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**Evaluating the Long Term Effects
of Crude Oil Exposure in Sea Otters:
Laboratory and Field Observations**

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Summary

Sea otters continue to inhabit the area that was impacted by the March 1989 Valdez oil spill. Their presence raises questions about the long term effects of the spill and the ability of the habitat to support sea otters.

To examine the long term effects of exposure to petroleum hydrocarbons on sea otters, we monitored the general health, fur condition, blood parameters, and behavior of twelve sea otters placed in captivity following the Valdez oil spill. We compared the behavior and coat conditions of these captive otters to wild otters residing in previously oiled areas of Prince William Sound and the Kenai Peninsula, Alaska. In addition, we evaluated the current suitability of the area's habitat for sea otters.

Major findings of the study are:

1. The current prognosis for captive otters that survived the initial effects of hydrocarbon exposure is good.

2. Petroleum hydrocarbon levels in the blood of captive otters returned to background levels within twelve months of the spill.

3. Clinical evaluations indicate that captive otters gradually recovered from poor thermoregulatory control, respiratory distress, and anemia which were associated with short term exposure to crude oil.

4. Eighteen months after the Valdez spill, crude oil was found only on isolated sections of shoreline. In addition, the oil had weathered so that it presented little if any threat to sea otters.

5. Several hundred sea otters were observed eating, grooming, and resting in previously oiled areas of Prince William Sound and the Kenai Peninsula.

6. Captive and wild otters were reproductively active. One captive female, which had been moderately oiled, gave birth to a pup within one year of the spill. Many females with pups were observed in previously oiled areas of Prince William Sound.



Sea otters continue to inhabit the area of the 1989 oil spill.

Introduction

Previous studies have demonstrated the vulnerability of sea otters (*Enhydra lutris*) to the detrimental effects of crude oil (Costa and Kooyman, 1982; Davis *et al.*, 1988). Sea otters face two types of problems from exposure to crude oil. While external contamination of the fur results in decreased insulation (Williams *et al.*, 1988) and potentially lethal hypothermia, internal exposure to petroleum hydrocarbons contributes to another set of health problems. We diagnosed pulmonary emphysema, anemia, and hypoglycemia (low blood sugar) in otters contaminated during the Valdez spill. The incidence and severity of such problems correlated with the date of exposure, and hence, with the extent of oil weathering.

From the perspective of the otters' health problems, the Valdez spill can be divided into two distinct periods. During the initial phase, lasting



Many females with pups were observed in the previously oiled areas.

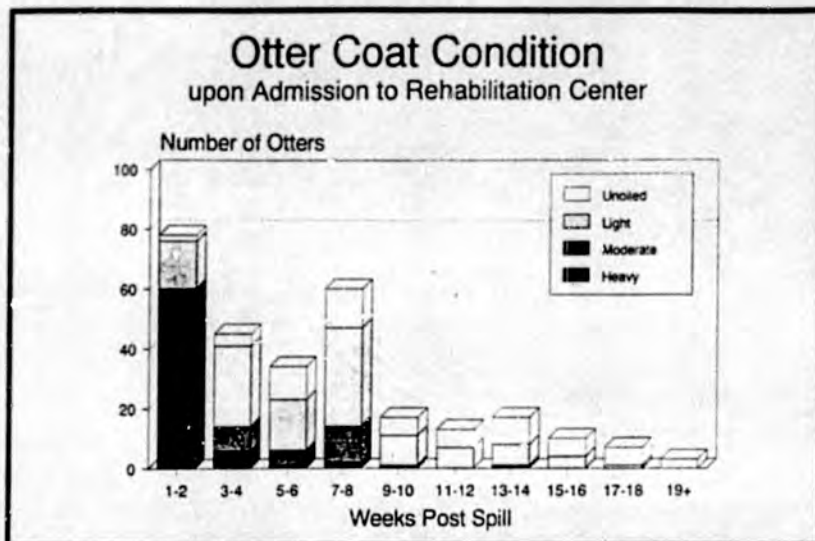


Figure 1. Degree of oiling on the coats of sea otters on admission to the rehabilitation centers. Note the rapid decline in the number of heavily and moderately oiled otters 3 weeks after the spill.

three weeks, contaminated otters showed the greatest degree of oiling (figure 1), encountered oil containing the most harmful crude oil components, and had the highest incidence of severe health problems. Although the rehabilitation program lasted several months, 54% of the deaths in the centers

occurred during this initial three week period. During the second phase, 70% of the otters captured within the spill area were lightly oiled or uniled. These animals were probably exposed to fewer harmful components, had comparatively fewer health problems and, hence, greater survivorship (Williams *et al.*, 1990).

Primary concerns for otters that survived the initial phase of the spill included: 1) long term effects due to oiling; and 2) continued contamination from residual oil in the habitat or in food items. To examine the former, we monitored the health of eight adult sea otters (seven females, one male) and four otter pups (three females, one male) for twelve months after the spill. For each



Nearly 18 months after the spill, oil was limited to isolated sections of beach. In most areas the oil had weathered so that it presented little threat to the otters.

animal, we determined: 1) general health; 2) body weight; 3) coat condition; 4) blood chemistry and hematology; and 5) reproductive status.

All of the adults in this study were captured during the initial phase of the spill and were severely impacted by the oil. Unlike the 197 adult otters that had been released back into the wild after rehabilitation, these eight adults were judged unsuitable for release and were therefore transferred to seaquariums for long term care. The pups used in this study were admitted to the centers during the latter phase of the spill (table 1).

We determined the likelihood of continued oil contamination in wild otters by assessing habitat conditions in previously oiled areas of Alaska. Major goals were to evaluate the condition of beaches and coastal waters in terms of the potential impact on sea otters, and to observe the general behavior of sea otters in previously oiled areas. These August 1990 field studies were qualitative observations; we do not consider them comprehensive. However, when compared to observations made in 1989, these studies provide a framework for evaluating general changes in the sea otters' habitat.

Status of Captive Otters

The condition of captive otters is summarized in Table 1. Several factors were assessed including coat condition, health status, body weight, blood parameters, and reproductive status.

Coat Condition

A water repellent coat is essential for thermoregulation in sea otters. Therefore, it was important to thoroughly clean fur that had been contaminated with oil. The amount of oil on the fur (i.e., the degree of oiling) varied among the otters brought to the rehabilitation centers (figure 1). Rankings ranged from heavily oiled to unoiled, and reflected the percentage of surface area covered and

the level of saturation to the skin (Williams et al. 1990). Five of the eight adult otters used in the study arrived at the rehabilitation center moderately or heavily oiled; the two other adults were lightly oiled.

All of these animals were cleaned successfully using methods outlined in Williams et al., 1988 and Davis et al., 1988. Because of the loss of natural oils from the washed fur (Davis et al., 1988), a seven to ten day delay in the recovery of water repellency is normal. Grooming activity by otters facilitates restoration of the oils and the original condition of the fur. Until then, areas of the pelt will saturate when the otter enters the water.

In the rehabilitation centers the pelt restoration process was complicated in clinically ill sea otters. These animals were often lethargic, did not groom, and therefore had consistently poor coats. Following several weeks in the rehabilitation centers, the animals began grooming and coat condition gradually returned to normal. By August 1989, coats of the adult otters had fully regained water repellency and remain in good condition.

The fur of sea otter pups also requires constant care to remain in good condition. In the wild, this

activity is provided by the mother; in the rehabilitation center members of the husbandry staff groomed the pups. In contrast to the adult otters, the sea otter pups were captured 5-10 weeks after the oil spill and arrived unoiled at the center. Although washing was not necessary, towel drying and daily combing were required to remove mats from the pups' hair. After two months in captivity the pups were able to groom themselves, no longer requiring human assistance. The fur of the pups has remained in good condition since December 1989.

Health Status

During the initial phase of the spill, sea otters showed a wide range of health problems (i.e., gastrointestinal dysfunction, respiratory distress, poor thermoregulatory control, anemia, and fluctuations in blood cell counts). Some of these conditions, particularly gastrointestinal problems and stress, may have been complicated by captivity. In addition, oiled sea otters often demonstrated chronic bloody nasal discharge, lethargy, diarrhea, and poor motor coordination in the water (Wilson *et al.*, 1990). Subcutaneous emphysema was documented in three study animals that were heavily oiled and captured during the initial phase of the spill (table 1). Body temperature was variable in these animals and ranged from 95°F to 102°F during the first few days in the rehabilitation centers (Body temperatures of healthy otters range from approximately 99°F to 100°F; Costa and Kooyman, 1982).

The general health of the captive adult otters improved after cleaning. Thermoregulatory problems were quickly corrected as water repellency of the coat was regained. Recovery from the initial health problems occurred within 3-4 months of exposure to the oil. Gastrointestinal problems were often corrected after two weeks, but the anemia persisted for over 45 days (see below). Corneal damage in one otter and minor locomotor problems in two

other animals persist, but these conditions do not appear to impair their general health. Treatments (i.e., antibiotic therapy) were unnecessary after September 1989. The appetite and activity level of the captive otters remain high, and indicate a general recovery from the initial effects of oil contamination. Currently, the health status of captive otters is good.

Unlike the adults, the sea otter pups did not appear oiled when captured. Often the pups were found abandoned in the wild or were separated from female otters that were unable to care for them. Despite the absence of surface oiling, general health of the pups on arrival at the rehabilitation centers was varied. For example, body temperatures were highly labile, three of the pups were underweight and anemic, and diarrhea was common until the diet was adjusted.

Raising sea otter pups in captivity is difficult because of their nutritional and thermoregulatory demands. Yet, the appetites of the otter pups indicated that the animals quickly acclimated to the seaquariums. The health of the pups has been good to excellent from October 1989 to present. One exception is a male pup (SW106) which has experienced several health problems (hypoglycemia, ane-



Sea otter fur usually returned to normal within a couple of weeks after cleaning. The lighter brown areas indicate water repellency.



Good appetites and increased body weights indicated that the otters were in good health.

mia, acute nephritis, and a moderate nonspecific infection). This animal showed a gradual improvement during November and December 1989, but it remains in guarded condition. Reasons for his persistent problems include oil related and non-oil related factors (i.e. an individual response to captivity or susceptibility to disease, exposure to petroleum hydrocarbons before or after birth). Because of the absence of surface oiling, it is unlikely that external contamination was a significant factor.

Body Weight

The resting metabolic rate of sea otters is almost 2.5 times higher than other similarly sized mammals (Costa and Kooyman, 1982), and results in a voracious appetite. Inadequate food intake or poor nutrition will accelerate weight loss in these animals. Because lethargy and anorexia were common problems in oiled otters, many of the animals were underweight during the first weeks in the rehabilitation centers. As the otters' health improved, their appetites returned and body weights quickly

increased. From March 1989 to January 1990, the body weight of non-pregnant adults increased an average of $12.3 (\pm 2.3 \text{ lb.})$ (figure 2). Similarly, all of the pups gained weight, with a mean increase of $22.6 (\pm 3.0 \text{ lb.})$ during the first nine months in captivity.

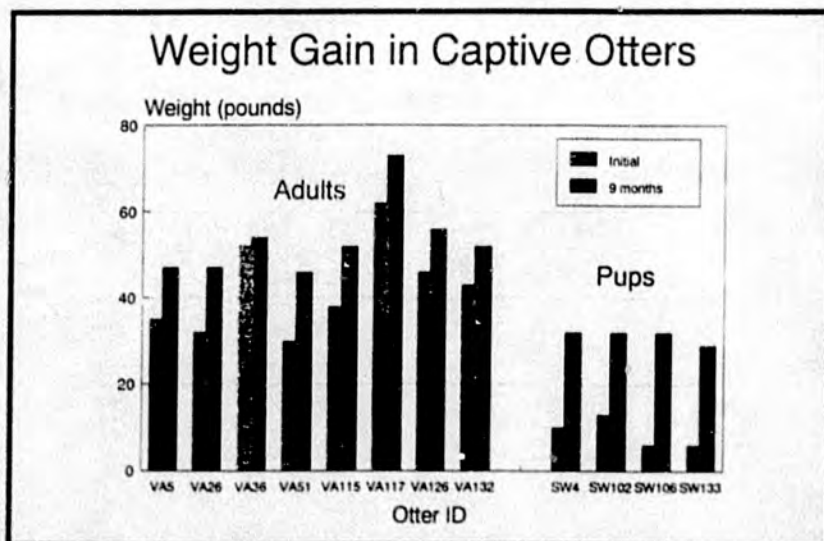


Figure 2. Changes in body weight of adult sea otters and sea otter pups on arrival at the rehabilitation centers and after nine months in captivity.

Blood Parameters

Blood panels were generated throughout the rehabilitation process and provided a good indication of the health status of the otters. When the oiled animals arrived at the rehabilitation centers, hypoglycemia and anemia were commonly diagnosed (Wilson *et al.*, 1990). White blood cell counts were quite variable in many of the otters (table 1). Much of this variability was attributed to the presence of infection or possible immuno-suppression.

Recovery from many of these conditions occurred within four months of captivity. White blood cell counts, creatinine levels, and enzyme levels (SGOT, SGPT, and CPK) returned to normal in all of the study animals except for two adults and one pup. Periodic elevations in white blood cell counts and enzymes (CPK, LDH, and to a lesser degree SGOT) continued to occur in the two adults. The male pup shows consistently abnormal blood variables (elevated WBC and LDH, anemia) that are attributed to infection.

Anemia persisted for approximately eight weeks in 11 of the sea otters (figure 3). Hematocrit of these animals was less than 40.0% during the first two months after the spill. This increased to 47.4% during the third month and was within the normal range for captive otters (50.0 - 62.0%; T.D. Williams, per. comm.) after four months. The hematocrit of the otters has remained normal since that time.

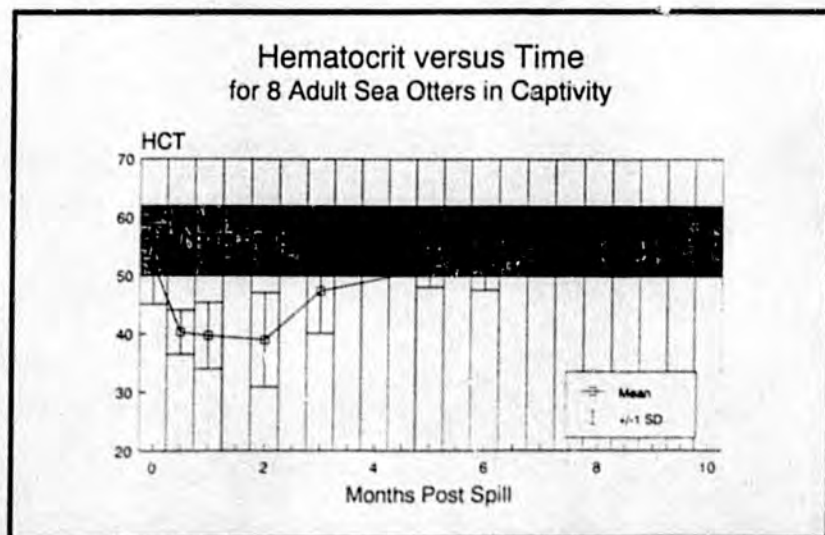


Figure 3. Hematocrit of adult sea otters in relation to time post spill. Mean values $\pm$ 1SD are indicated.

In addition to the routine analyses, we determined total paraffinic hydrocarbon (TPH) levels in the blood samples. TPH in blood provides an indication of systemic exposure to oil and can be useful in predicting survival (Williams, in prep.). We found that the concentration of TPH ranged from 19 ppm to 800 ppm in sea otters admitted to the rehabilitation centers (Williams *et al.*, 1990). Levels above 80 - 120 ppm were associated with mortality. In comparison, total paraffinic hydrocarbon concentration was insignificant in captive sea otters examined twelve months after exposure to crude oil (table 1). Such levels are well below the lethal threshold range determined for otters.

Reproductive Status

Mating among rehabilitated otters was often observed when male and female adult otters were housed together. The male in this study, VA117, became active in August 1989 after introduction into the female pool. He recently sired a healthy pup (see below). The females varied in responsiveness to VA117; although two animals were reproductively active in August 1989, the remaining females did not appear active for three to four months after that date. Currently, all of the captive adult otters are demonstrating reproductive behaviors.

In the wild, female sea otters are capable of producing a pup annually. However, age of the female, health, previous pregnancy among other factors will affect the reproductive interval (Rotterman and Simon-Jackson, 1988). The average time between copulation and birth ranges from 6-7 months for wild otters to 7-8 months for captive animals. Two of the four reproductively active females showed evidence of pregnancy during the study period. Serum progesterone for one animal sharply increased to approximately 6500 pg/ml in August 1989, and indicated possible pregnancy. However, serum levels gradually decreased over the following two months with no ensuing gain in weight. Another female, VA126, had

serum progesterone levels of 7630 pg/ml in April 1990 and gained 15 lb. in 2.5 months. On May 31, 1990 she gave birth to a healthy pup. The pup was nursing within three hours of birth and quickly gained weight. Although the pup had to be separated from its mother for several weeks because of poor fur condition, they have recently been reunited. Both animals are in excellent condition.

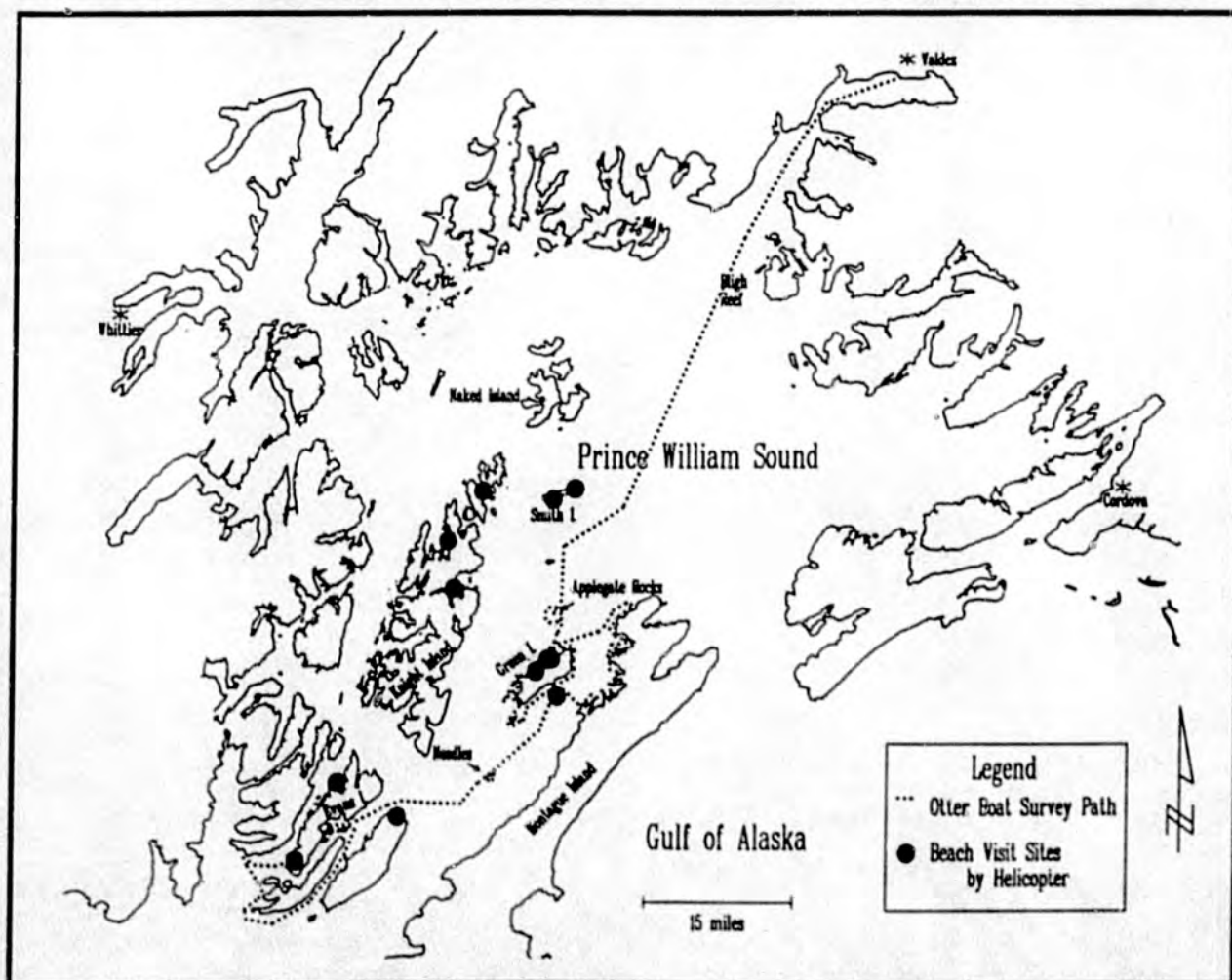
Field Observations of Wild Sea Otters

Approximately 100,000-200,000 sea otters are found throughout the coastal areas of Alaska (Rotterman and Simon-Jackson, 1988). Of these, almost 16,000 animals inhabited areas within the oil

spill zone (DeGange *et al.*, 1990). Currently, we do not know the exact number of otters impacted by the spill. Reliable counts show nearly 900 sea otter carcasses retrieved from the spill area (DeGange and Lensink, 1990) and 357 live animals brought into rehabilitation centers (Williams & Davis, 1990).

The degree of oiling and consequent mortality in wild otters declined early in the spill, paralleling the pattern observed in rehabilitation centers. By May 19, 1989 approximately 70% of the otter carcasses had been retrieved (DeGange and Lensink, 1990). An estimated 80% of these mortalities were attributed to exposure to oil. After the initial phase of the spill, the major concerns for wild otters were the possibility of fur contamination from residual oil on the water or beaches and toxicoses from

Field sites visited in Prince William Sound (below) and along the Kenai Peninsula (next page). Site selection was based on primary capture points for oiled sea otters during 1989 (Williams and Davis, 1990).



eating contaminated prey. A toxic response from direct contact with the oil was unlikely because most of the volatile petroleum hydrocarbons had dissipated within days of the spill (Neff *et al.*, 1990).

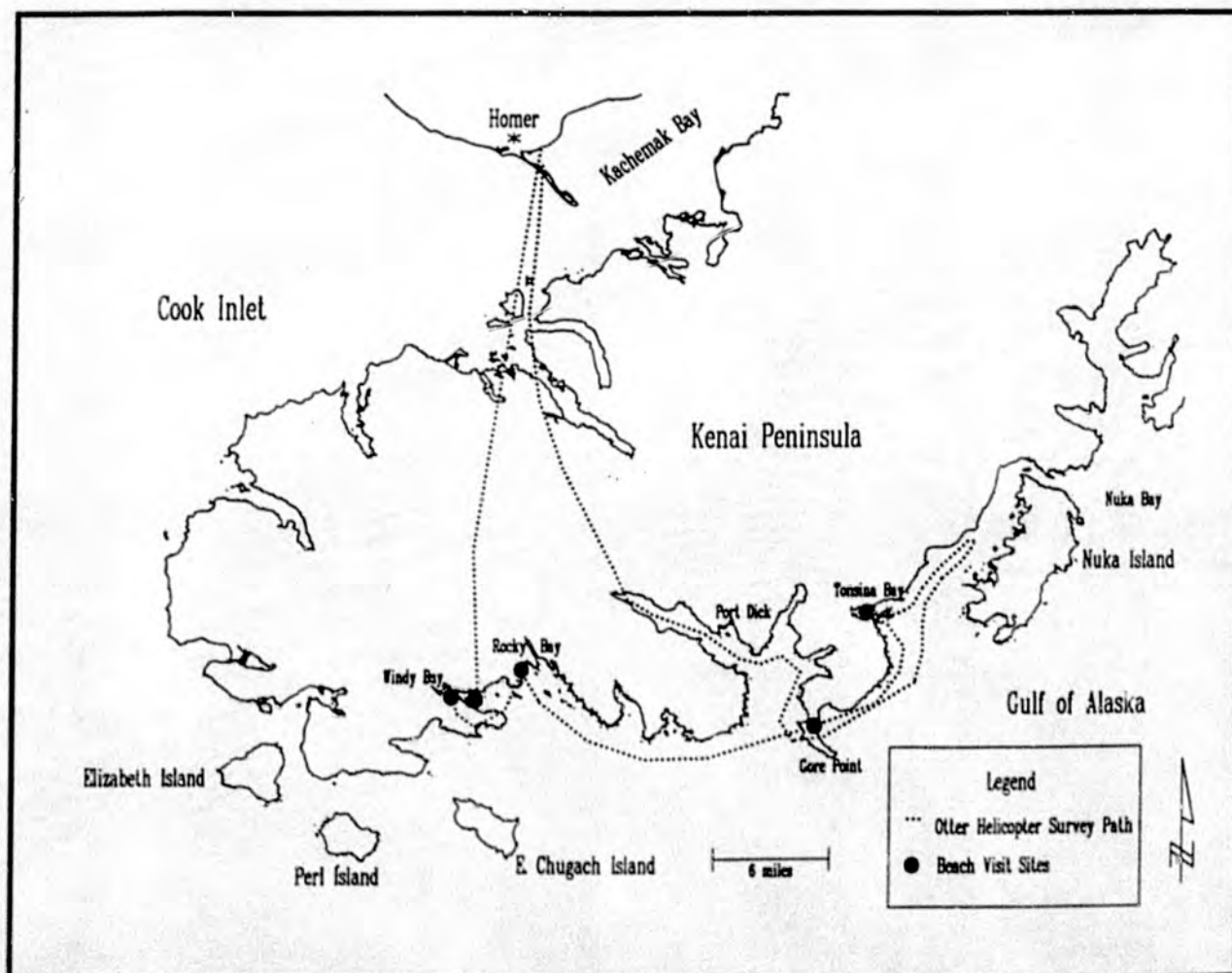
Field studies were conducted to determine the potential for continued contamination of wild sea otters. In early August, we examined areas of Prince William Sound and the Kenai Peninsula that were primary capture sites of oiled otters following the spill (see maps). These sites, which we visited by boat and helicopter, included Rocky Bay (40+ otters captured), Windy Bay (65 otters), Tonsina Bay (27 otters), Green Island and Applegate Rocks (40+ otters) and Evans Island (17 otters captured). The observation period for each was less than a day, therefore this is not considered an exhaustive otter count. In addition, we examined Sleepy Bay, a site of persistent heavy oiling and active cleanup activ-

ity during 1989 and 1990.

Each field site was evaluated in terms of its potential for contaminating sea otters. General beach condition was inspected visually for the presence and type of oil on the substrate. The presence of invertebrates and kelps was considered indicative of beach recovery. Evidence of sheen oil on the water surface was also determined.

Otters and Pups

We observed sea otters of all age classes (adults, juveniles, and pups) at many of the field sites in Prince William Sound and along the Kenai Peninsula. Other marine mammals, including Steller sea lions, harbor seals, killer whales and humpback whales were also seen in these areas. In this study, the largest concentration of otters (more than 150 adults and several pups) was found around Montague



Island. We saw 43 individuals in La Touche Passage, more than 80 otters, including juveniles and mother-pup pairs, near Green Island and Applegate Rocks, and small groups of otters along the Kenai Peninsula (including eight adults and two pups in Windy Bay and a raft of 40 animals by Gore Point). Few otters were seen in areas around the northern islands of Prince William Sound (i.e., Herring Bay, Smith Bay, Prince of Wales Passage, Naked Island). We did not see any otters in Tonsina Bay, although other investigators have observed them at this sight.

The sea otters sighted within the spill area appeared healthy. All animals were active and displayed typical behaviors. When approached by boat, the animals slowly swam away. If observed from the shore, the otters groomed, dove, and rested on the water's surface. In general, the coats of the animals appeared water repellent; dark areas indicative of saturation were not observed.

Suitability of the Habitat for Sea Otters

Sea otters forage along shore areas and feed on a variety of bottom dwelling invertebrates, but rarely on fish in Prince William Sound (Estes, 1980). The otter's diet reflects local prey abundance; in Alaska, prey items include sea urchins, mollusks, crustaceans, and some epibenthic fishes. Because many of these prey items are located in shallow areas and are known to accumulate petroleum hydrocarbons (Neff, 1979), exposure to crude oil or toxic hydrocarbon metabolites through ingestion of tainted food was a concern for wild sea otters. Current evidence, however, indicates that contamination of invertebrates was limited to isolated areas of Prince William Sound, the Kenai Peninsula, and Kodiak (Varanasi *et al.*, 1989). In a previous study edible tissue samples of mollusks from a tour of 13 sites had levels of aromatic hydrocarbons in excess of reference samples taken from Angoon, an Alaskan village



Many of the prey items for sea otters are evident in the Prince William Sound area.

outside the spill area. Three of these four sites (Kodiak, Chenega Bay, and Old Harbor) appeared to have been contaminated by spilled or discharged marine fuels from boat traffic in the village areas. Windy Bay mollusks, which appeared to have been contaminated by the Exxon Valdez spill, showed light aromatic hydrocarbons up to 16,000 ppb and heavy aromatics up to 2,500 ppb (Varanasi, 1989). However, even these levels of contaminants, the highest measured in the Subsistence Food Safety Study, are not considered a serious health risk for human consumption (USFDA, 1990).

It is difficult to predict whether these isolated areas of contaminated prey pose a significant threat to wild sea otters. We do not know if otters can detect petroleum hydrocarbons in food or if they will avoid contaminated prey. Nor do we know the amount of contaminated prey that would have to be ingested before a toxic response occurs. In view of these uncertainties and isolated patches of oil, it is unlikely that a large segment of the Alaskan sea otter population would locate and digest harmful quantities of contaminated prey.

Although most of the harmful components of the oil dissipated within days of the spill (Neff *et al.*, 1990), residual oil on beaches or on the water

surface could be detrimental to the fur of sea otters. Previous studies have shown that weathered oil does not have as great an impact on insulation as fresh crude (Williams *et al.*, 1988) because it is more viscous, remains on the tips of the hair, and is less disruptive to the insulating air layer. Likewise, the absence of hypothermia in lightly oiled sea otters at rehabilitation centers indicates that sheen oil is less disruptive to insulation than fresh oil (pers. obs.).

For most of the sites visited, surface oiling was limited to isolated areas of beach. Usually, traces of oil had hardened in thin layers on rock, sand, or pebbles. We occasionally found spots of tar between rocks, but we saw little evidence of oil sheens

on the water. This situation contrasts with conditions on capture beaches in 1989, when fresh oil or thick layers of mousse covered the beaches to upper tidal lines and sheens of oil were often observed on the water surfaces near heavily oiled shores.

In 1990 we surveyed two beaches, Herring Bay and Sleepy Bay, that were exceptions. At these worst case sites, we saw areas of standing oil and some sheening in the water. Although sea otters were not seen near these beaches, the potential for contamination of their fur does exist. However, neither USFWS nor the cleanup crews at these beaches reported sightings of oiled sea otters.



We observed many other mammals, including these steller sea lions, during the field investigations.

Prognosis for Captive and Wild Sea Otters

Behavioral, medical, hematological, reproductive, and toxicological parameters indicate that it is possible for otters to recover from many of the initial effects of oil exposure. Except for one pup, the captive otters recovered from many of the effects of exposure within 3 - 4 months. Coat condition, a critical factor for sea otters, was restored to normal water repellency within one month of thorough cleaning. Petroleum hydrocarbon levels that were comparatively high initially, declined to insignificant levels after twelve months of captivity. Consequently, the current prognosis for the captive animals is good.

Because the otters in this study received veterinary care and ad libitum food, it is difficult to extrapolate to animals in the wild population. Preliminary results from radio-instrumented animals indicate that rehabilitated otters are initially vigorous following release into the wild. Mortality increased during winter months indicating a susceptibility to environmental factors (Monnett *et al.*, 1990). Before conclusions can be drawn, however, further research is needed to determine the relationship between the degree of oiling of otter fur and survivorship in free-ranging sea otters.

Conditions in Prince William Sound and the Kenai Peninsula have improved, in terms of the suitability of the habitat for sea otters. Water column and subsistence fisheries reports, as well as the present study, indicate few traces of oil. In many cases the oil is: 1) inert; 2) limited to isolated shores and prey items; and 3) in a form which poses little threat to sea otters. Large numbers of adults and pups were found in previously oiled areas, and they appear to feed and behave normally. These results suggest that many of the previously contaminated areas are able to support sea otters. With time and further research, we will be able to determine the resiliency of both the sea otter population and its habitat in Alaska.

Acknowledgments

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Table 1

Condition of Captive Sea Otters

Otter #	Coat Condition		Health Status		Weight			Blood Panel		Paraffinic Hydrocarb. in Blood
	Initial	Final	Initial	Final	Initial	Final	% Gain	Initial	Final	TPH (ppm)
Adults										
VA-5 (3-31)	Heavy Oil	Good	Lethargic Emphysema	Normal	35	47	34	Anemia elev. CPK	Normal	ND *
VA-26 (4-1)	Heavy Oil	Good	Emphysema	Normal	32	47	47	elev. WBC	Normal	ND
VA-36 (4-2)	Heavy Oil	Good	Lethargic Diarrhea	Normal	52	54	4	Normal	Normal	ND
VA-51 (4-4)	Not Documented	Good	Emphysema	Normal	30	46	53	elev. WBC, LDH & CPK	Normal	-
VA-115 (4-10)	Light Oil	Good	Fair	Normal	38	52	37	elev. SGPT, LDH & CPK	occasional elev. WBC	-
VA-117 (4-11)	Light Oil	Good	Fair	Normal	62	73	18	elev. WBC	Normal	ND
VA-126 (4-16)	Mod. Oil	Good	Emaciated	Normal	46	56	22		Normal	ND
VA-132 (4-17)	Heavy Oil	Good	Lethargic Diarrhea	Normal	43	52	21	elev. WBC	occasional elev. WBC	-
Pups										
SW-4 (5-1)	No Oil	Good	Thermal Instability	Normal	10	32	220	-	Normal	ND
SW-102 (5-10)	No Oil	Good	Underweight	Normal	13	32	146	Anemia	Normal	ND
SW-106 (5-21)	No Oil	Good	Underweight	Fair	6	32	433	Anemia	Abnormal	ND
SW-133 (6-13)	No Oil	Good	Hypothermic	Normal	6	29	383	Anemia	Normal	ND

* ND = Non Detectable Amounts

CPK = Creatine Phosphokinase

LDH = Lactate Dehydrogenase

WBC = Total White Blood Cell Count

SGPT = Alanine Aminotransferase

TPH = total paraffinic hydrocarbons

Note: TPH measurements were made at 12 months.



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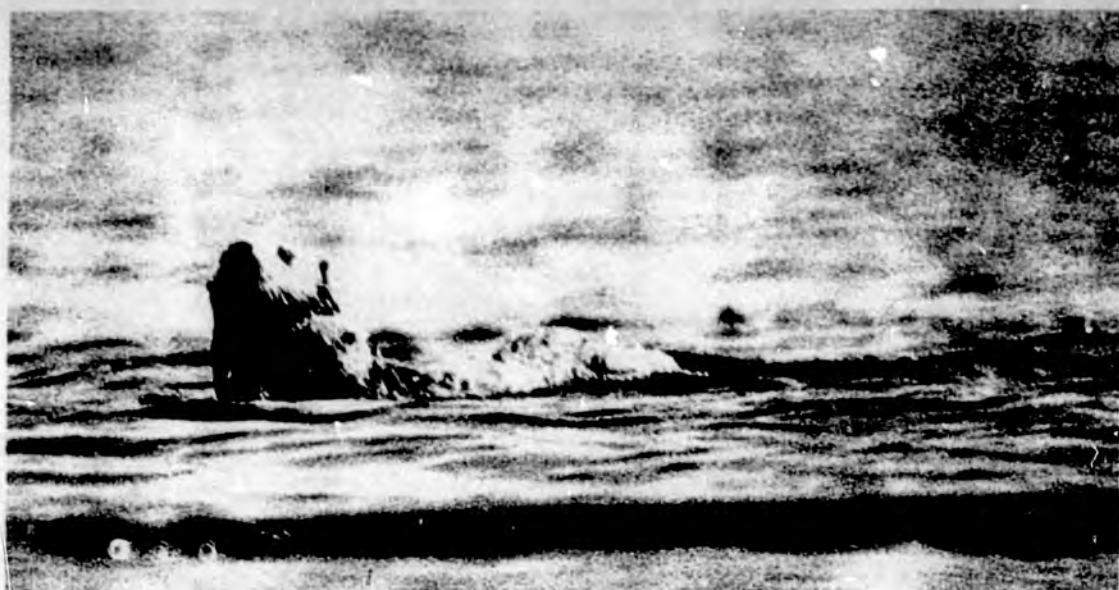
Sea Otters Thrive in Prince William Sound, Alaska



Sea otters are plentiful in the Prince William Sound area. Experts like Terrie Williams and Randall Davis have been working with the area's sea otters since the 1989 oil spill.

February 1991

Sea otter pups and juveniles make up a good portion of the area's population.



Thousands of sea otters inhabit the Prince William Sound area. Hundreds of pups were spotted in 1990.

Thousands of Otters and Pups

More than 16,000 sea otters inhabit the portions of Prince William Sound and the Gulf of Alaska that were impacted by the 1989 oil spill. Regrettably, more than 1,000 otters died at the time of the spill, but

recent observations confirm that the otter population is recovering. Hundreds of pups were spotted in 1990. A female adult otter typically produces a pup every year or so.



The area's sea otters rely on shellfish for food. They consume 25 to 30% of their body weight each day. Government studies conducted after the spill confirm that the area contains ample shellfish supplies safe for humans to eat.

A Diet of Shellfish

The area's sea otters rely on shellfish for food year round. The Sound's supply has helped sea otters recover from near extinction a few decades ago. They are no longer an endangered species in Alaska.

Immediately after the oil spill, there was concern over the area's shellfish. The State of Alaska and the Federal Government conducted extensive studies of all the area's subsistence foods, including shellfish. Their conclusion: ample stocks of shellfish were available and safe for humans to eat.



The majority of Prince William Sound was never oiled in 1989 and oil slicks are no longer on the water's surface.

Clean Water

Sea otters spend virtually all of their lives in and on top of the water. Any significant volume of oil on the surface of the water seriously threatens the otter habitat.

In 1989, oil from the Valdez spill remained on the water's surface only a short time, and only in a portion of Prince William Sound. Extensive tests have confirmed that the spill did not compromise the water column's quality.

Clean Beaches

The 1989 tanker spill impacted less than 20% of the area's shoreline. By the fall of 1990, about 1% of the shoreline still had patches of oily residue on it. The crude oil has "weathered" and lost its toxicity.

Following harsh winter storms and two summers of clean-up operations, Prince William Sound is essentially cleansed of oil from the Valdez spill.

Resilient Sea Otters

A sea otter, more than any other marine mammal, is vulnerable to oil spills because of its fur and lifestyle. Drs. Davis and Williams believe this animal's resiliency should not be taken for granted and everything reasonable to prevent another spill from happening should be done.

Fortunately, the evidence suggests that sea otters in the Prince William Sound area are thriving. Nearly two years have passed since the oil spill, and sea otters are plentiful throughout the region.

The evidence suggests that sea otters in the Prince William Sound area are thriving.



Drs. Terrie M. Williams and Randall W. Davis managed Exxon's Sea Otter Rescue and Rehabilitation Program following the 1989 Valdez oil spill. Because of those efforts, nearly 200 wild sea otters were treated and returned to Prince William Sound. Dr. Williams is currently a Research Physiologist with the Naval Ocean Systems Center in Hawaii. Dr. Davis is head of the Department of Marine Biology at Texas A&M University. In addition, they co-direct International Wildlife Research.

If you are interested in more details about the Prince William Sound sea otters, please write to: Exxon Company, U.S.A., Otters, P.O. Box 2180, Houston, Texas, 77252-2180 for a copy of two technical reports: "Evaluating the Long-term Effects of Crude Oil Exposure in Sea Otters: Laboratory and Field Observations" by Terrie M. Williams, Ph.D.; and "Advances in Rehabilitating Oiled Sea Otters: The Valdez Experience" by Randall W. Davis, Ph.D.



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The *Exxon Valdez* oil spill: Initial environmental impact assessment

Part 2 of a five-part series



FIGURE 1

(a) Smith Island, Prince William Sound, April 1989

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The March 24, 1989, grounding of the *Exxon Valdez* on Bligh Reef in Prince William Sound, Alaska, was unprecedented in scale. So too was Exxon's response to the oil spill and the subsequent shoreline cleaning program, including the employment of more than 11,000 people, utilization of essentially the entire world supply of containment booms and skimmers, and an expenditure of more than two billion dollars. In the days immediately following the *Valdez* spill, Exxon mobilized a massive environmental assessment program. A large field and laboratory staff of experienced environmental professionals and in-

ternationally recognized experts was assembled that included intertidal ecologists, fishery biologists, marine and hydrocarbon chemists. This field program to measure spill impacts and recovery rates was initiated with the cooperation of state and federal agencies. Although the agencies subsequently foreclosed cooperation in most of these studies because of litigation concerns, this comprehensive assessment program continues today. Through the end of 1989, this program has resulted in well over 45,000 separate samples of water, sediment, and biota used to assess spill impacts.

It is the intent of this paper to provide initial observations and preliminary conclusions from several of the 1989 studies. These conclusions are based on factual, scientific data from studies designed to objectively measure the extent of the impacts

from the spill. Data from these studies indicate that wildlife and habitats are recovering from the impacts of the spill and that commercial catches of herring and salmon in Prince William Sound are at record high levels. Ecosystem recovery from spill impacts is due to the combined efforts of the cleanup program as well as natural physical, chemical, and biological processes. From all indications this recovery process can be expected to continue.

Design of the assessment program

In early 1990 *Environmental Science & Technology* completed a four-part series on the state of the science of ecotoxicology and ecological risk assessment (1-4). This series reviewed the development and current regulatory applications of the ecological risk assess-



FIGURE 1
(b) Smith Island, June 1990

ment paradigm based on sequential comparisons of environmental fate (chemistry) data and biological effects data. Much of this literature is the basis for the Technical Information Documents (5, 6) being used by the regulatory agencies and supports the "Type B" natural resource damage assessment rules promulgated under Section 301 (C) of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).

The environmental fate-and-effects approach also forms the backbone of the *Exxon Valdez* environmental impact assessment program as discussed in these pages. Each of the studies was carefully designed to accurately measure hydrocarbon concentrations in the environment and within specific biological components as well as measure observed biological effects at individual trophic levels. For each program, environmental fate data complemented effects data to ensure that a fate-and-effects correlation could be developed to assess the likelihood of impacts.

Shoreline habitat

As a result of the grounding of the *Exxon Valdez*, some 258,000 barrels of crude oil were released into the water. Winds of more than 70 mph on the third day after the grounding rendered con-

tainment of oil on the water impossible, with the result that shoreline habitat in the southwestern segment of Prince William Sound was impacted. During the next few weeks more than 1100 miles of shoreline in south-central Alaska were impacted by oil to varying degrees (Table 1). These shoreline areas represented the habitat most obviously impacted by the spill where floating oil and mousse (oil-water emulsion) were washed ashore and deposited in intertidal areas. Indeed, it was these areas that were the focus of the two-billion-dollar cleanup program during 1989 and 1990.

It is important to point out that, at the very worst, slightly more than 10% of the shoreline habitat in Prince William Sound and the Gulf of Alaska was impacted by oil, thus leaving 85-90% of the regional shoreline untouched by oil (Table 1). Predictably, the percentage of impacted areas dropped significantly to approximately 1% of the shoreline areas in 1990 as a result of both the cleaning program and natural weathering. This effect can be seen most dramatically in the accompanying photographs of heavily impacted shoreline in 1989 and the comparatively clean state in the spring of 1990 (Figure 1).

Shoreline oiling data are the results of the 1990 spring shoreline assessment wherein 20 separate teams, each consist-

ing of state of Alaska agency, and Exxon representatives, assessed more than 120 miles of shoreline in Prince William Sound and the Gulf of Alaska. A total of 1,000 pits were assessed had no oil retained in < 10 cm. In addition, 5071 pits were dug to determine the extent of subsurface oiling. Subsurface oil (oil deeper than 10 cm) was found in 733 (14%) of the pits dug. The locations recommended for 1990 summer cleaning were widely scattered and generally confined to short lengths of shoreline; all identified areas received attention during the summer.

Biota. The shorelines of southern Alaska are predominantly large rocks, boulders, and cobbles with less than 10% of the area characterized as soft substrate. This rocky intertidal zone is an extremely harsh habitat. Plants and animals living there have evolved to cope with a highly unstable substrate, extremely forceful breaking waves, and wide seasonal and diurnal ranges in photoperiod and temperature. As a result, many of these areas have naturally low species density and diversity.

Barnacles, mussels, and rockweed (*Fucus*) are the best indicator species for visually estimating the biological condition of these intertidal habitats. They are good indicators of trends in recruitment

and loss caused by oil or other natural phenomena. In order to assess spill impacts on shorelines, these species and the intertidal communities in Prince William Sound and the Gulf of Alaska have been sampled via counts of species density, percent cover, species diversity, and photographic surveys of intertidal biota.

Results from the shoreline surveillance program indicate that although species densities were lower on heavily oiled shorelines, and high local mortalities occurred where oil coverage exceeded 90%, the species compositions of communities are similar to those that would be expected on comparable shorelines in the absence of oil. Even where oiling remains in sheltered, soft substrate areas, the spring and summer growing season of 1990 confirmed that continued recovery was under way (Figure 2). Thus, intertidal invertebrates and plants do survive on oiled shorelines, a point that supported spill researchers' conclusions that further extensive, intrusive shoreline cleaning efforts were unwarranted (7).

This recruitment trend also has been observed in several studies of other spills worldwide that showed that shoreline biota recovered in impacted areas within a few years after the spill (8).

Shoreline field-monitoring data support the following conclusions:

- The amount of oil remaining on the shoreline has continued to decrease since the summer of 1989. There are few oiled shores remaining.
- The impact to intertidal communities was a decrease in the number of individuals of each species and a change in the species composition of the communities. The intertidal communities are intact.
- Intertidal plants and animals were found settling and surviving on shorelines throughout Prince William Sound and the Gulf of Alaska throughout the 1989 and 1990 growing seasons.
- Even where residual oil remains on a few shorelines, biological recovery is taking place.

Water quality

Within six days of the spill, Exxon initiated an extensive water quality sampling program in Prince William Sound to help assess possible impacts on the marine biological community by measuring hydrocarbon concentrations in the waters of the Sound. Water samples were collected from March through October 1989 at 35 offshore locations.

More than 2300 water quality samples have been analyzed to date. Figure 3 is a plot of the average polycyclic aromatic hydrocarbon (PAH) concentrations through-

TABLE 1
Alaskan coast impacted by Exxon Valdez oil

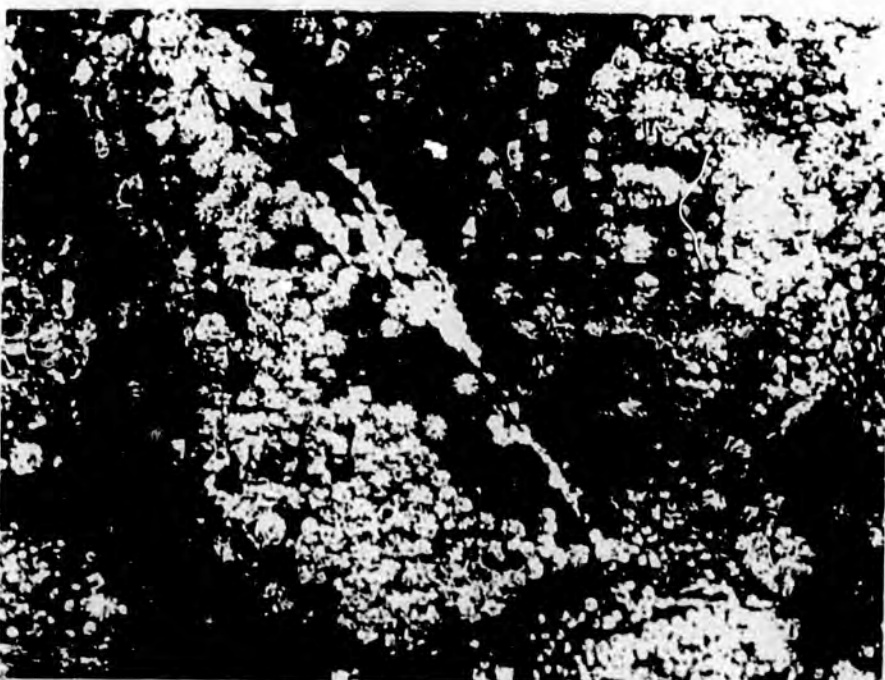
Area	Total coast (miles)	Oil-impacted coast, May 1989 (miles)	Total coast impacted, May 1989 (%)	July 1990 oil-impacted coast (miles) ^a	Total coast with oil, July 1990 (%)
Prince William Sound	3061	357	11.6	91.5	3.0
Gulf of Alaska	6345	732	11.5	24.1	0.4
Total (miles)	9406	1089		115.6	

^aJoint Exxon/Agency Spring Shoreline Assessment Teams (SSAT) report. Wide (>6 m), moderate (3-6 m), and narrow (<3 m) oiling widths combined.

FIGURE 2
Biological growth in Prince William Sound, spring and summer 1990



Grass sprouting from sediment containing residual oil



Barnacle growth on oil-stained rock

out the year. The three curves represent the averages of different sites. The solid line represents the three sites with the highest concentrations. These are three bays that were heavily oiled. Note that the average of these three worst bays peaked at only 7% of the Alaska state water quality standard. The average of all primary sites has a maximum value of about 1/50th of the state allowable limit for aromatics in water (9).

These data all confirm that average hydrocarbon concentrations measured in the water column for the more toxic components have consistently been well below state of Alaska standards and are 10–1000 times lower than levels lethal to plants and animals living in the water column, including commercially important fish species.

Fisheries

Pacific herring. Annually, millions of herring enter Prince William Sound to spawn in early April. This spawning period determines the commercial herring fishing season which had a 1988 value of \$12 million. The *Valdez* spill occurred approximately three weeks prior to the peak of the Pacific herring spawn. Concern regarding potential impacts to the herring resulted from the fact that herring deposit their eggs on kelp in the intertidal and shallow subtidal zones in Prince William Sound.

Results of aerial surveys flown by Alaska Department of Fish and Game field biologists confirm that both 1989 and 1990 spawning activity was comparable to historical averages for the years immediately preceding the spill (Figure 4). These data provide convincing evidence that herring spawning activity has been neither impaired nor delayed during the period when highest potential existed for exposure to spilled oil.

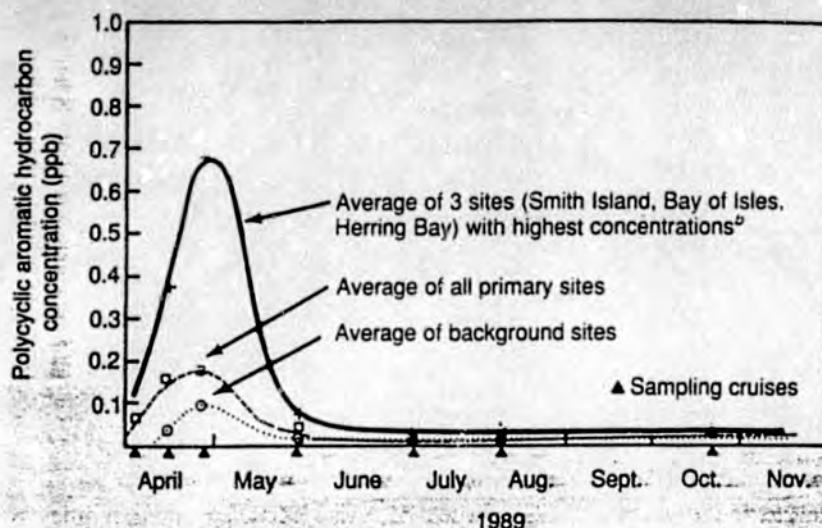
These conclusions were further substantiated by the results of the 1990 commercial herring fishing season wherein 8300 tons of herring were harvested during the 20-minute season opening on April 12th—an all-time record catch for the sac roe fishery. Record harvests were also achieved in the gill net and wild roe-on-kelp fishery seasons, thus minimizing concerns over long-term impacts on this important commercial fish species.

Pink salmon. The Prince William Sound salmon harvest is dominated by pink salmon and was worth approximately \$63 million in 1988. Data collected during the critical life stages of the pink salmon population show no significant oil-related effects.

An extremely healthy and vigorous zooplankton bloom supported the spring 1989 release of more than 600 million fry from the commercial fish hatcheries of Prince William Sound. Outmigrating

FIGURE 3

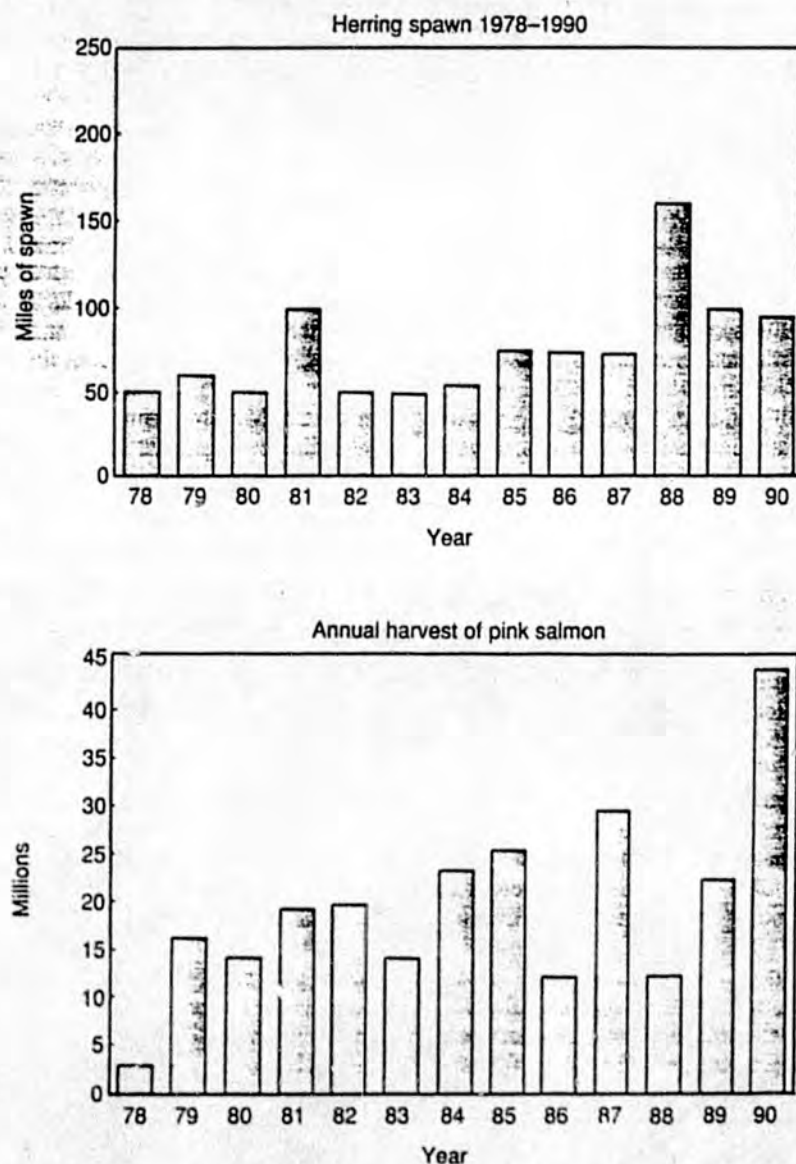
Average polycyclic aromatic hydrocarbon concentrations for Prince William Sound water^a



^aState of Alaska criterion for petroleum aromatics in water is 10 ppb.
^bPeak is 7% of state allowable limit.

FIGURE 4

Prince William Sound fish production



fry and juveniles in 1989 were apparently not impacted, and many juvenile salmon were observed in both oiled and unoiled nearshore areas. Similarly, field experiments with caged juveniles noted no differences in survival between oiled and unoiled sites. When the 1989 adults returned to spawn in the streams flowing into Prince William Sound, several streams had abnormally large numbers of returning fish. Also, no differences were evident in the spawning utilization of streams in oiled areas versus unoiled areas. Subsequent experiments in the fall with the incubation of eggs buried in gravel at several oiled and unoiled sites demonstrated similar survival and hatching rates between these sites, thus minimizing concerns over oil effects on this sensitive life stage.

Pink salmon have a short two-year life cycle. In the summer of 1990, those young salmon that left their native streams in April 1989, just after the spill, returned to spawn. The reported harvest through September 12, 1990, was 35.1 million fish taken, with an additional 8.9 million taken from hatchery stocks. This makes the 1990 commercial catch an all-time record, ahead of the 1987 record level by about 14 million fish and 163% ahead of the pre-season forecast for cumulative harvest. This strong run consists of both hatchery and wild-stock production (Figure 4).

In summary, there have been no indications of any significant pink salmon kills or effects on spawning activity related to oil exposure.

Shellfish and subsistence fisheries. Shellfish and crustaceans constitute a much less significant but still important part of Alaskan commercial, subsistence, and recreational fisheries. Several species of clams, mussels, crabs, and shrimp occupy the spill-impacted areas. Following the spill, a cooperative program was initiated between the Alaska Department of Fish and Game, the National Oceanic and Atmospheric Administration (NOAA), Exxon, and the subsistence communities to examine subsistence food safety in the spill-impacted areas. As of July 1990, more than 1300 samples of fish and shellfish have been collected in seven sampling cycles representing 23 species of fish and shellfish from 13 subsistence resource areas. They are being tested for aromatic hydrocarbons, the toxic components of crude oil most likely to be assimilated by shellfish.

The results of the chemical analyses received from the NOAA Seattle laboratory are very encouraging (Figure 5). Contaminant levels in all the fish and most shellfish were extremely low compared to data for fish and shellfish from a control site at the village of Angoon, outside the spill-impact-

ed area. Except for shellfish taken from Windy Bay and from Kodiak Harbor, all food resources sampled showed safe hydrocarbon levels.

A Toxicological Expert Committee consisting of representatives of the Food and Drug Administration, the National Institute of Environmental Health Sciences, NOAA, the National Marine Fisheries Service, the Alaska Department of Public Health, and various university and industry toxicologists reviewed the data and filed a consensus report. The committee concluded that finfish from anywhere in the study area are safe to eat in unlimited quantities, and except for those taken from obviously oiled shores, shellfish are also safe to eat (10).

Wildlife: Initial impacts

Some 36,000 bird mortalities were documented between March and September 1989. Although some of these were natural mortalities, no doubt exists that a significant number were caused by the spill. More than 1000 sea otter and 153 bald eagle mortalities were initially attributed to the spill. Although there was an extensive effort to recover dead birds and animals, actual mortalities were likely higher. When the above mortality figures are compared with total populations in the spill-impacted area—more than 10,000,000 sea birds; more than 30,000 sea otters; and 5000 eagles—indications are that recolonization has been rapid and robust, as it has been following other spills throughout the world (8).

Furthermore, extraordinary measures were taken to rescue and care for birds and otters that were impacted. From the period of March through September 1989, Exxon organized the largest and most comprehensive bird and sea otter rescue and rehabilitation programs ever attempted. Facilities for the holding, cleaning, and care of oiled birds and otters were built at Valdez, Seward, Homer, Kodiak, and Anchorage. During the program, more than 140 boats and 5 aircraft were used to retrieve oiled birds and otters from remote locations throughout Prince William Sound and the Gulf of Alaska.

Some 71 species of birds were handled during the six-month program. More than 1600 birds were brought in live for treatment with a release rate for the sea birds of 50%. Given the time of year of the spill, the climate, the remoteness of the site, and the logistics, this survival and release rate compares favorably with the 30–60% range of release rates seen in other spills. At its maximum, the bird program employed some 400 bird rescue personnel; this effort cost more than \$25 million.

Teams were hired to work in Prince

William Sound and Kodiak to collect eagles from heavily impacted areas to determine the need for potential rehabilitation. These capture teams caught 114 eagles; only 16 required treatment. (An additional 23 eagles caught by others not associated with the capture teams were also treated).

The staff of the otter centers grew to more than 320 specialists and volunteers at all three locations. From March 30 until September 15, 357 sea otters were treated and held at the centers; 223 (62%) were rehabilitated and released or placed in aquariums. Expenditures for this effort exceeded \$18 million.

Wildlife monitoring program

As noted, in spite of the best efforts by all parties, many birds died. To assess overall population-level impacts, comprehensive wildlife monitoring programs were initiated after the spill and are ongoing.

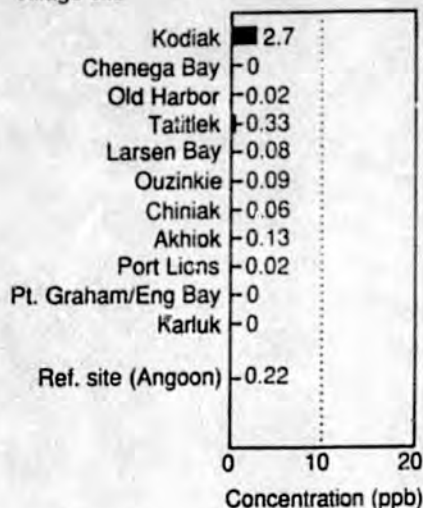
Bird monitoring. Results of Exxon and agency winter, spring, and summer wildlife monitoring programs are very encouraging. There have been no documented reports of oil or sheens affecting populations of birds or marine mammals, and there have been no confirmed mortalities attributable to oil since September 1989. Additionally, wildlife observations in Prince William Sound indicate that significant numbers of birds from 45 different species occupy previously heavily oiled areas and appear to be unaffected by existing conditions. Species diversity and density are similar for both oiled and unoiled areas.

The findings of the winter wildlife population surveys are most encouraging. These surveys of oiled areas in Prince William Sound show that bald eagle sightings have consistently increased since the summer of 1989. Data from cooperative U.S. Fish and Wildlife Service eagle surveys throughout the summer of 1990 indicate that 1031 of 2030 nests (51%) in the spill area were active and that 61 of 75 previously active nests (81%) in Prince William Sound cleanup sites produced an average of 1.4 eaglets per nest, thus matching historical productivity statistics for this area. It is recognized these data do not allow definitive statements of eagle population dynamics, but they do provide conclusive evidence that animal life is present in habitats previously impacted by the oil spill.

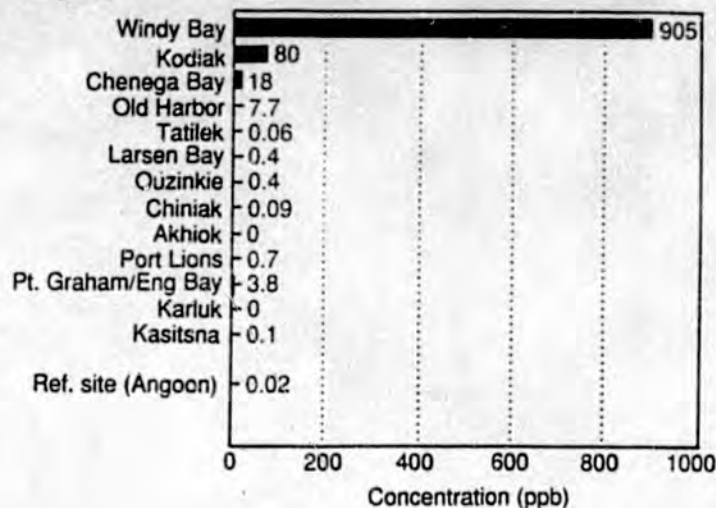
Mammal monitoring. Results of the winter mammal monitoring program indicate the presence and successful reproduction of mink and river otters in both oiled and unoiled bays of Prince William Sound. Sea otters are present in all areas in apparently equal numbers in oiled and unoiled bays.

FIGURE 5
Concentrations of aromatic hydrocarbons^a

In salmon
Village site



In mussels and clams
Village site



^aPrince William Sound, lower Cook Inlet, Kodiak.

Although census results of sea otters often vary widely, the Prince William Sound population has been generally estimated at 5000 to 6000 (11), with some recent estimates being as high as 9000. The Prince William Sound population was believed to be less than 50 individuals when commercial harvesting of this species ended in 1911. The population has increased very rapidly in recent years, and field surveys of 550 miles of oiled shorelines counted more than 350 sea otters. This survey covered about one-sixth of the total Prince William Sound shoreline. It indicates that otter populations are approaching pre-spill densities (11) and provides convincing evidence of sea otter recolonization of previously heavily oiled areas.

In summary, the monitoring data for mammals indicate that environmental conditions have improved enough that the spill should have no further substantive impacts on wildlife. Recovery is well under way.

Conclusions

A recently published Congressional Research Service report (12) carefully reviews the extent of environmental impacts from several previous oil spills worldwide and concludes that "To date, pollution from offshore petroleum activities has not appeared to be a significant threat to the survival of various species ... Despite short-term media attention to the catastrophic nature of major spill events, the chemicals contained in petroleum have long been part of the marine environment and physical impacts are likely to be temporary in the dynamic natural flux of the coastal environment."

Available data to date for the Valdez spill are consistent with the observations made in this report. Samplings of petroleum aromatic hydrocarbon concentrations in the waters of Prince William Sound clearly demonstrate that average levels have remained well below exposure levels known to cause acute and chronic effects to sensitive aquatic life. Field counts of plants, fish, and mammals from throughout the spill area provide convincing data that wildlife species are surviving and reproducing, thus confirming that biological recovery is rapidly taking place.

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NATURAL RECOVERY OF COLD WATER MARINE ENVIRONMENTS AFTER AN OIL SPILL

by

J. M. Baker, R. B. Clark, P. F. Kingston, and R. H. Jenkins

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NATURAL RECOVERY OF COLD WATER MARINE ENVIRONMENTS AFTER AN OIL SPILL

by

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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).

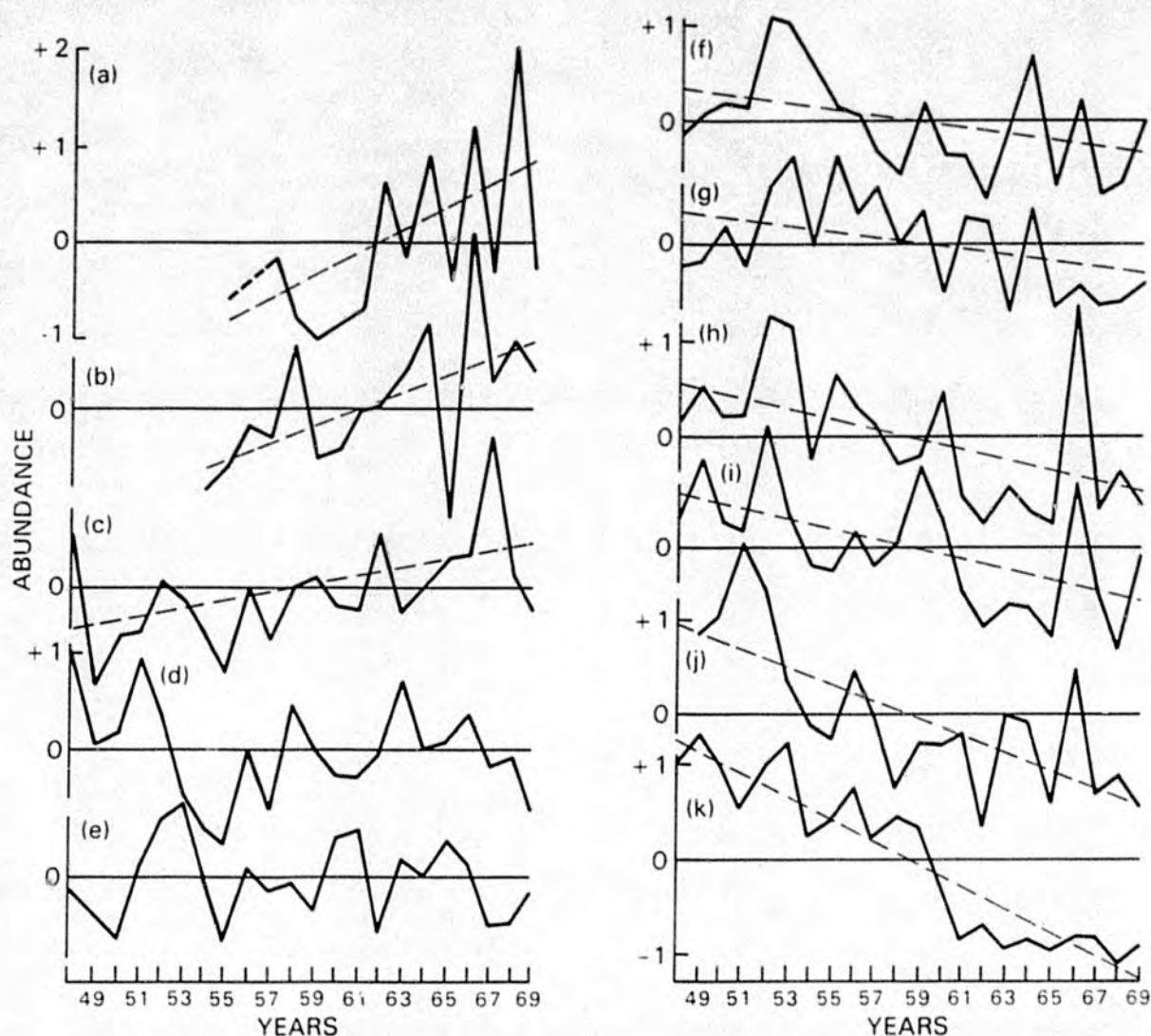


Figure 1: Changes in the abundance of zooplankton species in the North Atlantic. The organisms are: (a) *Pleuromamma borealis*; (b) *Euchaeta norvegica*; (c) *Acartia clausi*; (d) *Temora longicornis*; (e) *Clione lamacina*; (f) *Calanus helgolandicus* and *C. finmarchicus* stages V and VI; (g) *Metridia 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.

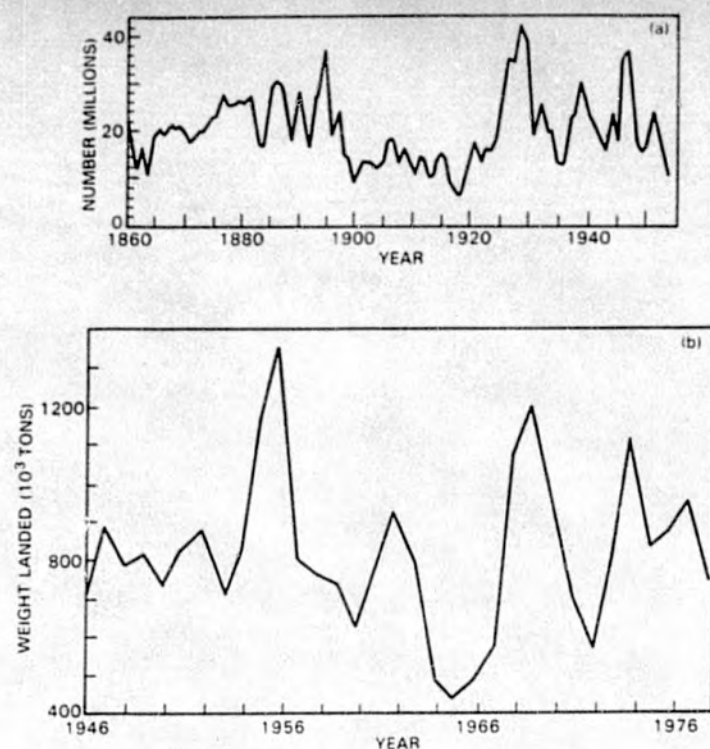


Figure 2: Landing of cod (a) in the Lofoten, Norway, fishery and (b) in the whole northeast Atlantic fishery (after Jones, 1982).

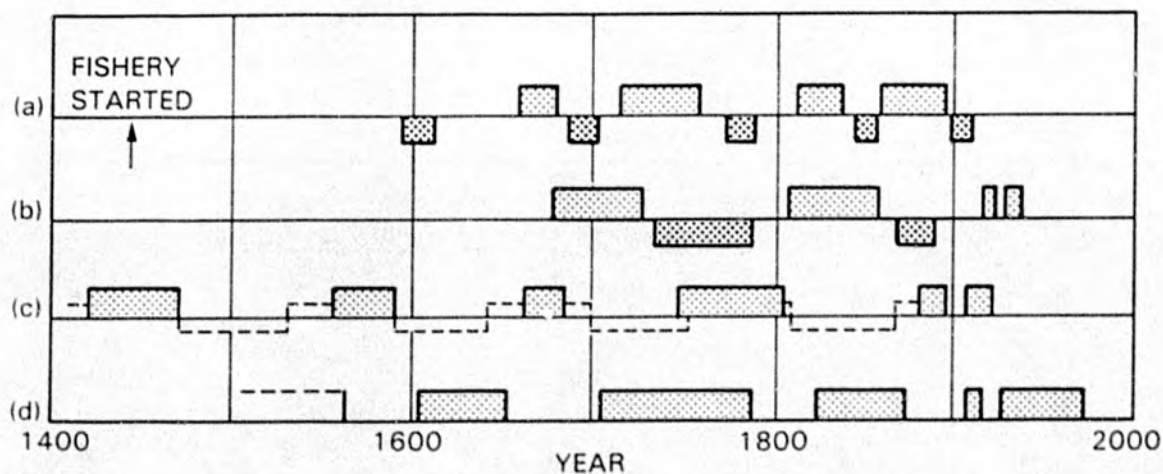
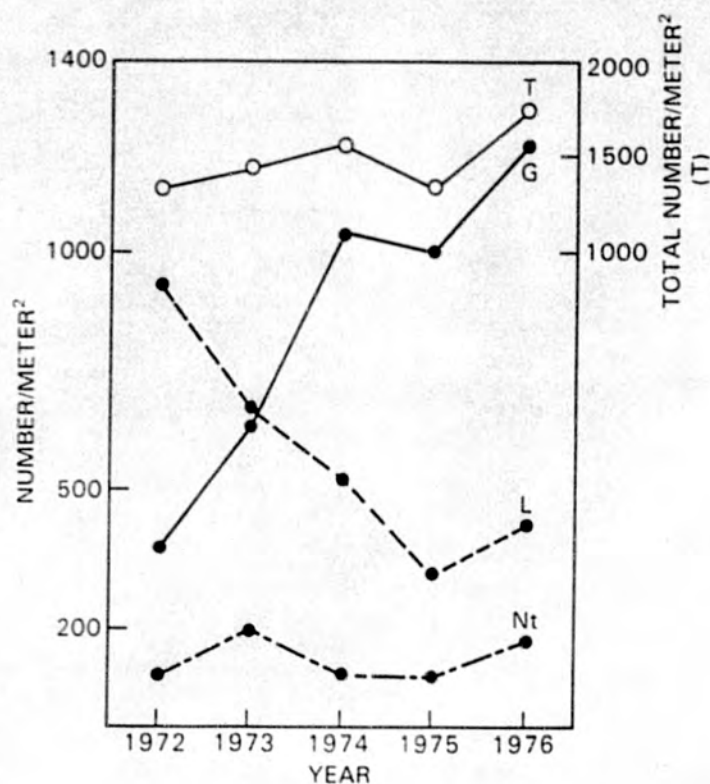


Figure 3: Periods of good and bad years (above and below line, respectively) for (a) Hokkaido herring; (b) Japanese sardine; (c) Bohuslan herring; (d) Atlantoscandian herring. (Dashed lines are speculative.) (After Jones, 1982.)



Gaining species 1972-6 (G)		Losing species 1972-6 (L)		Neutral species 1972-6 (Nt)	
Species	Gain/meter ²	Species	Loss/meter ²	Species	
<i>Paraonis gracilis</i>	238	<i>Spiophanes bombyx</i>	219	<i>Ampharete finmarchica</i>	
<i>Prionospio malmgreni</i>	144	<i>Chaetozone setosa</i>	129	<i>Glycera rouxi</i>	
<i>Myriochele oculata</i>	129	<i>Ampelisca tenuicornis</i>	112	<i>Goniada maculata</i>	
<i>Thyasira flexuosa</i>	115	<i>Amphiura filiformis</i>	50		
<i>Owenia fusiformis</i>	104	<i>Nephtys hombergi</i>	31		
<i>Magelona minuta</i>	100	<i>Scoloplos armiger</i>	26		
<i>Tharyx multibranchiis</i>	27	<i>Rhodine gracilior</i>	24		
		<i>Ammotrypane aulogaster</i>	17		

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).

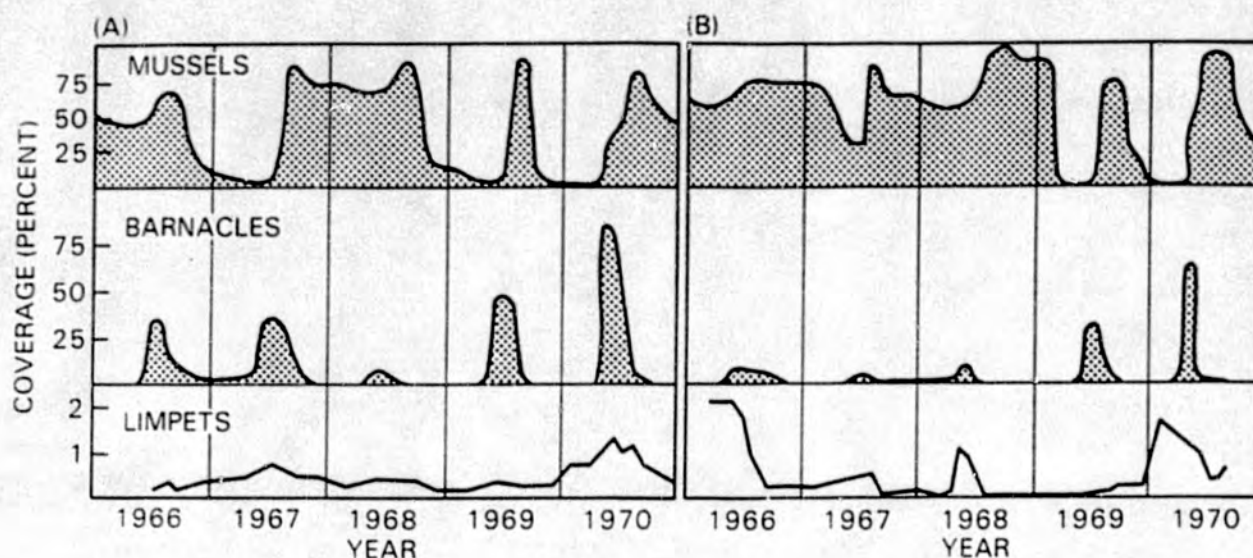


Figure 5: Fluctuations in the percentage cover on rocks by mussels, barnacles and limpets at two sites (A and B) on the north Yorkshire coast, 1966–1970 (after Lewis, 1972).

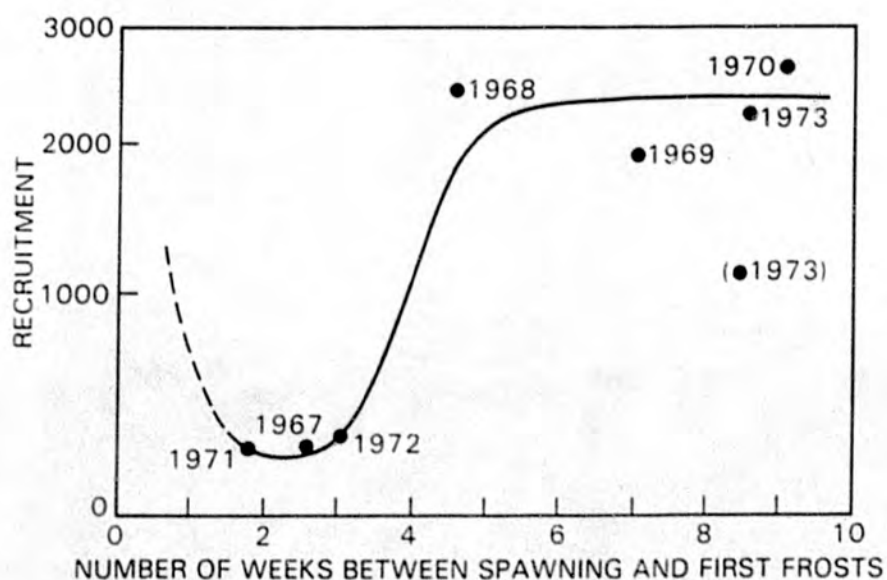


Figure 6: Relation between recruitment of limpets, *Patella vulgata*, and the duration of the frost-free period after spawning for the years 1967–1973. In 1973 there were two successional spawnings. (After Bowman and Lewis, 1977.)

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.

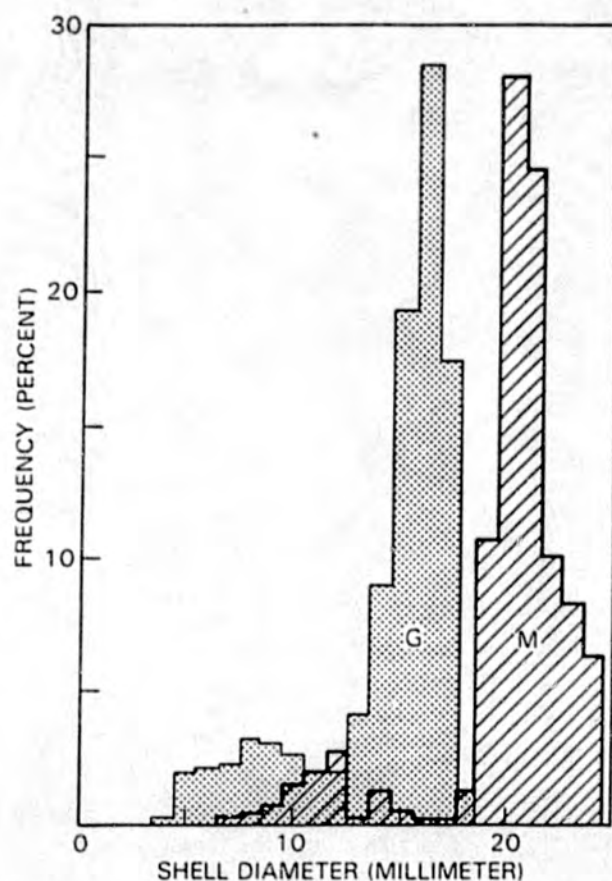


Figure 7: Size distribution of the intertidal gastropods *Gibbula* (G) and *Monodonta* (M) in a population on the Pembrokeshire coast, October 1981 (after Lewis, 1982).

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.

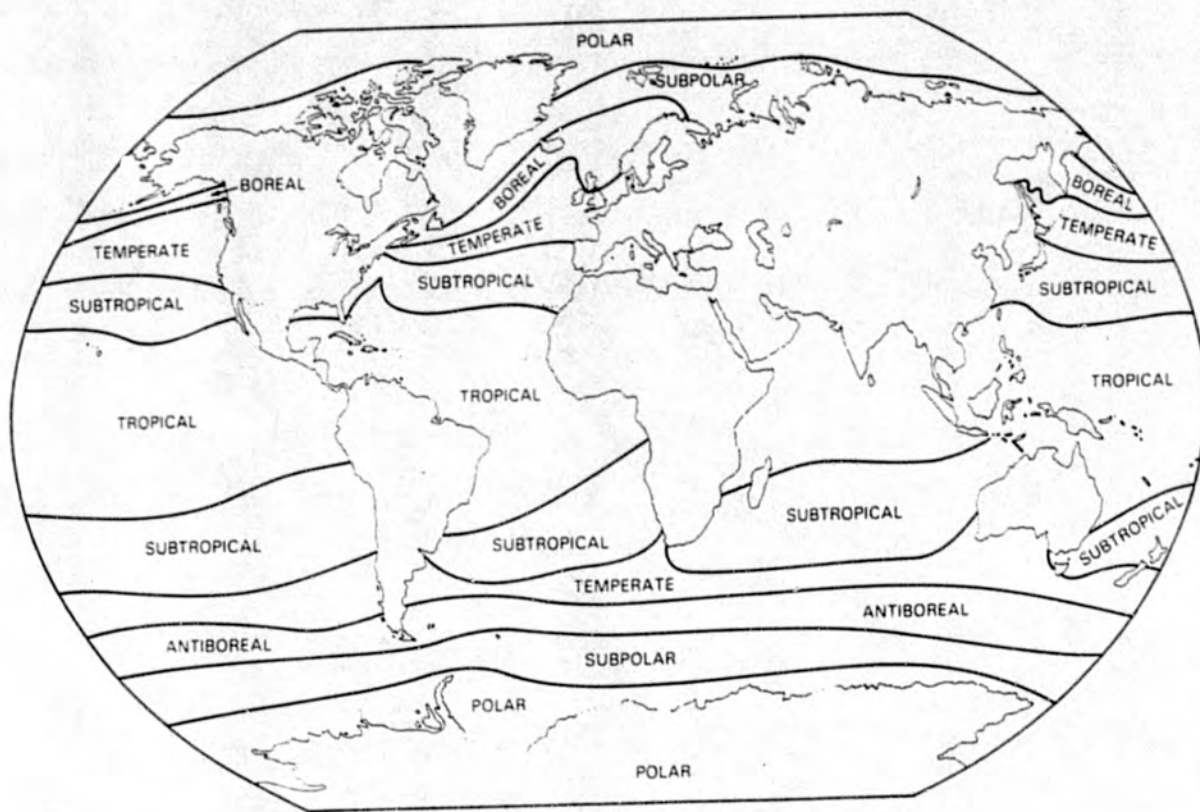


Figure 8: Map of the world showing the distribution of temperate to polar waters (Bogdanov, 1963).

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 $\mu\text{g}/\text{m}^3/\text{day}$ 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 murre, 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, 1973; 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 (*Parophrys 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 (CNEOX, 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 m³ 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 *Cerataulina* 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/meter²) 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).

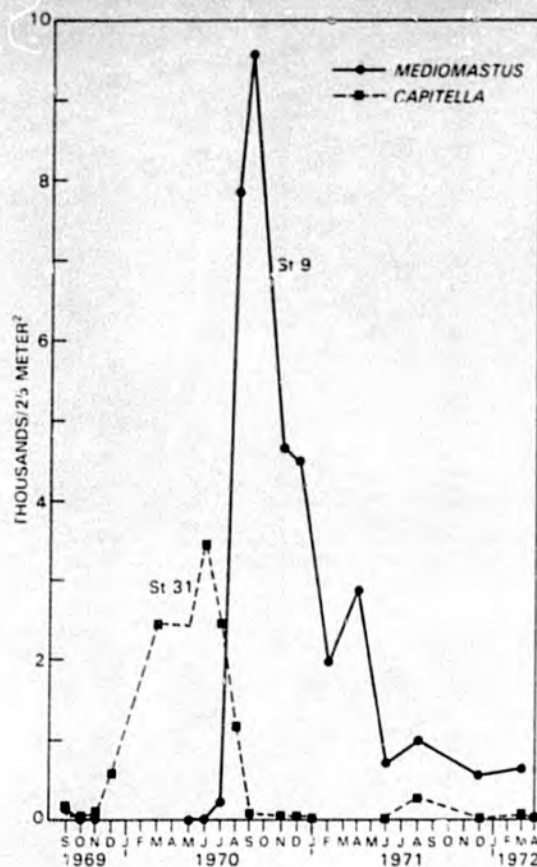


Figure 9: Succession of *Capitella* and *Mediomastus* nearshore (St 31) and offshore (St 9) after the Falmouth oil spill in Wild Harbor, Buzzards Bay, Mass. (Sanders *et al.*, 1980).

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 *Fontoporeia 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'.

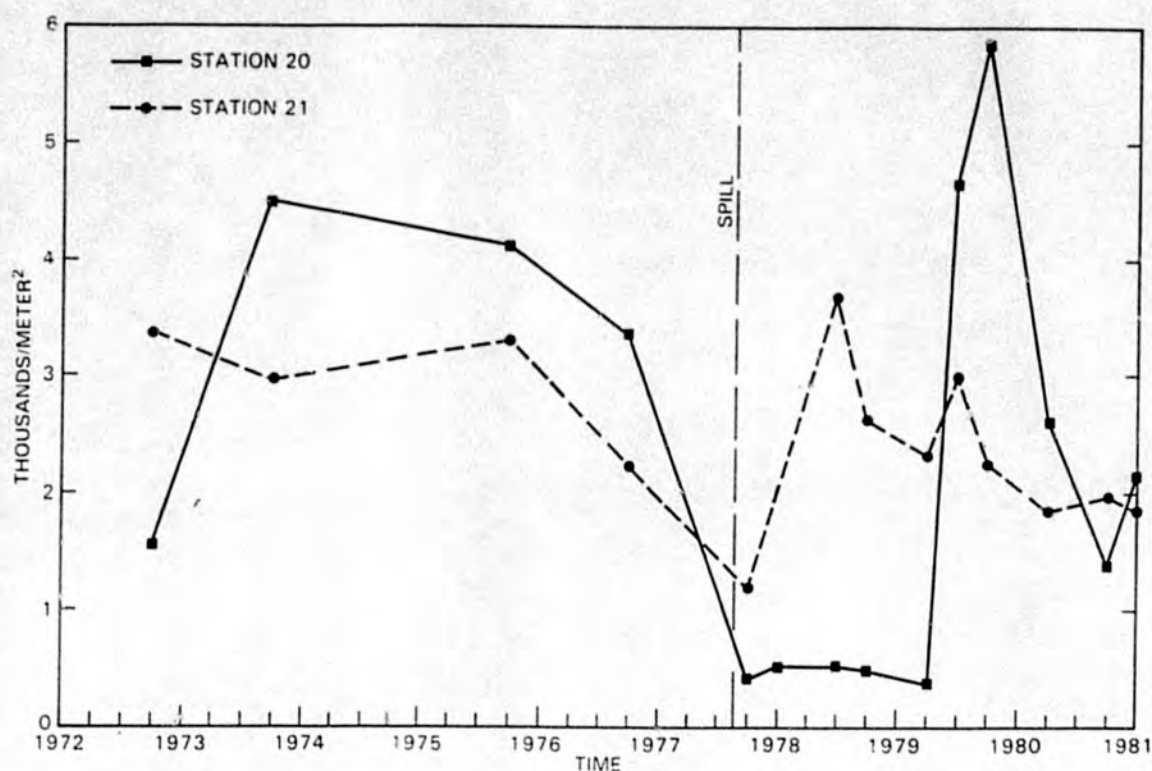


Figure 10: Changes in the faunal abundance at two localities affected by the *Tsesis* oil spill (Elmgren *et al.*, 1983).

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 *Beggiatoa 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.

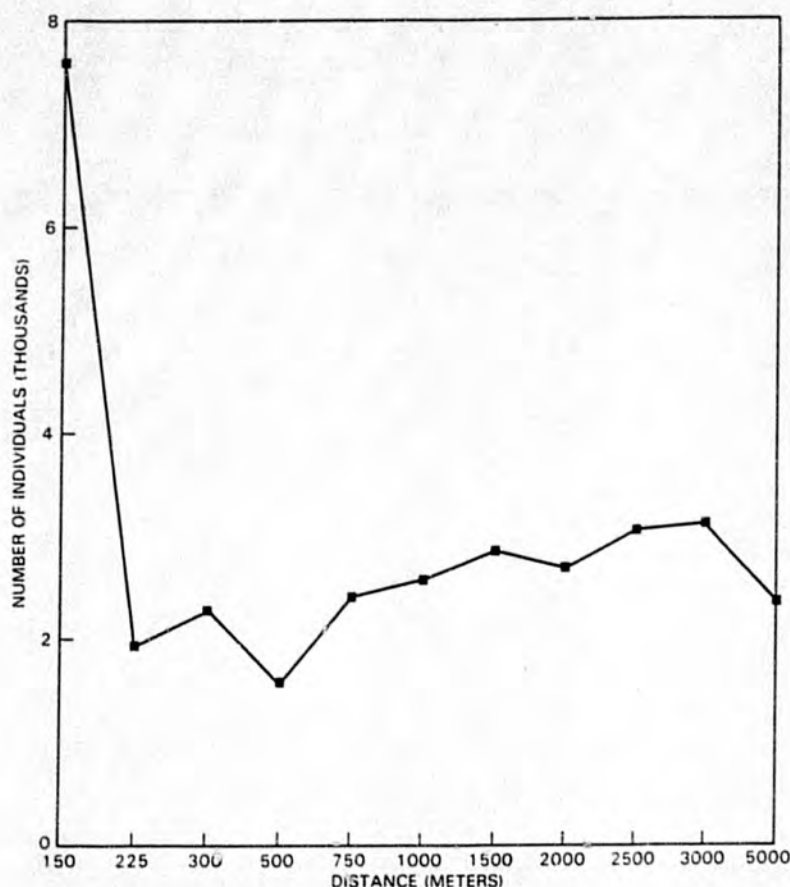


Figure 11: Changes in the number of individuals with increasing distance from the Statfjord Bravo production platform, 1984 (Matheson *et al.*, 1986).

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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 CC1 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.	Larson <i>et al.</i> (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 Daucledau, 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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (1980)
Scottish coast	Average of 109 ppb PNAH from 'clean' (sparsely populated) sites, 1127 ppb from industrialized sites.	Mackie <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>Torrey Canyon</i> 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 <i>et al.</i> (1984)
Exposed beach of mobile sand and shingle	Corton, E England	Heavy fuel oil and mousse from <i>Eleni V</i> 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 <i>Amoco Cadiz</i> 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 <i>et al.</i> (1979)
Variety of shores	Brittany	Light Arabian and Iranian oil/mousse from the <i>Amoco Cadiz</i> 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 <i>et al.</i> (1981)
A variety of shores along the Pink Granite coast	Brittany	<i>Amoco Cadiz</i> (1978) and subsequent spills	Contribution of remaining weathered residues of <i>Amoco Cadiz</i> oil to the hydrocarbon baseline was small in 1985 (7 years after the spill) compared with more recent inputs.	Page <i>et al.</i> (1988)
Variety of shores in small bay	British Columbia	No. 5 fuel oil from the <i>Irish Stardust</i> spill, 1973	After 4 years, no oil on stone, gravel residues and sand. UCM in sediment extracts.	Cretney <i>et al.</i> (1978)
Variety of shores in bay	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the <i>Arrow</i> spill, 1970	After 6 years, only a few locations visibly contaminated with large quantities of oil identifiable as <i>Arrow</i> Bunker C. These were upper high tide zones on 3 islands, where an oil/sediment pavement remained.	Keizer <i>et al.</i> (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 <i>Metula</i> 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 <i>et al.</i> (1987a, b) Baker <i>et al.</i> (1976); Hann (1977)
Sheltered beach of sand, shingle and rocks	Lowestoft harbour, E England	Heavy fuel oil and mousse from <i>Eleni V</i> 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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>Amoco Cadiz</i> , 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 <i>et al.</i> (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 <i>et al.</i> (1978)
Gently sloping shoreline of low-energy lagoon	Chedabucto Bay, Nova Scotia	Bunker C fuel oil from the <i>Arrow</i> spill, 1970	Stranded oil persists after 7 years, continuously leaching into beach substrate where weathering occurs throughout the top 15 cm.	Vandermeulen <i>et al.</i> (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 <i>et al.</i> (1989)
Salt marsh	Brittany	Crude oil remaining after <i>Amoco Cadiz</i> 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 <i>Arrow</i> 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 <i>Arrow</i> 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 <i>et al.</i> (1978)
Beaches in the Straits of Magellan	Light Arabian oil/mousse from the <i>Metula</i> 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 <i>et al.</i> (1978)
Ile Grande marshes, Brittany	Light Arabian and Iranian oil/mousse from the <i>Amoco Cadiz</i> 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 <i>et al.</i> (1984)
Aber Benoit estuary, Brittany	Light Arabian and Iranian oil/mousse from the <i>Amoco Cadiz</i> 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 <i>Amoco Cadiz</i> 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 <i>et al.</i> (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 <i>et al.</i> (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 (<i>in situ</i> 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 <i>et al.</i> (1987)