controlled reactions in a variety of organisms, notably microorganisms. The toxicity of oil can vary widely, depending on the nature of the source and the extent of degradation. Oil reaching the shore or the seabed may already have been partly degraded, and will continue degrading at rates dependent on local environmental conditions. The main process involved is oxidation by microorganisms.

Chemical Degradation

Abiotic chemical reactions are usually catalysed by light (photooxidation) and lead to the formation of a variety of oxygen-containing derivatives, including alcohols, ethers, dialkyl peroxides, and carbonyl compounds. Further details are given by Malins (1981), who also makes the points that the transformations of petroleum create a formidable challenge for the analytical chemist and that relatively little is known about the toxicity of oxidation products to various forms of marine life.

Factors affecting chemical oxidation include light intensity and duration, aeration, and oil thickness. Burwood and Speers (1974) found that aeration increased the chemical oxidation of crude oil under both subdued and direct sunlight. Riley *et al.* (1980) tested the effects of different simulated environmental conditions on the weathering of Prudhoe Bay crude over 24 days, and found that a combination of sunlight and water sprayed upon the surface of the oil (to simulate rough conditions) produced the largest decreases of the volatile saturated compounds and most of the aromatic compounds.

The complete degradation of some hydrocarbons may involve a combination of photooxidation and biodegradation. Hinga *et al.* (1986) studied the degradation of a polynuclear aromatic hydrocarbon, 7,12-dimethylbenz(a) anthracene (DMBA), in a marine microcosm. They concluded that the initial transformations, which affected almost all of the DMBA within hours, were primarily the result of photodegradation, but at least some of the photooxidation products were then subject to biodegradation resulting in production of carbon dioxide.

Biological Degradation

Microbial Degradation. Degradation occurs through oxidation reactions, and microbial degradation is an important process involved in the eventual disappearance of oil from the marine environment (Karrick, 1977). Reviews of microbial degradation of hydrocarbons are included in CONCAWE (1979), Jordan and Payne (1980), and Atlas (1985). Over 200 species of microorganisms are capable of degrading hydrocarbon compounds, and the species concerned are widely distributed bacteria and fungi, including those found in polar environments. Representatives of bacterial genera such as *Pseudomonas*, *Mycobacterium*, and *Nocardia* are capable of degrading both aliphatic and aromatic hydrocarbons (CONCAWE, 1979), as are some fungi such as *Cladosporium resinae* (Walker *et al.*, 1973). Other species are more restricted in their hydrocarbon utilisation. Details of species, hydrocarbons utilised, and metabolic pathways are given by Jordan and Payne (1980). CONCAWE (1979) identifies *Desulfovibrio desulfuricans* as an anaerobic hydrocarbon-degrading bacterium, though its degradation power is very slight (Wallhauser, 1967).

A number of observations on microbial degradation are summarised in Table 6. As exemplified in the Stewart and Marks (1978) study, hydrocarbon-utilising microorganisms multiply rapidly following an input of oil and subsequently decline as the oil is degraded. Degradation rates vary considerably, and a number of rate-limiting factors can be identified. Experimental work (*e.g.*, Colwell *et al.*, 1978, and the Baffin Island experiments summarised by Atlas, 1985) shows that nitrogen and phosphorus nutrients can be limiting. Biodegradation can be stimulated by the right kind of fertiliser application, even in polar and subpolar conditions.

Because degradation is oxidative, it proceeds slowly in low oxygen concentrations. These may come about in various ways. For example,

- Reduced accessibility because of relatively large amounts/thicknesses of oil (Colwell *et al.*, 1978; Fusey and Oudot, 1984)

- Consumption of oxygen by normal organic decay processes such as occur in piles of rotting seaweed (Sveum and Sendstad, 1985)

- Reduced penetration of oxygen into deeper sediments (Mille *et al.*, 1984)

- Limited circulation of interstitial water in fine-grained sediments

It follows that natural processes that can increase aeration, such as the worm activity described by Gordon *et al.* (1978), or some types of natural physical redistribution as described in the section on physical removal, can increase microbial degradation.

Colwell *et al.* (1978) concluded that temperature was not a limiting factor for degradation of hydrocarbons in the Straits of Magellan; however, there is some evidence that it can be elsewhere. Arhelger *et al.* (1977) found that dodecane oxidation rates were 0.7 g/litre/day in Port Valdez, 0.5 g/litre/day in the Chukchi Sea, and 0.001 g/litre/day in the Arctic Ocean. This difference with latitude is consistent with the experimental results of Haines and Atlas (1982), who found evidence of biodegradation only after one year in the case of Prudhoe Bay crude in Beaufort Sea sediments. Temperature limitation may operate seasonally. For example, the average degradation rate for n-hexadecane in experiments on Baffin Island (Atlas, 1985) varied from 9.5 to 43.8 $\mu g/m^3$/day for water, with the highest rates occurring in early August.

Petrogenic hydrocarbons can accumulate in subtidal sediments in various ways. Microbial activity may be present, but unable to cope with a high rate of continuous hydrocarbon input (*e.g.*, an area of the North Sea described by Massie *et al.*, 1985b). This example is relevant to possible post-spill scenarios, *e.g.*, continued leaching of relatively fresh oil from thick upper shore deposits to areas further down the shore.

The sediment may be oxygen- and/or nutrient-limited for various reasons, or if the route of the hydrocarbons to the seabed has been long and aerobic, the residues entering the sediments may be mainly complex high molecular weight compounds, which are relatively resistant to biodegradation. For example, in Milford Haven bottom sediments the bulk of the hydrocarbons is from degraded petrogenic sources, possibly representing the degradation residues of a wide variety of oil types (Little *et al.*, 1987).

Other Pathways. Many marine organisms including fish, crustaceans, and molluscs have been shown to have the capacity to convert hydrocarbons to metabolites; aromatic compounds have been studied in particular. The conversion of aromatic compounds to oxygenated derivatives takes place via enzyme (mixed-function oxygenase) systems. Malins (1981) gives examples of metabolic pathways and their products, which include glucuronides, glycosides and mercapturic acid derivatives.

Several studies reviewed by Malins (1981) have indicated that hydrocarbons are readily depurated from body tissues if organisms are placed in 'clean' environments; see also Page *et al.* (1987). However, the hydrocarbon metabolites tend to increase or remain constant in organisms for long periods. These may affect the organisms concerned; for example, mussels from the *Amoco Cadiz*-oiled sites had increases in polar aromatic compounds, and this was associated with observed cellular damage, *e.g.*, increases in lipid and lysosomal granules (Malins, 1981).

The metabolic activities of marine organisms can transform hydrocarbons (so removing some of them from the environment). Relatively little is known about the environmental significance of the oxidation products, some of which are also toxic compounds. However, because of the rapid dilution of the compounds in the water column compared with their slow rate of production, they are unlikely to have significant ecological impacts.

NATURAL RECOVERY

BIRDS

Seabirds are among the most conspicuous casualties of oil slicks and, as such, attract considerable public attention. But there is no reason to suppose that, from a biological point of view, this mortality is damaging to seabird populations. Arctic and sub-Arctic seabirds also suffer heavy mortality from natural causes and from fishery practices. Even the auks, which because of their very low reproductive rate might be expected not to be able to make good these losses, have sustained their population; and there is no evidence that other seabirds with a greater reproductive potential have declined in numbers.

Losses of Seabirds from Oil

Estimates of the number of casualties of seabirds from oil slicks are highly speculative. The only firm figures are counts of the number of oiled birds coming ashore, but these are subject to severe limitations depending on the intensity of the search, accessibility of the shoreline to observers, *etc.*, and some carcasses may have become oiled after death from natural causes. An unknown number of oiled birds may die at sea and not reach the coast. Tests by the RSPB (1979), when marked seabird corpses were released at sea, gave variable recovery rates on the shore, depending on the distance and direction of the coast, wind speed and direction, and accessibility of the shoreline to observers. Sea conditions are also likely to have a radical influence on the number of birds coming ashore.

There is little relation between the size of an oil spill and the number of seabird casualties. At least 12,000 birds of various species were killed when one or more oil slicks (which were never positively identified despite aerial surveillance, and on that account were probably small) moved up the coast of northeast England and east Scotland in January and February 1970 (Greenwood *et al.*, 1971).

One of the largest kills of seabirds by oil on record was in the Skagerrak (strait between Denmark and Norway) in January 1981, when some 30,000 oiled birds appeared on neighbouring beaches (Mead and Baillie, 1981). This was caused by a relatively small amount of oil released by perhaps two ships (RCEP, 1981). On the other hand, the greatest volume of oil released in any shipping accident, following the wreck of *Amoco Cadiz* on the Brittany coast in March 1978, when 230,000 tons of crude oil and bunker fuel were released over a period of 3 to 4 weeks, caused the known death of only 4,572 birds (Hope-Jones *et al.*, 1978), though the actual total was undoubtedly higher.

The greatest concern must be about the repeated losses of seabirds from oil pollution in areas where oil slicks occur frequently and pollution verges on chronic. The English Channel and North Sea are among the most heavily trafficked sea lanes in the world, and casual oil discharge from shipping has been a regular feature since the 1920's. The surrounding coasts are populated with bird-lovers, and monitoring of the numbers of oiled birds coming ashore has been unusually complete (Bourne, 1976).

Experience of the incidence and consequences of oil pollution in the northeast Atlantic offers the best available factual base for an assessment of the impact of oil on seabirds. The claim of Morzer-Bruyns and Tanis (1968) that 150,000 to 450,000 seabirds are killed annually by oil in the North Sea and North Atlantic has a slender factual base and may be an order of magnitude too high (Dunnet, 1982; Clark, 1987); but this chronic and small-scale oil pollution may be responsible for at least as many seabird deaths as those resulting from spectacular accidents (Croxall, 1977). However, if the annual mortality from oil is of the order of tens of thousands of seabirds a year, this must be set against a calculated annual mortality from natural causes of well over one million (Dunnet, 1982).

Other Causes of Mass Seabird Mortality

Many seabirds suffer mass mortality from causes other than oil. Harsh weather in winter and prolonged storms prevent their feeding and they become severely emaciated; if they do not starve, they are at risk from any additional stress. 'Wrecks' (mass mortality) of seabirds are erratic but frequent and may result in losses comparable to those following major oiling incidents. In autumn 1969, some 12,000 seabirds, mainly murres, died in the Irish Sea and western Scotland. Some were slightly oiled, some contained high levels of PCB's in their body fat, but all were severely emaciated and the most significant cause of death appears to have been prolonged storms (NERC, 1971). The wreck of the *Amoco Cadiz* appears to have coincided with a 'wreck' of puffins on the Brittany coast from natural causes, which augmented the casualties (Hope-Jones *et al.*, 1978). Even larger 'wrecks' of guillemot (100,000+ birds) have been reported from Alaska (Bailey and Davenport, 1972).

Failure of the food supply is a natural hazard of seabirds at high latitudes. Croxall and Prince (1980) reported that the absence of swarms of krill around South Georgia in 1977 to 1978 (krill are shrimp that are a major food source for many animals in the Antarctic) resulted in the failure of krill-eating birds, mainly gentoo penguin and black-browed albatross, to rear their chicks. The reproductive failure of puffins on the island of Rost in the Lofoten Islands in most years of the 1970s appears to have been due to a shortage of young herring, sand eel, and sprats, which are the principal food of the young (Mills, 1981).

The use of mono-filament gill nets is responsible for heavy seabird losses. These nets may be 30 or more kilometres long and trap not only fish, but also diving seabirds and sea mammals (seals, dolphins, *etc.*), which then drown. Nets that break free may drift at sea for a very long time (they are virtually indestructible) and 'fish' passively throughout that period. It has been estimated that the Danish salmon drift-net fishery in the North Atlantic killed 250,000 to 750,000 seabirds, mostly Brunnich's guillemot, each year between 1965 and 1975 when the fishery was exploited. The Japanese salmon fishery in the North Pacific and Bering Sea is estimated to have taken a toll of between 214,500 and 715,000 seabirds per year between 1952 and 1975, and between 350,000 and 450,000 per year during the 1975 to 1978 period (Clark, 1989). These figures are of the same order of magnitude as the most pessimistic estimate of the annual mortality of seabirds from oil in the northwest Atlantic and are quite possibly ten times greater.

Population Effects

While the death of oiled seabirds attracts a great deal of public concern, from a purely biological point of view what matters is not these deaths but the number and fate of the survivors. Most animals overproduce young, often on a colossal scale, and nearly all of them die before reaching the age of reproduction. Additional deaths from exposure to oil may be insignificant in comparison with the mortality from natural causes. Animals with a large breeding potential may rapidly make good such losses. Mortality is only significant if it results in a substantial decrease in the population, particularly in the breeding population (McIntyre *et al.*, 1978).

Auks present the worst-case scenario. They spend nearly all their time on the surface of the sea in dense flocks. Not only are they particularly at risk from floating oil, but casualties are likely to be large. More importantly, their breeding biology does not suggest that they would be able to replace losses from oil rapidly. Murres do not breed until they are 3 to 7 years old and then do not breed every year. They nest on precarious cliff ledges and lay only one egg. The eggs or chicks often fall off the ledges or are taken by predators. In one Scottish colony, common guillemots had only a 20 percent chance of successfully rearing a chick (Southern *et al.*, 1965), and a similar low success rate has been found in colonies of Brunnich's guillemot in Canada (Tuck, 1961). If the egg or chick is lost, the birds abandon breeding for that year. Once a chick successfully reaches the sea, it has few predators and has an average annual survival rate of over 90 percent (Dunnet, 1982). Thus the larger auks and a number of other seabirds have a life expectancy of 20 years or more.

A decline in the southerly populations of several species of auks that suffer heavy casualties from oil, including murres and razorbills (Tuck, 1961; Clark, 1978) and puffins (Bourne, 1971; Kress, 1977), has taken place on both sides of the Atlantic. This apparently confirmed the prediction that birds with such a low replacement rate could not sustain repeated losses from oil and that this population decline was the result. However, detailed censuses have shown that while colonies at the southern fringe of their geographical range have been declining, others (some equally exposed to oil) are either stable or have shown a dramatic increase in recent years (Harris and Murray, 1981; NERC, 1977). These population changes are thought to be due to the amelioration of climate in the North Atlantic during this century and the northward shift of the main centres of population of sub-Arctic species (Clark, 1984). Oil has not been a factor.

The growth of some colonies has been so rapid as to confound previous predictions, and how it has come about is still uncertain (Dunnet, 1982). Leslie (1966) calculated, from what was known about replacement and survival rates, that it would take a colony of guillemots 53 years to double in size. In other words, if half the colony were killed by an oil spill, restoration of the numbers would take half a century by natural growth. More sophisticated simulations specifically related to guillemot populations in the Bering Sea and Gulf of Alaska yielded comparable predictions of recovery time of 20 to 40 years, depending on the size and location of an oil spill (Ford *et al.*, 1982), and 70 years (Samuels and Lanfear, 1982). Clearly, the increase in northern populations of auks has been so widespread that the numbers in a depleted colony cannot have been made good simply by immigration from other colonies and at their expense. One explanation may be that young birds are entering the breeding population at an earlier age than previously (Dunnet, 1982). The large population of young, non-breeding adults therefore provides a reservoir from which depleted breeding colonies can be replenished if necessary.

Sea ducks, the other major casualties from oil, have much greater recovery potential. They start to breed at a younger age, lay much larger clutches of eggs, and replace lost eggs (Dunnet, 1982).

Only diving sea ducks suffer casualties from floating oil on a scale similar to that of the auks, but they have a much higher reproduction rate and seem well able to withstand this mortality without affecting the population size. Reports of a decline in the numbers of old squaw (long-tailed duck) and velvet scoter migrating through the Baltic (Bergmann, 1961; Lemmetyinen, 1966), which was attributed to the consequences of exposure to oil in the North Sea winter quarters, were based on estimates, not reliable censuses. Whatever the value of these observations, such a claim has not been made since then.

An example of the recovery potential of sea ducks is given by the experience of an oil spill in Finnish waters in the northern Baltic that affected a colony of eider duck. These birds have a relatively small clutch size compared with other sea ducks and tend to remain attached to a particular site. Following the grounding of the tanker *Palva* in 1969, 25 to 33 percent (2400 to 3000 birds) of the local eider colony were killed, but in the following year, the number of breeding birds was fully restored; indeed, the eider population is said to have been exceptionally large (Leppakoski, 1973).

Cleaning and Rehabilitation

Cleaning and rehabilitation of oiled seabirds are possible if adequate facilities are available. These efforts may be carried out on humanitarian grounds but have no ecological value (Clark, 1978; NRC, 1985).

Polar Conditions

Almost all the reliable scientific information about the impact of oil on seabirds is, understandably, from the northeast Atlantic and the North American coast south of Newfoundland. At higher latitudes, colonies of auks are an order of magnitude larger in size than colonies farther south, and flocks on the water are correspondingly greater. Although there is no evidence, it must be expected that casualties in an oil spill incident would be greater in higher latitudes than those in northeast Atlantic waters. However, Arctic populations have not suffered from recurrent exposure to oil for the last 70 years in the same way. Arctic populations of auks are said to be already in decline (NRC, 1985), but in the absence of censuses (impracticable under the circumstances), this claim, although it may be true, is without firm foundation.

The reader is referred to Appendix C for details concerning effects of oil on birds.

FISHERIES

There is a general consensus among fishery authorities that oil spills are damaging to fin-fisheries by excluding fishermen from fishing grounds for the period while oil is on the water, by the fouling of fishing gear, but most importantly by the public's fear of tainting of the fish which can have a serious effect on the market. Mobile fish species appear to be able to avoid oiled areas, and fish kills among them have not been recorded. Non-mobile, inshore rockfish may be killed, and, of course, fish in mariculture enclosures cannot escape and are likely to be killed if exposed to oil.

There can be a heavy loss of pelagic eggs and fish larvae if these are present at the time of an oil spill, although this has rarely been observed directly. In most cases this mortality has had no detectable impact on the fish stocks available to the fishing industry. Annual recruitment to these stocks fluctuates naturally, and the size of the catchable stock is determined more by the activities of the fishing industry (*e.g.*, overfishing) and by climatic changes than by any other factor.

Shellfish (*e.g.*, clams, oysters) in inshore or intertidal sediments are very vulnerable to oil, and severe and protracted damage may be caused to them by an oil spill.

Interference

The immediate effect of an oil spill is the exclusion of fishermen from fishing grounds where there is a risk of encountering floating oil. Fishing gear cannot be shot, or if an oil slick drifts through an area where fishing is in progress, the fishing gear is likely to become fouled with oil. The catch is then valueless and the gear has to be replaced. Fishing vessels are likely to be fouled also, but they can be cleaned.

Gear fouling by oil is most damaging in the case of fixed installations, though it is possible to protect them by the deployment of booms, *etc.*

Tainting

The greatest commercial impact of any resulting from oil spills probably comes from the public's fear of tainting (see, for example, Fairhall and Jordan, 1980; Clark, 1989).

Tainting is the change in the characteristic flavour or smell and may be caused by petroleum hydrocarbons being taken up in the tissues or contaminating the surface of the catch. Light to middle boiling range oils are the most potent source of taint, but it can be caused by any oil (Whittle, 1978). The concentration of oil required to cause tainting varies widely with the oil and the fish concerned (McIntyre, 1982). Fatty fish, such as salmon or herring, develop taint more readily than non-fatty fish; but even salmon lose their taint after 4 weeks, despite continuous exposure to oil (Brandal *et al.*, 1976).

In some countries a particular hazard occurs when shellfish (shrimp, molluscs) are boiled in bulk before marketing. One or two specimens with adhering oil can then taint the whole batch.

Commercial shellfish and finfish markets are subject to irrational reactions in a way not experienced by other food markets. Even the suspicion that seafood may be tainted is sufficient to depress the market significantly. Fish sales on the Paris market fell by half during the period of the *Torrey Canyon* oil spill, regardless of the quality or origin of the fish (Korringa, 1968).

Crude oil and refined products contain polyaromatic hydrocarbons (PAH), some of which are carcinogenic to mammals. Molluscs, in particular, are efficient accumulators of PAH, though they rapidly lose them when transferred to clean water. Human exposure to PAH through the consumption of contaminated seafood is not regarded as a public health risk, partly because of the public rejection of tainted seafood, but also because the total exposure to PAH from this source is small compared with that from other foods (cabbage, spinach, smoked or grilled fish or meat) (King, 1977; RCEP, 1981). Cabbage, for example, can contain twice the concentration of benzo(a)pyrene (a carcinogenic PAH) as clams from relatively polluted waters (RCEP, 1981).

Sublethal Effects

Experimental studies have shown that various pathological conditions, including fin erosion, ulceration of the integument, liver damage, and lesions in the olfactory tissue, can be induced in fish exposed to oil. However, it is not clear that any of these conditions, when observed in fisheries, are related specifically to exposure to oil; they appear to be a response to pollution-induced stress in general.

Fin erosion affects the caudal (tail) fin of mid-water fish such as mullet, or the posterior fins most in contact with the seabed in bottom-living flatfish (Desaunay, 1981). Fin erosion is often accompanied by the development of abnormal bent fin rays. While a proportion of fish from any waters show these conditions, an abnormally high incidence of the condition has been reported in flatfish living on contaminated sediments in many parts of the world (Sindermann, 1982).

Following the wreck of the *Amoco Cadiz* on the Brittany coast, three species of flatfish (plaice *Pleuronectes platessa*, sole *Solea vulgaris*, and dab *Limanda limanda*) caught in two heavily oiled bays showed the following proportions with fin erosion: April 1978 (immediately after the oil spill), 0 percent; December 1978, 90 percent; May 1979, 73 percent; October 1979, 2.5 percent (Desaunay, 1981). It is not known if the decline in the proportion of affected fish was due to the recruitment of a new year-class, immigration from outside the affected area, recovery of the affected animals, or differential mortality of them.

Minchew and Yarbrough (1977) studied fin erosion in mullet kept in experimentally oiled brackish-water ponds. Caudal fin erosion was first noticed 12 days after the spill, and by 13 days all samples of exposed fish had fin erosion. Thirty-four days after the spill, some regeneration of fins was noted and this was still in progress on the 56th day.

Six months after the wreck of the *Amoco Cadiz*, 50 to 80 percent of a catch of mullet (*Mugil cephalus*) were found to have ulcerated bodies (Balouet and Baudin-Laurencin, 1980). There is no positive evidence to link this condition with the previous oil spill, but mullet maintained in experimentally oiled ponds (see above) developed integumental lesions (Minchew and Yarbrough, 1977).

Liver pathology has been demonstrated in English sole (*Parophyrys vetulus*) after being maintained for 4 months on experimentally oiled sediments (McCain *et al.*, 1978), and a similar condition developed in Atlantic croaker (*Micropogon undulatus*) exposed to water-soluble fractions of crude oil (Eurell and Haensley, 1981). This condition, like fin erosion, is unlikely to be particularly related to exposure to oil, but is a general indicator of stress (McCain *et al.*, 1978). Liver lesions have been reported in killifish (*Fundulus heteroclitus*) sampled in an area that had been affected by an oil spill 8 years previously (Sabo and Stegeman, 1977), but there is no direct evidence to connect the pathological condition with the oil.

Pathological and degenerative changes in the olfactory membranes have been recorded in Atlantic silversides (*Menidia menidia*) (Gardner, 1975) and larval sand soles (*Psettichthys *melanostichus*) (Hawkes, 1980) experimentally exposed to water-soluble fractions of crude oils. These pathological conditions have not been reported in fish in the circumstances of oil spills, but no particular search has been made for them.

Eggs and larvae are more sensitive to the toxic effects of oil. Johannessen (1976) showed that the water-soluble extracts of Ekofisk crude reduce the hatching success of fertilized capelin eggs at concentrations of 10 to 25 ppb, and Tilseth *et al.* (1981), using the same oil, found reduction in growth and change in buoyancy of cod eggs and larvae after 14 days at 50 ppb. Larvae exposed to 250 ppb developed malformations of head and jaws that interfered with feeding.

There are numerous reports of petroleum-induced abnormalities in embryos and larvae of fish (Sindermann, 1982). These include malformed jaws, flexures of the vertebral column, reduced heart rate, loss of coordination and equilibrium, and degeneration of neurosensory cells. Following the *Argo Merchant* oil spill off Nantucket Island, eggs of cod and pollock showed cytological abnormality of the embryo's cells and arrest of cell division (Longwell, 1978).

Conan (1982) reported non-pathological sublethal effects in mullet and flatfish in the heavily oiled estuaries of the Brittany coast following the wreck of the *Amoco Cadiz*. Sediments in these low-energy embayments continued to release oil for two or three years following the spill, and during this period the fish showed reduced growth, fecundity, and recruitment.

Experience in Oil Spills

Following the wreck of the *Amoco Cadiz*, there was an immediate kill of several tons of rockfish at the site (CNEXO, 1981); but generally fish appear able to leave an oiled area, and kills of adults from the effects of oil are rare. Indeed, during the period when oil slicks were in the Santa Barbara Channel following the blowout in 1969, fish shoals were observed from the air by professional fish spotters in areas not covered by oil, and no heavy mortality of fish was recorded (Abbot and Straughan, 1969). After the *Tsesis* oil spill in the Baltic, and the wreck of the *Betelgeuse* in Bantry Bay, Ireland, herring (and also sprat in Bantry Bay) migrated through the oiled areas and spawned normally (Linden *et al.*, 1979; Grainger *et al.*, 1980).

The only important casualties from oil spills are of shellfish (crabs, clams) in shallow-water or intertidal sediments. The West Falmouth, Massachusetts, diesel spill in 1969 caused great initial mortality, and effects on crab populations were still obvious 7 years later (Burns and Teal, 1979). Populations of the clam *Mya arenaria* were still adversely affected 6 years after the grounding of the *Arrow* in Chedabucto Bay, Nova Scotia (Thomas, 1978).

Although recorded as 'sublethal', developmental abnormalities must be expected to lead to the early death of the young fish. Since they and damaged eggs sink to the bottom and rapidly decompose, it is difficult to estimate these losses in any oil spill. However, Longwell (1978) reported that following the *Argo Merchant* oil spill, 20 percent of cod eggs and 46 percent of pollock eggs in the spill zone were dead or moribund. Smith (1970) reported that 90 percent of pilchard eggs were killed in waters exposed to oil from the *Torrey Canyon*, compared with a 50 percent mortality more distant from the oil; but the situation in that case is complicated by the extensive use of very toxic dispersants in that incident. On the other hand, after the explosion of the *Betelgeuse* in Bantry Bay, Ireland, oil leaked from the wreck for 18 months and dispersants were frequently used. During the spring, sprats and whiting spawned in the area, but no adverse effects on eggs or larvae were detected (Grainger *et al.*, 1980).

The loss of fish eggs and larvae from oil exposure must be seen against the normal mortality, which is on a colossal scale. Only a minute proportion of larval fish survive to an age when they reach a commercial size. Furthermore, most fisheries are based on fish of various ages, and if the size of one year-class is reduced, that is unlikely to have more than a marginal effect on the commercial catch. Calculations made by Johnston (1977) suggest that, making the most pessimistic assumptions, even a catastrophic oil spill (400,000 tons) in the North Sea would be responsible for a loss, from all causes, of 13,000 tons of fish. Since the annual commercial catch is 4.36 million tons, this shortfall would be hard to detect, particularly against the natural fluctuations in fish abundance. In fact, the only case in which an oil spill seems to have caused a shortage of finfish was following the wreck of the *Amoco Cadiz*, when the 1-year-old class of flatfish was thought to have been reduced (CNEXO, 1981).

SEA MAMMALS

There is remarkably little reliable information in the scientific literature on which to make an assessment of the threat of oil exposure to marine mammals. Most species appear to ignore floating oil and are unharmed when they encounter it.

Sea mammals include cetaceans (whales, porpoises, and dolphins), the tropical sirenians, seals and sea lions, sea otters, and, to a degree, polar bears. A number of land mammals (mink, rats, *etc.*) regularly forage on the shore and are in part dependent upon the sea.

Cetaceans and sirenians are entirely aquatic; seals, sea lions, and sea otters spend most of their time at sea but return to land at least for breeding; polar bears are primarily terrestrial mammals, but readily enter the sea.

All sea mammals are air-breathing and must surface from time to time, thus risking exposure to floating oil. Polar bears, other shoreline foraging mammals, and seals and their pups in the breeding colonies on shore are also at risk from stranded oil.

Most seals are colonial breeders and very large numbers may be exposed to a single oil slick that impacts a rookery. Cetaceans often travel in groups (pods) and numbers of them may encounter a single oil slick.

Effects of Oil

Sea otters and polar bears rely primarily upon their dense fur to provide thermal insulation. As with seabirds, if the pelage is matted with oil, water penetrates to the skin and the animals rapidly lose body heat. Kooyman *et al.* (1977) and Costa and Kooyman (1980) report a 5 to 10 percent decrease in subcutaneous temperature below experimentally oiled areas of fur and almost a doubling of the body metabolism to preserve body core temperature. Oil coating of isolated polar bear fur led to a tripling of conductance of heat across the skin (Hurst *et al.*, 1982; Oritsland *et al.*, 1981), and the effect was even greater in the presence of wind. Subcutaneous temperatures fell, and metabolic rate increased to preserve body core temperature, in the same way as in sea otters. Clearly, as with oiled seabirds, this situation, if prolonged, could lead to death from hypothermia. However, sea otters (Williams, 1978) and polar bears (Oritsland *et al.*, 1981; Engelhardt, 1981) readily groom oiled fur; and while this results in the ingestion of oil with undesirable consequences, it is likely that death from hypothermia can be avoided except for heavily oiled animals.

Other marine mammals rely upon a layer of subcutaneous fat (blubber) to provide thermal insulation and are not vulnerable in the same way. Oil adheres to the fur of seals but is readily washed off by immersion in the sea. The fur of ringed seals that had been experimentally exposed to crude oil for 24 hours was clean again within 24 hours after return to clean water (Smith and Geraci, 1975; Geraci and Smith, 1976), and 58 free-ranging elephant seal pups that had initially been more than 75 percent covered with oil were, with one exception, found to be clean one month later (Le Boeuf, 1971). Sea lions, walruses, and cetaceans have little or no body hair, and oil is unlikely to adhere to them, although the NRC (1985) report speculates that rugosities, roughened skin, and encrusting barnacles on some whales might provide a site for adhering oil.

Baleen whales feed by filtering plankton from the water; if oil is present, the baleen plates collect oil particles or become coated with oil. Accumulations of oil on bowhead baleen reduce its filtering efficiency by 10 percent when coated with Prudhoe Bay crude oil, and by 85 percent when coated with a waxier oil (Braithwaite, 1983). However, this accumulation is unlikely to be important in the long-term feeding strategies of baleen whales (Geraci and St. Aubin, 1982).

If sea mammals surface through an oil slick, the nostrils and eyes are likely to be coated with oil. The former is not a problem, but eye damage, such as conjunctivitis, has been reported in oiled seals (Smith and Geraci, 1975; Geraci and Smith, 1976; Nelson-Smith, 1970; Morris, 1970), although the condition is rapidly reversed. Eye damage is, in any case, common in natural populations of seals (King, 1964; Ridgway, 1972).

Avoidance of Oiled Waters

Captive bottle-nosed dolphins are able to detect and avoid oil on the water surface (Geraci *et al.*, 1983). However, there is abundant evidence that sea otters (Williams, 1978), seals

(Spooner, 1967), and a variety of cetaceans (Goodale *et al.*, 1981; Geraci and St. Aubin, 1982) do not actively avoid oil slicks, whether or not they have the ability to detect them.

Casualties in Oil Spills

Although Nelson-Smith (1970) reported the death of grey whales, a dolphin, northern fur seals, Californian sea lions, and northern elephant seals as a result of oiling following the Santa Barbara blowout, the link has been questioned by Simpson and Gilmartin (1970), Brownell and Le Boeuf (1971), and Le Boeuf (1971). Mortality of seals following oil spills has been attributed to the effect of oil on the coast of Wales (Davis and Anderson, 1976) and following the *Torrey Canyon* (Spooner, 1967), *Arrow* (Anon., 1970), and *Kurdistan* (Parsons *et al.*, 1980) oil spills, and an oil spill in the Gulf of St. Lawrence (Warner, 1969). It is possible, of course, that these attributions were correct, but in no case was the cause of death clearly established. The numbers of animals involved in all these instances were very small.

PELAGIC ENVIRONMENT

Plankton

> *Although the toxic components of petroleum oils can kill planktonic organisms, the losses are rapidly replaced by immigration from outside the affected area or by the reproduction of survivors, most of which have a short lifespan and a very high replacement potential. Because of the complexity of the planktonic ecosystem, its rapid changes, patchy distribution, and movement by water currents, it is impossible to detect more than very transient effects on plankton by even the largest and most damaging oil spills. There is no evidence that even these effects have any ecological significance.*

Since plankton plays a critical role in most marine food chains, it is important to know the effect of oil spills on the planktonic community. Because the plankton occupies the upper layers of the sea, it is particularly exposed to toxic water-soluble elements leaching from floating oil as well as to microscopic droplets of emulsified oil. However, realistic study of the effect of oil on plankton is extremely difficult (Dicks, 1976; Colebrook, 1979), and most investigations have suffered from various disadvantages.

The planktonic ecosystem is very complex, containing as it does a large variety of single-celled plants (phytoplankton), many small crustaceans, the larvae of perhaps the majority of marine invertebrates as well as adults representing virtually all invertebrate groups, and the eggs and larvae of many fish species. Plankton is also very dynamic, with huge and rapid seasonal changes in species composition and the diurnal vertical migrations of many of the zooplankton (Davenport, 1982). It is also extremely patchy in its distribution (Cushing, 1953). All these factors make comprehensive field studies difficult.

Laboratory Studies. Toxicological studies in the laboratory have yielded little information that can be related to the consequences of oil spills for marine plankton. As Davenport (1982) points out, many of the organisms are small and delicate, and zooplankton species can show rapid changes of structure, physiological state, and nutritional requirements. Toxicological studies have therefore tended to focus on easily managed, robust species such as barnacle larvae, larger copepod species, fish eggs, or pure bacterial or algal cultures. It is unlikely that these are representative of a wider range of plankton species. Because of the rapid turnover of some of the species that have been used, or changing developmental stages of others, long-term exposure is generally impracticable and the experiments have therefore tended to be very short-term.

If the organisms are too varied to permit a 'representative' selection of species to be studied, the toxins in crude oil present a similar problem because they are equally varied. In most experiments, whole oil or water-soluble extracts are used and contain an unknown quantity of usually unknown and variable toxic constituents. Often, little regard is taken of the fact that crude oils from different oilfields differ greatly in their constituents. The alternative approach, that of using defined hydrocarbons in toxicity tests, gives greater precision but does not address the situation represented by an actual oil spill.

Crude oil and oil fractions are toxic to a variety of planktonic organisms. Oils with a high aromatic content are usually more toxic than others (Anderson *et al.*, 1977); weathered oil is much less toxic than fresh crude (Lee and Nicol, 1977); crude oils from different oilfields vary in their toxicity to developing eggs of cod, herring, and plaice (Kuhnhold, 1972). Lee and Nicol (1977) showed that coastal plankton is more resistant to fuel oil contamination than oceanic plankton. At low concentrations of petroleum hydrocarbons, below 30 to 50 ppb, photosynthesis is stimulated, presumably because of a nutritive effect; above 50 ppb it is progressively reduced (Gordon and Prouse, 1973). Feeding was depressed and food selection changed in two species of copepod at concentrations of about 250 ppb (Berman and Heinle, 1980). A variety of other sublethal effects have been recorded in zooplankton, but at concentrations close to lethal limits (Anderson *et al.*, 1977; Davenport *et al.*, 1979).

On the other hand, a number of planktonic crustaceans accumulate oil constituents rapidly from solution, evidently without coming to harm, and lose these contaminants when returned to clean water (Lee, 1975; Corner *et al.*, 1976). Parker *et al.* (1971) found that barnacle larvae and also a copepod, when maintained in a suspension of crude oil, swallowed droplets of crude oil which then appeared in the faeces, apparently unchanged, without harming these crustaceans.

Experiments with Large Enclosures. The acknowledged severe limitations of laboratory studies on marine plankton have led to the use of large enclosures in an attempt to bring greater realism to controlled experiments. These are large plastic enclosures ('big bags') of about 100 m^3 capacity. These enclosures are anchored in sheltered waters and isolate the natural plankton community, which can then be subjected to appropriate experimental procedures under reasonably natural conditions. Several such enclosures were constructed in the Saanich Inlet, Vancouver Island, and Loch Ewe, Scotland (an issue of the *Bulletin of Marine Science*, Volume 27, Part 1, 1977, was devoted to an account of the initial experiments, concerned with metal pollution, at both sites). Steele (1979), in a review of the achievements of such experimental ecosystems, says in summary that "the experiments conducted so far in large experimental ecosystems have probably taught us more about the general ecological interactions in such systems than about subtle long-term effects of pollutants."

Large experimental ecosystems offer substantial advantages over laboratory studies in that they contain a representative sample of the local plankton and permit interactions between different organisms in the planktonic ecosystem. The critical disadvantage is that by enclosing a section of the environment, horizontal movement of plankton is prevented, and that can be a critical departure from the natural situation, depending on local tidal flows and currents. The growth of organisms on the walls of the enclosure produces abnormal conditions at the periphery. While the larger the enclosure, the more realistic the conditions, the patchiness of plankton makes reliable sampling more difficult; furthermore, since large enclosures are very costly to build and maintain, fewer experiments can be carried out (Davenport, 1982).

In a series of Canadian experiments (Lee and Takahashi, 1977; Lee *et al.*, 1977), 10 ppb, 20 ppb, and 40 ppb of non-volatile hydrocarbons were added to enclosures. The lowest concentration had no effect, but at the two higher concentrations, there was a considerable increase in phytoplankton production and some change in species composition. There was a decline in the population of the diatom *Ceraulina* but a massive increase in the numbers of the micro-flagellate *Chrysochromulina* and also of the rotifers that feed upon it. The larger zooplankton species were unaffected.

Similar experiments, using a larger enclosure and an initial concentration of 100 ppb hydrocarbons of North Sea oil extract, were carried out in Scotland (Davies *et al.*, 1980). In this case, no effect on phytoplankton production or species composition was found, but copepod populations fell dramatically. This was due to a direct effect on the adults and also to the failure of eggs to develop.

Experience with Oil Spills. Toxicological studies and even experiments in large experimental enclosures give no realistic guidance as to the likely impact of an actual oil spill on the planktonic community. The enormous variety of organisms contributing to the community, the patchy distribution of them, and the rapid changes in species composition that occur, all make its study very demanding in manpower, resources, and time, so that field studies of acute impact on the plankton are rare and no definite statement can be made from them.

No effect on plankton was noted after the wreck of the *Torrey Canyon* (Nelson-Smith, 1970), the Santa Barbara blowout (Straughan, 1972), or the *Argo Merchant* oil spill (Kuhnhold, 1978), but these incidents were not followed by a detailed study of the plankton, and only gross changes are likely to have been detected.

A detailed study of plankton was made after the spillage of 1000 tons of No. 5 fuel oil from the tanker *Tsesis* in the northern Baltic Sea (Johansson *et al.*, 1980). Approximately 700 tons of the oil were collected, but the remaining 300 tons were deliberately left untreated and the effects monitored for one month. Close to the wreck, where oil concentrations were highest, the zooplankton biomass declined dramatically, but it is not known if this was due to mortality or avoidance of the area. Substantial recovery occurred within 5 days. Phytoplankton biomass and productivity increased in the area, probably because of the absence of grazing zooplankton.

Following the *Amoco Cadiz* oil spill, which occurred on March 16, 1978, the composition of zooplankton in impacted coastal areas was normal in early April. The spring phytoplankton bloom appears to have been depressed; but by June no differences in population composition could be detected, possibly because of replenishment from outside the affected area (Laubier, 1978). The sheltered inlets ('abers') with low water exchange, which were heavily contaminated with oil, may have taken longer to recover.

BENTHIC ENVIRONMENT

Littoral Zone (Intertidal Zone)

For all shoreline types, biological recovery processes have been observed in the intertidal zone. The recovery processes occur concurrently with weathering of the oil, over periods of time that commonly range from 1 to 10 years. The time scale is influenced by shoreline characteristics (especially energy level and substrate grain size), oil type and concentrations, and the biological characteristics of the shore.

Several studies that have investigated longer-term effects of oil on a variety of shores are summarised in Table 7. In general, recovery from most oilings has been good within the time scale of 1 to 10 years. Within this time scale, rocky shores appear to recover more quickly than soft-sediment and salt-marsh shores. It is usual for exposed rocky shores to show good recovery within 2 years—the exception in Table 7 (Westwood *et al.*, 1989) concerns the more lengthy successional changes that occur after drastic cleaning. Sheltered sediment shores may retain oil to a much greater extent, and some reports (*e.g.*, Gilfillan and Vandermeulen, 1978; Krebs and Burns, 1978; see Table 7) show that recovery can take more than 6 or 7 years. Brittany salt marshes that were oiled following the *Amoco Cadiz* spill took 5 to 8 years to recover (Baca *et al.*, 1987). Variations within this time scale depend on marsh type, degree of initial oiling, and cleanup treatments.

Recovery may take longer than 10 years in exceptional circumstances, for example, if extensive asphalt pavements are present, or if relatively toxic oil becomes trapped in anaerobic sediments. Asphalt pavements are commonly 5 to 10 cm thick and 1 to 30 m wide; an exceptional one following the *Metula* spill was 400 m wide (Hann, 1977). Pavements develop a weathered crust, but the oil inside may remain relatively unweathered for long periods of time. The pavement provides a relatively stable substratum (in contrast to more mobile sand and gravel), and the weathered surface allows colonisation by algae and invertebrates, but this community is different from the 'normal' community for such a beach. Pavements are gradually eroded; they persist longest on the upper shore, where they can constitute a physical barrier that restricts recolonisation by plants such as grasses and shrubs (Guzman and Campodonico, 1981).

Oil may be incorporated into anaerobic sediments, penetrating along pathways provided by the burrows of worms, molluscs and crustaceans, and the stems and root systems of marsh plants. Under normal conditions these pathways allow the penetration of oxygen into sediments that would otherwise be anaerobic. A possible problem following oiling is that there is subsurface penetration of oil, followed by death of the organisms that normally maintain the pathways. The pathways then collapse; *e.g.*, burrows become filled in with sediment from the top if they are not actively maintained. Thus oil can be trapped in anaerobic sediment where its degradation rate will be very low, and organisms trying to recolonise may encounter toxic hydrocarbons. Under these conditions oil-tolerant opportunistic species are favoured, *e.g.*, the worm *Capitella capitata*. The activity of such organisms helps the process of oil degradation, so that eventually more sensitive species can return, *e.g.*, the fiddler crabs (Krebs and Burns, 1978).

Although long-term studies are incomplete, some authors have speculated concerning recovery rate; *e.g.*, Conan (1982) suggests that as much as 30 years or 3 to 6 generations are required for recovery of a normal age distribution of clams. Another limitation of the available information, as pointed out by Vandermeulen (1982), is that only selected biological effects and recovery rates have been studied out of a very broad range of possibilities. For example, it is known that there is a dose-response relationship between aromatic hydrocarbons in mussel tissues and physiological responses such as feeding rate, respiration, and 'scope for growth' (Bayne *et al.*, 1982; Widdows *et al.*, 1987), but the ecological significance of these physiological changes in natural populations is not well-researched. These limitations in information should be borne in mind when considering overall recovery rates.

Two observations concerning the presence of residual oil as related to biological recovery can be extracted from Table 7. On the one hand, removal of oil by using drastic physical methods, beyond initial bulk oil removal, does not necessarily improve biological recovery times; on the contrary, it may prolong recovery (Baca *et al.*, 1987; Westwood *et al.*, 1989) if the cleanup removes living organisms and alters the habitat. On the other hand, the presence of residual oil does not necessarily prevent biological colonisation. For example, the growth of a number of organisms on asphalt pavement (Guzman and Campodonico, 1981) demonstrates the low toxicity of weathered residues. It may be justifiable to remove heavy deposits of oil where they have killed the underlying organisms and the shore is not being recolonised readily; a salt-marsh example is given by Guzman and Campodonico (1981). Transplants should be considered in such cases.

Animals with body burdens of hydrocarbons are likely to have measurable physiological changes, but they can survive and eventually depurate the hydrocarbons (Boehm *et al.*, 1982; Page *et al.*, 1987).

Different species may have different sensitivities and recovery times. For example, fucoid algae have been hardly affected by some oiling (*e.g.*, Notini, 1978), possibly because of their mucilage coatings and the frequency of tidal washings. Salt-marsh plants such as

Spartina may be more sensitive because they have oleophilic cuticles and occur higher up the shore, where tidal washing is less prolonged. Different species may recolonise at different rates because of different mobilities or life cycles. An example is the recolonisation of the Fawley salt marsh, Southampton Water (a sheltered mud shore with high biological productivity), following improvements in refinery effluent quality. Both annual plants (notably *Salicornia*) and the perennial grass *Spartina anglica* recolonised sediments in which oil residues remained. However, the *Salicornia* spread much faster through tidal distribution of abundant seeds than the *Spartina* did with its relatively poor seed production and its slow vegetative recolonisation (Dicks and Levell, 1989).

There is also the question of the distance of the 'reservoir' of the recolonising species from the area to be recolonised. If this is great, as may be the case for rare species or if the spill is large and has affected many contiguous miles of coastline, then recolonisation times may be relatively long.

The season of the spill can affect recovery rates. For example, winter oiling of a salt marsh can affect seeds and reduce germination in the spring. Marked reduction of flowering can occur if plants are oiled when the flower buds are developing; even though there may be good vegetative recovery, there is a loss of seed production for that year (Baker, 1971).

Sublittoral Zone (Subtidal Zone)

> *Subtidal benthos may be impacted when oil is carried from the sea surface to the seabed adsorbed to particulates or by the re-distribution of oiled beach sediments. Impact on the subtidal benthos is usually less than on the shores and takes place after the initial intertidal impact as a result of the cushioning effect of the overlying water. Once the physical and direct toxic effects of the oil begin to decline, microorganism, meiofaunal and macrofaunal activities are able to utilise petroleum hydrocarbons as an energy source through a variety of pathways. Thus post-impact response can be typical organic enrichment where there is a population explosion of species able to use the increase in nutrients. Once the damaging effect of the oil is sufficiently reduced, the continuous nature of the marine environment ensures that eventual community recovery is inevitable. This usually takes 1 to 5 years.*

There is a considerable body of information on the impact and recovery of intertidal benthos impacted by oil spills. Impact and recovery of the subtidal environment are less well studied largely because onshore impact causes more public concern and because of the direct physical impact, which causes an immediate and obvious effect. Although the overlying water column protects the subtidal benthos from physical impact, oil or its components may be carried to the seabed by a variety of means. In addition to the vertical mixing of the more water-soluble hydrocarbons into and throughout the water column, there is evidence that oil may be carried to the seabed by fine particulate matter. For example, estimates from sediment-trap data after the *Tsesis* spill in the Baltic Sea showed that at least 20 tons of oil (equivalent to 0.5 g/meter^2) reached the benthos in this way (Johansson *et al.*, 1980).

Oil may also directly come into contact with benthos through subsea leakage of installations such as storage facilities or pipelines and through natural seeps, the latter having been the subject of many studies of long-term effects of hydrocarbon contamination (Allen *et al.*, 1970; Spies and Davis, 1979; Davis and Spies, 1980; Kennicutt *et al.*, 1985; Hovland and Judd, 1988).

The other major source of oil on the seabed is from discharge of oil cuttings during offshore drilling operations (Davies *et al.*, 1984). Although, strictly speaking, this is not spilled oil, the large amount of accumulated information on the effects of hydrocarbon contamination could be useful in assessing the longer term recovery of oil-contaminated benthos.

The initial impact of a large oil spill on the subtidal benthos, as has been already said, will be cushioned by the overlying water, which has the effect of delaying the impact and reducing its intensity. A good illustration of this is the time lag between the impact of the *Florida* No. 2 fuel oil spill on the inshore fauna at Wild Harbor, Buzzards Bay, Massachusetts, and the fauna further offshore, which were affected a few days later as the oil gradually spread seawards (Sanders *et al.*, 1980).

Subtidal species vary in their sensitivity to oil. Some, like certain bivalves such as *Macoma balthica*, appear quite resilient to relatively high levels of petroleum hydrocarbons (Elmgren *et al.*, 1983). In contrast, certain crustaceans such as the Amphipoda appear to be particularly vulnerable. Huge populations of the amphipods *Apseudes*, *Bathyporeia*, and *Urothoe* were killed by the *Amoco Cadiz* oil spill in 1978 (Conan, 1982), with *Ampelisca* being later totally eliminated from the area (Cabioch *et al.*, 1980). Ampeliscids were also found to be particularly vulnerable to No. 2 fuel oil after the *Florida* spill at Buzzards Bay, Massachusetts, in 1969 (Sanders *et al.*, 1972). However, they were not the only susceptible species; within 48 hours of the oil's arrival in the bay, there were heavy mortalities of the benthos, including worms, other crustaceans, and fish which were up on the shores (Hampson and Sanders, 1969). Amphipods were again found to be amongst the most vulnerable species to oil pollution after the *Tsesis* spill, when two species, *Pontoporeia affinis* and *Pontoporeia femorata*, almost completely disappeared, together with the scaleworm *Harmothoe sarsi* (Elmgren *et al.*, 1983). In the Baffin Island Oil Spill Study at Cape Hatt, the amphipod *Gammarus setosus*, the only species severely affected by the experimental oil spill, washed up in the intertidal zone two days after oil had been introduced into the bay (Cross and Thomson, 1981, 1982).

There is evidence that where there is not a direct physical or acute toxic effect, the fauna may be indirectly killed by being driven from the protection of the sediment and then immobilised (Prouse and Gordon, 1976; Percy, 1976, 1977). For example, large numbers of subtidal heart urchins and razor clams were killed during the first weeks following the *Amoco Cadiz* spill when they were washed up onto the shore after leaving their burrows (Hess, 1978). During the Baffin Island oil spill study, sea urchins were observed feeding on bivalves that had left their burrows and had been incapacitated by the oiling (Cross and Thomson, 1981).

The impact of an oil spill varies greatly with the nature of the spill, the nature of the receiving environment, and the type of biological community affected. The *Florida* oil spill (630 tons of No. 2 fuel oil) had a major impact on the benthos (Sanders *et al.*, 1980) as did the *Amoco Cadiz,* which released more than 230,000 tons of oil (Hess, 1978). In both cases weather conditions were such that the oil was directed shorewards into shallow water and ultimately onto the shore. In the case of the *Argo Merchant* oil spill (28,000 tons), wind prevented the oil from beaching (Grose and Mattson, 1977) and there was no significant impact on the benthos (Pratt, 1978). These two extremes serve to illustrate the importance of establishing the starting point when assessing recovery and the difficulty of generalising the process.

Probably the most detailed series of studies of benthic communities following an oil spill were those carried out after the *Florida* oil spill in Buzzards Bay, Massachusetts in 1969. (Hampson and Sanders, 1969; Sanders *et al.*, 1972; Michael *et al.*, 1975; Sanders, 1978; Sanders *et al.*, 1980). The faunal changes observed could be linked to the duration and severity of the oil dose and the extent of weathering. Following immediate effects of the oil impact, there was a reduction in the benthic fauna; opportunistic species such as the polychaete worm *Capitella* then dominated (Figure 9) and continued to do so for 11 months after the spill (Sanders *et al.*, 1972). The huge *Capitella* population was then rapidly replaced as the indigenous species of the area re-established themselves. This process continued over several years, but as late as 1974 the numbers of species had not reached those of the control area set up at nearby Sippewisset Marsh (Michael *et al.*, 1975).

Whilst the inshore area (Wild Harbor Bay) had been dominated by the worm *Capitella*, the offshore part of Buzzards Bay, Massachusetts, which later became contaminated by hydrocarbons spreading seawards, experienced a population explosion of a closely related species, *Mediomastus* (Figure 9). This worm underwent a sequence of events similar to that of its inshore counterpart, finally being replaced by the natural population as the oil contamination abated (Sanders *et al.*, 1980). Spies *et al.* (1988) draw attention to the similarity of this community response to the effects described by Pearson and Rosenberg (1978) for other sources of organic materials in the marine environment.

Most of the evidence cited by Sanders *et al.* (1980) suggests that the study sites furthest offshore had recovered within a year of the incident, with the shallower nearshore stations taking three or four years to approach normality.

A similar faunal response was elicited in tidal rivers after the *Amoco Cadiz* spill in which, at high oil concentrations, the benthic population of the impacted area was dominated by opportunistic polychaete worms. The species of the worms appeared to relate to oil concentration. Thus at concentrations of oil between 100 and 1,000 ppm, opportunists of the spionid and cirratulid worm families established themselves. At oil concentrations over 10,000 ppm, cirratulids and capitellids dominated (Glemarec and Hussenot, 1981).

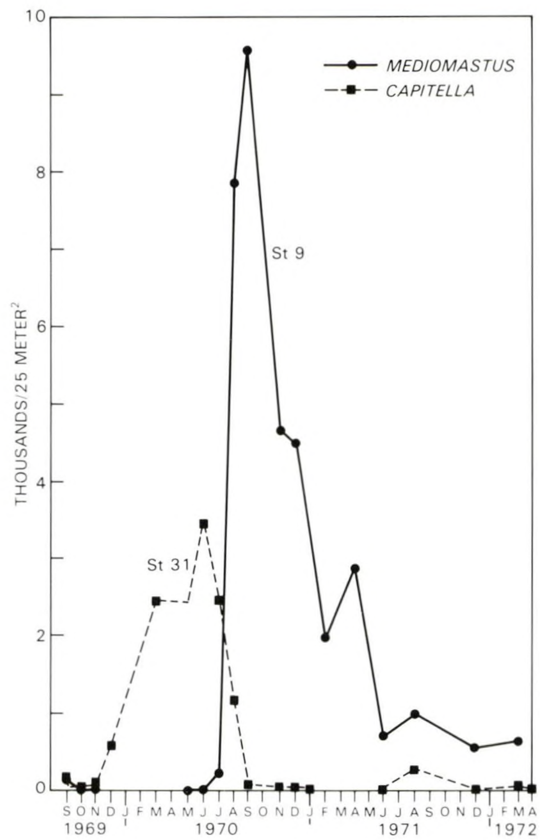


Figure 9: Succession of *Capitella* and *Mediomastus* nearshore (St 31) and offshore (St 9) after the *Falmouth* oil spill in Wild Harbor, Buzzards Bay, Mass. (Sanders *et al.*, 1980).

Although the presence of large numbers of opportunistic species clearly indicates highly disturbed environmental conditions, the increased total biomass that goes hand in hand with the increased numbers is indicative of enhanced productivity, the oil providing the source of organic enrichment. Similar community responses have been recorded where oily drill cuttings have been discharged (Davies *et al.*, 1984; Matheson *et al.*, 1986; Kingston, 1987). The occurrence of such opportunistic species under these circumstances may play a significant role in the active breakdown of the hydrocarbons.

Not all oil spills result in an obvious organic enrichment effect. Six years after the *Arrow* oil spill in Nova Scotia (1970), species diversity was still lower at oil-impacted sites than at the unoiled controls, and biomass was only one-third what it was at the controls (Thomas, 1978). Interestingly, however, populations of the lugworm *Arenicola* were more abundant in oiled sediments in 1976 than anywhere else in Nova Scotia (Gordon *et al.*, 1978). After the *Arco Anchorage* spill near Ediz Hook in Port Angeles in 1985, average total infaunal density, biomass, and diversity were highly variable over the shore and immediate sublittoral (Blaylock and Houghton, 1989). This variability persisted in the two years following the initial spill, although overall numbers of species and diversity increased over this time. Assessment of recovery at this site was complicated by the fact that the area was already under environmental stress from local industrial developments; however, the data available have been interpreted as indicating that recovery was well under way by winter 1988 (Mancini *et al.*, 1989).

The vulnerability of certain species of Amphipoda to oil has already been discussed. The recovery process, once the physical and toxic effects of the oil ameliorate, will in the early stages be dominated by opportunistic species. Most benthic species that include a pelagic larval stage in their life cycle may be considered potentially opportunistic. The extent to which such species realise their potential will depend upon their ability to tolerate environmental insult and to out-compete other species.

The occurrence of nearby unaffected populations of organisms that are eliminated in a major spill is important if recovery is to be rapid. Thus where a species has been eliminated over an extended geographical area, as happened to the amphipod *Ampelisca* after the *Amoco Cadiz* spill, full recovery may be slow, but it is still achievable. In most cases there is a sufficient pool of adults nearby to provide a supply of larvae for rapid recruitment should the levels of oil concentration permit it. For example, in the summer following the *Arco Anchorage* spill, settlement of young bivalves was reported at several of the study sites that had previously been affected by the oil spill (Blaylock and Houghton, 1989). Recruitment after the *Tsesis* incident was not as immediate, but in one area the amphipod *Pontoporeia femorata*, which suffered heavy mortality at the time of the spill, had recovered to pre-spill values after about 9 months (Elmgren *et al.*, 1983).

Recently Dauvin and Gentil (in press) found that most of the pericarid amphipod populations that were suspected of having been irretrievably lost after the *Amoco Cadiz* spill in 1978 (Cabioch *et al.*, 1980) had completely recovered by 1988. Dauvin and Gentil (in press) also report the reestablishment of benthic communities in the Aber Wrac'h Channel, Bay of Morlaix, and Bay of Lannion of a similar specific composition and pattern of dominance as those found before the spill.

The *Tsesis* oil spill study (Elmgren *et al.*, 1983) is one of the few thorough sublittoral benthic studies carried out on a spill where dispersants were not used and there was good pre-spill information about the receiving environment. Figure 10 shows the number of individuals of benthos from two sites, one in the middle (Site 20) and one on the eastern part of the spill area. The figure demonstrates the natural variability of the benthic community data and allows it to be related to the impact of the oil spill and the return of the system (the picture is further complicated by the increasing eutrophication in the area resulting from nearby sewage input). One of the features of the community response to the spill was the survival of the bivalve *Macoma baltica* and the priapulid *Halicryptus spinulosus*; both animals have low mobility and both showed increased abundance after the spill. Although the impacted faunal community was recovering, Elmgren *et al.* (1983) claim that the dominance of *Macoma baltica*, a species with a long life span, will persist, maintaining a 'disturbed' community for several years and that 5 to 10 years may be needed for 'full recovery'.

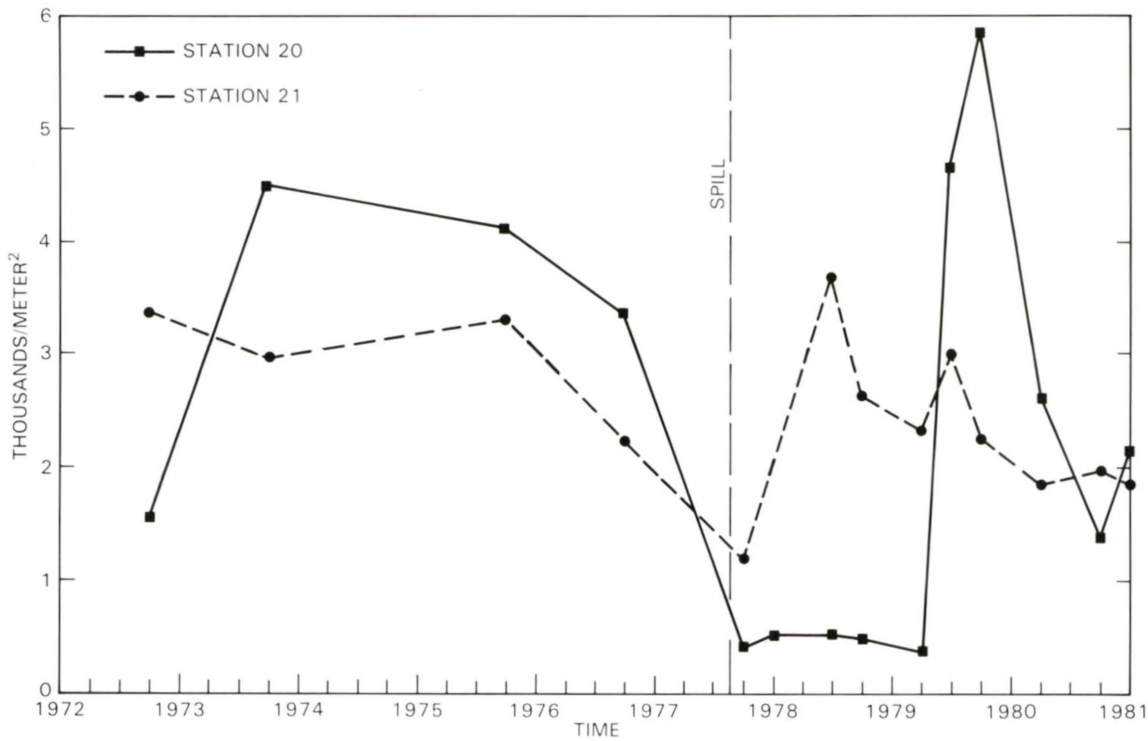


Figure 10: Changes in the faunal abundance at two localities affected by the *Tsesis* oil spill (Elmgren *et al.*, 1983).

There are few oil spill incidents where there has been no use of dispersants, no attempts to clean up, and no other anthropogenic activity. No cleanup effort was launched following the *Metula* spill in the Straits of Magellan. As has already been mentioned, hydrocarbons do, however, find their way to the seabed in other ways—through natural seepages and by deliberate discharge. Under these circumstances their presence on the seabed may be the only source of environmental disturbance.

The best known natural seepages are those that occur off the California coast, and these have formed the focus of many studies on hydrocarbon-based communities (Allen *et al.*, 1970; Straughan, 1976; Spies and Davis, 1979; Spies *et al.*, 1980; Davis and Spies, 1980). More recently, interest has been directed at hydrocarbon seeps in the North Sea (Hovland and Judd, 1988; Hovland and Thomsen, 1989).

Bacterial mats and other marine life associated with seepages were first studied by Spies and Davis (1979). Surveying oil and gas seeps near Santa Barbara, California, they found consistently greater densities of organisms around seep locations, compared with nearby areas. Where the seeps were particularly intense, the seabed was covered with white mats of the bacterium *Beggiatoa sp.* Similar mats have been found in the North Sea where the seepages are usually associated with pock marks caused by the sudden release of gas.

Beggiatoa sp. is known as an H_2S oxidiser that requires anaerobic conditions (Spies and Davis, 1979). It has been shown that there is a trophic pathway from petroleum oils through sulphate-reducing bacteria, to H_2S, to *Beggiatoa sp.*, to *Nematoda*, and thence other infauna (Spies and Des Marais, 1983). Similar *Beggiotoa sp.* mats have been observed associated with oily cuttings piles discharged onto the seabed during offshore drilling operations (Gillam *et al.*, 1986) together with greatly enhanced numbers of opportunistic species (Matheson *et al.*, 1986; Kingston, 1987; Mair *et al.*, 1987). These blooms of species, which often include *Capitella* (see Figure 11), are very similar to those recorded for spills such as the *Florida* (Sanders *et al.*, 1980), and it is tempting to draw parallels.

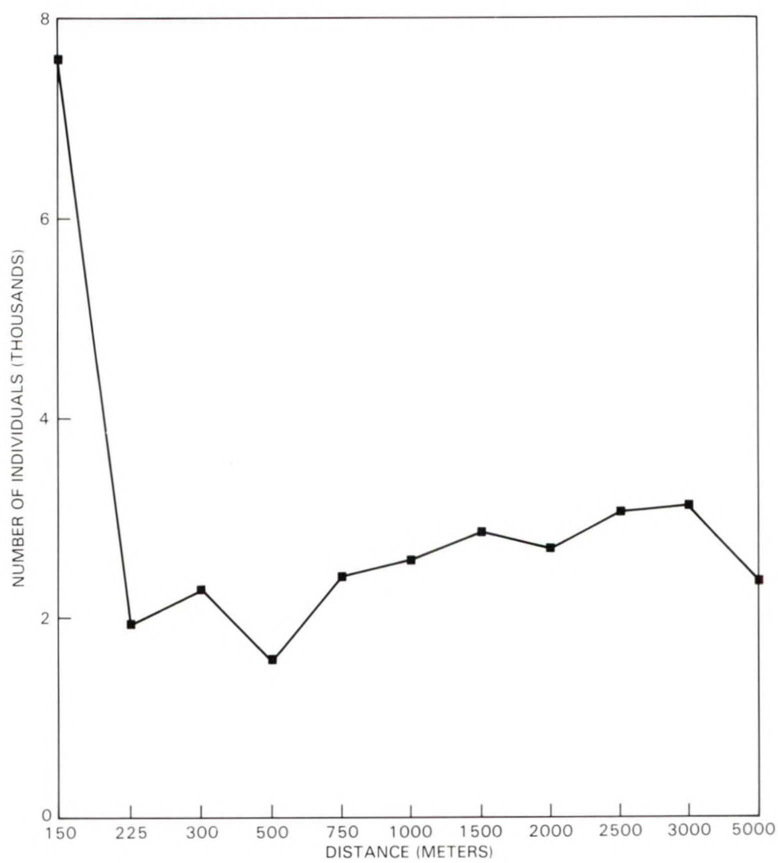


Figure 11: Changes in the number of individuals with increasing distance from the Statfjord Bravo production platform, 1984 (Matheson *et al.*, 1986).

Although there are no reports in the literature of subtidal *Beggiatoa sp.* mats associated with oil spills, it is likely that processes similar to those described above take place. Certainly elevated numbers of nematodes are a feature of many faunal responses to an oil spill. Large nematode populations were reported after the *Amoco Cadiz* spill (Chassé, 1978), and nematodes have been found to maintain substantial populations after spills such as that of the *Tsesis* (Elmgren *et al.*, 1983), and the *Venpet* and *Venoil* spills in South Africa (Fricke *et al.*, 1981).

It is apparent that once the physical and direct toxic effects of the oil begin to decline, microorganism, meiofaunal, and macrofaunal activities are able to utilise petroleum hydrocarbons as an energy source through a variety of pathways (Knap *et al.*, 1979; Spies and Des Marais, 1983). The implication of this is that high faunal abundance under these conditions indicates more active biodegradation of the hydrocarbons than if the fauna were impoverished.

ACKNOWLEDGMENTS

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TABLE 1

Examples of Background Concentrations* of Hydrocarbons in Water

Location	Hydrocarbon Type and Concentration	Reference
Ocean waters throughout the world	Most surface and near surface waters have 1–10 ppb total hydrocarbons. Both biogenic and petroleum hydrocarbons appear to be ubiquitous.	Myers and Gunnerson (1976)
Baffin Bay and eastern Canadian Arctic	Floating particulate residues rare. Extractable residues in surface microlayer 3–1726 ppb and in water column 0–87.5 ppb (solvent = $CC1_4$).	Levy (1981)
Arctic Ocean north of Svalbard	Total hydrocarbons (THC's) in surface water under ice, 0.1–0.6 ppb Kuwait crude oil equivalents with respect to light molecular weight compounds and 0.05–0.2 ppb with respect to heavy molecular weight compounds.	Fogelqvist *et al.* (1982)
UK marine waters (North Sea, English Channel and Irish Sea)	Total hydrocarbons (THC's) at 1 m, 1.1–74 ppb Ekofisk crude oil equivalents, all values greater than 3.5 ppb occurring inshore. Mean THC's offshore 1.3 ppb N. North Sea, 1.5 ppb W. Channel, 2.5 ppb E. Channel and S. North Sea, and 2.6 ppb Irish Sea.	Law (1981)
Northern North Sea	THC's at 1 m, 1978 'clean' station at least 80 km from land or from oil platform, 7.2 ppb. 1980 clean stations within 25 km of oil activity, 1.2–2.4 ppb. Outer Forth 7.4 ppb, mid Forth 9.4 ppb, inner Forth 23.2 ppb. Total aryl hydrocarbons at 1 m, 3.2 to 32 km east of Brent 0.032–0.109 ppb; outer Forth 0.064 ppb; mid Forth 0.064 ppb; inner Forth 0.042 ppb.	Massie *et al.* (1985a)
English Channel	THC's at 1 m, less than 0.3–14 ppb Ekofisk crude oil equivalents. The higher concentrations were in Southampton Water and to the S and E of the Isle of Wight (higher shipping activity).	Fileman and Law (1988)
Southern Baltic Sea	THC's at 10 m, Slupsk (offshore) 2.0 ppb Ekofisk crude oil equivalents, Gdansk basin (offshore) 34 ppb, Puck Bay (inshore) 130 ppb.	Law and Andrulewicz (1983)
Southern Baltic Sea	C15-C32 n-alkanes at 0 m, 9–1744 ppb, C15-C32 n-alkanes at 5 m, 10–219 ppb, benzo(a)pyrene at 0 m, 0.03–2.9 ppb, benzo(a)pyrene at 5 m, 0.04–3.1 ppb.	Grzybowski *et al.* (1987)
Southern Baltic Sea	Total polycyclic aromatic hydrocarbon (PAH) compounds (sum of 11 PAH) at 9 m, 9–143 ppb. Total PAH at 5 m, 10–180 ppb.	Lamparczyk *et al.* (1988)

* All concentrations have been converted to ppm or ppb from the reported values. A water density of 1 gram/cc has been assumed.

TABLE 2

Examples of Background Concentrations* of Hydrocarbons in Sediments

Location	Hydrocarbon Type and Concentration	Reference
Baffin Bay and eastern Canadian Arctic	Extractable residues 1–41 ppm. Solvent CCl_4.	Levy (1981)
Gulf of Maine	16 polycyclic aromatic hydrocarbon (PAH) compounds, total concentrations 10–512 ppb (dry weight). These concentrations are an order of magnitude lower than those observed in the coastal zone, but higher than those on Georges Bank.	Larsen *et al.* (1986)
Boston Harbour	14 PAH compounds, total concentrations 483–718,364 ppb (dry weight). The highest concentrations from the inner harbour almost an order of magnitude higher than previously reported concentrations from a range of urban estuaries.	Shiaris and Jambard-Sweet (1986)
Narragansett Bay	Alkanes and aromatics. 50–120 ppm south, 130–440 ppm central, 500–700 ppm north.	Farrington and Quinn (1973)
UK marine sediments	THC's (Ekofisk crude oil equivalents) 0.27–340 ppm, with the highest concentration being in the entrance to the Mersey.	Law (1981)
English channel	THC's (Ekofisk crude oil equivalents) dry mass, 0.3–5.60 ppm. Highest in Southampton Water.	Fileman and Law (1988)
Milford Haven and Dauccleddau, SW Wales (subtidal)	THC's (ppm) commonly 100–300 up to 700. Contaminants concentrated in areas of fine sediment (high in organic and clay content).	Little and McLaren (1989)
Milford Haven (Angle Bay intertidal), SW Wales	THC's (ppm dry weight) in fine sand (0–5 cm layer), 18–70.	Howard *et al.* (1989)
Danish marine sediments	Alkanes and aromatics. 5–390 ppm Danish coast, 46–1800 ppm Copenhagen Sound.	Jensen (1981)
Southern Baltic Sea	THC's (Ekofisk crude oil equivalents) dry mass, 4.0–140 ppm, with the highest concentration occurring in the Gotland and Gdansk basins.	Law and Andrulewicz (1983)
Southern Baltic Sea	C15-C32 n-alkanes, 626–23,040 ppm benzo(a)pyrene, 2–49 ppm, with the highest concentrations occurring in Gdansk and Pomerania Bays.	Lamparczyk *et al.* (1988)

* All concentrations have been converted to ppm or ppb from the reported values.

TABLE 3

Examples of Background Concentrations* of Hydrocarbons in the Tissues of Mussels

Location	Hydrocarbon Type and Concentration	Reference
Northeast Gulf of Alaska	17.6 ppm dry mass Unresolved Complex Mixture (UCM).	Wise *et al.* (1980)
Yaquina Bay, Oregon	Total concentrations of 15 polynuclear aromatic hydrocarbons (PNAH) measured. Average 986.2 ppb in industrialized area, 273.9 ppb in more remote area.	Mix and Schaffer (1983a)
Gulf of St. Lawrence	Most concentrations of benzo(a)pyrene below detection limit; high concentrations of 24 and 28.5 ppb (dry mass) at mouth of Saguenay Fjord.	Picard-Berube and Cossa (1983)
Various U.S. mussel watch stations	3–298 ppm dry mass UCM.	Farrington *et al.* (1980)
Scottish coast	Average of 109 ppb PNAH from 'clean' (sparsely populated) sites, 1127 ppb from industrialized sites.	Mackie *et al.* (1980)
N. Sea oil production platform	46–77 ppm dry mass UCM.	Rowland and Volkman (1982)
Copenhagen Sound	800–3500 ppm dry mass THC (calculated from micrograms per gram of lipid, assuming 8% lipid in body tissues).	Jensen (1981)

* All concentrations have been converted to ppm or ppb from the reported values.

TABLE 4

Residence Times of Oil on Shores

Shore Type	Location	Oil	Residence Time	Reference
Steep exposed bedrock	Hurlstone Point, SW England	Experimental application of Forties crude	40–50% cover remained after 11 days; thin brown stain remained after 1 month.	Baker *et al.* (1984)
Steep exposed bedrock	Hurlstone Point, SW England	Experimental application of Flotta residue and mousse	Most mousse washed off after 2 tidal immersions; most residue was washed off after 6 immersions.	Baker *et al.* (1984)
Exposed rock platform just above high water	Near East Prawle, Devon, England	Spots of oil thrown up by spray	Shrinkage of oil patches after 1.2–3.3 years, thinning after 1.9–5.3 years, varying with type and thickness of oil. Maximum duration of oil spots on this shore 17–18 years.	Mottershead (1981)
Rock reef	Godrevy Point, Cornwall, SW England	Kuwait mousse from the *Torrey Canyon* spill, 1967 (no direct dispersant use on this moderately oiled shore)	Traces of oil visible 1 year after, no visible oil 2 years after.	Southward and Southward (1978)
Rock platform	Watchet, Somerset, SW England	Experimental application of Forties crude	40–50% cover remained after 11 days, no visible oil after 1 month.	Baker *et al.* (1984)
Exposed beach of mobile sand and shingle	Corton, E England	Heavy fuel oil and mousse from *Eleni V* spill, 1978	Mechanical clearance of surface oil and massive resorting of beach material during winter storms left beaches visually clean after 1 year, except along tide lines. Hydrocarbon concentrations in inshore water and mussels returned to background levels about 2 years after the spill.	Blackman and Law (1980a) Blackman and Law (1980b) Blackman and Law (1981)

TABLE 4 Continued

Shore Type	Location	Oil	Residence Time	Reference
Exposed sand and gravel beach	Queen Charlotte Islands, Canada	Spill of diesel fuel and gasoline	Traces of fuel remained within 300 m of the spill site after 60 days.	McLaren (1985)
Variety of shores	Brittany	Light Arabian and Iranian oil/mousse from the *Amoco Cadiz* spill, 1978	Exposed rocky coasts and wave-cut platforms were cleaned of very heavy doses of oil within a few days. Oil penetrated deeply (30 cm) into gravel beaches.	Hayes *et al.* (1979)
Variety of shores	Brittany	Light Arabian and Iranian oil/mousse from the *Amoco Cadiz* spill, 1978	2 years after, oil persisted primarily as tar blotches and black staining along exposed rocky shores and as oil-contaminated interstitial water in intertidal flats.	Gundlach *et al.* (1981)
A variety of shores along the Pink Granite coast	Brittany	*Amoco Cadiz* (1978) and subsequent spills	Contribution of remaining weathered residues of *Amoco Cadiz* oil to the hydrocarbon baseline was small in 1985 (7 years after the spill) compared with more recent inputs.	Page *et al.* (1988)
Variety of shores in small bay	British Columbia	No. 5 fuel oil from the *Irish Stardust* spill, 1973	After 4 years, no oil on stone, gravel residues and sand. UCM in sediment extracts.	Cretney *et al.* (1978)
Variety of shores in bay	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	After 6 years, only a few locations visibly contaminated with large quantities of oil identifiable as *Arrow* Bunker C. These were upper high tide zones on 3 islands, where an oil/sediment pavement remained.	Keizer *et al.* (1978)

TABLE 4 Continued

Shore Type	Location	Oil	Residence Time	Reference
Sand/gravel/ cobble beaches, often with low tide terraces, wide range of energy levels	Straits of Magellan	Arabian crude and mousse from the *Metula* spill, 1974	12 years after, remaining oil was mainly a narrow strip that was laid down by high spring tides. Major exception in Puerto Espora area where extensive asphalt pavements remained in intertidal and supratidal areas over shoreline length of several km. Surface of pavements with weathered crust, but oil within of low viscosity in many cases (early post-spill descriptions for comparison).	Owens *et al.* (1987a, b) Baker *et al.* (1976); Hann (1977)
Sheltered beach of sand, shingle and rocks	Lowestoft harbour, E England	Heavy fuel oil and mousse from *Eleni V* spill, 1978	Very little change in appearance and composition of oil after 1 year; still little change after 2 years under a surface crust of weathered oil.	Blackman and Law (1980a) Blackman and Law (1981)
Range of beaches	Cape Hatt, Baffin Island	Experimental application of aged Lagomedio crude (BIOS programme), 1981	On exposed beach, 99% of oil removed within 48 hours. On partially exposed beach, most oil removed within first open water season of 6–8 weeks. With sheltered beaches, cleaning rate was slower on pebble/cobble beach than on flatter fine-grained beach.	Sergy (1985)
Range of sediment shorelines	Wales and SW England	Experimental applications of Nigerian crude	Residence times ranged from 3 days to more than 1 year, depended upon energy level, drainage, and sediment textural gradients.	Little and Scales (1987) Little (1987)

TABLE 4 Continued

Shore Type	Location	Oil	Residence Time	Reference
Sheltered gravel beach	Cape Hatt, Baffin Island	Experimental application of aged Lagomedio crude	Initial volume of oil retained on beach (5.3 m^3) reduced to 1.3 m^3 after 4 years. Nearly 30% of this was in an asphalt pavement, relatively unweathered.	Owens *et al.* (1987)
Sheltered beach	Cape Hatt, Baffin Island	Experimental application of aged Lagomedio crude	Over a 4-year period, migration of oil from the beach into nearshore subtidal sediments.	Owens *et al.* (1987)
Intertidal sand flat	Willapa Bay, Washington	Experimental application of North Slope crude	Fauna (mainly burrowing crustaceans) capable of introducing measurable amounts of oil into the subsurface, where it was retained long after the rest of the stranded oil had washed away.	Clifton *et al.* (1984)
Simulated intertidal flat conditions	Laboratory experiments (OILME) project	Experimental applications of weathered crude	Results show that there is greater retention with greater loading thicknesses, with longer sediment emergence periods and with lower mud content.	Harper *et al.* (1985)
Lower shore sediments, relatively low energy	Baie Verte Newfoundland	Spill of diesel oil, 1982	Relatively high concentration of aromatics still present after 27 months. After 39 months no PAHs found except fluorene.	Kiceniuk and Williams (1987)
Low-energy beach of pebbles and sand	Spitzbergen	Spill of diesel oil, 1978	High concentrations of oil in sediment 2 years after spill.	Gulliksen and Taasen (1982)
Sandy pocket beach	Long Cove, Nova Scotia	Experimental application of Scotian Shelf condensate (1982)	C8 hydrocarbons present after 3 months in upper intertidal surface sand. No loss of volatiles over 6 months in subsurface sand from upper and middle intertidal.	Strain (1986)

TABLE 4 Continued

Shore Type	Location	Oil	Residence Time	Reference
Sandy beaches	Brittany	Oil/mousse from the *Amoco Cadiz*, 1978	After 3 years, oil in discrete layers 1–2 cm thick persists in several beaches. These can migrate downwards, as far as the water table.	Long *et al.* (1981)
Low-energy fine sediments	Long Cove, Searsport, Maine	Jet fuel and No. 2 heating oil spill from fuel depot, 1971	5 years after spill, area estimated to contain roughly 20% less oil than in 1971.	Mayo *et al.* (1978)
Gently sloping shoreline of low-energy lagoon	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	Stranded oil persists after 7 years, continuously leaching into beach substrate where weathering occurs throughout the top 15 cm.	Vandermeulen *et al.* (1977)
Salt marsh	Milford Haven, SW Wales	Heavy fuel oil spill, oil up to 5 cm thick stranded, 1969	Oil has persisted for 18+ years with little diminution (protected by a layer of post-spill sediment).	Baker *et al.* (1989)
Salt marsh	Brittany	Crude oil remaining after *Amoco Cadiz* cleanup (1978)	Nuisance amounts of oil remained 5 years after.	Seip (1984)

TABLE 5

Summary of Shoreline Classification in Order of Increasing Vulnerability to Oil Spill Damage (Gundlach and Hayes, 1978)

Vulnerability Index	Shoreline Type	Comments
1	Exposed rocky headlands	Wave reflection keeps most of the oil offshore—no cleaning necessary.
2	Eroding wave-cut platforms	Wave-swept; most oil removed by natural processes within weeks.
3	Fine-grained sand beaches	Oil does not usually penetrate far into the sediment, facilitating mechanical removal if necessary; otherwise, oil may persist several months.
4	Coarse-grained sand beaches	Oil may sink and/or be buried rapidly, making cleanup difficult; under moderate- to high-energy conditions, oil will be removed naturally within months from most of the beach face.
5	Exposed, compacted tidal flats	Most oil will not adhere to, nor penetrate into, the compacted tidal flat; cleanup usually unnecessary.
6	Mixed sand and gravel beaches	Oil may undergo rapid penetration and burial; under moderate- to low-energy conditions, oil may persist for years.
7	Gravel beaches	Same as Index 6; a solid asphalt pavement may form under heavy oil accumulations.
8	Sheltered rocky coasts	Areas of reduced wave action; oil may persist for many years. Cleanup is not recommended unless oil concentration is very heavy.
9	Sheltered tidal flats	Areas of low wave energy and high biological productivity; oil may persist for many years. Cleanup is not recommended unless oil accumulation is very heavy. These areas should receive priority protection by using booms or oil-sorbent materials.
10	Salt marshes and mangroves	Most productive of aquatic environments; oil may persist for many years. Cleaning of salt marshes by burning or cutting should be undertaken only if heavily oiled.

TABLE 6

Microbial Degradation

Site/Samples	Hydrocarbons	Observations on Microbial Degradation	Reference
Shallow and deep water sediments, Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	In July and November 1970, hydrocarbon utilising microorganisms (HCUs) comprised up to 15% of the total population. In 1971 and 1972, HCU numbers at around 1%, a value obtainable routinely in 'clean' areas. In 1976 comparable with 1972/1972–77 out of 79 samples were comparable to areas described as 'clean'.	Stewart and Marks (1978)
Lugworm bed sediments, Black Duck Cove, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	Hydrocarbon concentrations were substantially lower in worm casts than in initial sediment. This loss can be accounted for by microbial degradation, which is stimulated by the worms' activity.	Gordon *et al.* (1978)
Beaches in the Straits of Magellan	Light Arabian oil/mousse from the *Metula* spill, 1974	Oil degradation proceeded relatively slowly, with marked persistence of oil 2 years after the spill. Experimental work suggested that the limiting factors were low concentrations of nitrogen and phosphorus and the inaccessibility of hydrocarbon components inside asphalt-like concretions. Temperature did not seem to be a limiting factor.	Colwell *et al.* (1978)
Ile Grande marshes, Brittany	Light Arabian and Iranian oil/mousse from the *Amoco Cadiz* spill, 1978	Evolution of hydrocarbons studied 1978–1980. Biodegradation important in superficial sediments, with preferential degradation of n-alkanes. Aromatics seemed not to have altered after 3 years. Degradation of percolated hydrocarbons slower than in surface layer. Numbers of degrading bacteria decreased when n-alkanes disappeared.	Mille *et al.* (1984)
Aber Benoit estuary, Brittany	Light Arabian and Iranian oil/mousse from the *Amoco Cadiz* spill, 1978	By 1986, all but the most heavily oiled locations had assemblages of biogenic hydrocarbons similar to the reference sites. Measurable residues of weathered *Amoco Cadiz* oil remained only in isolated soft sediment locations serving as repositories for fine sediment from other parts of Aber Benoit. These predicted to reach a background state as microbial activity continues to degrade any petroleum fractions remaining.	Page *et al.* (1989)

TABLE 6 Continued

Site/Samples	Hydrocarbons	Observations on Microbial Degradation	Reference
Nearshore sediments (top 5 cm) of Beaufort Sea. Ice-free, –1.8°C year-round	Experimental application of Prudhoe Bay crude	Studies for 2 years. Oil degraded slowly; only after 1 year's exposure was biodegradation evident. Aliphatic compounds were not preferentially degraded over aromatics. C17 and lower molecular weight alkanes were preferentially degraded over high molecular weight alkanes.	Haines and Atlas (1982)
Estuarine sediments, Brittany	Experimental applications of Arabian light crude	As long as the amounts of oil remained higher than a threshold value, biodegradation was inhibited, presumably due to oxygen and/or nutrients limitation.	Fusey and Oudot (1984)
Samples of water and sediments from North Sea oilfields	Experimental additions of aromatic hydrocarbons	Microorganisms in the samples had the potential to degrade smaller aromatic hydrocarbon molecules rapidly. Degradation of larger aromatic molecules as exemplified by benzo(a)pyrene was minimal.	Massie *et al.* (1985b)
Seaweed heaps on Arctic and sub-Arctic shores	Experimental applications of weathered Statfjord crude	In Norway (subpolar and temperate), biodegradation of oil in heaps requires artificial supply of oxygen, as heaps become anoxic within days under normal conditions. Seaweed heaps in Arctic Spitzbergen provided more favourable conditions for oil biodegradation, because of more heterogeneous conditions in terms of oxygen availability.	Sveum and Sendstad (1985)
Intertidal mud, Jadebusen, W Germany	Experimental applications of Statfjord crude (*in situ* and with mud samples in laboratory)	Aerobic mud was highly active in hydrocarbon biodegradation. Anaerobic experiments confirmed the common experience of low or zero rates. Hydrocarbons in an adsorbed state are more easily biodegradable than as a floating layer or droplets.	Hopner *et al.* (1987)

TABLE 7

Recovery of Littoral Benthos

Shore Type	Location	Oil	Observations on Recovery	Reference
Rock reef	Godrevy Point, Cornwall, SW England	Kuwait mousse from the *Torrey Canyon* spill, 1967 (no direct dispersant use on this moderately oiled shore)	Good recovery after 2 years.	Southward and Southward (1978)
Rock shelf	NW coast of the State of Washington	Navy Special fuel oil from *General MC Meigs* spill, 1972	Effects on urchins and algae for at least 1 year. No marked long-term effects on community balance. Hydrocarbon residues present in mussels after 5 years; attributed to recontamination by winter discharges from wreck.	Clark *et al.* (1978)
Rock/boulders/ cobbles	Sullom Voe, Shetland	Heavy fuel oil from the *Esso Bernicia* spill, 1978	9 years after, 5 km of shore still showed successional changes (resulting mainly from use of bulldozers in cleanup).	Westwood *et al.* (1989)
Rock/boulders/ cobbles	Sullom Voe, Shetland	*Esso Bernicia* oil and subsequent inputs	Winkles *Littorina littorea* collected 1981 had significant enzyme changes indicating lysosomal destabilization.	Moore *et al.* (1982)
Exposed rock	Hurlstone Point, N Somerset, England	Experimental applications of Forties crude, 1979	Reductions in limpets and small littorinid winkles during the year following treatment.	Crothers (1983) Baker *et al.* (1984)
Exposed rock	Hurlstone Point, N Somerset, England	Experimental applications of Flotta residue and mousse, 1981	Only significant change after 6 months was an increase of colonisation by barnacle larvae in oiled plots.	Baker *et al.* (1984)
Rock platform	Watchet, N Somerset, England, 1979	Experimental applications of Forties crude	Main change in all plots including controls was increase of fucoid algae, interpreted as a continuation of a long-term trend.	Baker *et al.* (1984)

TABLE 7 Continued

Shore Type	Location	Oil	Observations on Recovery	Reference
Rocky shores	Swedish archipelago, Baltic Sea	No. 5 fuel oil and some bunker oil from the *Tsesis* spill, 1977	Dominant alga *Fucus vesiculosus* not affected. Faunal density within algal zone was 8–10% of previous level 2 weeks after spill. Recovery began within 2 months, with normal densities after 1 year at some sites. Recovery varied depending on severity of oiling and species involved. After 1 year hydrocarbon concentrations in mussels approached normal conditions except at the most heavily oiled sites, though there were still elevated levels of substituted naphthalenes.	Notini (1980) Linden *et al.* (1979) Boehm *et al.* (1982)
Rocky shores	Atland, Finland, Baltic Sea	Crude oil from the Soviet tanker *Antonio Gramsci*, 1979	By 4 months after spill there was new settlement of barnacles and mussels close to remaining oil on rocks. Little difference between oiled and non-oiled areas in *Cladophora* belt; in *Fucus* belt there was some decline in mussels and mobile crustaceans.	Bonsdorff (1981)
Rocky, moderately exposed shore	W coast of Norway	Iranian crude oil from 1976 spill	13 months after spill, significant amounts of hydrocarbons remained in winkles *Littorina littorea.* No detectable effects on fertilization, but hatching success significantly less in oiled population.	Staveland (1979)
Beaches with sand, stones and rock	Gastviken Bay, Musko Island, Baltic Sea	Medium and heavy fuel oil from the *Irini* spill ,1970; mechanical cleanup	No oil-associated effects on *Fucus vesiculosus*; severe initial depletion of fauna. Recolonisation by most species occurred within 1 year, but low population densities during the 2nd year. After 4 years, no significant evidence of lasting detrimental effects, when natural annual variations taken into account.	Notini (1978)

TABLE 7 Continued

Shore Type	Location	Oil	Observations on Recovery	Reference
Variety of shores in bay	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	6 years after, lower diversity, lower biomass of flora, smaller clams (*Mya*) and larger winkles (*Littorina*) at oiled sites compared with controls. Abundant lugworms (with elevated hydrocarbon concentrations) in oiled sediments.	Thomas (1978) Gordon *et al.* (1978)
Variety of shores in bay lagoons	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the *Arrow* spill, 1970	*Mya* remained under continued stress in oiled lagoons 6 years after spill.	Gilfillan and Vandermeulen (1978)
Variety of shores	Straits of Magellan	Light Arabian oil/mousse from the *Metula* spill, 1974	1977 data suggested continuing impact in areas still heavily oiled, and recovery of invertebrates in areas which had lost oil.	Straughan (1978)
Variety of shores	Straits of Magellan	Light Arabian oil/mousse from the *Metula* spill, 1974	Asphalt pavement recolonised by algae and (more slowly, from lower intertidal levels) animals, mainly mussels and the gastropod *Siphonaria lateralis*. 5 years after spill, community still relatively simple in composition.	Guzman and Campodonico (1981)
Variety of shores	Straits of Magellan	Light Arabian oil/mousse from the *Metula* spill, 1974	Persistent effects on flora and insect fauna of heavily oiled salt marsh.	Guzman and Campodonico (1981)
Variety of shores	Brittany	Light Arabian oil/mousse from the *Amoco Cadiz* spill, 1978	Marshes severely affected, with no recovery in 2 years at the most heavily oiled sites. 5 years after, Cantel marsh which was oiled but had no cleanup was essentially restored by natural processes. 8 years after, the Ile Grande marsh which had heavy cleanup was well recovered, facilitated by artificial plantings.	Gundlach *et al.* (1981) Baca *et al.* (1987)

TABLE 7 Continued

Shore Type	Location	Oil	Observations on Recovery	Reference
Variety of shores	Brittany	Light Arabian oil/mousse from the *Amoco Cadiz* spill, 1978	9 months after, hydrocarbons from the spill identified in limpets from rocky shores and *Mya* from mud flats. Heavier molecular weight aromatics present in *Mya*, not in limpets.	Vandermeulen *et al.* (1981)
Variety of shores	Brittany	Light Arabian oil/mousse from the *Amoco Cadiz* spill, 1978	Japanese oysters taken from the oiled Aber Wrac'h in 1979 had reached a steady state with respect to environmental exposure to weathered oil; they were able to reach background levels during a 96-day depuration period (in Maine).	Page *et al.* (1987)
Variety of shores	Brittany	Light Arabian oil/mousse from the *Amoco Cadiz* spill, 1978	Delayed effects: declines in populations of clams *Tellina* and nematodes reported 1 year after spill at St. Efflam and Bay of Morlaix, respectively.	Conan (1982)
Sediments in experimental trays	Sequim Bay, Washington	Experimental application of Prudhoe Bay crude, 1976	Initial concentrations of oil in sediments upon field emplacement were up to 5000–6000 ppm; no substantial inhibition of recruitment by benthic organisms.	Anderson *et al.* (1978)
Salt marsh	Buzzards Bay, Massachu-setts	No. 2 fuel oil from the *Florida* spill, 1969	Recovery of fiddler crab population was correlated with the disappearance of naphthalene in sediments; not complete after 7 years.	Krebs and Burns (1978)
Stonework jetty	Buzzards Bay, Massachu-setts	No. 2 fuel oil from the *Florida* spill, 1969	Reasonably functioning population of oyster drill *Urosalpinx cinerea* re-established at Wild Harbour by 1975, but with greater year-to-year variation in genetic structure than at reference site.	Cole (1978)
Salt marsh	Steart, Somerset, England	Experimental applications of Forties crude, 1979	No detectable retention of experimental oil in sediment samples of May 1981, but flora did not recover to control levels during 1981 growing season.	Baker *et al.* (1984)

TABLE 7 Continued

Shore Type	Location	Oil	Observations on Recovery	Reference
Salt marsh	Steart, Somerset, England	Experimental applications of Forties crude, Flotta residue and mousse, 1981	Recovery of all treated plots started during 1982, but *Spartina* density remained low in the Forties crude plot throughout 1982.	Baker *et al.* (1984)
Intertidal sea grass beds	Angle Bay, Milford Haven, Wales	Experimental applications of Nigerian crude	No visual effects on sea-grass following tidal removal of oil, but no increase in cover during growing season following treatment (compared with increase of cover in control).	Howard *et al.* (1989)

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APPENDIX A

COLD WATER ECOSYSTEMS

Marine cold water ecosystems are characterised by high seasonal temperature fluctuations relative to tropical and sub-tropical waters. As a result, the diversity of the fauna and flora is generally lower than at lower latitudes. Because of the better mixing of the water layers in cold water areas, productivity is still relatively high. Fauna of cold water ecosystems also tend to adopt reproductive strategies that involve a higher degree of brood protection and thus are believed to recover more slowly from major ecological damage.

PHYSICAL CONDITIONS

Cold water ecosystems, in the context of the marine biospheres, include those marine habitats found in temperate, boreal/antiboreal, subpolar and polar seas. Circumglobal cold water habitats also exist in the deep sea; however, since this review is primarily concerned with the natural recovery of coastal littoral and sublittoral environments, discussion of the deep sea environment is excluded.

The extent of the cold water habitats in the world's oceans is shown in Figure 8. In the southern hemisphere, the boundaries between the regions are relatively uncomplicated, with the transition between the polar/subpolar, antiboreal and temperate zones following latitudinal parallels across the oceans in the southern hemisphere. The boundaries between the temperate and tropical zones are displaced south on the western boundaries of the Indian and Atlantic Oceans and north on the eastern boundaries as a result of the interaction between the continental masses of South America and Africa and the main global oceanic currents (Bogdanov, 1963).

In the northern hemisphere, the boundaries between the temperate, boreal and polar seas are more complicated, since there is no direct latitudinal interconnection between the oceans. The displacement of the boundaries is most marked in the northern Atlantic Ocean, where the Gulf Stream pushes the boreal/subpolar boundary 30° further north on the eastern margin of the ocean. A consequence of the greater preponderance of land masses in the northern hemisphere and the greater displacement of zonal boundaries is that there is a much greater cold water coastline in the north than the south. For example, antiboreal coastline is represented only by the southern tip of South America and a few southern ocean islands (see Figure 8).

Cold water ecosystems range in annual mean temperature from around 18°C in the temperate zones to 5°C at the edge of the cool or boreal/antiboreal zones. At higher latitudes (polar and subpolar), surface temperatures are below 5°C. Probably of more ecological significance is the difference in annual temperature range between the different zones. In tropical regions, surface water temperatures show an annual variation of no more than 2°C

(Ekman, 1967), whereas in temperate to subpolar waters this may reach up to 10°C (see table below). Temperature changes are seasonal in cold water ecosystems and are thus important in controlling cyclical biological events such as reproduction. As a result of thermally induced mixing of water layers, seasonal temperature changes are an important factor in enhanced productivity in these regions (Koblentz-Mishke *et al.*, 1970).

In addition, temperate to subpolar regions correspond to a band of greater atmospheric turbulence and experience many storms, especially during winter, with gales and heavy precipitation (Gross, 1987). This turbulence also aids water mixing and hence nutrient dispersion, resulting in a greater likelihood of high-energy shores on coastlines in this region.

Mean Surface Temperatures in the Coldest and Warmest Months and the Annual Mean at the Boundaries of the Atlantic Boreal Region

	Temperature, °C		
	February	August	Annual Mean
Murman coast, Kola peninsula	1	9-10	4
Northern Iceland	0-1	8-10	3-4
Bear Island (between Spitzbergen and Norway)	0-1	5	2-3
Southern tip of Greenland	1	5	3
English Channel, southwestern entrance	9	17	12-13

(from Ekman, 1967)

FAUNA AND FLORA

The cold water environment embraces a range of temperature regimes and other climatic factors that place stress on the plant and animal communities. These stresses are manifested principally in a reduction of coastal species diversity that broadly forms a gradient from low to high latitudes (Sanders, 1968). Another consequence of the more stressful conditions is greater instability in the communities, which can vary both in species composition and abundance from year to year, relying heavily on successful recruitment of short-lived organisms each season. Such reproductive strategies theoretically favour rapid recovery after environmental impact. However, reproductive strategies, particularly of the fauna, also relate to the severity of the environment. Marine fauna may reproduce either by viviparity (the bearing of live young), direct development (directly from egg to miniature adult), or by larviparity (the egg develops into a larval form that spends some time in the water column before settling and metamorphosing into a miniature adult). In the latter case the larvae may be planktotrophic, in which case they feed on other organisms in the plankton, or lecithotrophic where they are nourished by a yolk supplied by the parent.

Planktotrophic larvae hatch from small eggs, since they do not carry food reserves, whereas other reproductive strategies are characterised by large egg production to accommodate nourishment for the developing embryos (Thorson, 1957). Because the energy available to a parent for egg production is limited, the larger the egg the smaller is the number of eggs produced. Thus, the potential for a population of non-pelagic breeders to recover quickly from a mass mortality is considerably less than for one producing large numbers of planktotrophic larvae.

In the higher latitudes, where the summers are short and the plankton production curtailed, there is a tendency for fewer animals to reproduce with planktotrophic larvae and a preponderance of species with viviparous or direct development (Thorson, 1950) or a brooding habit (Wells and Percy, 1985). This is illustrated for marine snails in Figure A1.

The implication is that communities from higher latitudes (subpolar and polar) may recover more slowly from environmental damage than those from farther south, where planktotrophic reproductive strategies predominate (Dunbar, 1968; Clarke, 1983).

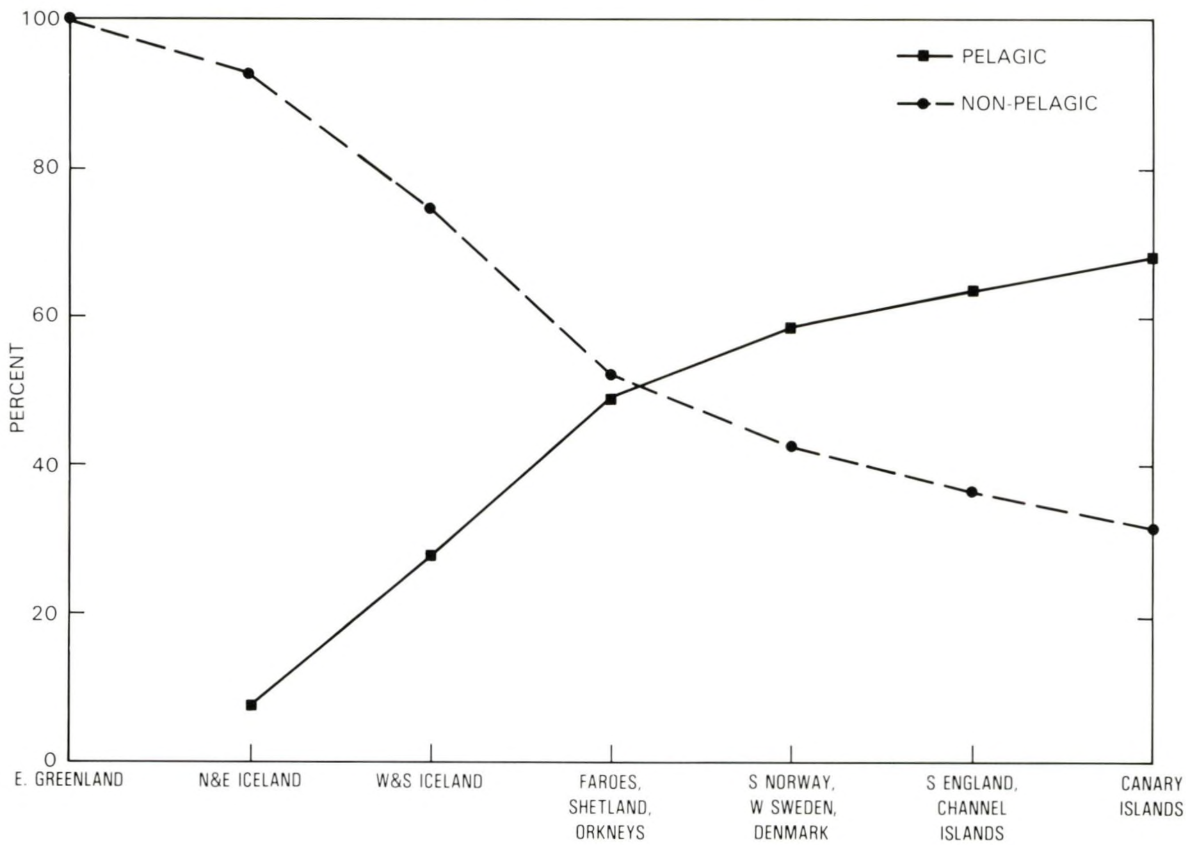

Figure A1: The relations between a pelagic/non-pelagic larval development in prosobranchs throughout the North Atlantic (Thorson, 1950).

APPENDIX B

PHYSICAL AND CHEMICAL BEHAVIOUR OF SPILLED OIL

BEHAVIOUR AT SEA

The marine environment may be divided into the open water or pelagic division and the seabed or benthic division. The benthic division extends from the top of the shore to the deepest part of the ocean. These divisions form various ecological zones (Figure B1) and broadly follow a scheme first proposed by Hedgpeth (1957).

In oceanic waters, impact from oil spills is likely to be minimal and confined more or less to the surface and near-surface water layers. The greatest impact is usually in coastal waters and on shores (littoral zones).

The shoreline, where land, air and sea meet, is the most dynamic part of the ocean (Gross, 1987). Its nature is determined by tides, winds, waves. The most obvious of the factors are waves which, together with offshore currents, are most influential in determining the type of intertidal substratum. Where the shore is sheltered from rigorous wave action (so-called low-energy shores), sedimentary beaches will form. These may range from highly sheltered or estuarine mud flats, through sand beaches, to gravel or boulder shores where wave and current action is strong enough to wash fine particles away faster than they are brought in. Rocky shores exist where the effect of waves on the coastline is mainly erosive, wearing down the softer materials and carrying them away, leaving the hardest rocks exposed. Rocky shores are biologically zoned, with the zones varying in extent depending upon the slope of the rock surfaces, the tidal range, and exposure to wave action. Thus for a given tidal range and shore gradient, the range of the littoral zone increases with increasing wave exposure (see Figure B2).

When crude oil and petroleum products are released at sea, they are subject immediately to a series of complex interactions of physical, chemical and biological processes (Koons, 1987). These processes are collectively called 'weathering' and include

- Spreading
- Evaporation
- Sinking
- Dissolution
- Natural dispersion into the water column
- Emulsification
- Biodegradation and photolysis

The time-frame following an oil spill over which the above processes occur is illustrated in Figure B3.

The significance of these natural processes in determining the fate of spilled oil is determined by a number of factors, such as the type of oil spilled and the oceanographic and meteorological conditions. A brief discussion of the above processes follows.

Spreading

Spilled oil initially spreads over the sea surface to form a relatively thin slick. Under the influence of 'weathering' and winds, thicker layers of oil/mousse may be formed.

Under the influence of gravity and the difference in density between oil and water, the spilled oil will spread over the surface of the sea. Amongst the mathematical models which describe this process, the best known are those of Blokker (1964) and Fay (1969).

Several models are available to predict the movement or drift of the surface slick under the influence of winds, surface currents, and waves. There is a tendency, with time, for the oil slick to become non-homogeneous in thickness, and to form lenses of thicker portions of oil which align themselves along the direction of the wind to form 'windrows'. The formation of thicker lenses of oil becomes more significant as the viscosity of the oil increases due to 'weathering'. Hollinger and Mennella (1973) have reported that as much as 90 percent of the oil in the slick (after evaporation) may be associated with these thick patches, although the major area of a slick is composed of a relatively thin sheen.

Observations made at the Ixtoc 1 oil well blowout (Atwood *et al.*, 1980) indicated that near the blowout, the oil slick thickness ranged from 1 to 4 mm. However, at a more distant location, long lines of thicker material were observed surrounded by thin films of oil. At times, the winds rolled the portions of the slick upon itself, and rafts of agglomerated material up to 1 m thick were seen 5 to 15 nautical miles from the spill site.

Evaporation

Most of the toxic components in crude oil rapidly evaporate after spillage, and disperse into the atmosphere to background levels. Evaporation is a major process for the removal of oil from the sea surface for lighter crudes.

When oil is released, the volatile components rapidly evaporate. For example, estimates from major incidents such as the Ekofisk Bravo, *Amoco Cadiz*, and Ixtoc 1 indicated that 30 to 50 percent of the spilled oil evaporated within the first few hours or days. This process is especially dominant in the initial phase following a spill and is a major process that removes oil from the sea surface—particularly for light crudes.

The rate of evaporation is dependent upon

- Spreading (the increase in surface area of the oil will facilitate evaporation)
- Temperature

- Wind speed
- Sea state conditions
- Oil composition

Spills of light petroleum products such as liquefied petroleum gas (LPG) and gasoline will evaporate very quickly, leaving virtually no residual surface slick.

The volatile components in crude oils which evaporate include the toxic hydrocarbons benzene and the mono-nuclear aromatics. After evaporation, these organics disperse into the atmosphere and are rapidly diluted to 'background' levels. Payne *et al.* (1980) found that air samples taken immediately above the surface slick of freshly spilled oil at the Ixtoc 1 oil spill had concentrations of benzene below the level for detection (1 ppb) by selected ion monitoring.

For spills of relatively non-volatile materials such as heavy fuel oils and Bunker C, the evaporation losses from the slick would be much lower. Studies following the *Potomac* spill of Bunker C oil in Melville Bay, Greenland (Petersen, 1978), attributed additional factors such as light winds, low water temperatures, thick oil slicks, and the oil composition to the low evaporation rates experienced.

Sinking

> *Most crude oils have little tendency to sink to the seabed even after extensive weathering and associated density increases. Oil concentrations in the water column measured below the surface slicks at sea trials and oil spill incidents have been very low. Similarly, samples of the seabed taken at offshore oil spill sites have been only lightly oiled. Some oil may reach the seabed associated with particulates. This process is unlikely to be significant for spills offshore unless the seawater is highly turbid. Spills of very dense oils* (e.g., *heavy fuel oil) may weather to form slicks of sufficient density to sink to the seabed.*

Wilson *et al.* (1985) conclude that most crude oils have little tendency to sink to the seabed even after extensive weathering and the density increases associated with emulsion formation. They attribute reports of oil sinking to the ability of heavy oil slicks to adopt a position just below the sea surface and so become undetectable by remote sensing systems such as IR.

Sea trials carried out with Ekofisk crude oil (Cormack *et al.*, 1978) showed a rapid decrease in oil concentration, falling to 0.02 to 0.08 ppm at 10 to 15 m below the oil slick. Measurements taken approximately 50 hr after the well had been capped indicated oil concentrations as low as 0.055 to 0.118 ppm at a 1-m depth below oil sheens (Cormack and Nicholls, 1978).

Maximum subsurface concentrations of oil measured at major spills are 0.30 ppm for Ekofisk Bravo (Grahl-Nielsen, 1978), 0.35 ppm for *Amoco Cadiz* (Calder *et al.*, 1978), and 0.45 ppm for *Argo Merchant* (Grose and Mattson, 1977).

At the Ixtoc 1 oil well blowout, the hydrocarbons were released subsea which resulted in relatively high concentrations (10.6 ppm) of oil droplets measured in the top 20 m of the water column near the spill site (Fiest and Boehm, 1980). Boehm and Fiest (1980) concluded that only 1 to 3 percent of the oil spilled partitioned into the sediments. However, the area experienced a hurricane before the samples were taken. This might have significantly affected the contamination levels of the sediments.

The sediments sampled from the seabed in the area of the Ekofisk Bravo blowout, had relatively low concentrations of oil immediately after the blowout and fell to background levels 4 to 6 weeks after the well had been capped (Johnson *et al.*, 1978). Studies of the biological populations in the sediments (Addy *et al.*, 1978) were not able to attribute any specific changes to the blowout.

Significant levels of oil contamination of sediments are most likely to occur in the nearshore environments, where substantial quantities of sediments are suspended in the water column by the wave energy (see later section entitled Behaviour Near Shores).

Petersen (1978) studied the spill of Bunker C heavy fuel oil in Melville Bay, Greenland, when a combination of low water temperatures (3 to 4°C), light winds (0 to 7 knots) and calm seas resulted in relatively slow evaporation, biodegradation and dispersion rates. However, because of the special conditions described above, this can be viewed as an exceptional situation. Pancakes of spongy material ranging from 8 to 15 cm in diameter arranged in windrows on the sea surface were observed 15 days after the spill. It was estimated that most of the residues eventually sank in less than 50 days.

Dissolution

> *Less than 1 percent of spilled oil is likely to be dissolved into the seawater. Although the water-soluble constituents of oils are relatively toxic, they are rapidly diluted to very low concentrations in the water column.*

The components of oil which dissolve in the seawater are the volatile, low molecular weight compounds which are relatively toxic. However, on physico-chemical grounds it is expected that the evaporation rates of such compounds would be two orders of magnitude greater than rates of solution (Cormack, 1983). Furthermore, a substantial proportion of these dissolved hydrocarbons are likely to be evaporated from the water column.

Because of their toxicity, the dissolved components pose a potential threat to marine life. However, this toxic effect is short-lived because the soluble components are quickly dissolved from the fresh oil and rapidly diluted in the water column to very low concentrations. Measured concentrations of dissolved hydrocarbons underneath slicks were less than 0.1 ppm (McAuliffe, 1986). Payne *et al.* (1980) measured up to 0.1 ppm of benzene, toluenes, xylenes and other low molecular weight aliphatic and aromatic hydrocarbons in seawater samples taken immediately beneath the surface slick near to the Ixtoc 1 oil well blowout.

It has been estimated (Mackay and McAuliffe, 1989) that less than 1 percent of the crude oil spilled would be likely to dissolve into the seawater.

Natural Dispersion into Water Column

Natural dispersion is generally recognised to be a major process for the removal of oil from a surface slick by the formation of small droplets. The process is favoured by high wave energy conditions and for low-viscosity oils.

Natural dispersion is the process whereby small oil droplets (0.01 to 0.1 mm) are generated from an oil slick by wave energy. Because of their small size, the droplets disperse speedily into the body of water, resulting in ever-decreasing concentrations.

Natural dispersion is generally recognised to be a major process for the removal of oil from the sea surface. This is particularly the case for low-viscosity, lighter oils which do not form highly viscous weathered materials. Studies carried out following the Ekofisk Bravo blowout (Cormack, 1983) indicated that only about 1 percent of the initial quantity of oil spilled would remain as a surface slick after about 80 hours. Sea trials with the Ekofisk crude oil indicated that even in moderate sea conditions, natural dispersion represented a major process which contributed to the complete removal of oil from the sea surface.

The rate of dispersion of oil is enhanced by high wave energy conditions and is inhibited with increased viscosity of the oil slick. Evaporation of the volatile components leads to an increase in viscosity of the remaining slick, but the most significant increases can result from the formation of water-in-oil emulsions (discussed below). Slicks of highly viscous oils/emulsions are relatively persistent due to their cohesiveness (Mackay and McAuliffe, 1989). Floating slicks of viscous emulsions are slowly broken up into smaller rafts or pancakes of material, then finally form tarballs. When this material reaches the nearshore, where sediments are suspended in the water column, the sediments can become enmeshed into the residual oil to form asphaltic deposits or pavements. Floating tarballs on the sea surface are relatively innocuous to marine life, but may persist for many months or years.

Emulsification

The formation of water-in-oil emulsions (mousses), which commonly occurs following marine oil spills, can have a very significant effect on the fate and behaviour of spilled oil. Some crude oils rapidly form stable emulsions of very high viscosity and form slicks which are relatively persistent.

The formation of water-in-oil emulsions (mousses) can have a significant effect on the fate and behaviour of spilled oil. Some crude oils and fuel oils have been shown to form very stable emulsions that contain 20 to 80 percent seawater. These emulsions have densities approaching that of seawater and viscosities exceeding 100,000 cP (EEC, 1986). Slicks of highly viscous material are relatively persistent (*i.e.*, resistant to natural dispersion).

The key compositional factors in the oil which favour the formation of stable/viscous emulsion are not known, but are believed to include asphaltenes, waxes, natural surfactants and surfactants formed by photolysis. Emulsification is also enhanced by low temperatures and by high-energy conditions.

The formation of mousse is a very common feature of marine oil spills. The emulsification process leads to an increase in the quantity of material in the slick, its density, its viscosity and, hence, its persistence on the sea surface. Light refined petroleum products such as gasoline usually do not form water-in-oil emulsions.

Biodegradation and Photolysis

> *Biodegradation is regarded as the ultimate fate of much of the dissolved and dispersed oil, but the process generally has a long time scale. The rate is controlled by oxygen and nutrient availability, temperature, chemical composition, and the surface area of the oil. Photolytic oxidation of some oil components occurs under the action of ultraviolet radiation (in sunlight). The oxidation products from both biodegradation and photolysis may be relatively toxic, but are biodegradable, water-soluble and rapidly dilute to very low concentrations in the sea.*

Naturally occurring populations of bacteria, yeast and fungi which are present in seawater (and in other environments) are capable of degrading petroleum hydrocarbons to oxidation products. Biodegradation is regarded as the ultimate fate of much of the dissolved and dispersed oil, but the process takes place over a long time scale. The rate of biodegradation is controlled by temperature, oil composition, surface/volume ratio of the oil, and the supply of oxygen and of nutrients such as nitrogen and phosphorus. Bacteria preferentially attack the low molecular weight components in the oils and the normal paraffins.

Water samples taken at the mouth of Aber Wrac'h following the *Amoco Cadiz* spill indicate a large reduction in nC-17/pristane and nC-18/phytane ratios and an increase in the content of branched, cyclic and aromatic hydrocarbons compared with samples taken near the spill site (Calder and Boehm, 1981). From this, the authors conclude that biodegradation was occurring at a faster rate than evaporation or dissolution.

The biodegradation of the mousse formed from the Ixtoc-1 blowout was found to be extremely slow (Atlas *et al.*, 1980), and the supply of nutrients was suggested as the limiting factor. Higher rates of biodegradation of mousses and of stranded tarballs when they are associated with decaying plant material have been observed (Boehm and Fiest, 1980; Blumer *et al.*, 1973), presumably due to nutrients. The biodegradation of oil in sediments is discussed later in the report.

Under the action of ultraviolet radiation (in sunlight) and the presence of air, some components present in oil are oxidised by photolysis. These oxidation products include phenolic and acidic compounds that are water-soluble and biodegradable. The oxidation products derived from photolytical reactions of spilled oils are relatively soluble. However, their rates of generation at the surface of spilled oil are slow because they are controlled by diffusion. Consequently, if these products are leached from the surface of the oil and enter the water column, they are rapidly diluted. Although the effects on marine life due to the formation of these oxidation products have not been quantitatively assessed, they are unlikely to have significant ecological impact.

BEHAVIOUR IN ICE

> *Oil may be entrained within a developing ice field under freezing conditions. This process would significantly reduce or even stop evaporation, dissolution, biodegradation and dispersion of the oil.*

From a series of field experiments, Wilson and Mackay (1986) concluded that significant quantities of oil may be entrained within a developing ice field under freezing conditions. The extent of oil incorporation was enhanced by

- A high oil density and/or viscosity occurring naturally or induced by weathering
- The presence of sufficient turbulence to disperse the surface oil and to induce mixing within the ice field
- The formation of emulsions of seawater and/or ice in oil
- The formation of small oil droplets
- The formation of coalesced ice particles measuring about 5 mm in diameter

The authors also concluded that the densest oils would be released relatively slowly from a frozen pancake during thawing. Once the oil begins to collect on the surface of the ice, solar radiation will tend to hasten the thawing process.

From a series of experiments in flow-through tanks, Payne *et al.* (1987) observed that extremely rapid formation of stable water-in-oil emulsions occurred under ice-forming conditions. Emulsions of Prudhoe Bay crude oil were formed within four hours of initiating waves in the presence of grease ice and breaking or rotting ice floes. They attributed this rapid emulsion formation to the low water temperatures and to the micro-scale turbulence created by the grinding of the grease ice crystals, which injected small water droplets into the viscous oil. On thawing, these emulsions were sufficiently dense to cause them to reside immediately below the grease ice. With continued agitation and melting, the emulsions eventually surfaced into patches of open water between the individual ice floes.

When oil is trapped into ice, weathering processes such as evaporation, dissolution, biodegradation and dispersion are significantly retarded or even stopped. Hence when released after thawing, the oil will be relatively 'fresh'.

In addition to being rapidly formed, emulsions formed in the presence of developing ice have significantly higher viscosities than those formed under ice-free conditions.

BEHAVIOUR NEAR SHORES

The coastline exposure and geomorphology are very important factors which influence the retention and dispersal of oil in the nearshore areas. The churning action of waves in shallow waters may result in the oil being incorporated into sediments. Beached oil may be washed off and may also become incorporated into subtidal sediments. Alternatively, the oil on the beach may be refloated and transported to impact other stretches of the coastline, or dispersed into the water column as fine droplets. Oiled sediments on beaches could be buried under fresh deposits of sand. However, oil transported from beaches is usually weathered and less toxic than fresh oil. High-energy rocky shores tend not to accumulate oil and, if impacted, are rapidly cleaned by wave action.

Because of the turbulence in shallow inshore waters and the churning action of waves, the relatively high sediment burden of these waters can result in oil being incorporated into the bottom sediments to a greater extent than in the offshore environment.

The oil stranded on the intertidal coastal areas may be partially removed by subsequent tides and transported to unimpacted stretches of the coastline. During the first two weeks following the *Amoco Cadiz* oil spill, a total of 72 km of coastline was heavily impacted (Hayes *et al.*, 1979). One month after the incident, 84 percent of the stranded oil was estimated to have been naturally removed from the shoreline. But the impacted area had increased to 213 km of lightly oiled and 107 km of heavily oiled beaches.

Hayes *et al.* (1979) also report that the geomorphology in the coastal zone was very important in the dispersal and accumulation of oil once it came onshore. Exposed rocky coastlines did not accumulate significant quantities of oil. Mousse and oil stranded on fine-grained sandy beaches or sediments did not usually penetrate far below the surface.

Sandy beaches may undergo cycles of erosion and deposition in response to changing wave conditions. Hence, high-energy beaches may rapidly self-clean or, alternatively, the oiled sediments may become buried under fresh deposits of sand, dependent upon the phase of the beach cycle.

Local entrapment of oil was found to occur between rock crevasses, in marsh pools and in scour pits around boulders. Penetration of oil into coarse-grained sandy beaches, mixed sand and gravel beaches, and gravel beaches may occur rapidly.

Oiled sediments washed from the beaches can be a major contributor to oiling of subtidal sediments. This is not an important process for high-energy, rocky coastlines.

Harper *et al.* (1985) studied the retention and residence times of oil in low-wave-exposure, tidal flat environments. Their main conclusions were that

- Retention of oil in the sediments was proportional to the loading thickness, up to a limit beyond which the retention increased very little. This limiting thickness for oil loading was thought to be a function of the sediment size.

- The retention was proportional to the sediment emergence time. Intertidal sediments which are exposed longer have greater oil retention times.

- The retention was inversely proportional to the mud content of the sediments. Even small amounts of mud may significantly reduce oil retention in sediments.

- Sediment size and composition were thought to be the most important factors which influence oil retention.

Mechanical removal of bulk oil from beaches is beneficial (1) by reducing a potential source of oiling of the subtidal sediments and other stretches of the coastline and (2) by enhancing the self-cleaning of intertidal sediments.

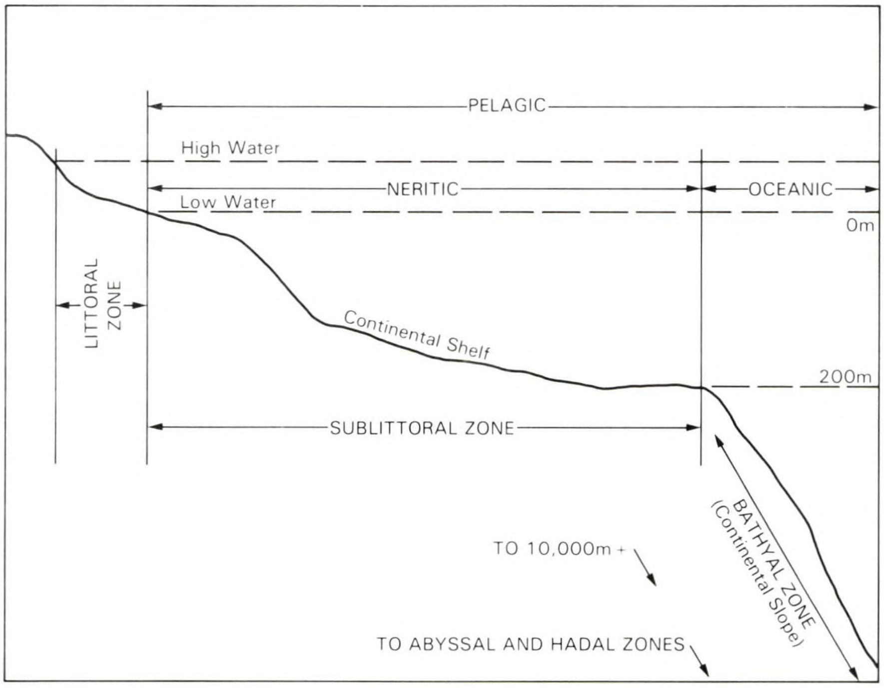

Figure B1: Divisions of the marine environment (modified after Hedgpeth, 1957).

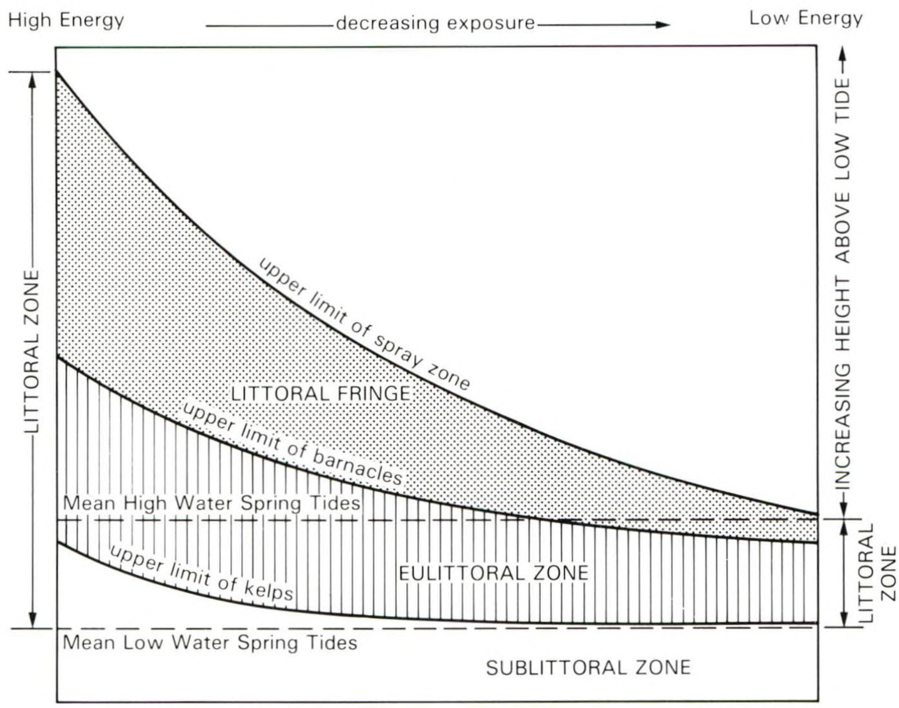

Figure B2: Effect of exposure on the width of the littoral zone of a rocky shore (modified after Lewis, 1964).

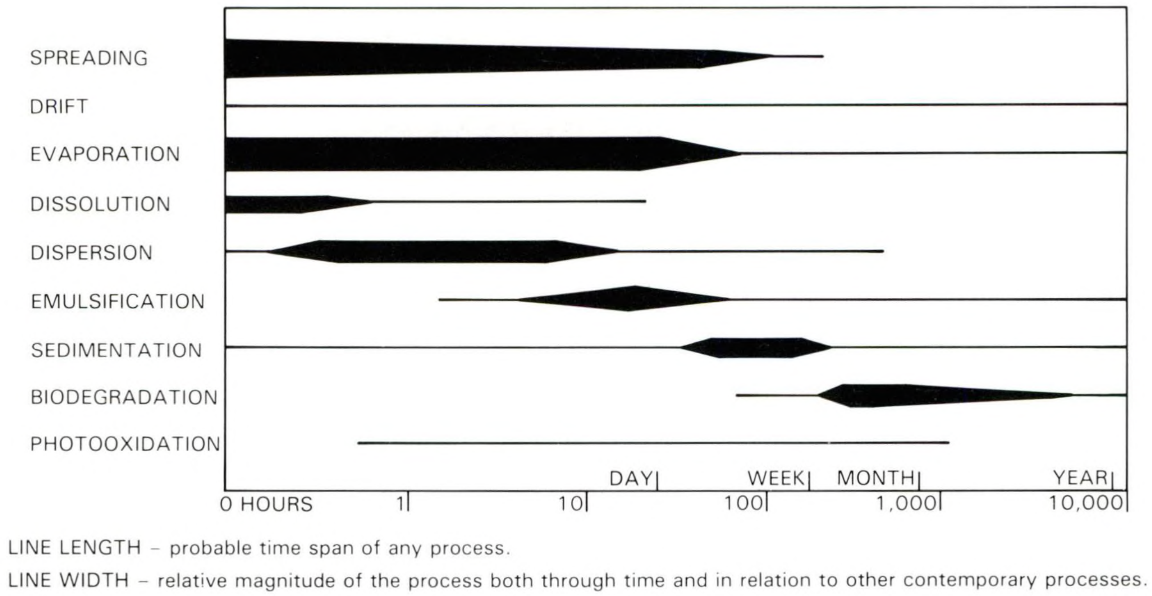

Figure B3: Time span and relative magnitude of processes acting on spilled oil (modified after Whittle *et al.*, 1982).

APPENDIX C

EFFECTS OF OIL ON BIRDS

The most serious effect of oil is on the bird's plumage. Oil mats the feathers and destroys their water-repellant properties. For aquatic birds, this results in water displacing the air normally trapped under the plumage. This air layer provides buoyancy and thermal insulation. Birds with a water-logged plumage may sink and drown, but will certainly suffer rapid loss of body heat; fat reserves are used to counter this, but within a short time these are exhausted and the birds succumb to hypothermia, pneumonia, or related diseases. It may be anticipated that birds are more likely to die after oiling in cold climates (Brown, 1982) or after prolonged stormy weather has prevented feeding and energy reserves are low (NERC, 1971).

Lightly-oiled birds are able to clean themselves by preening within about 2 weeks (Birkhead *et al.*, 1973), but in doing so swallow oil. Depending on the age and toxic properties of the oil, this may cause gastric disorders (Croxall, 1977) and may reduce egg-laying or decrease the fertility of eggs that are laid (Grau *et al.*, 1977). While these factors may result in additional deaths or depress reproduction for a time, they are insignificant compared with the direct mortality from oiling.

If oil is transferred from the plumage to incubating eggs, the developing embryo may be killed and breeding success reduced (Brown, 1982), although there is no evidence that this happens on a large or widespread scale.

Birds at Risk

The casualties of oil pollution (*e.g.*, bilge water, dirty ballast water, *etc.*, as well as oil spills) are overwhelmingly aquatic birds, though in exceptional circumstances shore birds or even land birds may be affected. The following birds are most at risk.

Alcids (auks)

These include murres, guillemots, razorbills, puffins, *etc.* They are weak fliers and spend virtually all their time on the surface of the sea, returning to land only for breeding. They hunt their food (small fish) under water and are very gregarious at all times of the year. Because of their habits, auks are more likely than most birds to encounter oil slicks and, since they dive rather than fly up when disturbed, may surface through the oil slick and become coated with oil. Because they occur in large, dense flocks on the water, casualties from an oil spill are likely to be high.

Diving Sea Ducks

Many species of sea duck disperse to land or fresh water for breeding, but occur in very large flocks in coastal waters during the fall and winter. Some species dive for their food in a similar manner to auks and are vulnerable to oil slicks in the same way. But for all species, casualties may be very high if oil impacts winter flocks in coastal waters.

Grebes and Loons

These, like auks, are divers and weak fliers. They breed in fresh waters but spend the winter in ice-free coastal waters. They are likely to become oiled if they encounter an oil slick, but since they occur in only small groups, casualties are usually small. However, because the world population of these birds is small, even low casualties may be significant.

Pelican, Gannet, Tern

These birds dive from the air into the sea to catch fish. If, as sometimes happens, they dive through an oil slick, they become coated with oil. Pelican and gannet casualties have been reported, though not in large numbers. Terns appear to avoid this hazard.

Other Birds

Almost any seabird or shore bird may encounter floating or stranded oil and suffer some oiling of the plumage, the extent of which is usually slight, and casualties are small.

BIOGRAPHICAL SKETCHES OF THE AUTHORS

DR. JENIFER M. BAKER

Dr. Baker is a biological consultant specialising in environmental impact assessment and oil spill response. Her doctoral studies at the University of Wales involved research on the effects of oil on and cleaning methods for salt marshes, and she has subsequently worked on oil pollution problems in many parts of the world. She was formerly Research Director of the UK Field Studies Council, and is presently a fellow of the Institute of Biology and of the Institute of Petroleum. Dr. Baker has published numerous papers in the scientific literature on the subject of the recovery of impacted shoreline ecosystems.

DR. ROBERT B. CLARK

Dr. Clark is currently Professor Emeritus of Zoology at the University of Newcastle upon Tyne. He received his Ph.D. from the University of Glasgow and a D.Sc. from the University of London. He has served as Honorary Director of the Seabird Research Unit of the British Advisory Committee on Oil Pollution of the Sea (1969–75), and as Director of the Natural Environmental Research Council (NERC) Research Unit on Rocky Shore Surveillance (1980–87). Dr. Clark also worked with numerous national and international bodies, including the United Nations group of experts on the Scientific Aspects of Marine Pollution, and the Royal Commission (UK) on Environmental Pollution. His extensive publications include the textbook *Marine Pollution*. He is founder (1969) and Editor of the *Marine Pollution Bulletin*.

DR. ROWLEY H. JENKINS

Dr. Jenkins became Deputy Director, in 1986, of the Institute of Offshore Engineering at Heriot-Watt University in Edinburgh, Scotland. He had earlier served 28 years in the petrochemical and oil industry in research and environmental areas. His current focus in oil spill research includes emulsification and weathering, as well as modelling and fingerprinting by GC/MS analysis. Part of his environmental experience involved a two-year secondment to Woodside Petroleum as Environmental Coordinator for the North West Shelf Gas Project in Western Australia. In his post as Oil Spills Divisional Manager for British Petroleum (BP), Dr. Jenkins provided the BP group worldwide with a wide range of services — *e.g.*, emergency response from the Southampton Oil Spill Response Base, equipment testing and appraisal, and oil spill response training. He was active in both CONCAWE and IPIECA as the BP representative.

DR. PAUL F. KINGSTON

Dr. Kingston obtained his doctorate from the University of London and, after a three-year post-doctoral research fellowship at the University of Newcastle upon Tyne, joined the staff at Heriot-Watt University in 1975 as lecturer in marine biology. He is currently the

Assistant Director of the Institute of Offshore Engineering. Earlier, he had acted as Consultant to the Institute on all offshore, inshore, and coastal environmental projects, and served as part-time Assistant Director with responsibility for biological studies. Dr. Kingston has had considerable experience in assessing the environmental impact of the offshore oil industry and has worked on most major North Sea developments. His research interests centre on the structure and dynamics of seabed communities. He has published extensively and currently serves as News Editor of the *Marine Pollution Bulletin.*

GLOSSARY

aerobic bacteria	Bacteria that can live only in the presence of free oxygen.
aliphatic hydrocarbons	Saturated straight-chain or branched-chain hydrocarbons.
anaerobic bacteria	Bacteria that can live in the absence of free oxygen.
antiboreal	Cool or cold-temperate regions in the southern hemisphere (see Figure 8).
aromatic hydrocarbons	Hydrocarbons containing one or more benzene rings in their molecular structure.
asphaltenes	High boiling point constituents of crude oils that are soluble in polar solvents such as benzene or methylene chloride but are not soluble in paraffin naphthas.
bacterial mat	A carpet of filamentous bacteria which can form over the surface layer of organically enriched sediments.
benthic fauna	Animals inhabiting the seabed.
benthos	Those forms of marine life that are bottom-dwelling; also, the ocean bottom itself. Certain fish that are closely associated with the benthos may be included (AGI).*
biodegradation	Breaking down of substances by bacteria.
biogenic	Originating from living matter.
biomass	The amount of living material in a particular area, stated in terms of the weight or volume of organisms per unit area or of the volume of the environment (AGI).
biota	All living organisms of an area; the flora and fauna considered as a unit (AGI).
bivalves	Molluscs having a shell in the form of two plates, *e.g.*, clams.
boreal	Cool or cold-temperate regions in the northern hemisphere (see Figure 8).

* American Geological Institute, *Glossary of Geology*, Third Edition

copepod	A subclass of *Crustacea*, generally of small size. These crustaceans are important members of the zooplankton and meiofauna.
crustaceans	A class of *Arthropoda*, mostly of aquatic habitat, which includes shrimps, prawns, barnacles, crabs, and lobsters.
depurate	To rid of contaminants, especially with relation to shellfish.
diatom	A microscopic, single-celled plant which grows in both marine and fresh water.
dispersants	Chemical mixtures containing surface-active agents which enhance the dispersal of oil in water.
ecosystem	Any area of nature which includes living organisms and non-living substances interacting to produce an exchange of materials between the living and non-living parts, *e.g.*, the sea, a pond, lake, or forest. It comprises four constituents: abiotic substances, producers, consumers, and decomposers, and is the basic functional unit in ecology.
emulsification	A colloidal suspension of one liquid in another, *e.g.*, mousse.
epifauna	Fauna living upon rather than below the surface of the seafloor (AGI).
eutrophication	Depletion of oxygen in the water column in response to an increase in nutrients and associated elevation in primary production.
fauna	A collective term denoting the animals occurring in a particular region or period.
fucoid algae	Brown seaweed belonging to the family *Fucaceae*.
gas chromatography	An analytical technique for the separation of components in a mixture on the basis of boiling point. The use of very sensitive detectors enables this form of chromatography to be applied to microgram quantities.
gasoline	A distillate of crude oil (boiling temperature less than 190°C), having a carbon number distribution between C5 and C10, that is used as a liquid fuel.
herbivore	An organism that feeds on plants.

infauna	Those aquatic animals that live within rather than on the bottom sediment (AGI).
intertidal	The benthic ocean environment between high- and low-tide water levels (see Figure B1). Syn. *littoral.*
lecithotropic	A form of larval development whereby the eggs are dispersed in the water column and contain an independent food source for the developing larvae.
lipids	Generic terms for fats, waxes, and related products in living tissues.
littoral	See *intertidal* (see Figure B1).
lysosomal	Cellular particles intermediate in size between mitochondria and microsomes and which contain hydrolytic enzymes.
macrofauna	Animals large enough to be seen with the naked eye.
meiofauna	Organisms in the size range of 0.1 to 1.0 mm that live within sediments.
metabolic	The chemical and physical changes constantly taking place in living matter.
mousse	A viscous water-in-oil emulsion which is often brown in color (also: chocolate mousse).
n-alkanes	Normal alkanes; straight-chain aliphatic hydrocarbons; paraffins.
nematodes	A phylum of unsegmented worms with an elongate rounded body pointed at both ends. Included are round worms, thread worms, and eel worms.
neritic	That portion of the seafloor lying between low-water mark and the edge of the continental shelf, at a water depth of about 180 m (see Figure B1).
oleophilic	Having a strong affinity or preference for oil.
opportunistic species	Species which are able to quickly colonise and exploit environmentally disturbed areas.
paraffins	A whole series of saturated aliphatic hydrocarbons of the general formula C_nH_{2n+2}.
pelagic	Living in the middle depth and surface of waters of the sea.

petrogenic	Derived from rocks.
petroleum hydrocarbons	Liquids that are generated through the action of time and temperature on organic matter buried below the earth's surface. They consist of saturated and unsaturated structural groups.
phenolic compounds	Compounds containing a hydroxyl function (OH^-) substituted on a benzene ring.
photolysis	The decomposition or dissociation of a molecule as the result of the absorption of light.
phytane	A branched-chain saturated hydrocarbon containing 20 carbon atoms.
phytoplankton	Plant members of the plankton.
plankton	Animals and plants floating in the water of seas, rivers, ponds, and lakes, as distinct from animals which are attached to, or crawl upon, the bottom; especially minute organisms and forms.
planktotrophic	A form of larval development whereby the eggs are dispersed pelagically and the larvae then spend time as temporary members of the plankton where they feed and grow before settlement.
polar compounds	Molecules in which the electrical charge distribution is dipolar (like a bar magnet); commonly used to refer to components of crude oil containing the elements N, S, and O in addition to H and C.
polychaete	A type of segmented marine worm, *e.g.*, bristle worm.
polynuclear aromatic	Aromatic hydrocarbons containing two or more benzene rings.
pristane	A branched-chain saturated hydrocarbon containing 19 carbon atoms.
recruitment	Addition by immigration or reproduction of new individuals to a population.
saturated hydrocarbon	A general term including straight-chain, branched-chain, and ring structures in which all carbon-carbon bonds are covalent. Syn. *alkanes*, *paraffins*.

shingle	Coarse, loose, well-rounded, water-worn detritus, especially beach gravel, composed of smooth and spheroidal or flattened pebbles, cobbles, and sometimes small boulders, generally measuring 20–200 mm in diameter; it occurs typically on the high parts of a beach (AGI).
sublethal toxicity	The toxicity of a substance which is shown to cause deleterious effects in plants and animals, but not death.
substrata	Non-living materials to which a plant is attached and from which it obtains substances used in its nutrition.
subtidal	The benthic ocean environment below low tide which is always covered by water (see Figure B1). Syn. *sublittoral.*
tarballs	Water-insoluble, buoyant agglomerates formed in water from weathered crude; consist largely of the non-hydrocarbon fractions.
toxins	Poisonous substances of plant or animal origin.
zooplankton	Animal life that floats or drifts in water.