3

Excavation and Rock Washing Treatment Technology
Net Environmental Benefit Analysis

With Contributions From:

Exxon Company, U.S.A.
National Oceanic and Atmospheric Administration
State of Alaska

Compiled By:

Hazard Materials Response Branch
National Oceanic and Atmospheric Admin.
7600 Sand Point Way, N.E.
Seattle, Washington 98115

J. Michel
J. Robinson

July, 1990

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL OCEAN SERVICE
OFFICE OF OCEANOGRAPHY AND MARINE ASSESSMENT
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July 9, 1990

RADM David Ciancaglini, USCG
Federal On-Scene Coordinator
601 West Fifth Street
Anchorage, AK 99501

Dear Admiral Ciancaglini:

Enclosed is the Net Environmental Benefit Analysis (NEBA) on excavation and rock washing, my concluding summary, and responses to that summary from the State of Alaska and Exxon. We were delayed somewhat in transmitting this material, awaiting the response from Alaska Department of Environmental Conservation.

It is important to note that the summary and conclusions represent my best judgement as Scientific Support Coordinator for the EXXON VALDEZ spill response. The interpretations therein should not be construed to reflect the opinion of the Agency in any broader context.

As you know, the NEBA studies have been relatively short term and specifically directed to the questions involved in this analysis. In accomplishing this task, we did not have the benefit of the studies being undertaken through the damage assessment programs of the Trustee Agencies or Exxon. Nothing in the attached material should be interpreted to imply an opinion on my part, or on the part of other contributors to this report, as to the extent of damage that may have occurred or may occur in the future as a result of the spill.

Sincerely,

John H. Robinson, Chief
Hazardous Materials Response Branch

Attachments

U.S. Department of Transportation

United States Coast Guard

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16 July 1990

Mr. Otto Harrison
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Dear Mr. Harrison:

The Net Environmental Benefit Analysis (NEBA) Committee Report has been reviewed in its entirety. It is my understanding that the data contained in the report and submitted individually by NOAA, the State of Alaska and Exxon was included in toto for my consideration. Based on this data, I have determined that the benefits derived from the wide-scale use of the rock washer would be far outweighed by the detrimental effects caused by its usage. Although there is some disagreement regarding various conclusions in the report, there is sufficient basic information not in contention to support my decision. Therefore, as the Federal On Scene Coordinator, I will not authorize use of the rock washer. Essentially, the reasons for this decision are as follows:

A. With recolonization and a year's natural cleansing, the use of the technology would be detrimental to environmental recovery.

B. Considering the secondary environmental damage associated with the rock washer, damage to the lower intertidal zone and the mid intertidal zone habitats, and the unknown gain by using such an intrusive method, there are much better ways to improve the high human use and subsistence areas.

C. On those shorelines which still have significant subsurface oiling, the top 15cm of oiled sediment is of greatest concern. For those shorelines, other alternative treatment methods such as bioremediation, tilling, berm removal, etc. may be used to significantly reduce subsurface oil.

D. Based on the information contained in the NEBA report, the potential environmental hazard posed by the oil in its present state is far less than the certain ecological harm that would be incurred by rock washing. Biological recolonization is occurring, natural cleansing continues to remove oil, sheening is dramatically reduced, and commercial fisheries are taking place.

The process to develop the rock washer and study its effects on the environment is appropriate and consistent with the procedures employed prior to the use of other treatment technology used on this spill response.

A letter similar to this has been sent to ADEC and NOAA.

Sincerely,

D. E. CIANCAGLINI
Rear Admiral, U. S. Coast Guard
Federal On Scene Coordinator

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL OCEAN SERVICE
OFFICE OF OCEANOGRAPHY AND MARINE ASSESSMENT
ROCKVILLE, MARYLAND 20852

JUL 17 1990

RADM David Ciancaglini, USCG
Federal On-Scene Coordinator
601 W. Fifth Avenue, Suite 300
Anchorage, AK 99501

Dear Admiral Ciancaglini:

I have personally reviewed the recent report of your Scientific Support Coordinator (SSC), Excavation and Rock Washing Treatment Technology: Net Environmental Benefit Analysis, and the summary and conclusions in John Robinson's July 5 cover memorandum. As Mr. Robinson stated in his July 9 transmittal to you, the report and its conclusions were based on the best information available to the report authors and the SSC at the time of their preparation.

The summary and conclusion of the report should not be interpreted as conclusive about the extent of comprehensive damages that have occurred and may continue to occur as a result of EXXON VALDEZ oil spill. In addition, the conclusions of the report should not be interpreted as precluding the need for restoration activities to enhance natural recovery processes. The Natural Resource Damage Assessment Program of the Federal and State of Alaska trustees must make those determinations.

The National Oceanic and Atmospheric Administration fully supports the conclusions reached in Mr. Robinson's memorandum of July 5 with respect to the specific question of the environmental benefits of washing rocks to remove subsurface oil.

Sincerely,

Charles N. Ehler

Charles N. Ehler
Director

Excavation and Rock Washing Treatment Technology Net Environmental Benefit Analysis

With Contributions From:

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National Oceanic and Atmospheric Administration
State of Alaska

Compiled By:

Hazardous Materials Response Branch
National Oceanic and Atmospheric Administration
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July 1990

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
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OFFICE OF OCEANOGRAPHY AND MARINE ASSESSMENT
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July 5, 1990

MEMORANDUM FOR: RADM David Ciancaglini, USCG
Federal On-Scene Coordinator

FROM: John H. Robinson, Chief, Hazardous Materials Response Branch, NOAA

On April 27, 1990, you requested that NOAA conduct a Net Environmental Benefit Analysis (NEBA) comparing the benefits of excavation and rock washing with the benefits of natural cleanup as augmented by the approved 1990 cleanup methodologies. A committee of technical experts from NOAA, the State of Alaska, and Exxon, was formed to investigate and attempt to reach some resolution of the issues involved.

The committee has worked diligently to define the environmental factors important to the analysis, research the scientific literature to gain insight from past spills, and undertake field programs to obtain additional data where necessary.

The NEBA committee produced the attached report following a mutually agreed-upon outline and technical approach. All members of the committee were provided the opportunity to comment on a draft of the final report and the authors of each section were urged to consider committee member comments in their final manuscripts. The opportunity to append other viewpoints was provided. NOAA then prepared a summary of the report which is included as an attachment to this memorandum. The summary was forwarded to the State of Alaska and Exxon for review and comment; comments received were considered in the final draft.

Many key technical issues surrounding the analysis were partially resolved in the course of this undertaking and on some topics the range of disagreement was substantially narrowed. In accord with your instructions, NOAA attempted to reach a consensus on the question and reserved our position on the issue until it became clear that the major parties to the matter would be unable to reach agreement on the key issues.

Given the disagreement between the State of Alaska and Exxon on the conclusion of the analysis, we offer our opinion that there is no net environmental benefit to be gained from shoreline excavation and washing — in fact we believe that this technology has the potential of aggravating the injury to the environment caused by the oil spill.

We believe that the data gathered in the process of this analysis support the contention that deeply buried subsurface oil (i.e., oil beyond the reach of near-term bioremediation and other less-intrusive cleanup methods) poses little risk of causing further significant environmental injury. This finding is borne out by the following:

- There is evidence of a substantial reduction (up to 90%) in the top 20 cm of the shoreline since September of 1989. Subsurface oil should continue to be reduced gradually as fall and winter storms scour the coast.

- There is evidence that the frequency and magnitude of sheens have decreased significantly over the past several months and chemical changes in the subsurface oil have rendered it much less capable of sheening. Commercial fisheries closures attributed to sheening are infrequent.

- There is evidence that recolonization of biological communities is proceeding in candidate areas for rock washing. There is no evidence that subsurface oil is causing a serious impact to intertidal or subtidal organisms, either by direct contact or by indirect contact through adverse effects on nearshore water quality.

- There is evidence that the toxicity of subsurface oil is continuing to decline, primarily as a result of microbial degradation.

The case against excavation and rock washing is further accentuated by a review of the negative impacts of that technology:

- Excavation would alter the natural structure and profile of the shoreline, and require one to three years for a return to equilibrium.

- Excavation would eliminate virtually all plant and animal life indigenous to the mid- and upper-intertidal zone and may cause a significant adverse effect in the highly productive lower intertidal zone. Biological recovery of the shoreline following excavation would require four to eight years compared with two to six years if left to natural processes.

- Excavation would cause numerous large-scale secondary impacts associated with fuel consumption, incidental spills, noise, air pollution, siltation, possible archaeological disturbance, and human intrusion into the affected shoreline areas.

We believe that the only benefit to be derived through excavation and rock washing is the predictability that oil would be promptly removed from the subsurface environment, eliminating the potential for future contamination that might disrupt human use of the shoreline and nearshore waters. This potential benefit, in our opinion, is certainly far outweighed by the ecological disruptions that would be result from use of this technology.

We believe that the conclusions reached through this analysis strongly support natural cleanup processes and continuation of the approved 1990 cleanup program.

Attachments

SUMMARY

A technical committee was formed in May 1990 to investigate the environmental tradeoffs associated with excavating and washing oiled sediments remaining deeply buried along some sections of the Alaskan shoreline affected by the EXXON VALDEZ oil spill. The committee, comprised of Exxon, NOAA, and State of Alaska scientists, was charged with conducting a Net Environmental Benefit Analysis (NEBA) related to the advisability of excavation/washing in comparison to natural cleansing and application of treatment protocols approved by the Federal On-Scene Coordinator for use in 1990.

The NEBA committee selected two shoreline segments, Sleepy Bay on the north coast of Latouche Island and Point Helen on the southeast coast of Knight Island, as candidate locations to pursue field studies associated with the analysis.

The committee collectively outlined the range of environmental issues important to the analysis. The benefits and negative impacts of excavation and rock washing were researched in some detail, using both site-specific data from the candidate beaches and published studies on the fate and effect of oil and the impacts of physical disturbances to the shoreline. The results of these analyses can be summarized as follows.

How much oil is buried at what depths along the most severely affected sections of the coast? What are the prospects for removal by natural processes?

During the March/April 1990 shoreline assessment, 212 of the 1,134 shoreline segments surveyed had subsurface oil deposits thicker than 15cm and buried deeper than 15cm. Of 5,071 pits dug in the assessment, only 279 contained oil that had penetrated greater than 30cm, with 25 that had oil penetration greater than 60cm. The deepest occurrences of subsurface oil were found on coarse-grained beaches (boulder/cobble/pebble) along exposed shorelines.

NOAA studies at 18 stations in Prince William Sound over the six-month period from September 1989 to March 1990 indicated that, on the exposed shoreline segments most likely to be amenable to excavation, sediment reworking resulted in the removal of oil from the top 10-20 cm of

surface sediments. Data taken from Sleepy Bay and Point Helen reflected a 90% oil reduction in the top 30cm, but no discernible trend in oil reduction was evident below this depth. Exxon's winter monitoring study at 18 sites in Prince William Sound concluded that the concentration of total petroleum hydrocarbons (TPH) in subsurface sediments (generally >10cm) declined by 88% over the same period.

Using various survey methodologies, the estimated amount of oil remaining in subsurface sediments at Sleepy Bay, one of the candidate sites for excavation, was:

Gallons/meter of Shoreline	No. of Samples	Source
9.9	43	NOAA Oct-Feb Survey
5.0	13	Exxon Mar Survey
6.1	11	Exxon Jun Survey

Perhaps of more importance than the quantity of subsurface oil is the rate of natural removal which may be expected over the next few years. There have been numerous studies of previous spills in which the persistence of stranded oil has been surveyed. However, there are many factors specific to Prince William Sound that might alter the extrapolation of oil persistence curves from other spills to the EXXON VALDEZ. The factors that may speed natural removal processes in Alaska are:

1) Flocculation, the process by which fine-grained sediments adhere to the subsurface oil and make the oil less sticky and more biologically available.
2) Enhanced biodegradation, both naturally occurring and stimulated by the addition of nutrients.
3) Removal of surface oil during the 1989 treatment, which limited the formation of asphalt pavements.

The factors that may slow natural removal rates in Alaska are:

1) The relatively low wave energies relative to the grain size of shoreline sediments.
2) The initial high degree of contamination of some shoreline segments.

Oil removal is expected to be most pronounced early in the recovery period; this expectation is clearly being borne out by data gathered thus far through the various monitoring programs. NOAA estimates that virtually all of the oil buried in gravel shorelines should be removed, given no further treatment, in the periods of time indicated below:

Sheltered parts of Prince William Sound	10+ years
Sheltered outer Kenai	3-5 years
Exposed parts of Prince William Sound	2-4 years
Exposed outer Kenai	1-2 years

At what rate is subsurface oil being weathered?

Samples of subsurface oil were collected by NOAA from the two NEBA study areas and analyzed by gas chromatograph/mass spectrometry (GC/MS) to determine both chemical composition and weathering trends. Most of the subsurface oil in these samples was moderately to heavily weathered and altered significantly in both physical and chemical properties from the original state. Physically, the oil had become more viscous, less sticky, and would not readily sheen. Chemically, the oil had lost many of the light- to moderate-weight polynuclear aromatic hydrocarbons (PAHs), components which are of greatest toxicity to aquatic organisms. For example, the naphthalenes and phenanthrenes had been reduced by more than a factor of 100 when compared with fresh EXXON VALDEZ oil. Analysis of the saturate fraction of this subsurface oil showed strong evidence that microbial degradation had occurred. Overall, the subsurface oil had decreased in both concentration and toxicity and showed evidence of significant microbial degradation.

Will subsurface oil become exposed by wave action?

It is likely that there will be further removal of subsurface oil by wave action and shoreline erosion; however, these processes are most likely to occur in the winter when sediment abrasion and thus natural cleaning of the surface is at a maximum. There is no general agreement as to the rate at which the oiled subsurface sediments would be exposed by these processes. Larger cobbles exposed during severe winter storms may remain with some surface oil contamination; the smaller sand, granules, and pebbles would rapidly be cleaned.

To what degree is water within the beach being contaminated by subsurface oil and thus exposing organisms to toxic hydrocarbon compounds?

Water within the beach ("pore water") comes in contact with subsurface oil as it moves through the interstitial space among oiled sediment particles. The prospect that pore water may become contaminated was investigated by the committee.

An Exxon mathematical model was used to predict the concentration of PAHs in pore water, which contacts subsurface oil deposits, and the rate at which tidal flushing would leach the water-soluble fractions of the oil. Using data on concentrations of PAHs in sediments collected in March 1990, Exxon predicted that concentrations of total PAHs in pore water would be generally below 20 ppb. These model results were confirmed by Exxon analyses of field samples of pore water collected in March at eleven sites in Prince William Sound. Only one sample contained PAH levels greater than 10 ppb and only three reflected levels greater than 1 ppb. Two of the three highest values were for samples collected from Sleepy Bay. All observed levels of pore water contamination are below concentrations known to be acutely toxic to marine organisms.

An additional twelve water samples were collected by NOAA at the sediment/water interface during falling tides from Sleepy Bay in May 1990. Total PAH concentrations in these samples were all less than 1 ppb. The rapid dilution of the pore water as it mixes with clean interstitial and shallow subtidal waters should result in PAH concentrations that pose little or no toxic hazard to marine organisms on the lower shore and in subtidal waters.

To what extent can sheens be produced by subsurface oil? If such sheens are produced, what adverse effects can be anticipated?

The committee investigated the frequency, magnitude, composition, and effect of oil sheens likely to originate from subsurface oil. Observations by ADEC and Exxon show a declining trend in both sheen frequency and volume throughout Prince William Sound. The downward trend in Exxon data is more pronounced, possibly due to the elimination of non-Prudhoe Bay crude sheens from the data set. On a weekly basis, the volume of sheens from EXXON VALDEZ oil is now well below the present level of petroleum hydrocarbons introduced to surface water of Prince William Sound from other sources (vessel traffic, combustion, natural organics, etc.).

NOAA chemical analysis indicates that the natural weathering of subsurface oil has rendered the oil both less toxic and much less capable of producing sheen. As long as the oil remains physically stable in the shoreline, sheen frequency should be minimal. A temporary increase in sheening would likely be produced either by excavation or by intense winter storms. Sheens produced by excavation could be expected to be controlled in light to moderate weather conditions. While control of storm-induced sheens is unlikely, the range and effect of such sheens would be limited by natural dispersion during the storm event. NOAA studies in 1989 indicated that only in unusual weather conditions would convergence zones have the potential to concentrate free-floating oil into popweed windrows to the extent fishing gear might be contaminated. The threat of exposure to birds feeding at these windrows is likely to be small as exemplified by the lack of reports of oiled marine birds this year.

In summary, subsurface oil is relatively stabilized and insulated from exposure to vulnerable resources through sheening. Adverse effects on birds, fish, wildlife habitat, or human uses are not expected to originate from sheens produced by subsurface oil.

What is the near- and long-term fate of subsurface oil?

Subsurface oil that persists in the shoreline will eventually be degraded by indigenous bacteria. However, prior to the complete microbial breakdown of oil buried deep within the shoreline, other processes may be important in controlling the intermediate distribution and fate of the oil.

Flocculation is one process that has been proposed by Exxon as being an important mechanism for enhanced natural removal and dispersal of subsurface oil. The flocculation process has been demonstrated in laboratory studies, and in the field some subsurface oil appears to have taken on a more weathered, smeary texture indicative of floc formation. However, the rate of flocculation and its relative importance, particularly in the removal of heavily oiled subsurface sediments, is not known.

There is some concern that the fine-grained fraction of oiled subsurface sediments may be exposed by erosion and deposited in the nearshore subtidal environment. Since there are few data available to evaluate this potential risk, a diving survey was conducted by ADEC in June 1990 to collect subtidal sediments along a transect out to 100m off both Point Helen and Sleepy Bay. Total PAHs in these subtidal sediments were 0.1-9 ppb off Point Helen and 1-130 ppb off Sleepy Bay. This limited sampling would indicate that deposition of oiled sediments, derived from either subsurface or surface sources, may be minimal.

What are the potential effects of rock excavation on the shoreline substrate?

There is great variability in sediment grain size and shoreline exposure in Prince William Sound, and therefore the effects of excavation and rock washing on the structure of the intertidal zone would vary considerably from location to location. Shoreline segments that contain high concentrations of subsurface oil are also generally exposed to moderate or high wave energy. The upper part of the intertidal zone in these locations is affected by the erosion and deposition of high-tide berms, as shown through analysis of data obtained during the Exxon, NOAA, and ADEC winter monitoring programs. The middle and lower intertidal zones are less affected, particularly if a cobble/boulder armor is present, such as on Point Helen, one of the candidate sites for rock washing.

Using data from the winter monitoring programs and case histories from other studies, it is predicted that, if the material were removed from the upper one-third of the beach and replaced in approximately the same position from which it was taken, the upper berms at Point Helen would reestablish themselves after two or three medium-sized storms. The re-armoring process in the middle and lower part of the shoreline would require a few more storms, possibly covering a period of one to three years.

For Sleepy Bay, which is more sheltered but also is composed of finer-grained sediments, the upper surface sediment resorting should occur in less than one year. However, because the middle portion of the beach changes more slowly, complete reestablishment of the original profile and sediment resorting may require up to three years, though it may take place sooner. Recolonization by intertidal organisms would be delayed during these periods of sediment restabilization.

Loss of fines is not expected to be of concern, mainly because of the relatively small percent of sand-sized materials in the upper zone on most beaches in Prince William Sound. Because the sand exists primarily as a matrix to the already fixed framework of the beach, there should be no impact of its potential loss on beach morphology or erosion. There were concerns that the loss of sand might change the hydraulics of the beach, but the magnitude or consequences of this impact are unknown.

What are the effects of subsurface oil and excavation on the recolonization by intertidal communities?

In the areas where excavation is being considered, the intertidal zone can be divided into three subzones on the basis of the distribution of biological communities. The upper intertidal zone, generally the location of the highest concentrations of subsurface oil, is normally not inhabited by a very rich biological community because of relative dryness, sediment mobility, and lack of food.

The middle intertidal zone contains a more rich and diverse biological community. Observations made this spring and summer indicate that these communities are recovering from the oil and treatment activities. Oil concentrations of the surface sediments in this zone are generally low and, as discussed earlier, toxic impacts from the discharge of groundwater through oil-contaminated sediments across this zone are not expected to be significant. Because most of the plants and animals in this zone live in the top 10-15cm, there is no pathway of exposure to subsurface oil other than oil exposed during erosional events. To the extent the oil remains buried, it poses no serious risk to intertidal communities in this zone.

The lowermost intertidal zone has the greatest biomass and species diversity. In most cases, this zone is showing evidence of recovery and only very low concentrations of oil occur in the surface sediments. The lower intertidal zone would be least impacted by the residual subsurface oil.

With the information available on the likely zone of disturbance by excavation and rock washing, it is assumed that few organisms in the upper and middle intertidal zones would survive. There may also be impacts to these zones resulting from the traffic of equipment and people on the shoreline. It is likely that these surface impacts would extend to some degree into the relatively rich lower intertidal communities.

NOAA estimates that three to five years would be required for biological recolonization after the shoreline stabilizes. Adding one to three years for stabilization, the total recovery period after excavation is estimated to be four to eight years. If excavation does not occur, NOAA estimates recolonization will occur in two to five years at Point Helen and three to six years post-spill at Sleepy Bay.

What would be the secondary impacts from excavation and rock washing?

Increased sediment loading in the water column would be unavoidable during excavation and replacement. Coarse material would settle out readily, although it is likely that oiled silts and clays would settle slowly and might be transported along the beach by tidal currents.

Secondary spills of fuel and process water were estimated using the spill rate of cleanup activities in 1989. Assuming that four or five rock washing units were active for a two-month period, an estimated eight to ten fuel spills would occur.

Excavation and rock washing would disturb wildlife with work activity and noise for six weeks or longer in each work area. Sources of disturbance include personnel, vessels, equipment, and aircraft. Traffic in the intertidal area may further affect the recolonization process.

Exxon estimates that excavation and rock washing at Sleepy Bay would generate 12,000 gallons of skimmed oil/water mixture, 168,000 gallons of wastewater, 500,000 gallons of sludge waste, and 300,000 pounds of air emissions. Fuel consumption is estimated at 400,000 gallons.

What would be the impacts on human use of the shoreline and nearshore waters should subsurface oil not be removed? What would be the related effects of excavation/rock washing?

The State of Alaska indicates that any of the shoreline segments which are candidates for excavation and rock washing are in areas of moderate to high human use. These areas are used throughout the year for recreation, subsistence and commercial fishing, and are of significant cultural importance. Subsurface oil has some capacity to pose an episodic threat to these uses of the shoreline and nearshore waters.

Although sheens are declining in Prince William Sound, and the relationship between sheens and subsurface oil is tenuous in non-storm conditions, commercial fishing in the immediate vicinity could be affected in the event sheening does occur. Episodic releases of subsurface oil may contaminate subsistence fisheries, interfere with shore-based commercial and recreational fishing, and impair use of the shoreline for recreation. Large amounts of subsurface oil can impact recreation and subsistence users who dig fire pits in the upper intertidal zone.

Excavation/rock washing is one of the most predictable means of assuring that oil is removed from the environment rather than remaining as a potential source for episodic exposure during winter storms. The expected impacts of the excavation and cleaning process include potential for temporary sheening or resuspension of oiled particulates, creation of silt plumes, and disturbance by equipment and personnel. These impacts can be mitigated by proper containment of sheens and timing of operations to minimize impacts to resources and users. Recreational and some subsistence uses could resume soon after the treatment process. Other subsistence activities would be delayed four to eight years until natural recolonization of intertidal biota occurred.

FOREWORD

This report was only possible through the dedicated efforts of many individuals who worked extraordinarily long hours for several weeks in meeting the stringent deadlines associated with its production. The authors especially wish to thank Lori Harris of NOAA for her outstanding efforts in compiling both the draft and final manuscripts.

TABLE OF CONTENTS

II. **Environmental Considerations**, *cont.*

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LIST OF ABBREVIATIONS

ADEC	Alaska Department of Environmental Conservation
ADF&G	Alaska Department of Fish and Game
ADNR	Alaska Department of Natural Resources
cm	centimeter
GC/MS	gas chromatograph/mass spectrometer
km	kilometers
K_{ow}	octanol/water partition coefficient
mg/kg	milligram per kilogram
µg/l	micrograms per liter
m	meter
mm	millimeter
MLLW	mean lower low water
NEBA	Net Environmental Benefit Analysis
NOAA	National Oceanic and Atmospheric Administration
ng/l	nanograms per liter
ng/mL	nanograms per milliliter
PAH	polynuclear aromatic hydrocarbons
PCB	polychlorinated biphenyls
ppb	parts per billion
ppm	parts per million
SCAT	Shoreline Cleanup Assessment Team
SSAT	Spring Shoreline Assessment Team
TPH	total petroleum hydrocarbons

LIST OF FIGURES

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LIST OF TABLES

I. DEFINITION OF STUDY

I.A. OBJECTIVE

I. A.1 STATEMENT OF THE PROBLEM

Cleanup of the EXXON VALDEZ oil spill has required the use of various shoreline treatment techniques, many of which are new to oil spills. Water flooding, omnibooms, and application of fertilizers were innovative approaches developed and modified for use in addressing the cleanup problems in 1989. One of the shoreline treatment problems in 1990 is the presence of oil which has penetrated into porous sediments and persisted through the first year of fall/winter/spring storms. It is believed that the approved shoreline treatment techniques for 1990, namely, manual removal, spot washing, and bioremediation, will not effectively remove subsurface oil this year. Bioremediation has been shown in laboratory and recent field studies to induce higher respiration rates to depths of 15-30cm, yet the rates are lower than at the surface. The ongoing monitoring program of the use of Inipol and Customblen fertilizers will help answer the questions about the degree and depths of enhanced degradation resulting from nutrient augmentation.

In the event that none of the approved techniques adequately address the subsurface oil treatment problem, the Federal On-Scene Coordinator requested that Exxon determine the feasibility of use of an excavation/rock washing process. Exxon agreed to manage the design and demonstration of a rock washer for removal of subsurface oil as long as an assessment of the net environmental benefit was conducted simultaneously with the engineering development. A committee was formed with representatives from the National Oceanic and Atmospheric Administration (NOAA), State of Alaska, and Exxon, with NOAA designated as the chair, to conduct a Net Environmental Benefit Analysis (NEBA), working in close conjunction with the Engineering Process committee. The NEBA committee was charged with the task to determine if there were net environmental benefits from the excavation and washing of oiled sediments, and return of treated sediments to the excavated site over natural cleansing and the use of approved 1990 treatments.

I. A.2 GENERAL APPROACH

To determine the net environmental benefit of excavation and rock washing of oiled sediments, the NEBA committee developed preliminary selection criteria to identify the types of beaches that would be candidates for rock washing. Based on the assumptions that 1) approved or developing techniques would be adequate for treatment of surface oil, and 2) natural cleansing or bioremediation may be effective for removal of subsurface oil to depths of 10-15cm, this report focuses on deep subsurface oil. The following general criteria were used to identify candidate beaches:

1) The degree of subsurface oil contamination was classified as OP*, OL*, or OR* during the Spring Shoreline Assessment Team (SSAT) survey.
2) The depth of oil is >15cm.
3) The thickness of the oiled sediment zone is >15cm.
4) The substrate type is mostly sediment of cobble or finer grain size.
5) TAG recommendation was bioremediation or mechanical relocation/till.
6) The shoreline was a chronic source of sheens.

*OP = Oil fills pore spaces between sediment particles
OL = Lens or layer of buried oiled sediments
OR = Residual oil on sediments or in pore spaces but not saturated

The committee felt that it was important to identify specific beaches so that the actual operational constraints and environmental conditions of the sites would provide the basis for evaluation. Because so many parts of the evaluation had to be based on the literature and extrapolation of data, it was important that real sites be included. Three specific candidate beaches were to be identified as representative of the range of shoreline types being considered for rock washing treatment. These shoreline types were initially identified as:

- Exposed, outer beach with long stretches of relatively uniform sediments and little to no operational access problems.
- Moderate-energy shoreline with highly variable substrate.

- Sheltered, pocket beach with relatively small areas needing treatment and limited space for onshore operations.

It was agreed that one of the candidate sites should also include an anadromous stream mouth.

Using site-specific examples, the remainder of Section I describes the engineering process envisioned for excavation and rock washing and the existing physical, chemical, and biological conditions of the candidate sites. In the first part of Section II, the persistence and impact of the subsurface oil under a no-treatment condition is assessed. In the second part of Section II, the impacts associated with excavation and rock washing are assessed. Section III consists of a summary statement on the impacts and tradeoff considerations for excavation and rock washing as a shoreline treatment technique.

I.B. DESCRIPTION OF EXCAVATION ROCK WASHING

I.B.1 GENERAL TECHNOLOGY

The excavating rock washer is a mechanical process for removing oil from excavated beach rocks and sediments using heated sea water. All equipment components and specifications have not yet been determined, but most characteristics of the process can be identified. Conceptually, this process is intended to be a self-contained, permitted technology that would be capable of excavating and cleaning coarse and most finer beach sediments and replacing the clean material back on the beach. Process objectives include minimization and containment of process wastes and control of suspended sediment plume formation resulting from the excavation.

The equipment used in this process is based on that used extensively in placer mining throughout Alaska, and in gravel classification and washing in Alaska and throughout the United States. The method involves excavating the contaminated material, sorting it according to size, and running it through the washing or tumbling equipment, then replacing the treated and clean sediment near its original location. Generally, the finest sediments become waste and would not be returned to the beach.

To satisfy the need for self-containment and the minimization of wastes, this equipment is augmented with additional equipment capable of excavating oiled beach sediments, separating oil from the process water, cleaning the process water of entrained sediment, recycling the process water, and dewatering and replacing the excavated and cleaned sediment back on the beach. Since the entire process requires an assorted number of individual pieces of varying bulk, and since large volumes of oily wastes are anticipated, ancillary equipment is required. This ancillary equipment would be used to deploy materials onto and off the beach, berth and feed support personnel and store generated waste water, oily slop and contaminated sediment.

I.B.2. PROCESS FLOW DIAGRAM (PRELIMINARY)

See Figure I-1.

I.B.3. OVERVIEW OF PROCESS CAPABILITIES

This equipment does not yet exist. Although smaller-scale equipment developed primarily to clean sand or pebble beaches exists, nothing of the scale and complexity proposed for the present application has ever been built. The selected process is composed of field-tested components. However, operation as a combined unit in an adverse environment (on a barge and between barge and beach) make the following capabilities somewhat uncertain in the field.

a. The unit can operate at a process rate of 100 cubic yards per hour.

b. The unit is capable of handling sediment sizes from silt to 24" in diameter although it is not designed for excessive fine sediment or friable sediment loads.

c. The unit is capable of operating both onshore as well as from a mobile offshore platform, and can be deployed on any size beach where physical access is possible.

d. The process uses salt water maintained at 160°F or greater to clean the sediment.

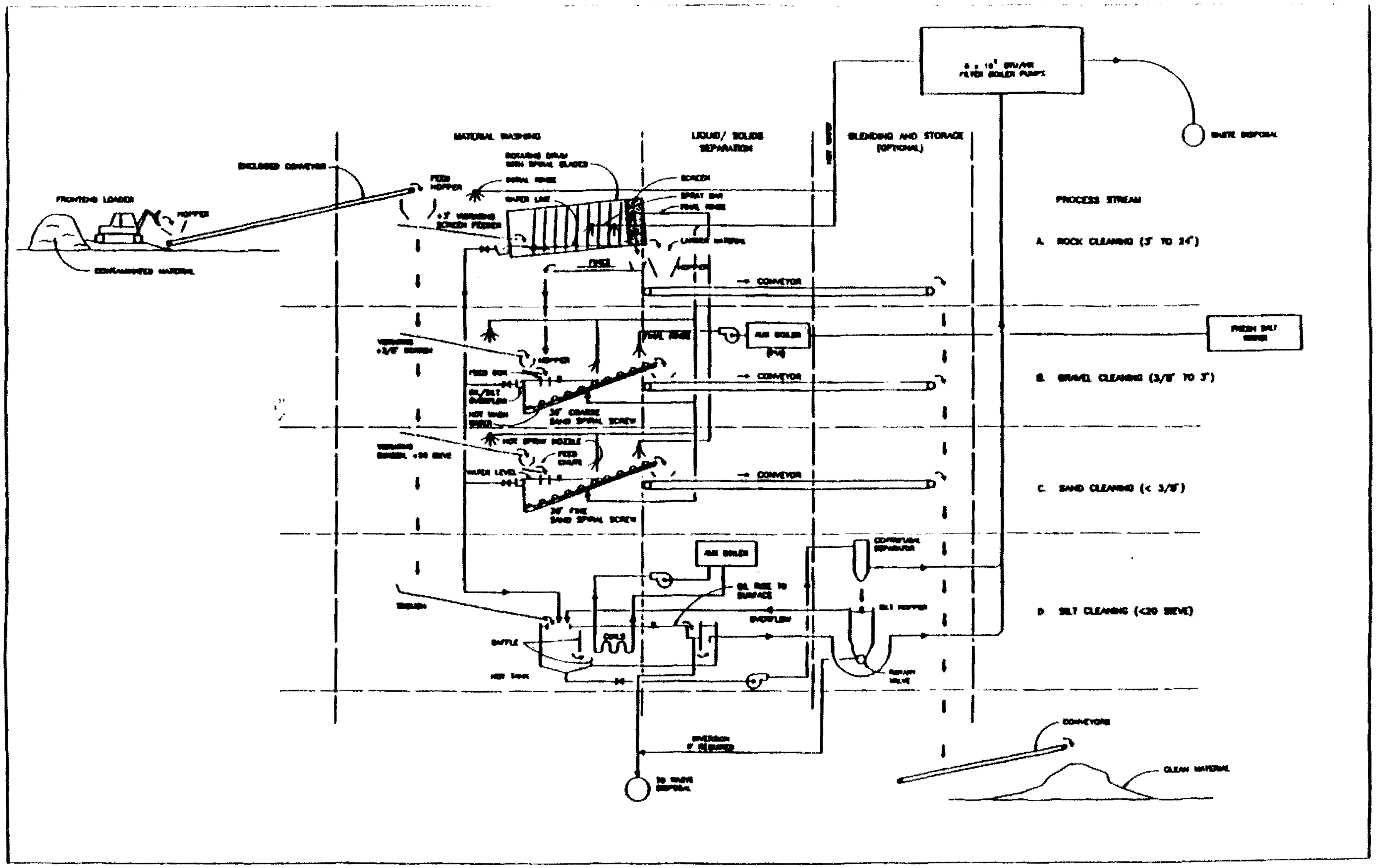

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Figure I-1. Preliminary process diagram (KCM, Inc. and Northwest Enviro Service, Inc., 1990).

e. The process design minimizes the recycle water rate and make-up water rate. Discharge of any process water as an effluent stream will not be considered as an acceptable option unless it satisfies all laws and regulations concerning discharge standards.

f. The process dewaters all processed sediment in order to minimize the water content of the processed sediment to an acceptable level for recontouring on the beach.

g. The unit is designed to minimize rock crushing or breakage. However, when sediments containing both sandstone and shale are processed, shales may be pulverized and could generate heavy sediment loadings when returned to a beach.

h. The oil content of the final cleaned rocks will be less than 800 parts per million (ppm) total petroleum hydrocarbon (TPH) on composite samples from a representative distribution of sediment sizes.

i. The process is designed to accomplish the job using only salt water and not any other chemicals to enhance the cleaning of rocks. However, the process has the flexibility to allow addition of chemicals to enhance treatment of recycled process water.

j. The process will release oil and fine particles during excavation that can be carried by waves and water around and away from the excavation site. These releases must be controlled by other means, such as booms and/or sediment curtains.

I.B.4. OPERATIONAL CONCEPTS AND LOGISTIC CONSIDERATIONS

The excavating rock washing operation entails a sequence of activities and events that must be considered in any evaluation of environmental impact:

Site Selection and Pre-Operational Preparations

The selection of a beach site for excavation rock washing would entail a beach characterization based on existing data, field observation, and measurement. This characterization will yield operational parameters, such as work area, slope and stability of beach, needs for site preparation, volume and distribution of sediments to be excavated, working conditions, and access to areas to be excavated. The characterization will help determine how the equipment will be mobilized and deployed. In particular, it will help determine which equipment components will be employed (and which operations will be conducted) on-beach and which off-beach. The process is capable of both modes of operation, although off-beach would be generally preferred due to greater mobility and flexibility and lower intrusiveness.

Mobilization

All necessary equipment, barges and boats involved with the excavation project would be marshalled at a nearby port, Valdez, Seward or Whittier, with industrial capabilities. Portable equipment would be loaded onto the appropriate transportation and the process components would be assembled and checked. All elements would be tested to ensure each one is operational and last-minute modifications made. Pollution prevention controls on the barges would be ascertained. A review of all safety, operational, logistical, weather, and pollution plans would be accomplished and last minute changes incorporated.

Site Preparation

Double oil containment booms would be put in place around the area to be excavated and the waste storage and process transporting barges to contain any released oil or sheening from the work site. A crew with boats and cleanup equipment would be on standby for any spill cleanup. Engineering controls would be in place to reduce silt loadings from the excavation. Prior to any process excavation, large debris, such as logs, or any other non-processable material would be gathered and stockpiled on the beach for later replacement or disposal. Clean sediments extending to a depth of approximately 30cm, as much as practical, could be set aside on the beach for later replacement.

Sediment Washing, Dewatering, and Quality Control

Oiled sediments, as well as sediments outside the oiled area (due to a need for a working area and excavation layback), would be excavated.

Rocks larger than the equipment capacity would be separated and stockpiled on the beach for later replacement. Rocks too large to move with the equipment being utilized will remain in place.

The excavated sediments would be size classified then dumped into primary washing units and washed with sea water heated to 160°F or greater. The water would then be recycled to reduce heating requirements and minimize wastewater volumes. The processed sediment would then be dewatered and, if the TPH content is at or less than 800 ppm as determined by an agreed upon test protocol, the processed sediments would be returned to the beach. Dewatering produces the low water content necessary to maximize the stability of returned sediments. Excess water and the oil removed, plus a portion of the fine glacial silts that cannot be processed to the 800 ppm TPH level (probably sediments less than 1mm in size), would be separated for eventual disposal.

The most probable mode of operation would have earth moving equipment and conveyor stations on the beach and all other process equipment on a barge off the beach. Another choice would be operation of the entire process on the beach and do away with the necessity of an offshore processing unit. This scenario is unlikely due to the actual size of the components of the process equipment necessary, the need for mobility to "follow" excavation equipment as it progresses down the beach and the real possibility of severe weather damage and destruction to the processing equipment.

At a minimum, the equipment on the beach would be a couple of backhoes and/or front end loaders and the termination of the belt conveyor systems. The remainder of the rock wash process equipment would be based on a barge. This would eliminate any construction pads (leveled areas to place the process on) on the beach and would reduce the impact to non-oiled beach segments.

<u>Oil/Water/Solids Separation and Quality Control</u>

The separated oil, sludges, contaminated sediments and oily water would be transferred to a tank on a U.S. Coast Guard-certificated petroleum/waste oil tank barge. When full, the barge would be taken to a nearby shore facility to further process the waste. No secondary concentrating of oily waste is anticipated to be done on-site.

Handling, Transfer, Storage, and Disposal of Waste

All wastes generated would be transported to a land-based facility for treatment, recovery, or further dewatering and concentration. It is anticipated that the oily solid wastes generated by this process would be transported to an approved landfill.

Control of Oil Leaching, Sheens, and Glacial Fines

A double boom will be in place around the beach and the process and waste storage barges to contain sheens and sediment released as a result of the beach excavation. Pads and pompoms can be used to "wipe up" sheens or concentrations of oil.

Backfill and Beach Recontouring

To the extent that the working area allows, and the rate of return of washed material, portions of the excavation could be backfilled during the processing operation. Prior to demobilization at the particular site, the beach would be re-contoured to original conditions as much as possible.

Frequency of Movements of Equipment

Work on the beach will be governed by the weather conditions encountered and the tide cycle. Assuming that the weather is not a factor, it is anticipated that the work day could be anywhere from eight to fourteen hours. Movements by vessels along the beach would be a function of the excavation progress being made. It is anticipated that the repositioning of equipment would be made at high tide and generally once or twice in a 24-hour period. Depending on the availability of suitable waste storage barges and their capacities, movement to the shore processing facility would be made to coincide with demobilization or when the barge is loaded to its design capacity. A possibility exists, due to barge capacities and the amount of waste that might be generated, that in order to continue uninterrupted excavation operations, more than one waste oil barge would be necessary.

Demobilization

On complete excavation and cleaning of a segment and final beach re-contouring, with agreement by concerned parties, the equipment will be moved to the next beach segment targeted for cleaning and the process will begin again. On termination of all cleaning operations, all vessels will either be released from the site or returned to the port of embarkation for final debrief, unloading and cleaning of the process components.

I.B.5. EXPECTED PROCESS AND ANCILLARY EQUIPMENT

Equipment: Purpose	Type/Example/Size
Debris Removal	Frontend loader tracked w/5 yd^3/bucket
Conveyor(s)	Covered, w/hand cap. up to 24" rock, 100'-120' length
Water Heater/Boiler(s)	@ 65 mbtu, Area: 15' x 40'
Rock Washer	30' x 7' trommel w/supports and 3-stage vibrating screen on 10'x 20' base
Oil/Water/Solids	2 to 4 sand screws 25' length, settling tanks, separation & dewater pumps, sieves, hydroclones, hoses, pipes; Area 140'x 20'
Beach Regrading	Bulldozer, w/gen. purpose blade
Excavation and Washer Feed Loading	Backhoe(s) and front end loader(s) w/4 or 5 cubic yds bucket
Waste Tank Barge(s)	30k-100k bbl capacity approx 50'x 200' dimension
Crane	10-60 ton capacity
Landing Craft(s)	Handle beach excavation equipment, approx. 3 @ 70'x 30'

Equipment: Purpose, cont.	Type/Example/Size
Deck Barge/Liftboat	50'x 200'; house process equipment
Oil Containment Boom-36"	5,000'-10,000'; around process barge and beach excavation area
Generators	Power auxiliary equipment/lights
Support Vessels and Aircraft	
Tugboats (Ocean)	(2) @ 1800-5000 hp; standby/handle/move barge(s)
OSV/Crewboat	150-180'; equipment and berthing/messing and A/C landing pad
Workboats	15-25 ft, transportation to and from beach, boom deployment
Helicopter	Medical evacuation and rapid transportation
Personnel	20+ pers to run process/excavation; 50+ pers Support/Vsls/Contract/Sup/ Regulatory

Positions, Locations of Process and Ancillary Equipment

Debris Removal	Beach
Excavation	Beach
Conveyor(s)	Beach excavation area to barge and back
Water Heater/Boiler(s)	Deck barge/liftboat
Rock Washer	Deck barge/liftboat
Oil/Water/Solids	Deck barge/Liftboat (separation & de-water)
Beach Regrading	Beach
Waste Tank Barge	Seaward of Deck barge/Liftboat

Positions, Locations of Process and Ancillary Equipment, *cont.*

Crane	Deck barge/OSV
Landing Craft(s)	Alongside OSV/Deck barge
Oil Containment Boom-36"	OSV/Deck barge/Deployed around the work area
Tugboats (Ocean)	Seaward of deck barge
OSV/Crewboat	Seaward of deck barge
Workboats	OSV/Deck barge/Beach
Helicopter	OSV
Personnel	3-7 pers beach; remainder on vsls/barge
Generators	Process barge/beach

I.B.6. OPERATIONS

Equipment Position

It is anticipated that the deck barge (draft approximately six to eight feet) handling the process equipment will be anchored an expected 50-100 feet from the beach to safely support the anticipated draft in the expected sea, wind and current conditions. Other vessels will be anchored or stationed in the immediate vicinity to best support the process. If a lift boat or jack-up barge is utilized, the distance to the beach will be less than 50 feet, with the distance used depending on bottom conditions such as sediment type and slope. See Figure I-2 for a schematic representation.

Beach Progress

Assuming a 12-hour work day with no breakdown and weather considerations and an excavation cross-section of 25 yards and 0.66 yard depth (60 cm) at a rate of 100 cubic yards per hour, progress up the beach would be at a rate of about 6 yards per hour or about 250 feet per day. Equipment would probably be moved every 12- to 24-hour workday. In actuality, allowing for equipment breakdown, maintenance, and weather-affected delays, progress would likely not exceed 100-150 feet per day.

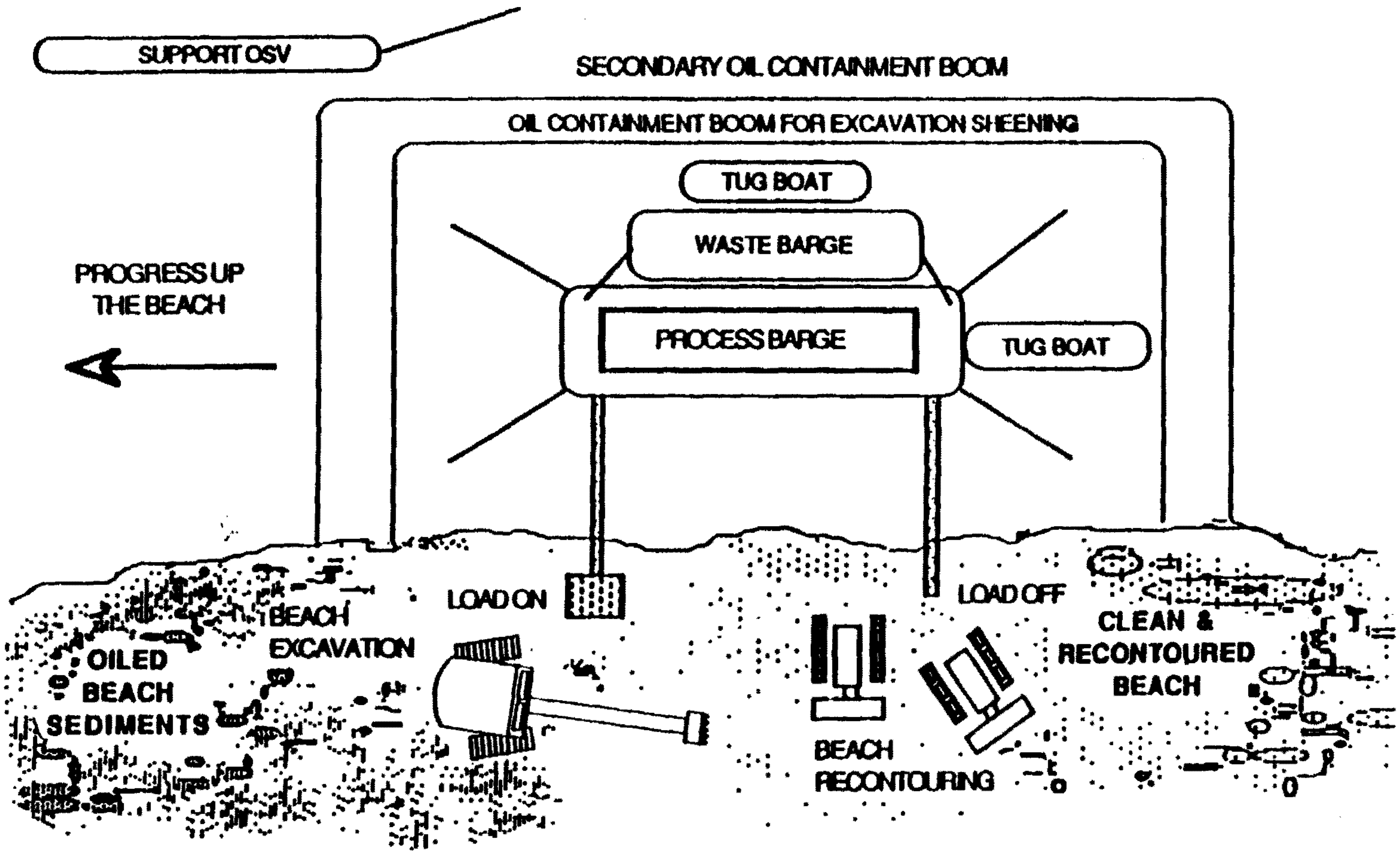


Figure I-2. Beach excavation rock washing project.

Weather Considerations

If the equipment is barge mounted it is anticipated that the weather window would be limited to seas four feet or less and winds less than 20 knots. Barge stability is important for process water clarification purposes and any gravimetric laboratory sampling analysis. A comprehensive heavy weather plan would have to be devised and implemented with contingent "safe anchorages" identified to which the process equipment, waste storage barges, and auxiliary vessels could be moved.

Oil Spill Contingency

Any sheens generated by the excavation process or resulting from handling of oily sediments over water and conveying could be handled with pads and pompoms. However, the concentration of oil in the process water could result in a more significant release through equipment failure.

USCG Marine Safety Office Valdez, Alaska Oil Spill Contingency Plan should be reviewed and implemented as applicable.

State of Alaska Oil Spill Contingency Plan should be reviewed and implemented as applicable.

A comprehensive survey review of the tidal current conditions and bottom composition, contour, and sediment conditions would have to be made to provide for the safe anchoring and navigation of process equipment and all auxiliary vessels.

Secondary Impact: Waste Generation, Air Emissions, Fuel Consumption, and Noise Generation

Machinery involved with the process would produce noise at a level of over 110 decibels (dB). This is a high level of noise that would persist in an operating area during daily work periods of between 8 to 14 hours. At about 1/2 mile away, the perceived noise level would be over 62 dB, well above levels regarded as potentially disturbing to eagles based on 1/4 restrictions for skiffs that produce 90 dB.

It is anticipated that oil/water/sediment wastes will be generated at a rate of up to one barrel per cubic yard of sediment processed. These wastes will include removed oil, oil sludges, oil/sediment complexes, suspended sediments and water. About one-half gallon per barrel of this waste will constitute removed oil.

These wastes would likely be transported to Valdez, Seward or Whittier ports for treatment (liquids) or for further processing and concentrating for ultimate disposal at a landfill (solids).

Each of these wastes have a potential to impact the environment as a result of spills or accidents causing contamination of beaches, surface waters, soil, and ground water with oil or fine sediment. See Section II.B.6 for a further discussion of potential waste impacts.

Fuel use and air emissions are also significant. For example, if all oiled subsurface sediments at Sleepy Bay were processed, a total of about 400,000 gallons of fuel would be used to run equipment, support vessels, and aircraft, and 300,000 pounds of regulated air pollutants would be emitted over the 4-6 week period required for processing. Waste generation, air emissions, and fuel consumption for Sleepy Bay (LA-16 through LA-20) are illustrated schematically below:

SEDIMENTS EXCAVATED	17,000 cubic yards
Fuel use	400,000 gallons
WASTES GENERATED	
Air pollutants	300,000 pounds
Wet skimmed oil	12,000 gallons
Wet, oily solid waste	500,000 gallons
Oily wastewater	168,000 gallons
Processed beach sediment	17,000 cubic yards

I.C. DESCRIPTION OF CANDIDATE BEACHES

I.C.1 SELECTION CRITERIA

Using the general criteria outlined above, members of the NEBA Committee developed a list of candidate beaches. The list was compiled using:

1) the SSAT database and search criteria on oil type, thickness, and depth;
2) the Alaska Department of Environmental Conservation (ADEC) beach profile database; and
3) personal observations of the committee members.

It was agreed that Point Helen and Sleepy Bay were the best candidates for the exposed outer beaches and moderate-energy shoreline with variable substrate, respectively (Figure I-3). Specific parts of each area were identified as study areas. However, the initial segment selected for the candidate for the sheltered, pocket beach, KN500, was subsequently rejected because of the presence of an active eagle nest at the site, which restricted all approaches. It was very important that the committee members be able to survey each site with rock washing issues in mind, so another candidate for the sheltered beach type was sought. Several recommended sites were evaluated and none was found to be adequate. After two weeks of effort to identify and field-check sites, it was decided to proceed with only two sites. During the two-week period, 22 sites were suggested, with none of them selected as an appropriate candidate site for the purposes of this study. Table I-1 lists the sites considered for the sheltered pocket beach candidate site (by segment number), the team who inspected or researched the site, and the reason for rejection. It should be noted that rejection of a site as a candidate beach for this study does not necessarily mean that it is not an appropriate site for rock washing.

In the following sections, each of the two candidate beaches is described in detail.

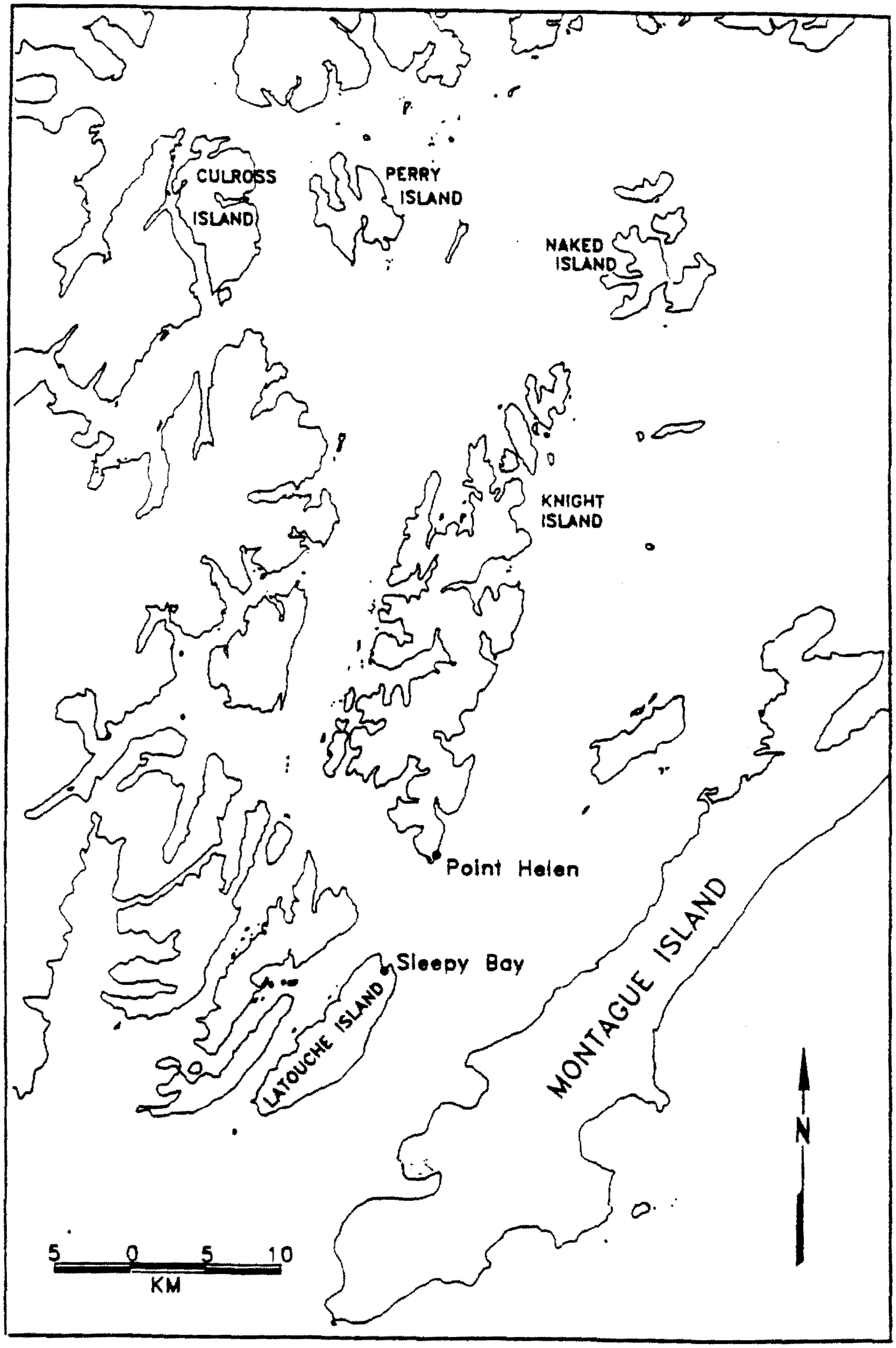


Figure I-3. Location map showing the two candidate beaches at Sleepy Bay and Point Helen.

Table I-1

Sites Considered for Inclusion as a Candidate Beach Representative of the Sheltered Pocket Beach Type for Rock Washing

Site	Rejected By	Reason for Rejection
AE-05	NOAA	Substrate
BP-16 (Marsh Lagoon)	Exxon	Lack of access
CR-05	NOAA	Shallow peat substrate
DI-62	ADNR	Lack of access
DI-67	ADNR	Insufficient oil
EB-11	Consensus	Insufficient oil
EL-10	NOAA	Unapplicable substrate
ER-20	Exxon	Eagle restriction
EL-56	NOAA	Insufficient oil
IN-20	Consensus	Insufficient oil
IN-22	Consensus	Applicable area very small; peat layer limits penetration
KN-113	Consensus	Insufficient oil
KN-121	Consensus	Lack of access
KN-132	NOAA	Unapplicable substrate
KN-134	NOAA	Oil not deep enough
KN-135	Consensus	Lack of oil penetration
KN-211	Consensus	Insufficient oil
KN-213	Exxon	High energy site
KN-500	Consensus	Eagle restriction
LA-17	Consensus	Not low energy
PR-16	Consensus	High/moderate energy Eagle restriction
SP-19	Consensus	Very small beach; limited depth, good bioremediation site

I.C.2. GEOMORPHOLOGY

Point Helen

Introduction. This beach is located on one of the more exposed sites in Prince William Sound. Consequently, it contains abundant coarse material—pebbles, cobbles and boulders—that shows signs of frequent transport by wave-generated currents (i.e., rounding and sorting). Due to uplift in the area on the order of 8 feet during the 1964 earthquake, the beach has not completely readjusted to an equilibrium profile. The specific part of Point Helen used as the candidate beach was 100m to the south and 200m to the north of NOAA Station N-1. Exxon's stations AP-9 and AP-10 are north and south, respectively, of the candidate beach section.

The Beach Profile. The field sketch in Figure I-4 and the profile in Figure I-5 show the morphology of the beach at Point Helen at NOAA's station N-1 on 24 May 1990, which is typical of the entire beach area under consideration in this report. There are three morphologically distinct components of the profile:

1) High-Tide Berms

The upper ±10m of the profile was host to a series of migrating spring-tide and storm berms between September 1989 and May 1990, as shown by data collected during NOAA's winter monitoring program (Advanced Technology, Inc. and Continental Shelf Associates, Inc., 1990). The finest surficial materials occurring on the beach (down to pebble size) are found in this area.

2) Stable Central Ramp

A cobble to boulder armor has formed over the surface of this central (10-25m) portion of the profile, which has shown almost no change over the past 9 months. The ratio of boulders to cobbles increases in a seaward direction.

3) Low-Tide Bar Zone

This zone, which extends from 25m to the low spring tide line (±60m), periodically contains asymmetric bars built by wave action called swash bars. These bars, which characteristically migrate toward the south, may attain heights up to 40 cm, as shown by the survey on

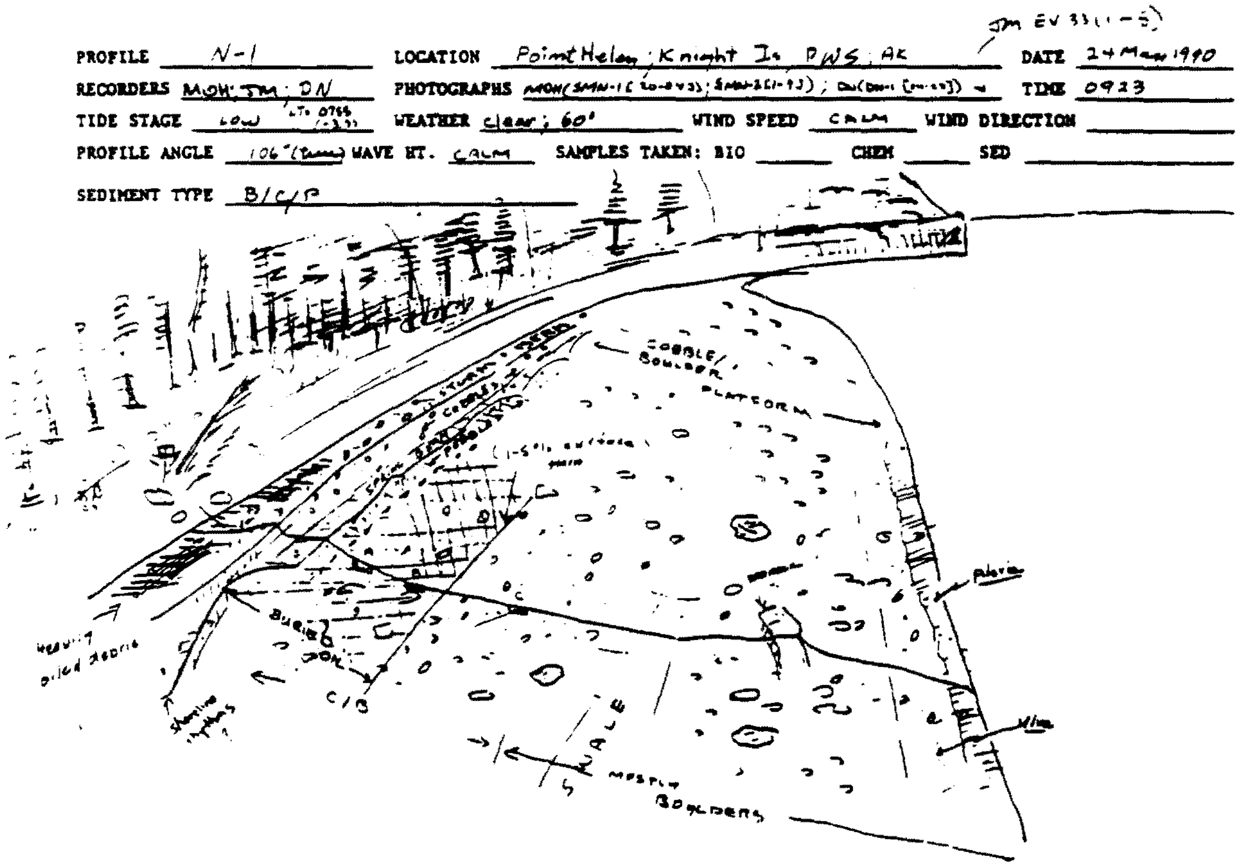


Figure I-4. Beach sketch of NOAA's station N-1 at Point Helen at low tide on 24 May, 1990. Note presence of storm and spring-tide berms near high-water mark and bedrock outcrop in lower portion of the profile. Buried oil occurs on the upper quarter of the beach.

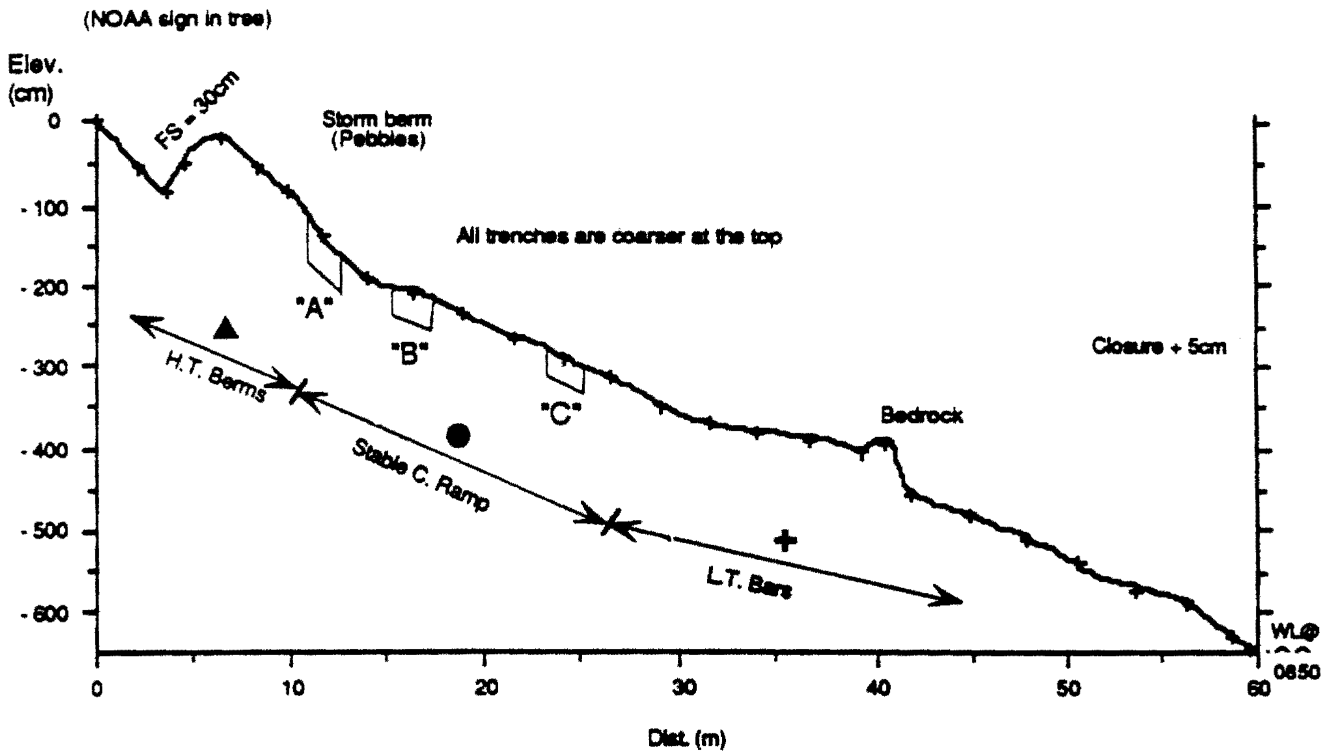


Figure 1-5. Beach profile at NOAA's station N-1 at Point Helen that was measured at low tide on 24 May, 1990. The beach is classified morphologically into three zones: (1) high-tide berms, an area subject to fluctuations of berm levels during storms and spring tides; (2) stable central ramp, bypass zone with an armor of cobbles and boulders; and (3) low-tide bars, boulder/cobble zone that sometimes contains gravel swash bars.

1 February 1990. The surface material is mostly boulders, but patches of bedrock also occur within this zone (Figs. I-4 and I-5).

Composition and Grain Size of Beach Clasts. - The pebbles, cobbles and boulders of Point Helen are no doubt derived locally, probably from outcrops exposed along the shoreline to the north. They are quite hard and do not crumble readily. The composition of the clasts is variable, with the following possibilities (in decreasing order of abundance):

1) Slightly metamorphosed deep-water sandstones.

2) Dark gray-black or reddish-brown hard siltstone and argillite.

3) Miscellaneous basaltic lava, metamorphosed conglomerate, and others.

NOAA's field team measured the detailed distribution of grain size along profile N-1, which is representative of the entire Point Helen area, on 24 May 1990. Careful estimates were made of relative abundance of clast type - pebbles (P), cobbles (C), boulders (B), and granule/sand (G/S) - at 25 evenly-spaced intervals along the profile. These results were plotted on the ternary diagram shown in Figure I-6. The distribution of clast sizes on the surface of the profile is shown on the map in Figure I-7. These diagrams indicate that the surface clasts of the high-tide berms are mixtures of cobbles and pebbles, the stable central ramp has a surface armor of cobbles and boulders, with cobbles predominating, and the low-tide bar zone is dominated by boulders, with cobbles typically making up 25-30% of the total.

Over the winter monitoring period, fourteen trenches were dug and described on this profile. Three were dug on 24 May 1990, and their descriptions are given in Figure I-8. These descriptions show that, in every case, an armor of coarse material overlies fine material at depth. The plot of the estimated grain sizes of the trench sediments on the ternary diagram in Figure I-9 emphasizes further the finer-grained nature of the deeper sediments (compare with Figure I-6).

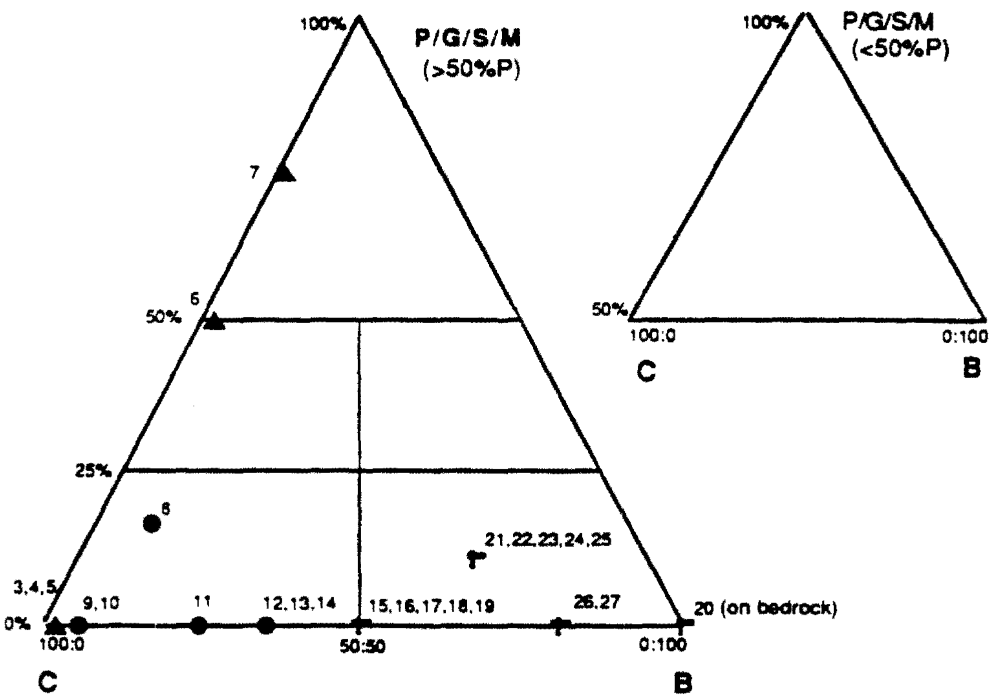
With regard to the rock-washing scenario, only the sediments in the zone between 12 and 21 m would be of interest, because that is where the buried oil occurs (see Figure I-10). The surface layer of sediments in that zone average: 6% B; 74% C; and 20% P. However, the subsurface

ZONAL GRAIN SIZE DESCRIPTON

PROFILE: N-1 LOCATION: POINT HELEN

DATE: 24 MAY '90

X PROFILE STATIONS -OR-
___ TRENCH DESCRIPTIONS



	PROCESS ZONES	PROFILE INTERVALS
▲	High-Tide Berms	3 - 8
●	Stable Central Ramp	9 - 15
✝	Low-Tide Bars	16 - 27

NOTES FOR VERBAL DESCRIPTION:

▲: Cobble storm berm with welded pebble spring berm.

●: Cobble ramp, becomes increasingly boulder-bearing seaward.

✝: Boulder platform, cobbly and pebbly, no relief in May but showing swash bar development earlier (e.g., Dec.'89 to Feb.'90).

Figure I-6. Plot of 25 estimates of grain size of surface sediments along NOAA's profile N-1 on 24 May 1990, with respect to relative amounts of boulders (B), cobbles (C), pebbles (P), granule (Gr), sand (S), and mud (M). The beach zones within which the estimates were made are indicated by symbols (▲, ●, ✝) and the numbers by the symbol represents the profile intervals at which the estimate was made. Profile intervals were usually 1.5-2.0 m apart. Note the clear segregation of size by beach zone, with a distinct increase in grain size in a seaward direction.

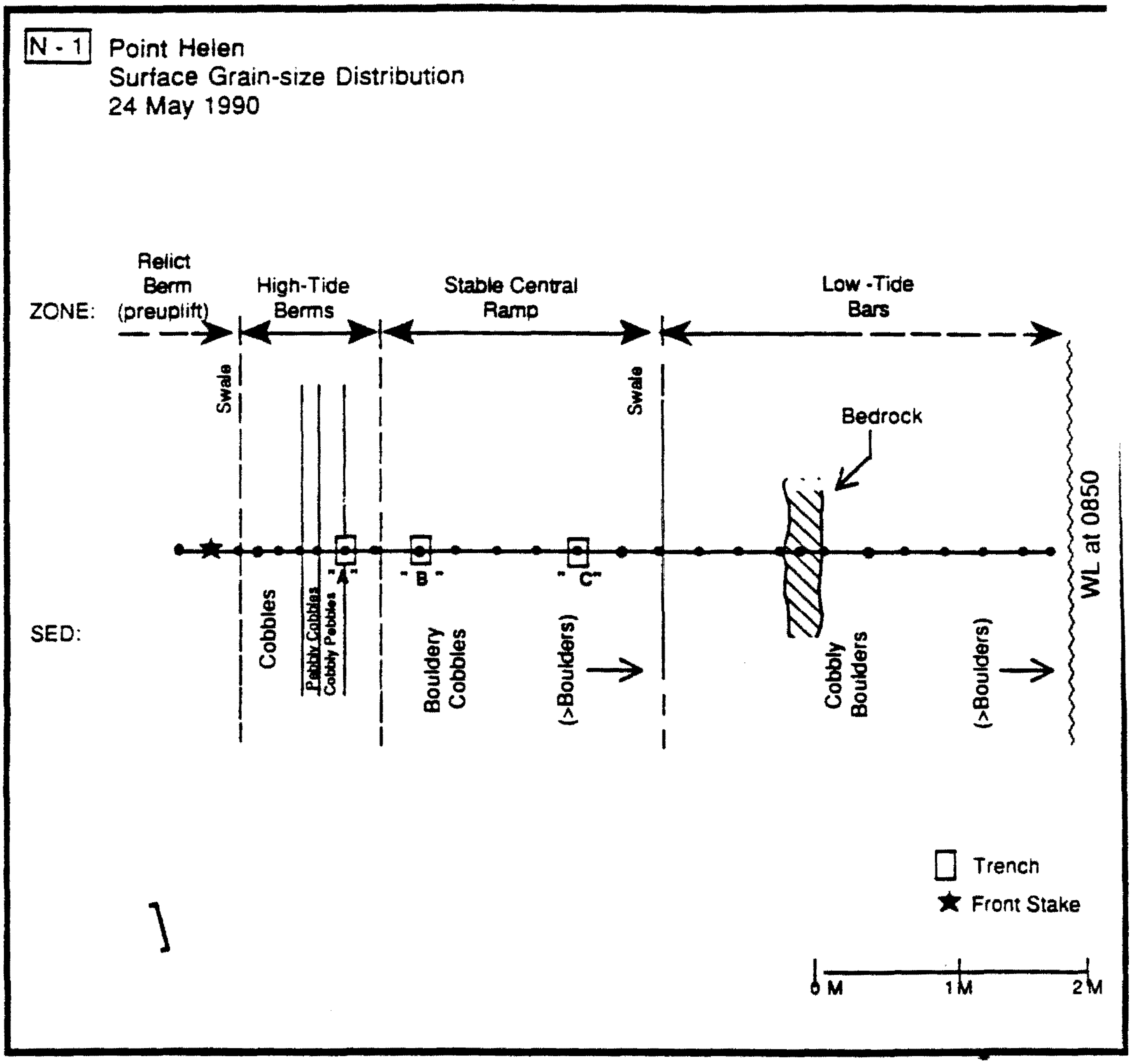


Figure I-7. Distribution of grain size of surface sediments adjacent to NOAA's profile N-1 on 24 May 1990. Dots along profile line indicate points where grain size estimates were made.

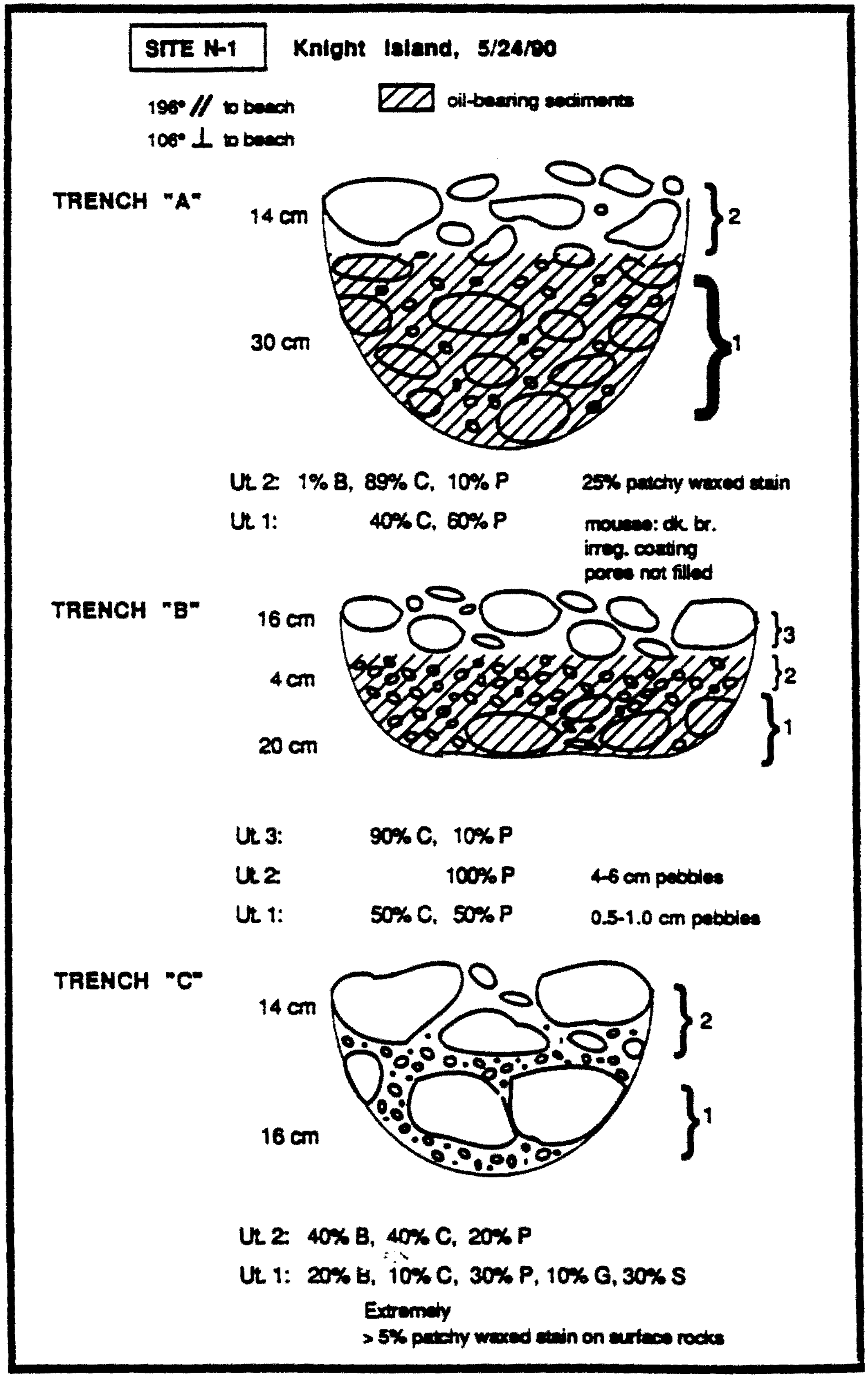


Figure I-8. Description of three trenches dug on NOAA's profile N-1 on 24 May 1990 (see Figs. I-4, I-5, and I-7 for location). Note tendency for upper units to be somewhat coarser grained than lower units. Also note that the upper sediments in trenches A and B are relatively free of oil as result of wave action during the non-summer months of 1989 and 1990.

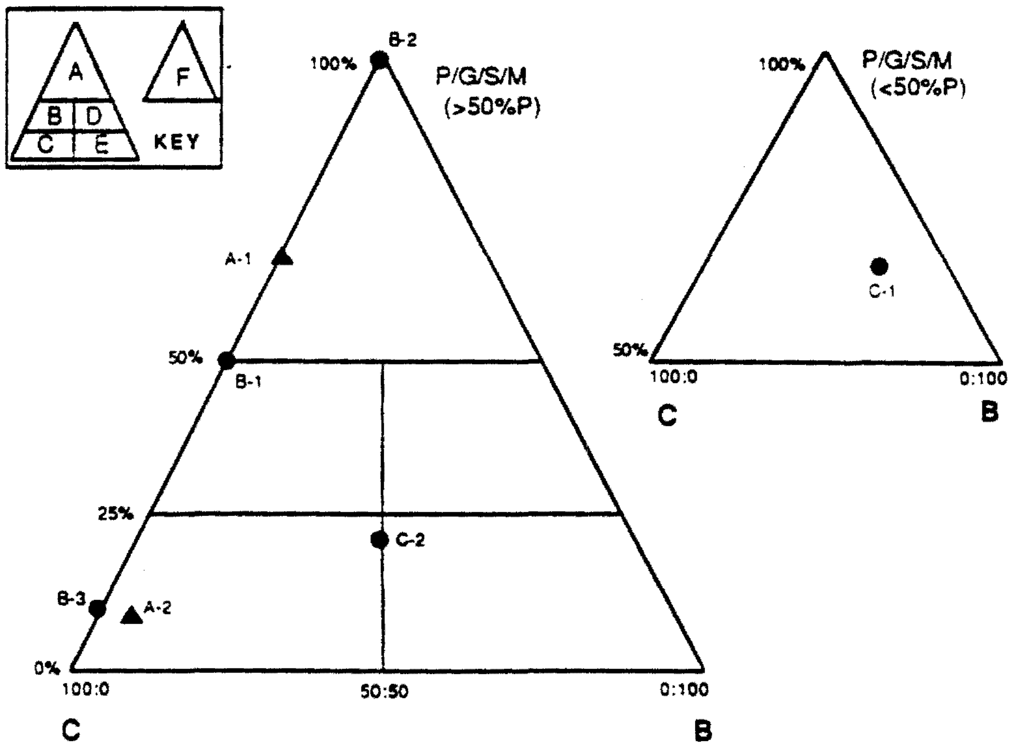
ZONAL GRAIN SIZE DESCRIPTON

PROFILE: N-1 LOCATION: PT. HELEN

DATE: 24 MAY '90

___ PROFILE STATIONS -OR-

_X_ TRENCH DESCRIPTIONS



	PROCESS ZONES	TRENCH
▲	High-Tide Berms	"A"
●	Stable Central Ramp	"B" & "C"
+	Low-Tide Bars	

NOTES FOR VERBAL DESCRIPTION:

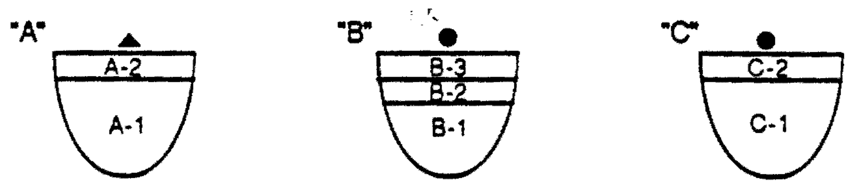


Figure I-9. Plot of grain size estimates for sediments from the trenches described in Figure I-8. These sediments are generally finer than the surface sediments for the entire profile (compare with Fig. I-6).

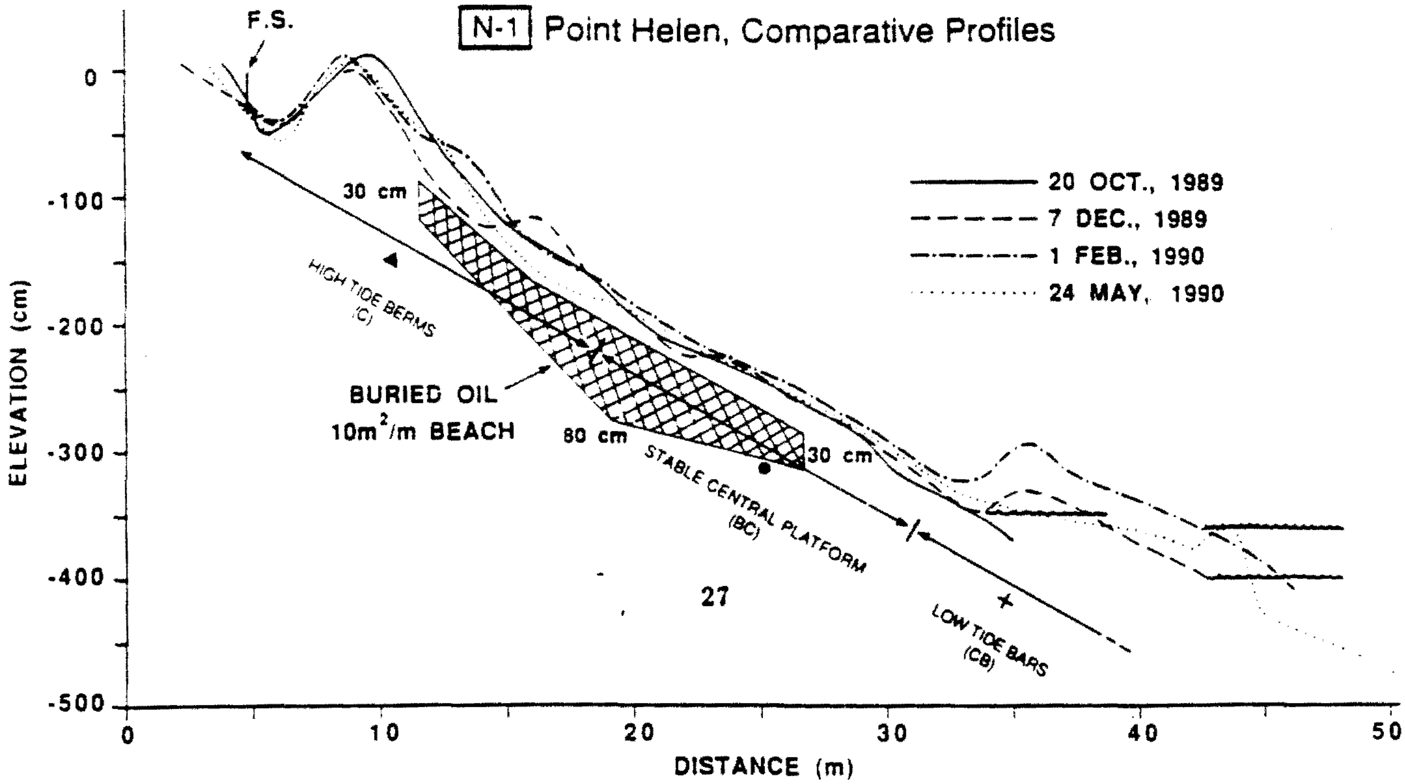


Figure I-10. Beach changes and buried oil at NOAA's profile N-1 at Point Helen. The plot of four profiles measured during the fall-winter-spring interval of 1989-90 reflects the morphological signature of the three designated beach zones: (1) migrating berms in the upper zone; (2) relative stability in the central zone; and (3) migrating bars in the lower zone.

sediments described in the trenches are considerably finer: 1% B; 58% C; and 41% P. If one assumes that in the rock washing process five times more material would be derived from the subsurface than from the surface, the sediments to be washed at Point Helen would have the following grain size:

Sediment Type	Percent	Average Size
Boulders	2	40 cm
Cobbles	61	13 cm
Pebbles	30	2 cm
Granule/Sand*	7	3 mm

Energy Level and Beach Dynamics. - This beach is exposed to the east, the direction from which the dominant winds blew during storms in the fall-winter-spring period of 1989-90, according to data gathered at NOAA's meteorological stations at Lone Tree and Danger islands. The effective fetch distance ranges from 15-20 km in a due easterly direction up to 45-50 km to the NNE. No seasonal measurements of wave or current conditions are available for the site.

As implied above, there is a significant amount of clast transport on this beach on a seasonal basis. The four seasonal profiles plotted on Figure I-10 illustrate the migration of berms in the high-tide portion of the beach and the presence of intermittent swash bars in the low-tide section (see photo in Figure I-11). Apparently, the stable central platform is a zone where finer clasts bypass over the coarse-grained surficial armor without disrupting it.

Sleepy Bay

Introduction. This gravel beach is located at the head of Sleepy Bay, an embayment on the north end of Latouche Island. It is similar to Point Helen in overall morphology, but the grain size of the mobile sediment is

* We assumed that the sediments designated P had a 20% matrix of granule/sand.

Figure I-11. Point Helen at low tide on 5 January, 1990. Note the presence of gravel swash bars in the intertidal zone that were obviously migrating in a southerly direction.

somewhat smaller because of its more sheltered location. The area of interest is bisected by a small, anadromous stream that is constantly shifting position as it builds a small delta. Due to uplift in the area on the order of 8-10 feet during the 1964 earthquake, the beach has not completely readjusted to an equilibrium profile. The specific area used as the candidate beach is shown in Figure I-12. This area includes NOAA's station N-18 and Exxon's station AP-13.

The Beach Profile. As shown on the map in Figure I-13, the NOAA team surveyed three profiles at the study site in May 1990. In addition, NOAA's permanent station N-18 was surveyed seven times between September 1989 and May 1990.

Exxon and ADEC also had a number of permanent profile sites in this area; thus, the dynamic changes of the profile of the beaches at this site are well documented.

NOAA's profile N-18, illustrated in Figures I-14 and I-15, serves as a basis for the following description:

1) High-Tide Berms

The upper ±10m of the profile typically consists of a number of storm and spring-tide berms, composed mostly of pebbles and cobbles. The berms at the top of the beach were deposited after the beach was flattened during a period of large waves and high tides in October 1989.

2) Stable Central Ramp

Just as at Point Helen, the central portion of the profile has shown little change over the fall-winter-spring interval of 1989-90, except for possibly a small degree of overall lowering. Mixtures of cobbles and pebbles, oriented in bands parallel with the shoreline, cover the entire surface of this zone.

3) Low-Tide Bars

Swash bars showing a range in grain size from sand to cobbles have been observed on the lower portion of the beaches in this area during both the Exxon and NOAA surveys. Many of these bars were associated with the delta at the stream mouth.

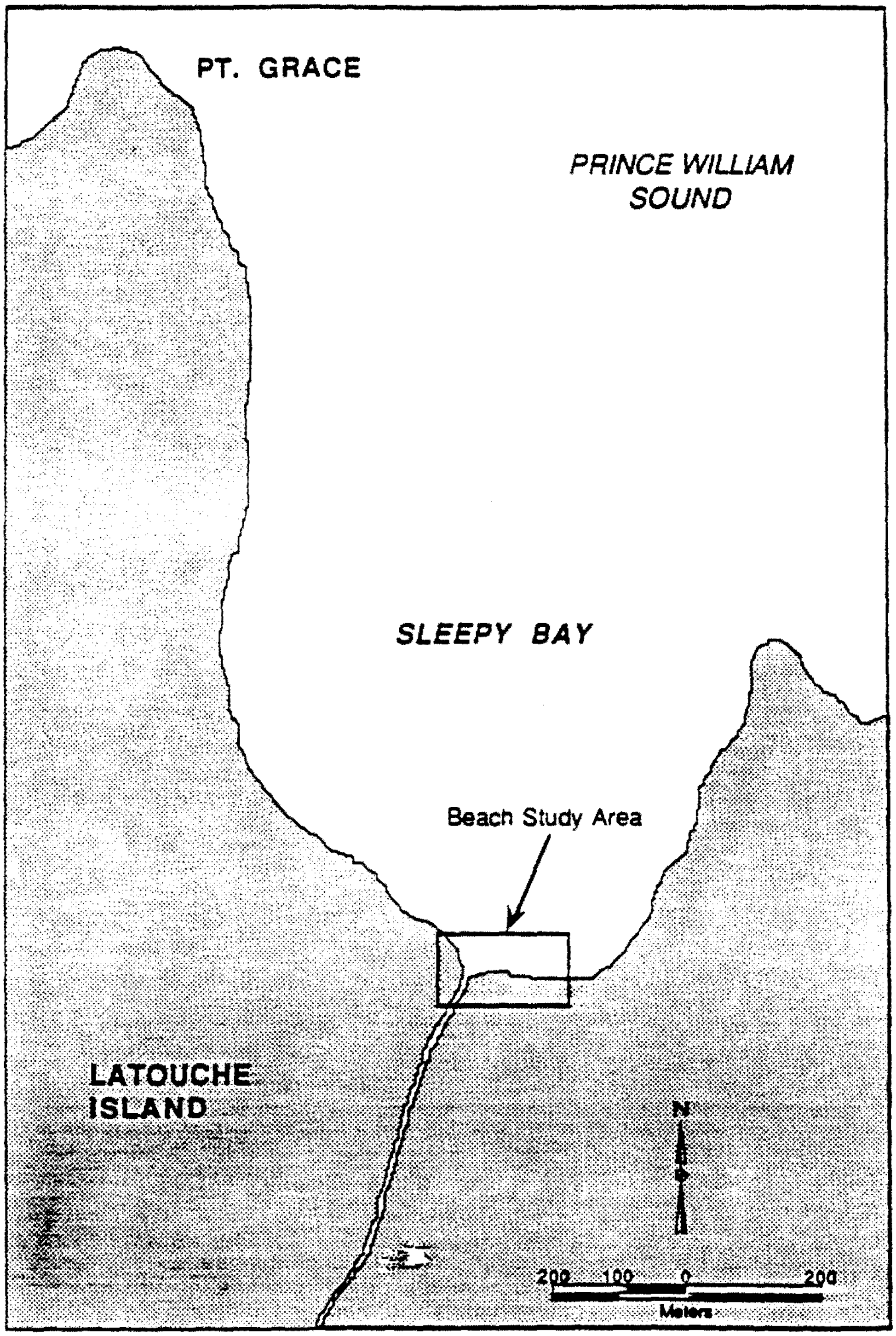


Figure I-12. Map of Sleepy Bay showing the location of the specific section of Sleepy Bay used as a candidate beach for this analysis.

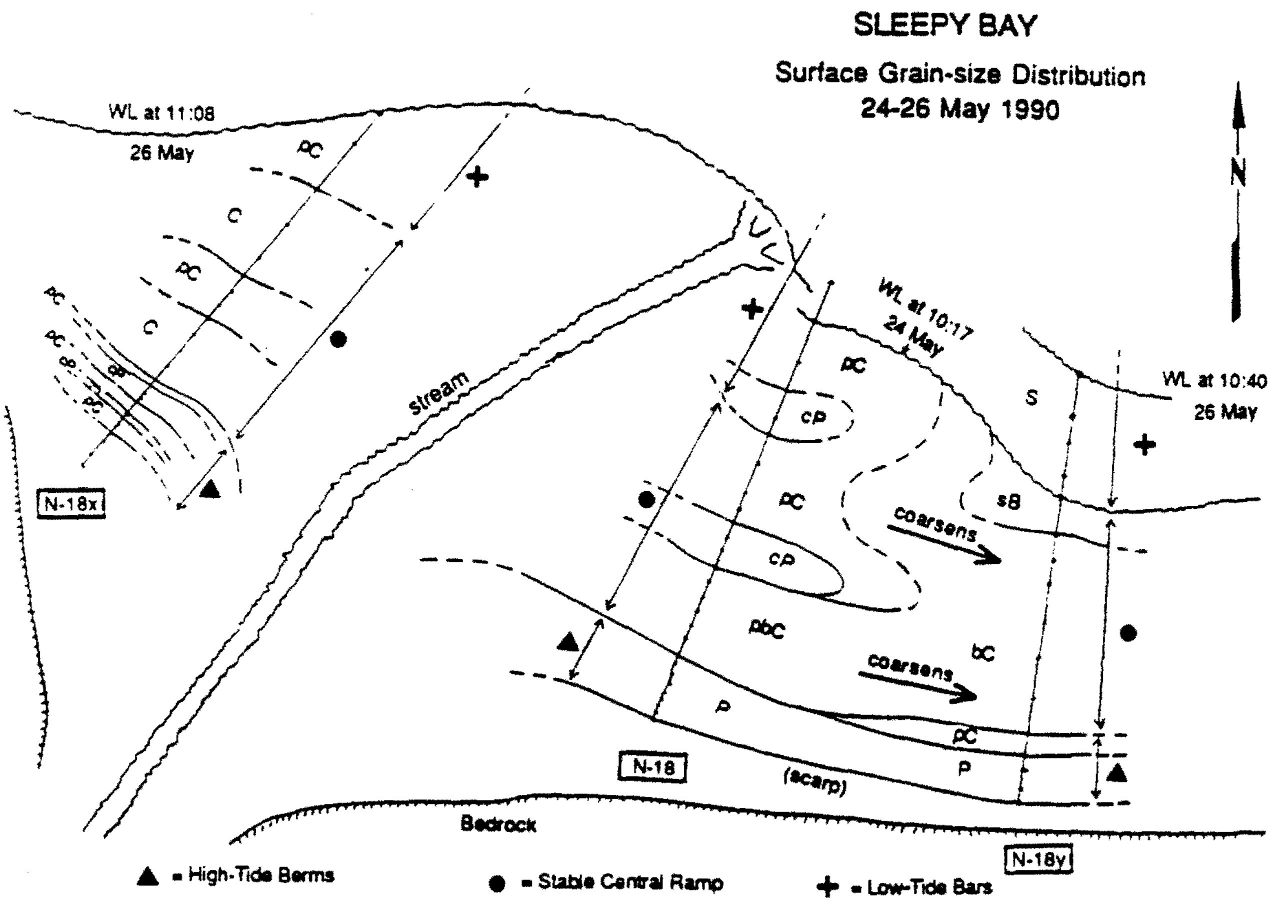


Figure I-13. Surface sediment distribution map at the head of Sleepy Bay on 24-26 May, 1990. Map is based on grain size estimates made along three beach profiles established by NOAA (N-18; N-18x; N-18y). The sediments are mostly parallel bands of different mixtures of pebbles and cobbles. The river mouth migrates constantly, thus these patterns will change in the future.

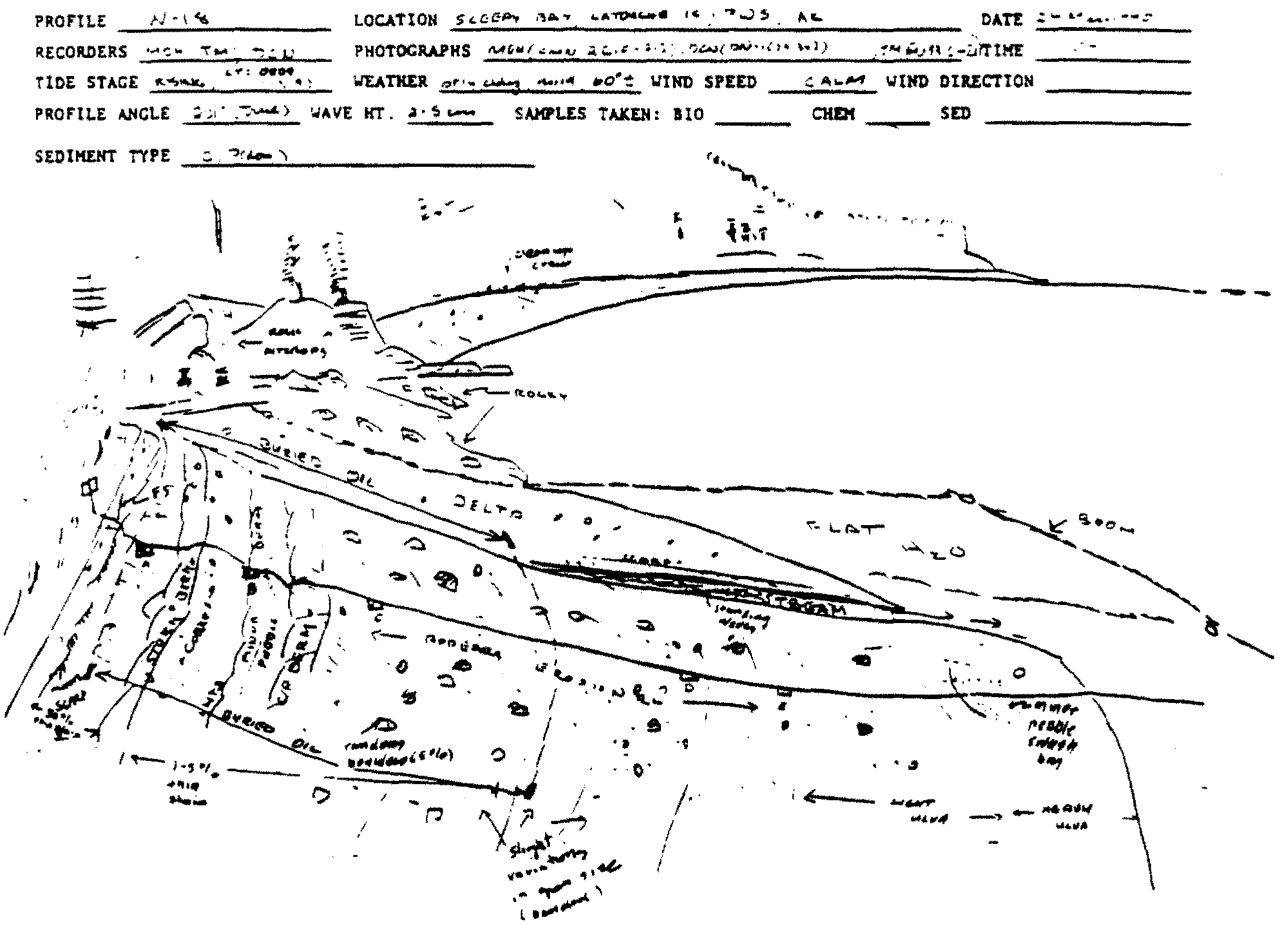


Figure I-14. Field sketch of NOAA's profile N-18 at Sleepy Bay at low tide on 24 May, 1990. Note the presence of multiple berms in the high tide area. Buried oil occurs in the upper third of the profile.

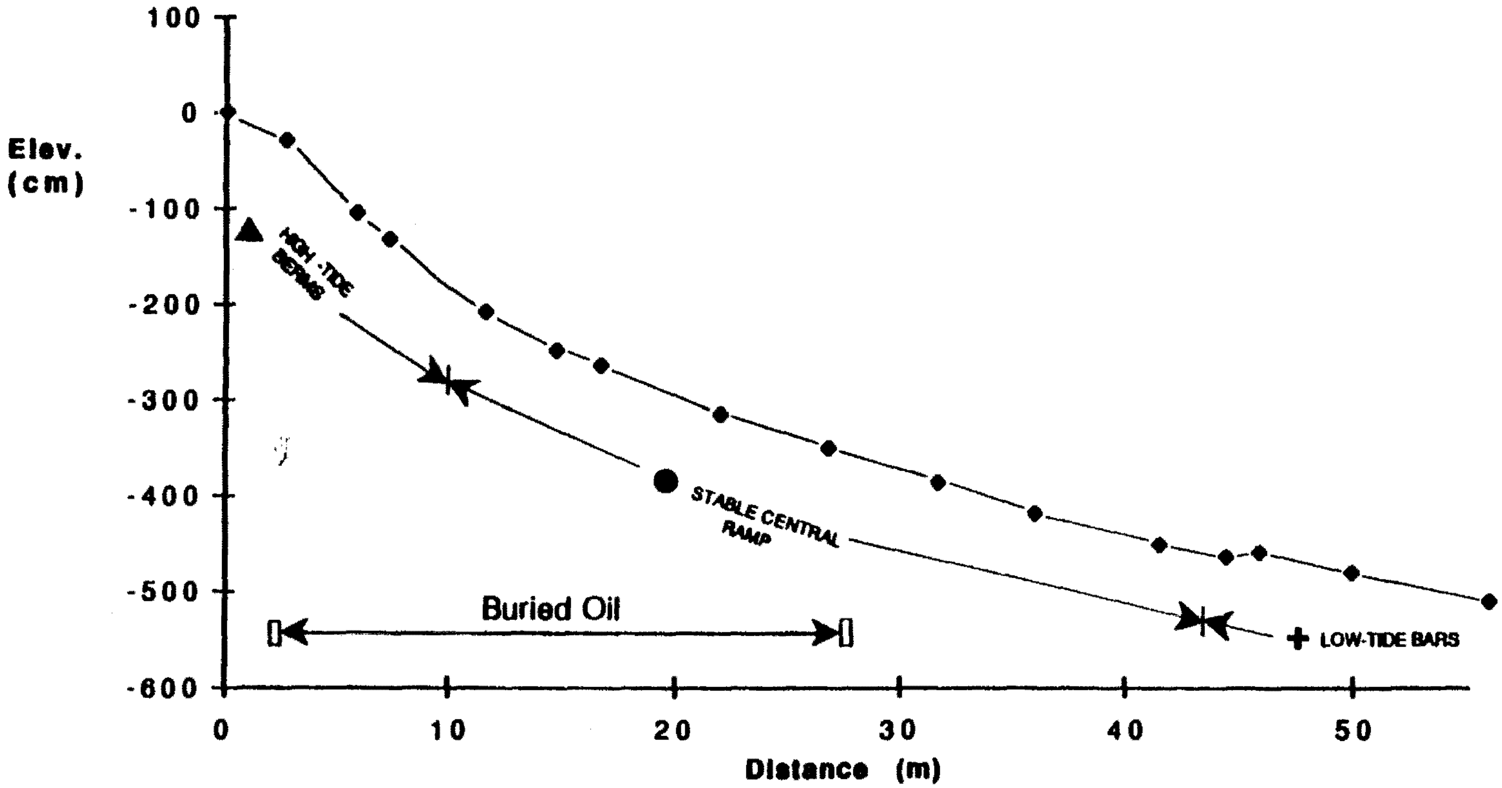


Figure I-15. Plot of NOAA's beach profile N-18 as it appeared on 1 February, 1990. During the fall-winter-spring of 1989-1990, the upper part of the profile was host to multiple migrating berms, the central part was relatively stable, and the lower part contained swash bars from time to time, some of which were associated with the river delta.

Composition and Grain Size of Beach Clasts. The sediments on the beaches at the head of Sleepy Bay are derived from two sources: (1) the rocky headlands located within and to either side of the bay; and (2) the stream and its delta. The coarse clasts, which are quite hard and resistant to breakage, have the same wide variety of composition that is present at the Point Helen site. Dark-colored metamorphosed sandstone, shale, and basaltic lava are common, as well as several other rock types.

A map of the distribution of grain size of the surficial beach materials at the head of Sleepy Bay is presented in Figure I-13. Note that most of the surficial sediments at the site are composed of pebbles and cobbles. Boulders are rare, except to the east of profile N-18y. A plot of 26 grain size estimates for the surficial sediments along NOAA's profile N-18 is given on the ternary diagram in Figure I-16.

Over 30 trenches have been dug by the NOAA team at this site. Two typical trenches from the oiled sector that were dug on 24 May 1990 are shown in Figure I-17. Armoring is relatively poorly developed at this site compared to many others in the Sound, the sediments at depth being only slightly finer than the surface layer (except for the absence of boulders at depth). This probably is a function of the finer-grained, more mobile character of the sediment. However, this beach sediment was mixed significantly during cleanup last summer, and it is possible that there has not been enough time for the armoring process to have matured.

If rock washing were to be carried out at this site, the focus would be on the buried oil, which is located between 3-28m along the profile. The surface layer of sediments in that zone average: 9% B; 40% C; 49% P; and 2% G/S. However, the subsurface sediments described in the trenches are somewhat finer: 1% B; 26% C; and 73% P. If one assumes that in the rock washing process five times more material would be derived from the subsurface than from the surface, the sediments to be washed at Sleepy Bay would have the following grain size:

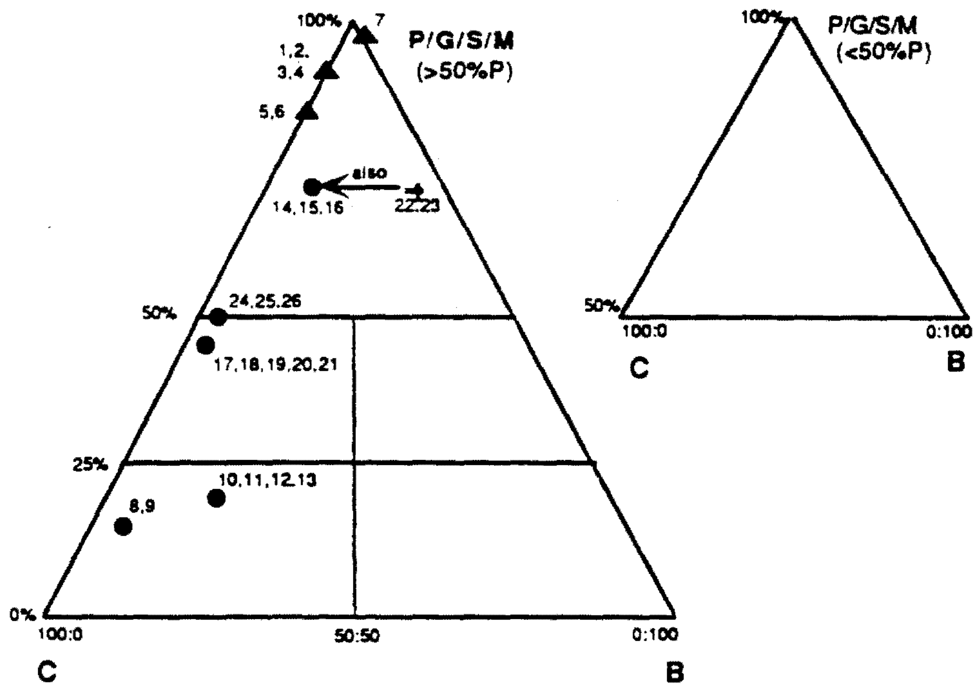
ZONAL GRAIN SIZE DESCRIPTON

PROFILE: N-18　　LOCATION: Sleepy Bay

DATE: 24 MAY '90

X PROFILE STATIONS -OR-

__ TRENCH DESCRIPTIONS



	PROCESS ZONES	PROFILE INTERVALS
▲	High-Tide Berms	1 - 7
●	Stable Central Ramp	8 - 21
┿	Low-Tide Bars	22 - 26

NOTES FOR VERBAL DESCRIPTION:

▲: Pebble berm.

● : Pebbly cobbles alternating with [illegible] pebbles; some boulders, especially higher on ramp.

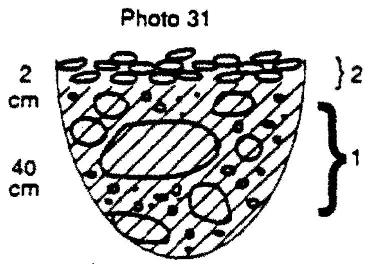
┿: Pebbly swash bar over pebbly cobbles.

Figure I-16. Plot of estimates of the grain size of the surface sediments at 26 intervals along NOAA's profile N-18 on 24 May 1990. Note that mixtures of pebbles and cobbles predominate.

SITE N-18 Sleepy Bay, 24 MAY 1990

oil bearing sediments

TRENCH "B"

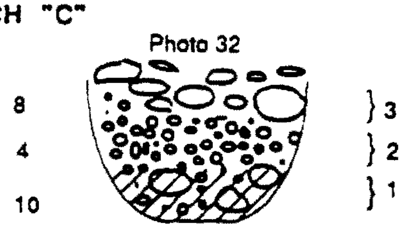


Ut. 2 70 P, 30 Gr V. discoidal

Ut. 1 5B, 40 C, 40 P, 15 Gr

Ut. 1 Coated with med. brn. mouse, not saturated
Ut. 2 clean
scattered spruce needles in Ut. 1.

TRENCH "C"



Ut. 3 50C, 50P clean

Ut. 2 50P, 40Gr, 10S clean, P's mostly <2cm

Ut. 1 10C, 50P, 40Gr mousse

V. dk. brn. mousse coating in Ut. 1, not saturated

Figure I-17. Description of sediments in two trenches dug on NOAA's profile N-18 on 24 May 1990. See Figure I-14 for location. Surface armoring is not as well developed in these trenches as it is elsewhere in Prince William Sound.

sediments described in the trenches are considerably finer: 1% B; 58% C; and 41% P. If one assumes that in the rock washing process five times more material would be derived from the subsurface than from the surface, the sediments to be washed at Point Helen would have the following grain size:

Sediment Type	Percent	Average Size
Boulders	2	40cm
Cobbles	61	13cm
Pebbles	30	2cm
Granule/Sand*	7	3mm

Energy Level and Beach Dynamics. - This beach is exposed to the east, the direction from which the dominant winds blew during storms in the fall-winter-spring period of 1989-90, according to data gathered at NOAA's meteorological stations at Lone Tree and Danger islands. The effective fetch distance ranges from 15-20 km in a due easterly direction up to 45-50 km to the NNE. No seasonal measurements of wave or current conditions are available for the site.

As implied above, there is a significant amount of clast transport on this beach on a seasonal basis. The four seasonal profiles plotted on Figure I-10 illustrate the migration of berms in the high-tide portion of the beach and the presence of intermittent swash bars in the low-tide section (see photo in Figure I-11). Apparently, the stable central platform is a zone where finer clasts bypass over the coarse-grained surficial armor without disrupting it.

Sleepy Bay

Introduction. This gravel beach is located at the head of Sleepy Bay, an embayment on the north end of Latouche Island. It is similar to Point Helen in overall morphology, but the grain size of the mobile sediment is

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* We assumed that the sediments designated P had a 20% matrix of granule/sand.

I.C.3. SEDIMENT OIL CONTENT AND COMPOSITION OF CANDIDATE BEACHES

Point Helen - Oil Content

There are several sources of data on the oil content of sediments along Point Helen. NOAA has a monitoring station about 1,500 meters north of the point. This station was occupied monthly from September 1989 to February 1990, and in May 1990. During these surveys, 37 sediment samples were collected and analyzed for TPH by both weight and volume. The measured concentrations of oil in these samples should be reviewed with caution because the substrate is very heterogeneous in both size and oil content. The samples represent only the finer-grained components since only sediment pebble-sized and smaller can fit into the sampling containers. Because the mass of these large-grained sediments is so large compared to their surface area, the traditional measure of oil contamination in milligrams per kilogram (mg/kg) by weight is not very meaningful. Measuring the oil content of the sediments by volume is a better approach, but it still suffers from disturbance of the sediment packing (thus volume) and poor representation of the entire substrate.

Figures I-18 through I-20 show the sediment analyses for samples collected along the NOAA monitoring beach profile at Point Helen for each month. The values shown are TPH by weight. The values shown on the profile line are surface samples; the samples shown below the line are placed at the depth of sampling. Although there are wide variations, several distinct trends are discernable. The oil was heaviest along the upper one-third of the intertidal zone, covering a width of about 12m. The remainder of the intertidal zone had concentrations in the 100-200 mg/kg range. The oil content of the surface sediments, down to about 30cm, was significantly reduced after September, with all samples except one below 300 mg/kg. Visually, the surface sediments at Point Helen appeared cleaner after the first few fall storms. Surface oil remains as a stain on the cobbles and boulders with heavier coating on the back side of boulders sheltered from waves.

The degree of oil contamination of the subsurface sediments has also visually improved. In the fall, sediments in trenches were described as

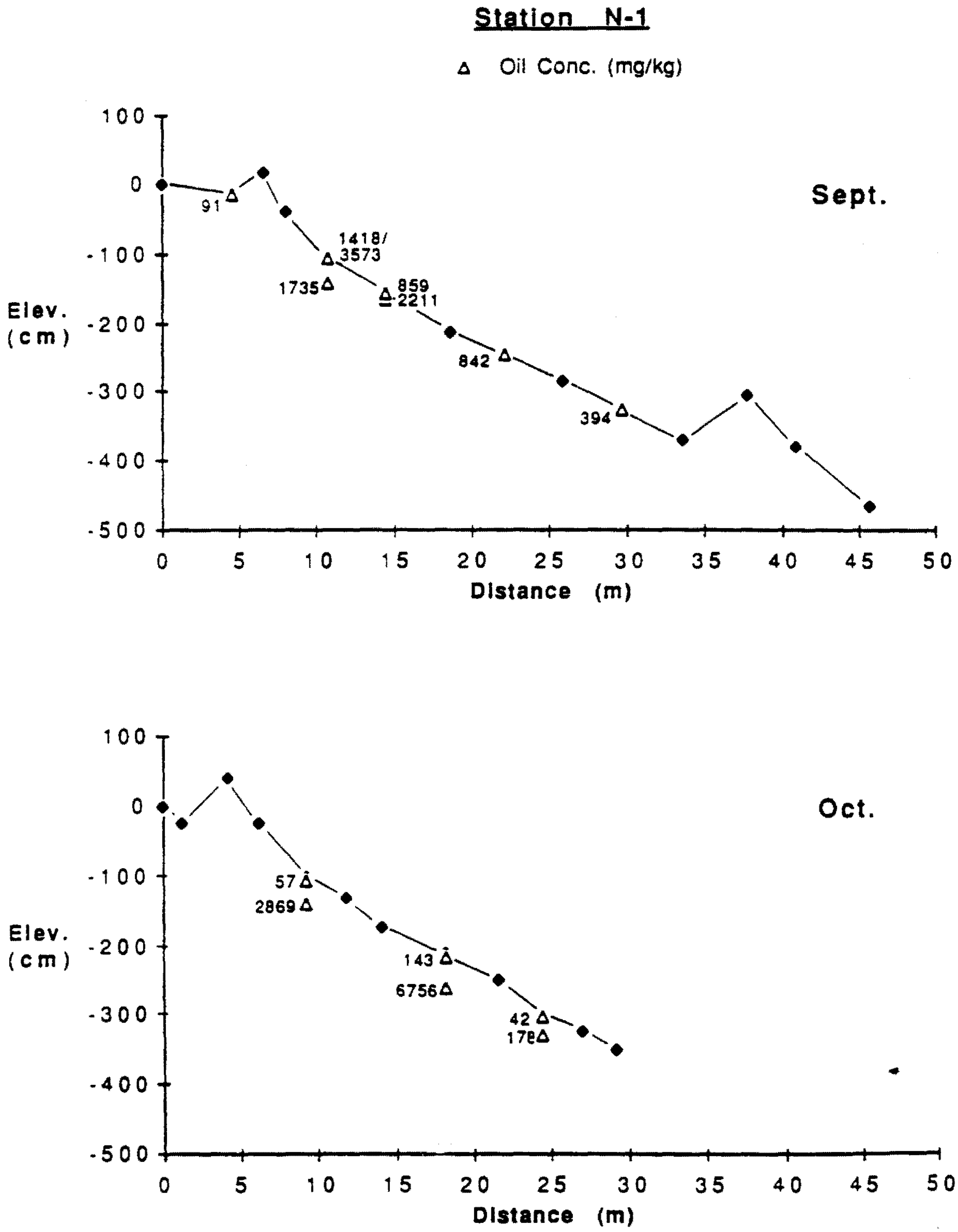

Figure I-18. Plots of the shoreline profile for Sept. and Oct., 1989, at NOAA Station N-1, Point Helen. Oil concentrations at the depths are shown in mg/kg.

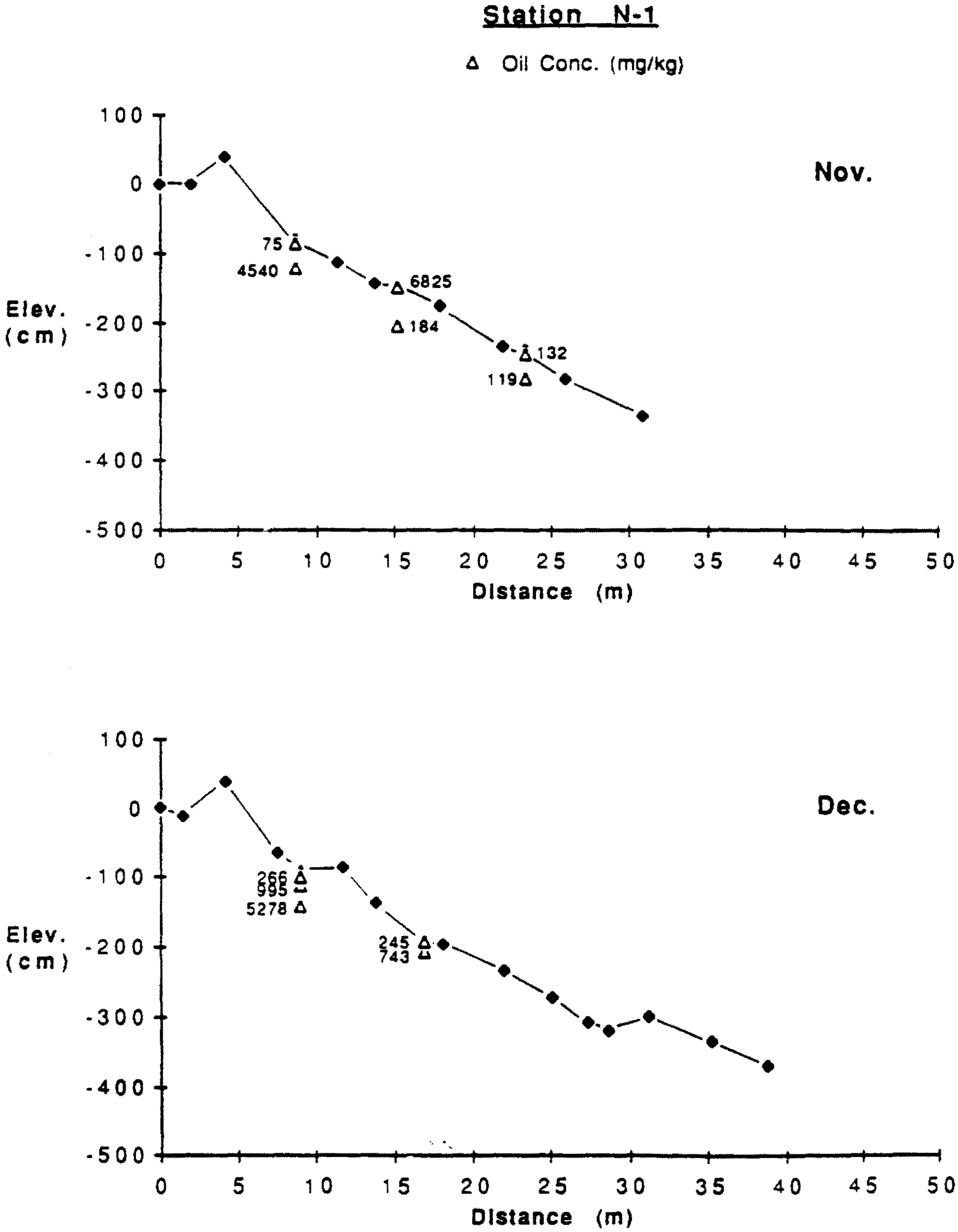

Figure I-19. Plots of the shoreline profile for Nov. and Dec., 1989, at NOAA Station N-1, Point Helen. Oil concentrations at the depths are shown in mg/kg.

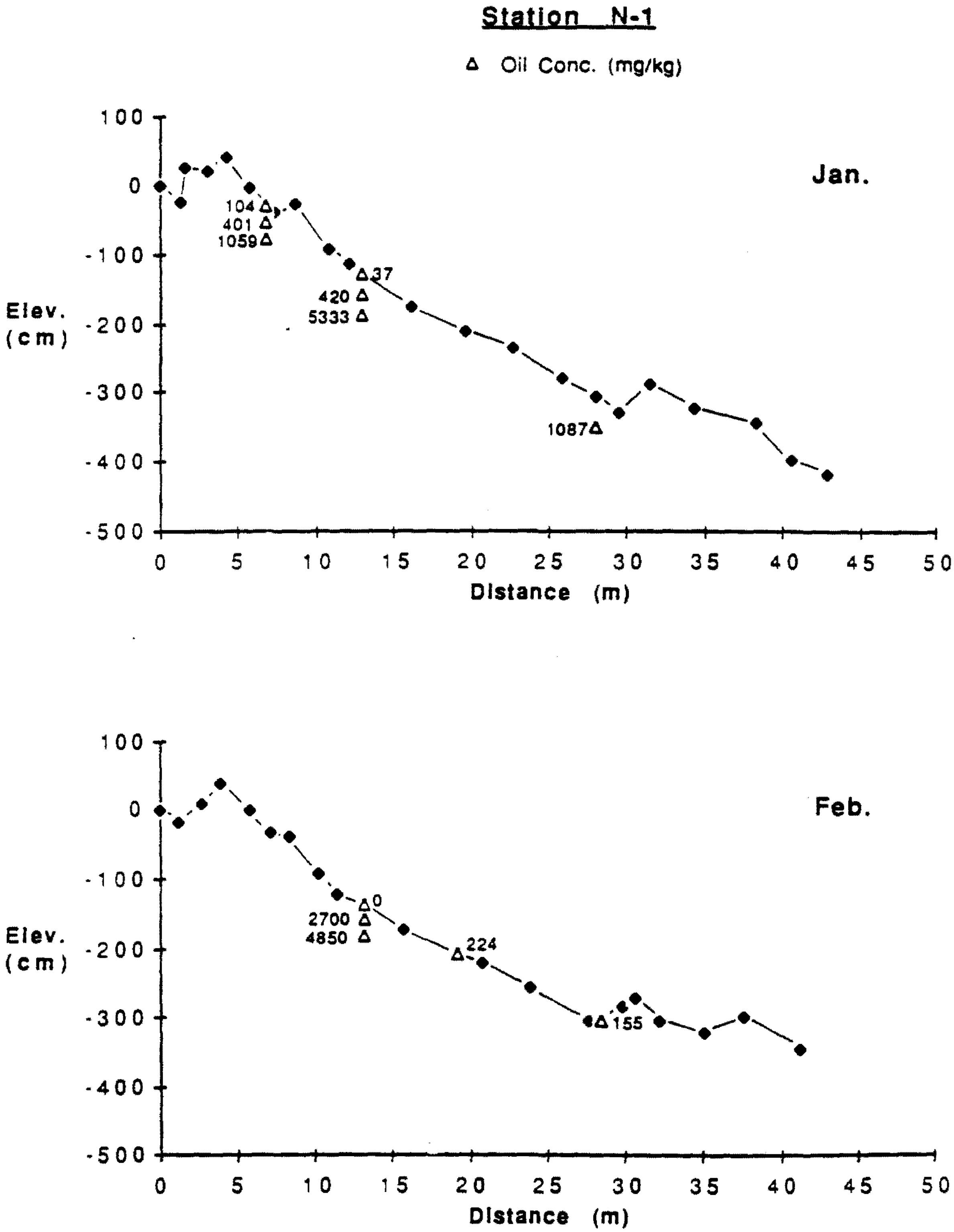

Figure I-20. Plots of the shoreline profile for Jan. and Feb., 1990, at NOAA Station N-1, Point Helen. Oil concentrations at the depths are shown in mg/kg.

heavily coated with liquid, black oil which floated on the water table. By spring, the oil was described as light to moderate. The average oil content of samples collected at depths greater than 30cm from the upper third of the beach ranged between 2,000 and 3,500 mg/kg, with no distinct trend over time. Large variations in the oil content of subsurface sediments are expected, so it will be difficult to rely upon analytical measures. Visual observations are very important, and visually there has been a significant reduction in the amount of oil in subsurface sediments, even below 30cm.

An estimate of the volume of oil in the candidate beach at Point Helen is based on the following assumptions:

1) Width of subsurface oil =12m, based on a maximum extent of >1,000 ppm by weight in any samples collected from NOAA's station between September 1989 and February 1990.

2) Depth of subsurface oil =100cm (average maximum)

   50cm (average at the edges)

   Top 20cm are "clean."

3) Oil content of subsurface sediments is determined from the following percent oil by by volume measurements of NOAA samples over the winter:

Month	Maximum %	Mean %
Feb	1.1	0.85
Jan	1.1	0.4
Dec	1.0	0.5
Nov	0.78	0.78
Oct	1.1	0.82
Mean	1.0	0.67 = 0.5%

These values are for the smaller components of the beach sediment (granule to pebble). According to grain-size descriptions of trenches at N-1 at Point Helen in May 1990, the subsurface sediments are about 50 percent cobble or greater and 50 percent pebble/granule/sand. It is assumed that the volumetric amount of oil on a cobble/boulder is negligible.

The cross-sectional area of oiled sediments is calculated as:

12 m x 1 m =	12 m^2	
	- 2.4 m^2	(reduction due to the clean top 20 cm)
	- 3.0 m^2	(due to thinner depth of oil at edges)
	6.6 m^2	oiled sediment per linear m of beach

The oil volume per linear meter of beach is calculated as:

6.6 m^2 x 0.5% oil by volume x 1m beach length x 0.5 (%<cobbles)
= 0.016 m^3 or 4.3 gallons per linear meter of beach

ADEC performed similar calculations based on an oiled zone at 30-120cm depth and estimated 6.2 gallons per meter of beach.

The area of Point Helen selected for review is 300m long, thus that shoreline is estimated to contain 1,290 gallons of oil. Assuming that the returned sediments still contain 800 ppm oil by weight (or an estimated 0.1 percent by volume), washing will remove approximately 1,030 gallons from this 300m stretch of shoreline. For all the area of Point Helen which has subsurface oil (an estimated 2,500m length), washing would remove an estimated 8,700 gallons from the subsurface sediments.

Sleepy Bay - Oil Content

There are three sources of data on subsurface oil volume at Sleepy Bay: NOAA's winter monitoring program; Exxon's March 1990 survey; and a June 1990 survey by Exxon. The oil distribution on the surface and in the subsurface is highly variable in Sleepy Bay, thus there were different results for each of the programs monitoring the degree and changes in oiled sediments. NOAA's monitoring station is located 50m to the east of the major stream. It has been occupied monthly between September 1989 and February 1990 and again in May 1990. A total of 43 sediment samples have been analyzed for TPH from this station, and the results are shown in Figures I-21 through I-23. The zone of subsurface oil is about 25m, wider than Point Helen. The oil concentrations in the surface sediments have been reduced from about 1,000 ppm in the fall to 100 ppm in the late winter.

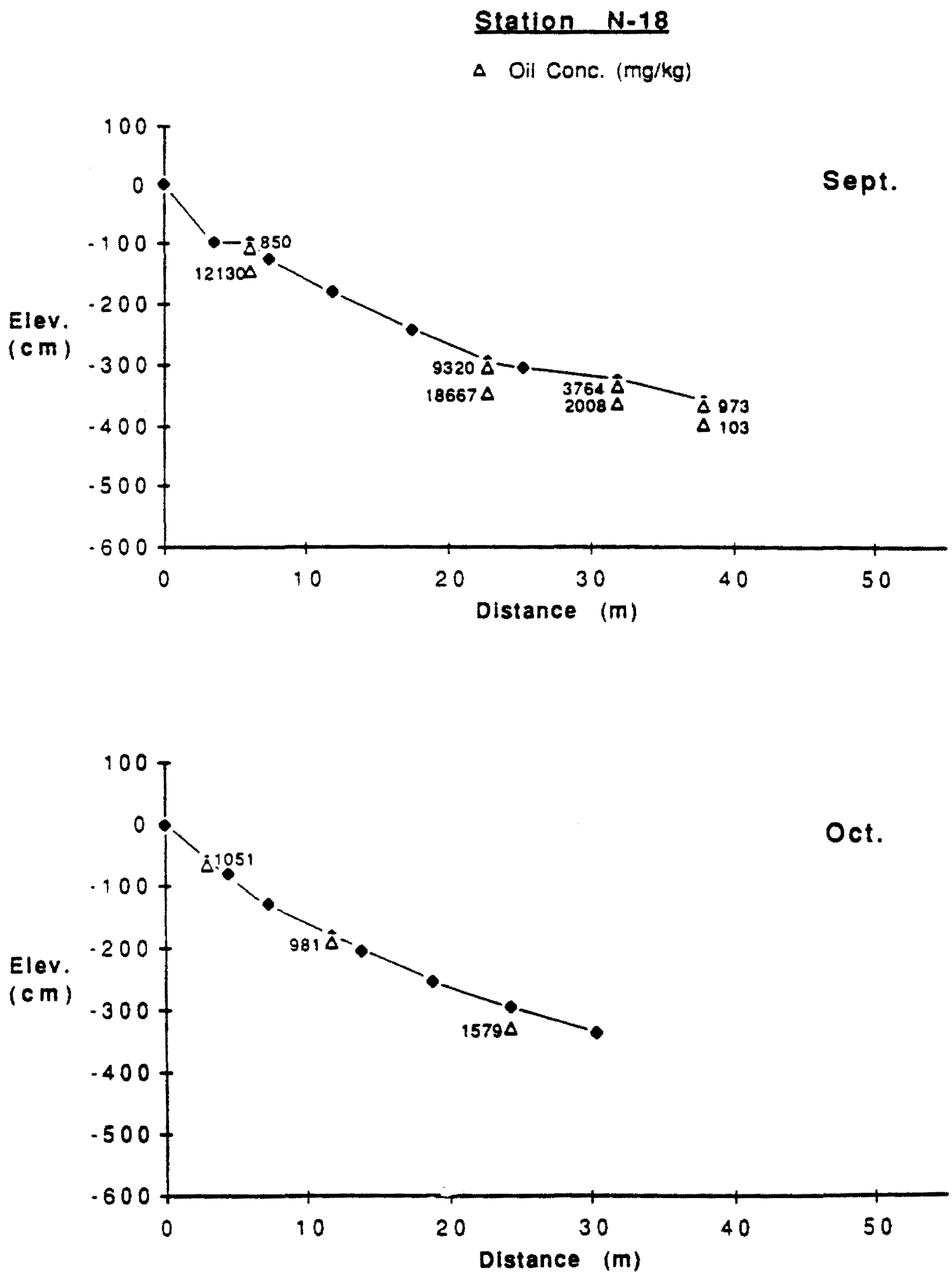

Figure I-21. Plots of the shoreline profile for Sept. and Oct., 1989, at NOAA Station N-18, Sleepy Bay. Oil concentrations at the depths are shown in mg/kg.

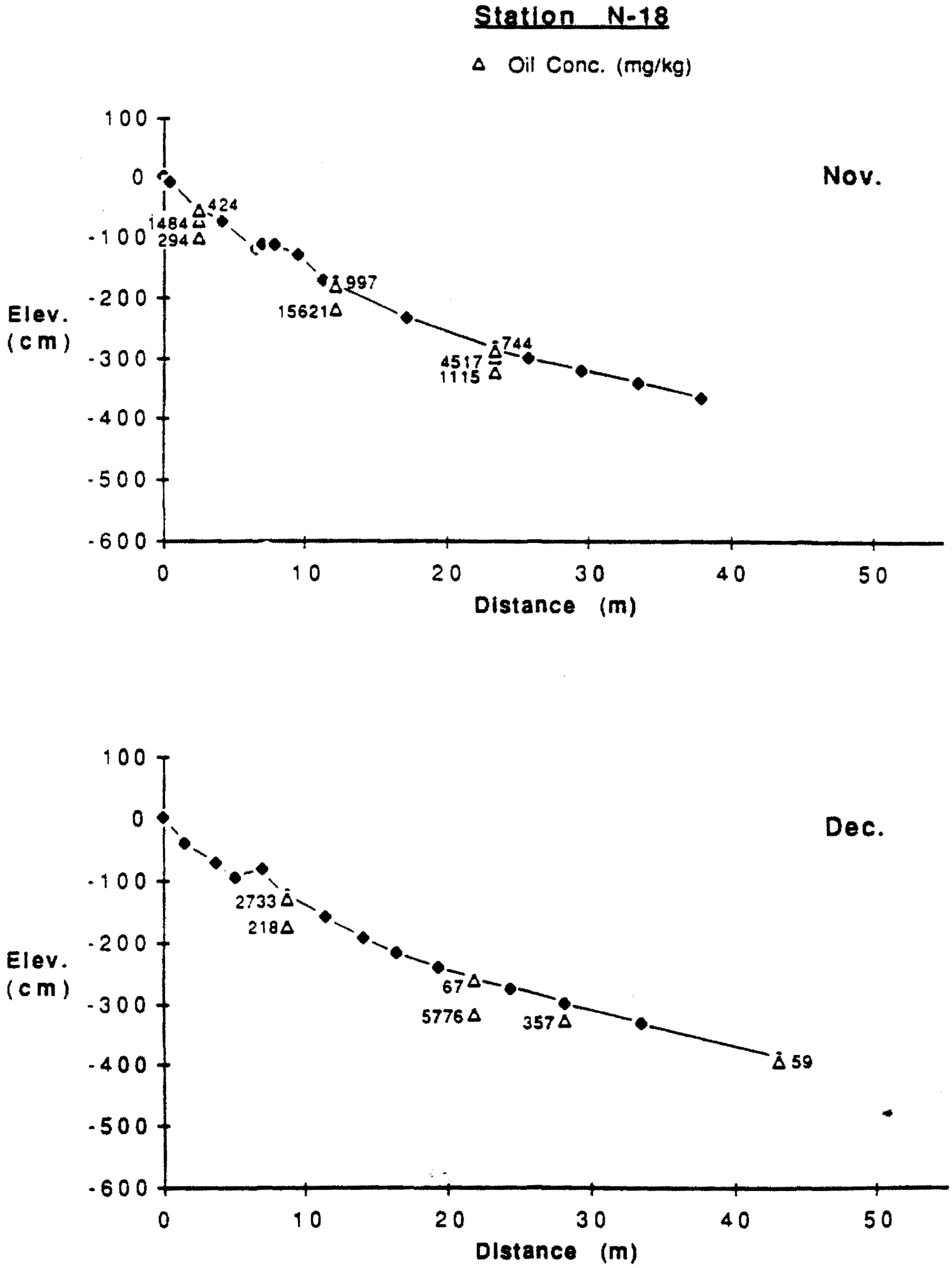

Figure I-22. Plots of the shoreline profile for Nov. and Dec., 1989, at NOAA Station N-18, Sleepy Bay. Oil concentrations at the depths are shown in mg/kg.

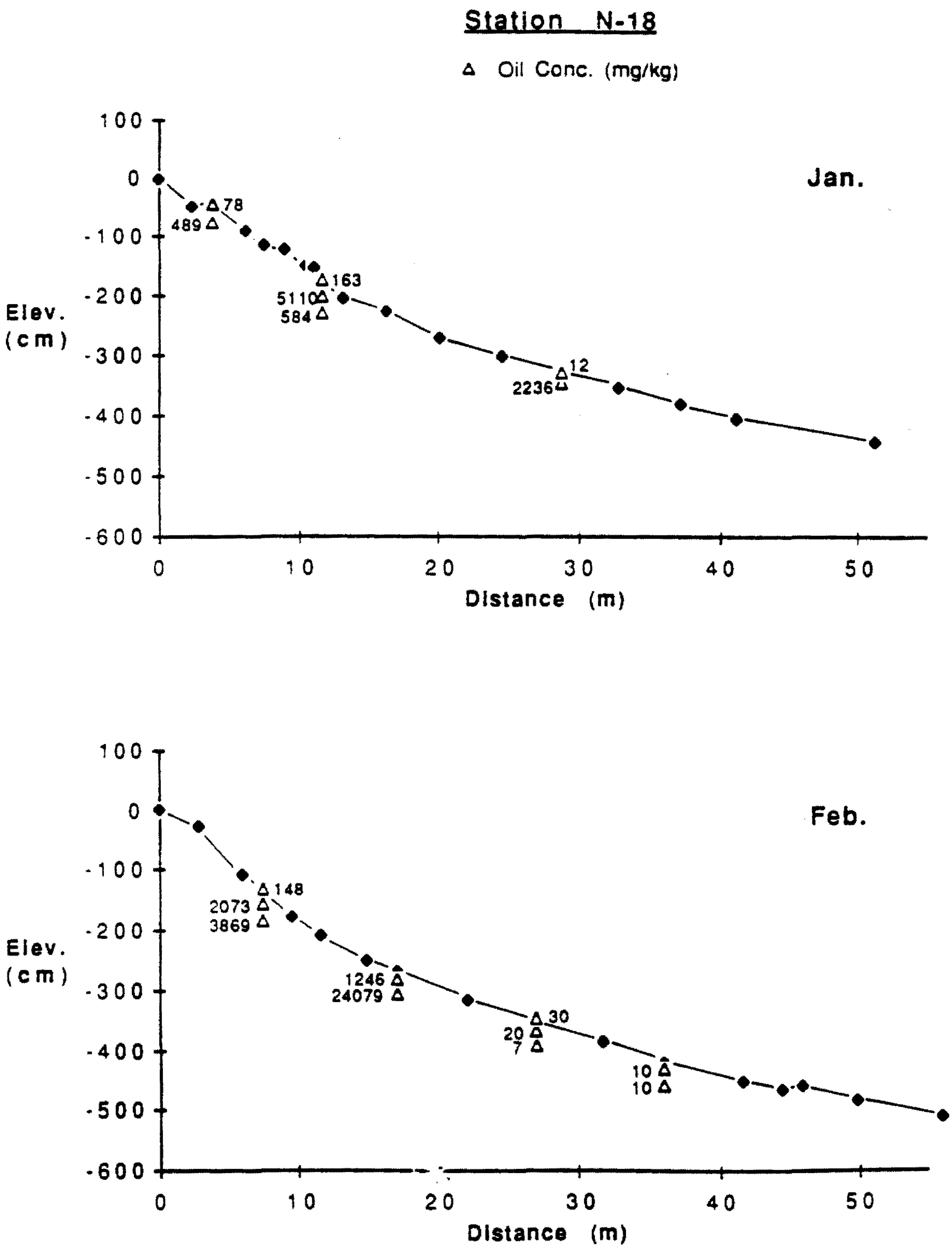

Figure I-23. Plots of the shoreline profile for Jan. and Feb., 1990, at NOAA Station N-18, Sleepy Bay. Oil concentrations at the depths are shown in mg/kg.

However, there are wide variations in oil distribution with depth and over time, as a function of sediment reworking. The initial heavy oiling of sediments at the high-tide berm (over 12,000 mg/kg in September) was greatly reduced as the berm (composed of pebble-sized sediments) was eroded, reformed, and migrated with changing neap to spring tides. The central part of the beach had high subsurface oil concentrations throughout the monitoring period, ranging from 5,800 to 24,000 mg/kg, with the natural variation masking any temporal trends. In fact, the highest value measured was for a February 1990 sample from the upper part of the central platform. The rest of the intertidal zone has relatively low levels of oil contamination, with concentrations in February of 50 mg/kg or less.

An estimate of the volume of oil in the candidate beach at Sleepy Bay using NOAA data is based on the following assumptions:

1) Width of subsurface oil = 25m
2) Depth of subsurface oil = 60cm (average maximum)
   30cm (average at the edges)

   Top 20cm are "clean."
3) Oil content of subsurface sediments is determined from the following percent oil by by volume measurements of NOAA samples over the winter:

Month	Maximum %	Mean %
Feb	4.5	1.4
Jan	0.7	0.4
Dec	1.0	0.4
Nov	2.6	1.0
Oct	0.9	0.3
Sept	3.3	1.4
Mean	2.2	0.8

These values are for the smaller components of the beach sediment (granule to pebble). According to grain-size descriptions of trenches in the candidate beach area of Sleepy Bay in May 1990, the subsurface sediments are about 25 percent cobble or greater and 75 percent pebble/granule/sand. It is assumed that the volumetric amount of oil on a cobble/boulder is negligible.

The cross-sectional area of oiled sediments at Sleepy Bay is calculated as:

$25\text{ m} \times 0.6\text{ m} = 15\text{ m}^2$

-5 m^2 (reduction due to the clean top 20 cm)

$-\underline{3.75}\text{ m}^2$ (due to thinner depth of oil at edges)

6.25 m^2 oiled sediment per linear m of beach

The oil volume per linear meter of beach is calculated as:

$6.25\text{ m}^2 \times 0.8\%$ oil by volume x 1m beach length x 0.75

(% <cobbles) =

0.0375 m^3 or 9.9 gallons per linear meter of beach

The area of Sleepy Bay selected for review is 300 m long, thus that shoreline is estimated to contain 2,960 gallons of oil, about 2 1/4 times more than the amount estimated for Point Helen. Again, assuming that the returned sediments still contain 800 ppm oil by weight (or an estimated 0.1 percent by volume), washing will remove approximately 2,600 gallons from this 300m stretch of shoreline.

Exxon survey teams collected thirteen samples from the eastern section of Sleepy Bay in March 1990 and eleven samples in June 1990. Exxon sampled relatively large intervals, such as from 20 to 40cm in their samples. In March, TPH concentrations in the upper 22m of the beach ranged from 234 to 7,254 mg/kg and averaged 2,363 mg/kg. In June, TPH concentrations in the upper zone ranged from 117 to 11,009 mg/kg and averaged 3,415 mg/kg. Using these numbers and the oil thicknesses measured in each pit, Exxon calculated oil loading as follows:

March 1990	5.0 gallons of oil/meter of beach
June 1990	6.1 gallons of oil/meter of beach

Chemical Composition of Stranded Oil at the Candidate Beaches

Field investigations and detailed gas chromatograph/mass spectrometer (GC/MS) analyses of representative samples collected during May 1990 field investigations of the candidate beaches (LA-18/19, Sleepy Bay, and KN-405, Point Helen) suggest a diversity in weathering processes (fates) and chemical composition of the stranded oil. Oil spilled into the environment is subject to physical and chemical changes. These changes are generally known as weathering and include physical transport mechanisms and chemical/biological alterations of the spilled oil. When spilled oil is stranded on a beach, these weathering processes continue but are modified by the environment (beach material, tides, wave action, etc.) on which it is stranded. The primary weathering processes include spreading, evaporation, dissolution, dispersion, photochemical oxidation, water-in-oil emulsification, adsorption onto suspended material (particulate and colloidal), biodegradation, and various shoreline interactions. Penetration of the oil into the subsurface of the beach and the fate of that oil is the focus of this investigation.

The detailed chemistry data obtained from the May 1990 investigations have been synthesized and presented to provide an understanding of the composition of the remaining oil. Discrete samples were chosen to reflect the different types of oil identified by visual inspection of the beach. The data are presented as histogram plots comparing the "fresh" EXXON VALDEZ cargo oil to May 1990, samples (e.g., mousse, tar mat, weathered oil residues). It should be noted that each of the compositional histograms (Figures I-24 through I-32) are limited in that they present only the relative abundances of the target compounds, namely the polynuclear aromatic hydrocarbon (PAH) compounds. PAHs represent only a small fraction of the "whole" oil remaining; PAHs comprise less than 1% of the fresh oil. The compounds targeted were chosen because of their relative abundance in the spilled oil, persistence in the environment, and toxicological concerns. The y-axis of the plot represents the concentration of individual target compounds in the sample (including the sediment) and are reported as ng of analyte per mg of sample weight (ppm).

Sleepy Bay. Sleepy Bay contains stranded oil characterized as moderate to heavily weathered on the surface with patches of less weathered mousse and fresher oil contained within exposed tar mats. The mousse found in Sleepy Bay is characteristically an oil-in-water emulsification which is very stable. Figure I-24 shows a comparison of EXXON VALDEZ cargo oil to mousse collected from the surface at Sleepy Bay 14 months after the spill. The slight differences noted between the spilled oil and the 14-month old mousse represent only part of the chemical changes that have occurred during its formation. The GC/MS analysis indicates evaporative and dissolution loss of the lighter constituents. Only slight preferential degradation was noted, indicating very little microbial degradation has occurred. Mousse in the environment is characteristically a very stable material that is slow to degrade without physical breakup since most of the weathering fates are surface dependant. Microbial and photooxidation of the mousse occurs only on the very surface. The resulting effect is the formation of a hard crust which encapsulates the remaining oil within, thus retarding further degradation. The mousse sample collected at Sleepy Bay did not readily sheen. If the emulsion is broken by physical action or even by elevating the temperature (mousse exposed to the solar radiation will act as a black body), the mousse will sheen.

Tar mats represent another pathway of oil degradation. As the tar mat continues to weather, it will ultimately form an asphalt pavement which is very resistant to continued dispersion and degradation. Both the mousse and tar mat are primarily surface impacts only. The recommended treatment is physical removal; therefore, they are not characteristic of the type of oil which the rock washer is expected to remove. Their inclusion in this study is to provide a comprehensive comparison of the types of oil present at the candidate sites, and for contrast with the subsurface oil. Figure I-25 shows a comparison of fresh EXXON VALDEZ oil to a sample of tar mat. Like the mousse sample shown in Figure I-24, the tar mat has undergone only slight weathering. The fresher oil contained within the tar mat will readily sheen.

An example of surface oil from Sleepy Bay that has been significantly degraded compared to EXXON VALDEZ cargo oil is shown in Figure I-26. The heavily weathered oil has lost all of the lighter constituents, the non-alkylated (parent) and C-1 and C-2 PAH homologs are significantly reduced relative to the more degradation-resistive C-3 and C-4 PAH homologs, and

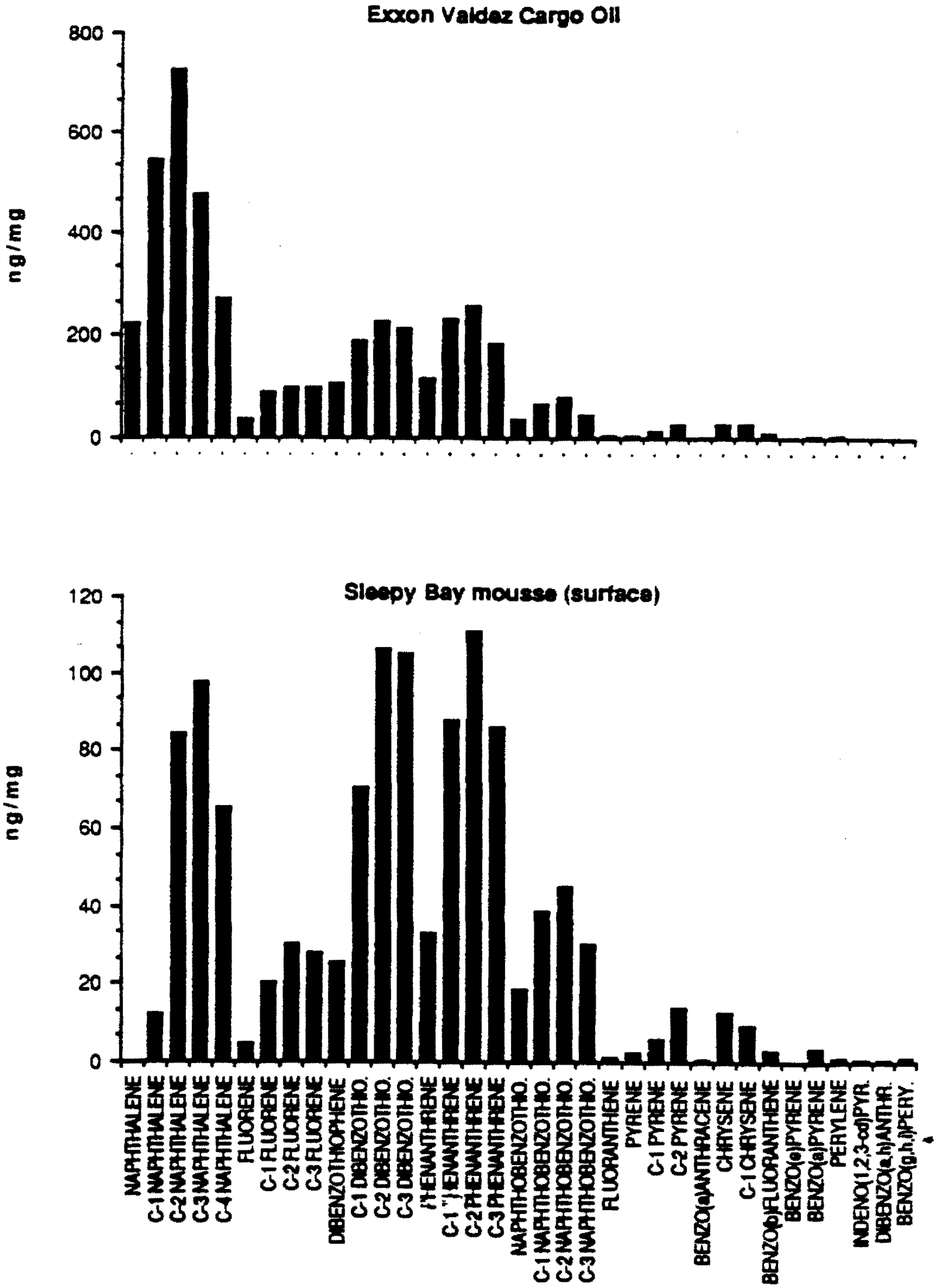

Figure I-24. Comparison of Exxon Valdez cargo oil to mousse collected at Sleepy Bay (May 1990). Note the difference in scale of concentration between the two plots.

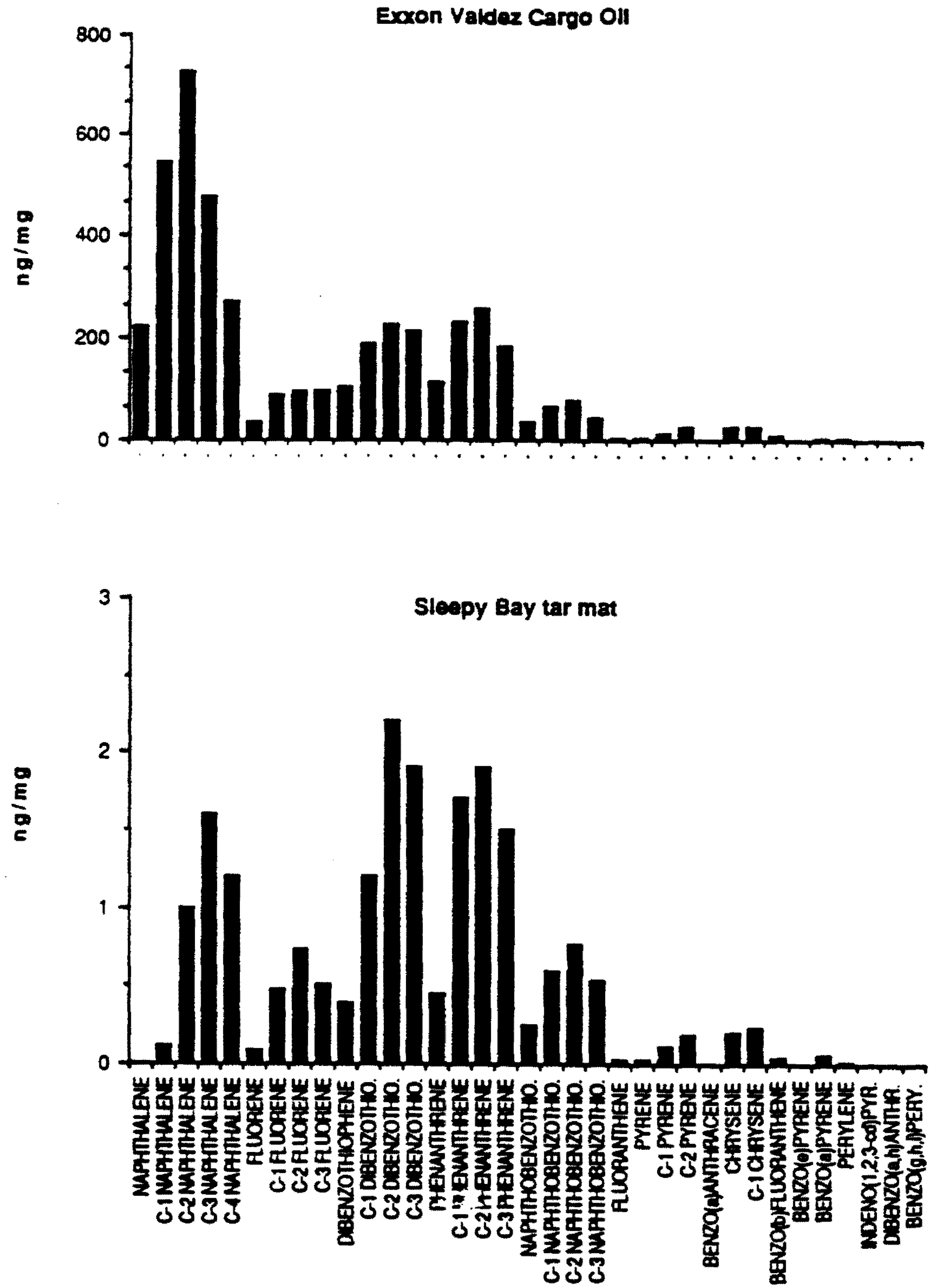

Figure I-25. Comparison of Exxon Valdez cargo oil to oil contained within a tar mat at Sleepy Bay. Note the difference in scale of concentration between the two plots.

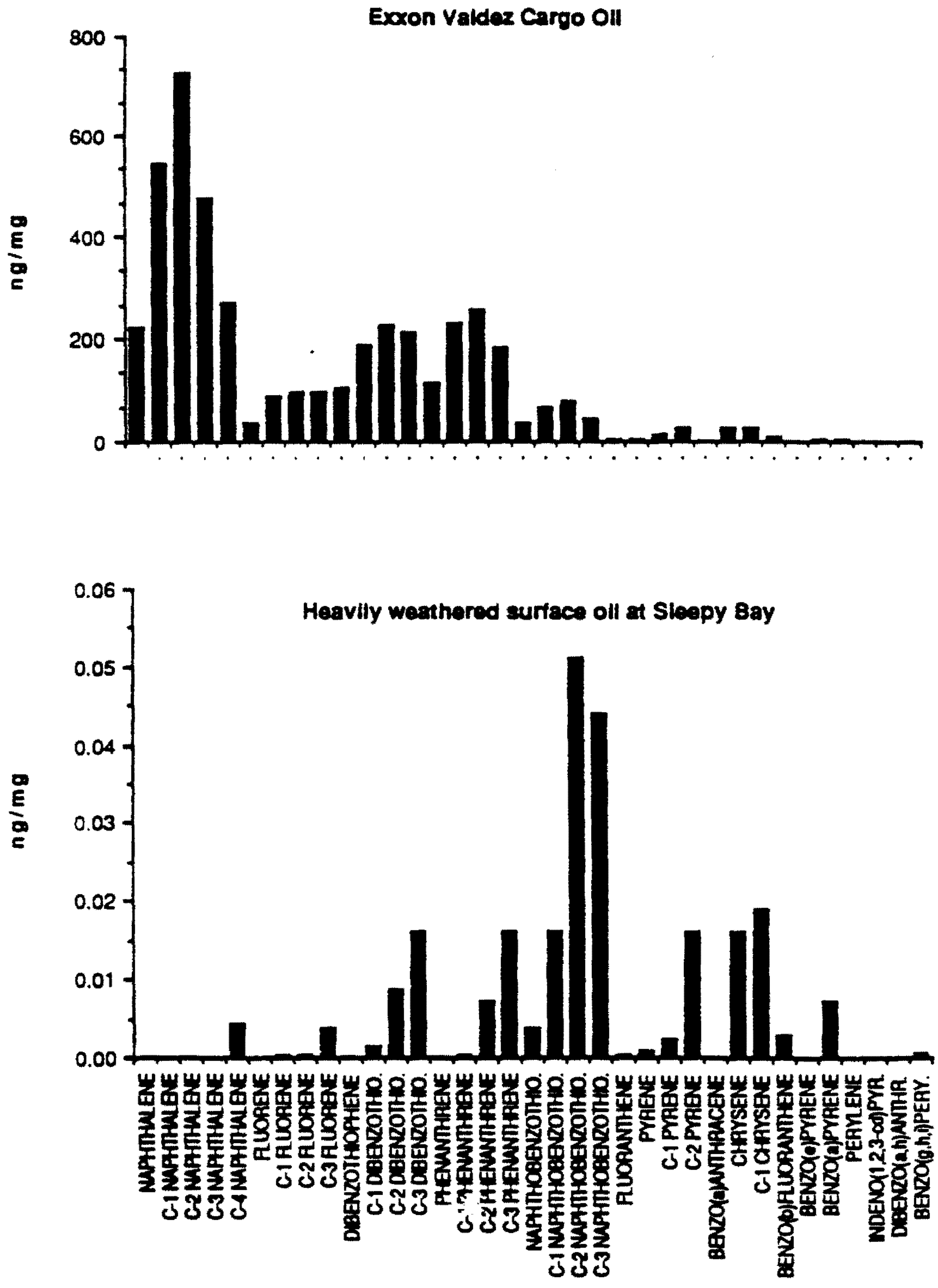

Figure I-26. Comparison of Exxon Valdez cargo oil to a surface sample from Sleepy Bay that is heavily weathered. Note the difference in scale of concentration between the two plots.

the sulfur heterocycle compounds generally show less reduction compared to PAHs with similar molecular weights (e.g., the C-3 dibenzothiophenes, MW=226, are slower to degrade than the C-3 phenanthrenes, MW=220). These changes are consistent with normal degradation in the environment and reflect loss of individual constituents by dissolution and evaporation of the lower molecular weight and more water-soluble compounds in addition to preferential degradation by microbial processes. The sulfur heterocycles (dibenzothiophenes and naphthobenzothiophenes) are becoming more predominant in the target compound profile; this pattern is suggestive of a resistance to microbial degradation relative to the other target compounds. Oil of this composition will not sheen.

Subsurface oil demonstrates a range of weathering similar to the surface. Based on data collected from the NOAA winter study in Prince William Sound and on field observations in May 1990, the absolute quantity of subsurface oil is generally higher in concentration. The subsurface weathering processes differ relative to the surface. The physical composition of the oil varies from a heavily weathered oil residue with a "peat-like" feel to a fresher oil partially emulsified with water and fine colloidal and particulate material. The subsurface oil penetrated the beach substrate while still a relatively fresh oil, filling in available pore spaces. Through continual interactions with the water column (tidal action, etc.) and exposure to fine sediments, organic colloids, natural surfactants, as well as microbial-mediated oxidation and surfactant action, the oil has become viscous, characteristically more "mousse-like," and less mobile.

Interestingly, the composition of the oil buried under the the spring tide berms at Sleepy Bay at depths to 30cm demonstrates significant weathering and is characteristically dissimilar in appearance to "fresh" crude oil or the surface mousse typically encountered in Prince William Sound and the Gulf of Alaska. This oil does not sheen. The degree of weathering is significant, being characterized as moderately weathered. It would appear that the beach is acting in a fashion analogous to a trickle filter system used for waste water treatment. Snow melt, rain, and tidal activities carry nutrients, bacteria, and detrital material from the upland area and the open bay to the berm where it trickles through the berm sediments which are partially coated with oil. This process is probably most effective during the late spring and summer months when the temperature is warmer and runoff is greater. The upper intertidal and middle intertidal subsurface oil is characteristically not as weathered as that in the spring berm. Factors which may contribute to the reduced degradation rates in the lower areas noted are the physical distribution of the oil (clumpy and "mousse-like") and higher overall oil concentration in the subsurface sediments, which reduces the effectiveness of natural dispersion and degradation activities since they are primarily active at the surface only. Figures I-27 through I-30 show compositional histograms of the target constituents in the oil monitored for the spring tide berm, and the upper, middle, and lower intertidal zones, respectively. Note the changes relative to reference EXXON VALDEZ cargo oil. Oil of this composition doesn't normally sheen.

Point Helen. Only a limited number of May 1990, samples were analyzed from Point Helen. An upper intertidal zone subsurface sample collected at the NOAA study station, N-1, suggest that significant weathering has occurred (see Figure I-31). Neither of the samples collected at Point Helen generated a sheen. The relative composition was similar to that found in the spring tide berms at Sleepy Bay and Point Helen shown in Figures I-27 and I-32, respectively. PAH concentration data of subsurface oil from the Exxon Winter study show similar degradative changes in composition. These values are included in Table I-2.

Very little attention has been given to the saturate fraction of the stranded oil. All of the subsurface samples analyzed showed marked decreases in the lower molecular weight (less than n-C20) aliphatic compounds with a marked preference for the non-branched relative to the branched hydrocarbons. The isoprenoid hydrocarbons norpristane, pristane, and phytane showed a marked resistance to change as well as did the heavier paraffins (greater than n-C20); this selective preference indicates microbial degradation has occurred.

In summary, oil stranded on the candidate beaches can be described as four discrete types: mousse, tar mat, moderate to heavily weathered surface oil residues, and subsurface oil. The composition of the subsurface oil suggests that significant weathering has modified the stranded oil. The current composition of the subsurface oil is moderately weathered and

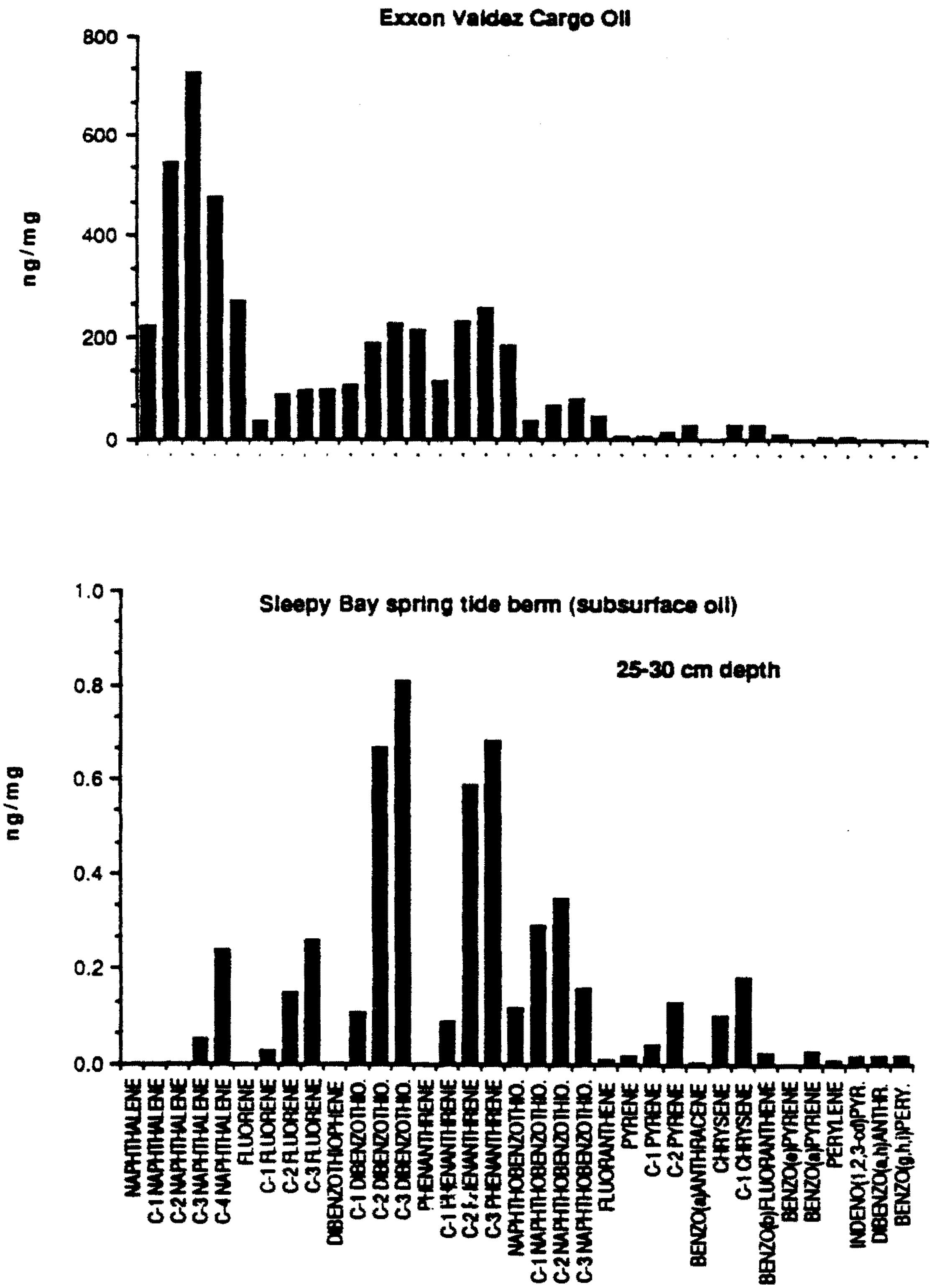

Figure I-27. Comparison of Exxon Valdez cargo oil to subsurface oil buried in the spring tide berm at Sleepy Bay (May 1990). Note the difference in scale of concentration between the two plots.

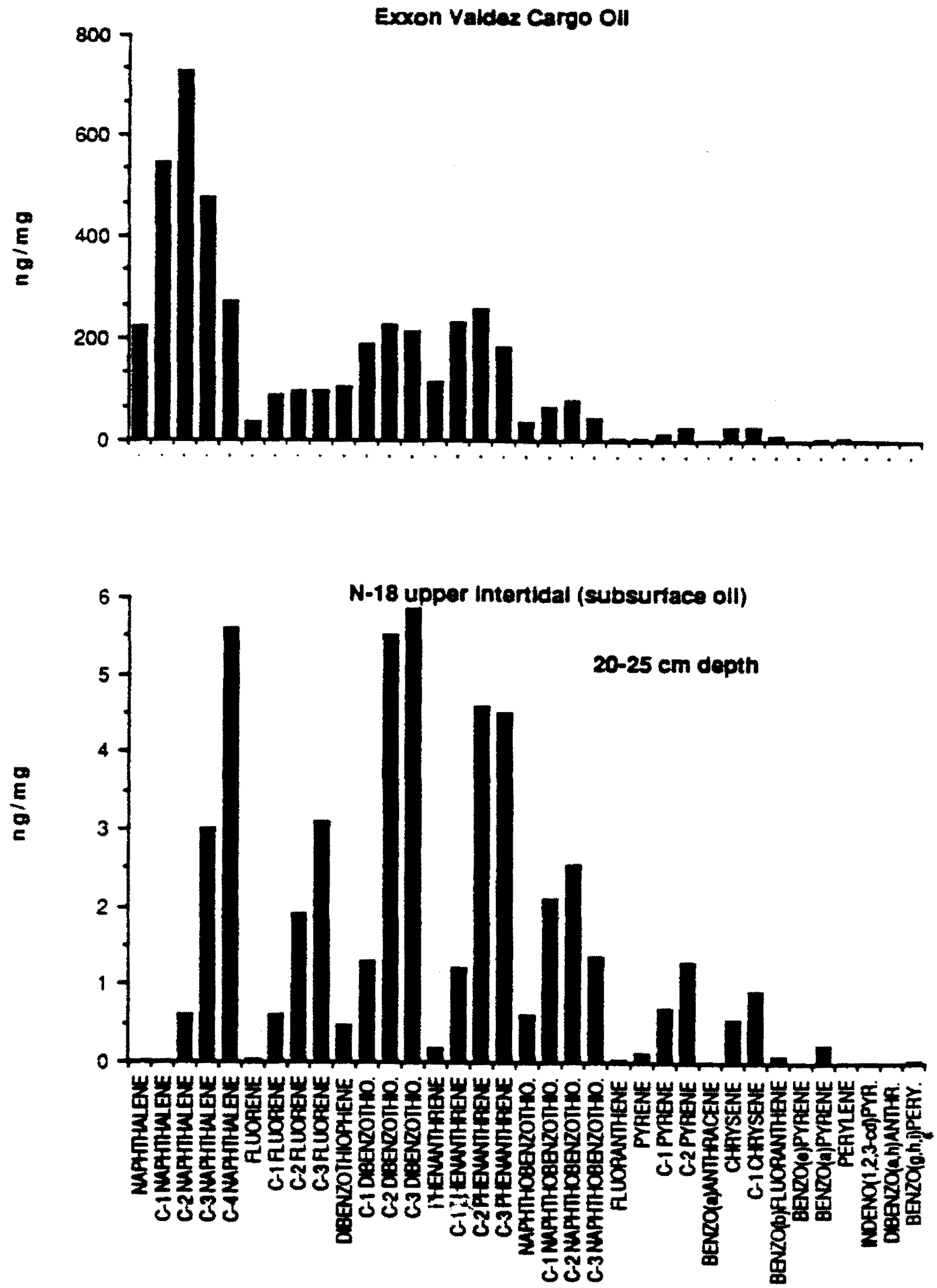

Figure I-28. Comparison of Exxon Valdez cargo oil to subsurface oil of the upper intertidal zone at NOAA study station N-18 (May 1990). Note the difference in scale of concentration between the two plots.

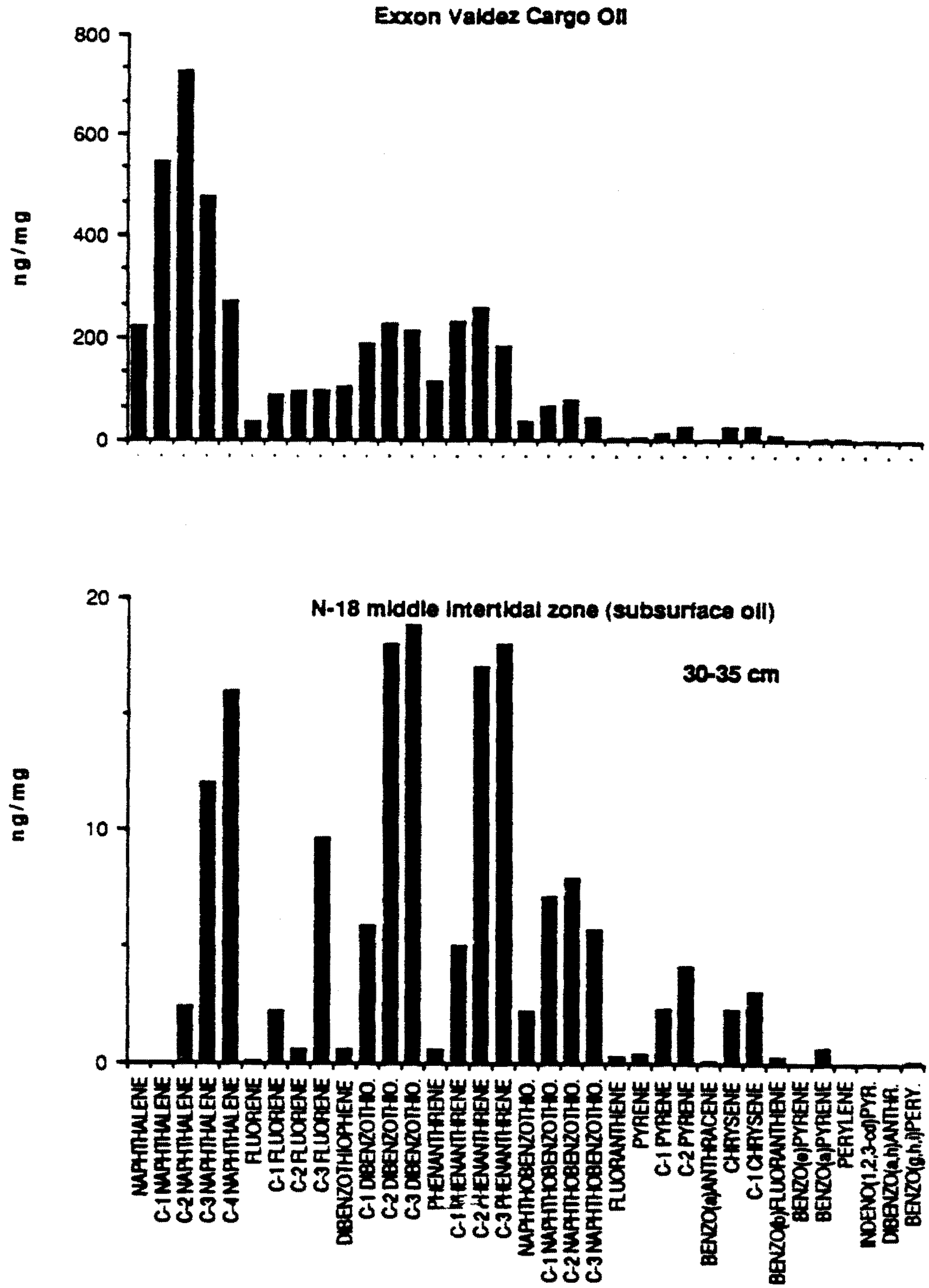

Figure I-29. Comparison of Exxon Valdez cargo oil to subsurface oil of the middle intertidal zone at NOAA study station N-18 (May 1990). Note the differences in scale of concentration between the two plots.

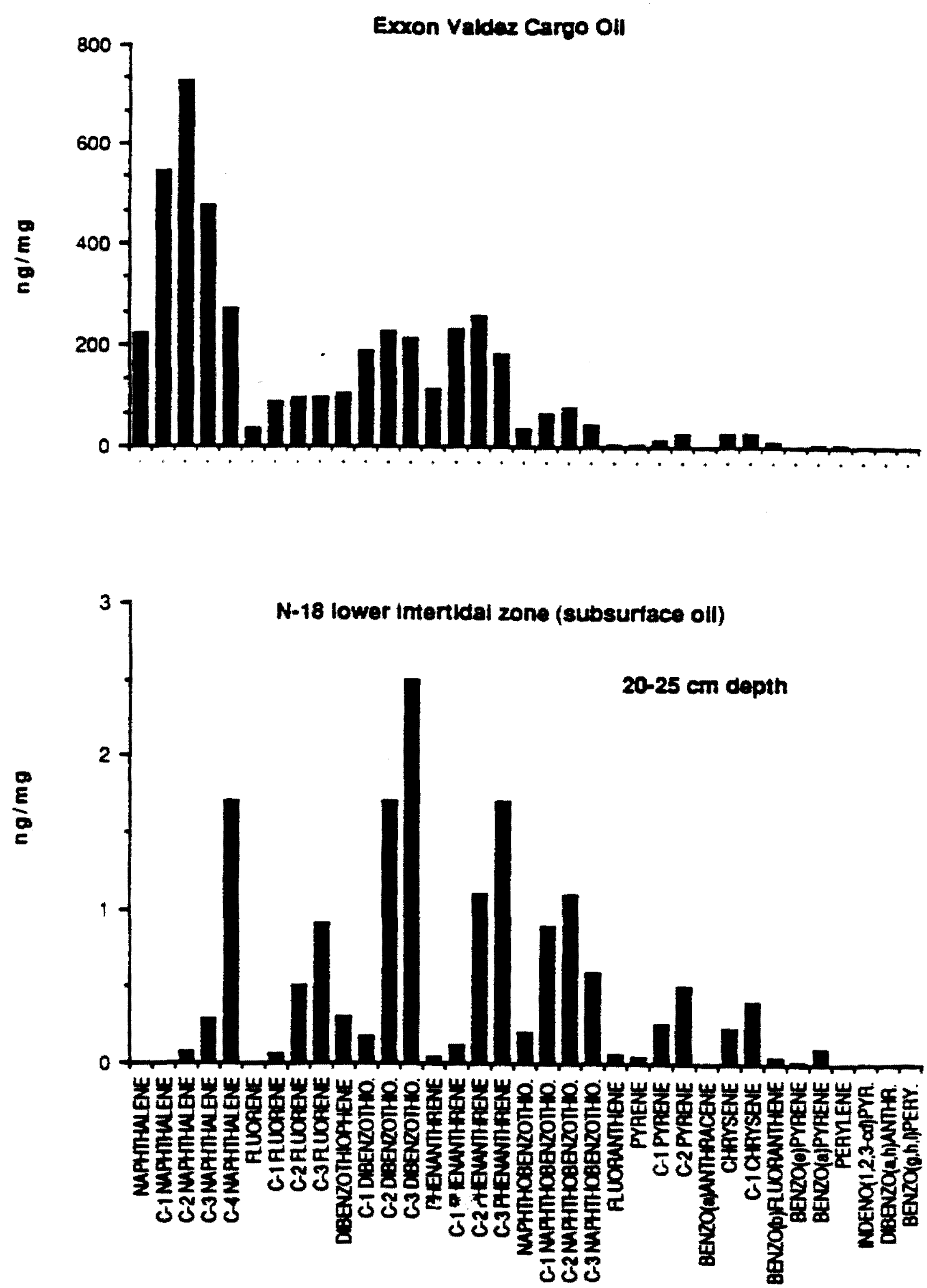

Figure I-30. Comparison of Exxon Valdez cargo oil to subsurface oil of the lower intertidal zone at NOAA study station N-18 (May 1990). Note the difference in scale of concentration between the two plots.

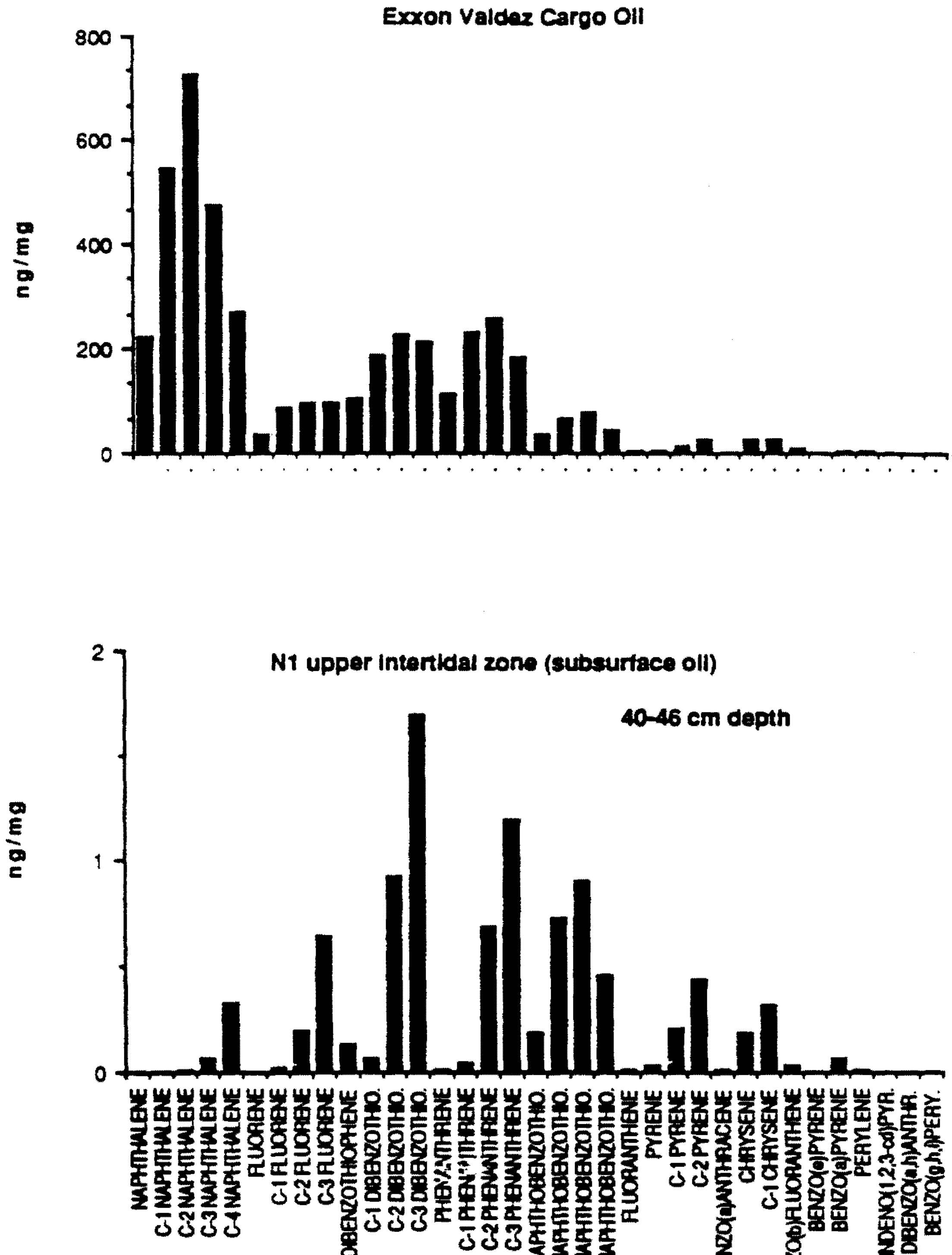

Figure I-31. Comparison of Exxon Valdez cargo oil to subsurface oil at NOAA study station N-1 upper intertidal zone (May 1990). Note the difference in scale of concentration between the plots.

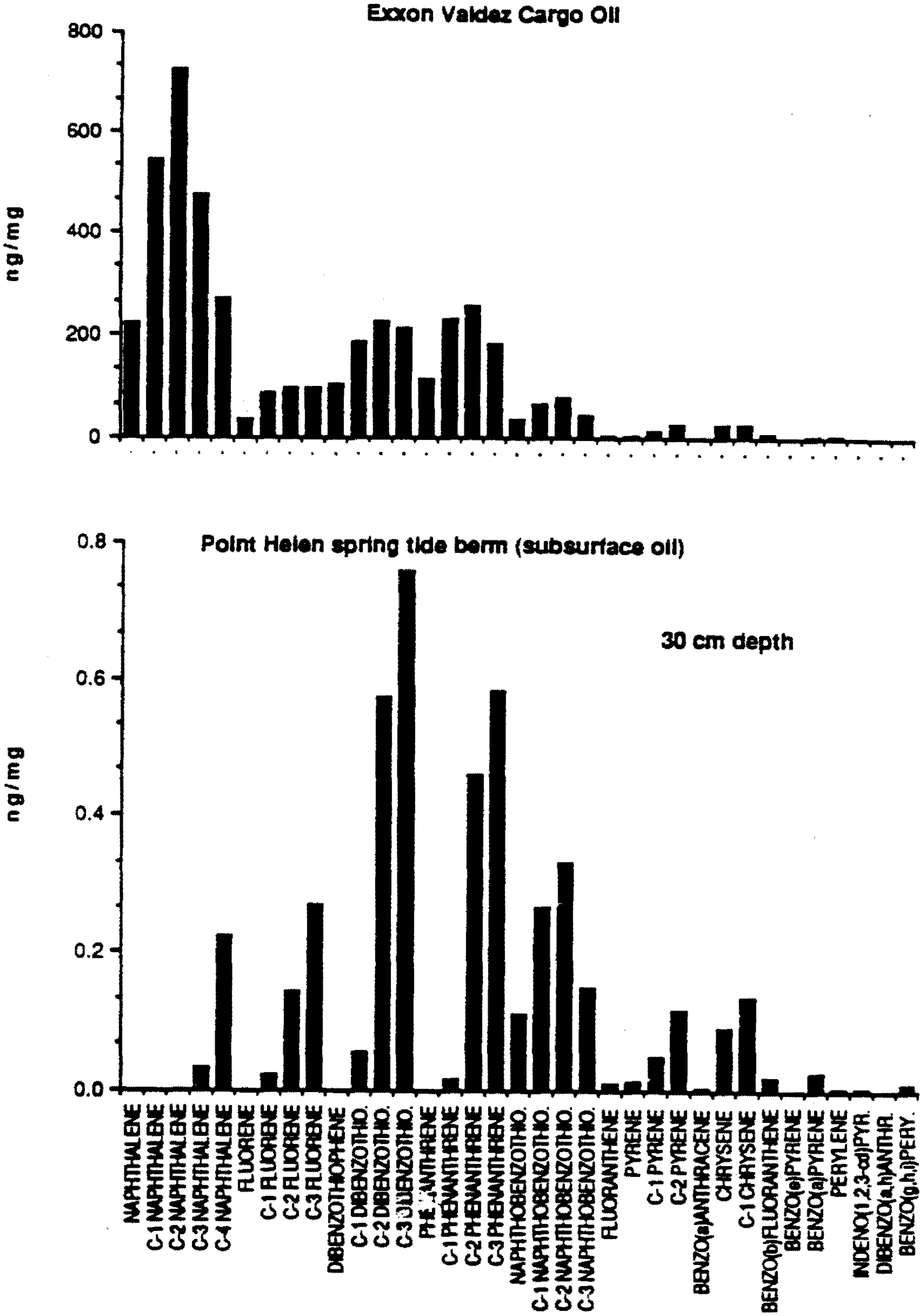

Figure I-32. Comparison of Exxon Valdez cargo oil to subsurface oil in the spring tidal berm at Point Helen (May 1990). Note the differences in scale of concentration between the two plots.

Table I-2. Concentration of Specific Compounds in Sediments at the Candidate Beaches.

GC/MS QUANT RESULTS					
STATION:	LA-19	LA-19	LA-19	LA-19	N18
SAMPLE TYPE:	mousse	tar/grav	grav./peb.	grav./peb.	grav./peb.
DEPTH:	0-5 cm	0-5 cm	0-5 cm	25-30 cm	20-25 cm
LAB ID:	N0143-2	N0143-5	N0143-6	N0143-1	N0145-10
MONTH SAMPLED:	MAY, '90	MAY, '90	MAY, '90	MAY, '90	MAY, '90
RELATED FIGURE:	F 1-24	F 1-25	F 1-26	F 1-27	F 1-28
COMPOUND	(ng/mg)	(ng/mg)	(ng/mg)	(ng/mg)	(ng/mg)
NAPHTHALENE	0.1400	0.0021	ND	ND	ND
C-1 NAPHTHALENE	12.0000	0.1200	ND	ND	ND
C-2 NAPHTHALENE	84.0000	1.0000	ND	0.0014	0.0700
C-3 NAPHTHALENE	98.0000	1.6000	ND	0.0540	0.2900
C-4 NAPHTHALENE	65.0000	1.2000	0.0042	0.2400	1.7000
FLUORENE	4.5000	0.0820	ND	0.0000	0.0000
C-1 FLUORENE	20.0000	0.4800	0.0003	0.0290	0.0600
C-2 FLUORENE	30.0000	0.7400	0.0004	0.1500	0.5100
C-3 FLUORENE	28.0000	0.5000	0.0037	0.2600	0.9100
DIBENZOTHIOPHENE	26.0000	0.3900	ND	0.0007	0.3100
C-1 DIBENZOTHIO.	71.0000	1.2000	0.0013	0.1100	0.1800
C-2 DIBENZOTHIO.	110.0000	2.2000	0.0087	0.6700	1.7000
C-3 DIBENZOTHIO.	100.0000	1.9000	0.0160	0.8100	2.5000
PHENANTHRENE	33.0000	0.4500	ND	ND	0.0400
C-1 PHENANTHRENE	88.0000	1.7000	0.0002	0.0870	0.1100
C-2 PHENANTHRENE	110.0000	1.9000	0.0073	0.5900	1.1000
C-3 PHENANTHRENE	86.0000	1.5000	0.0160	0.6800	1.7000
NAPHTHOBENZOTHIO.	18.0000	0.2500	0.0036	0.1200	0.2000
C-1 NAPHTHOBENZOTHIO.	39.0000	0.5900	0.0160	0.2900	0.9000
C-2 NAPHTHOBENZOTHIO.	45.0000	0.7600	0.0510	0.3500	1.1000
C-3 NAPHTHOBENZOTHIO.	30.0000	0.5400	0.0440	0.1600	0.5900
FLUORANTHENE	1.4000	0.0230	0.0002	0.0078	0.0600
PYRENE	2.0000	0.0350	0.0008	0.0210	0.0400
C-1 PYRENE	6.0000	0.1000	0.0023	0.0450	0.2600
C-2 PYRENE	14.0000	0.1900	0.0160	0.1300	0.5100
BENZO(a)ANTHRACENE	0.5200	0.0057	0.0000	0.6800	0.0000
CHRYSENE	13.0000	0.2000	0.0160	0.1000	0.2300
C-1 CHRYSENE	9.3000	0.2300	0.0190	0.1800	0.4100
BENZO(b)FLUORANTHENE	3.1000	0.0400	0.0028	0.0240	0.0400
BENZO(e)PYRENE	ND	0.0040	0.0000	0.0006	0.0100
BENZO(a)PYRENE	3.7000	0.0530	0.0073	0.0280	0.1000
PERYLENE	0.8800	0.0088	ND	0.0110	ND
INDENO(1,2,3-cd)PYR.	0.7700	0.0013	ND	0.0200	ND
DIBENZO(a,h)ANTHR.	0.5200	ND	ND	0.0160	ND
BENZO(g,h,i)PERY.	1.2000	0.0097	0.0007	0.0200	ND

All values are valid to two significant figures only.

Table I-2. Cont.

GC/MS QUANT RESULTS				
STATION:	N18	N18	N1	KN-405
SAMPLE TYPE:	grav./peb.	grav./peb.	grav.	grav./peb.
DEPTH:	30-35 cm	20-25 cm	40-46 cm	30 cm
LAB ID:	N0145-9	N0145-11	N0145-5	N0143-3
MONTH SAMPLED:	MAY, '90	MAY, '90	MAY, '90	MAY, '90
RELATED FIGURE:	F 1-29	F 1-30	F 1-31	F 1-32
COMPOUND	(ng/mg)	(ng/mg)	(ng/mg)	(ng/mg)
NAPHTHALENE	ND	ND	ND	ND
C-1 NAPHTHALENE	0.0400	0.0100	ND	ND
C-2 NAPHTHALENE	2.4000	0.6200	0.0100	0.0014
C-3 NAPHTHALENE	12.0000	3.0000	0.0600	0.0325
C-4 NAPHTHALENE	16.0000	5.6000	0.3300	0.2254
FLUORENE	0.1000	0.0300	ND	ND
C-1 FLUORENE	2.2000	0.6100	0.0200	0.0223
C-2 FLUORENE	0.6100	1.9000	0.2000	0.1444
C-3 FLUORENE	9.6000	3.1000	0.6400	0.2690
DIBENZOTHIOPHENE	0.6000	0.4600	0.1300	ND
C-1 DIBENZOTHIO.	5.9000	1.3000	0.0600	0.0554
C-2 DIBENZOTHIO.	18.0000	5.5000	0.9200	0.5739
C-3 DIBENZOTHIO.	19.0000	5.9000	1.7000	0.7578
PHENANTHRENE	0.6100	0.1600	0.0100	ND
C-1 PHENANTHRENE	5.0000	1.2000	0.0400	0.0172
C-2 PHENANTHRENE	17.0000	4.6000	0.6900	0.4598
C-3 PHENANTHRENE	18.0000	4.5000	1.2000	0.5818
NAPHTHOBENZOTHIO.	2.2000	0.6100	0.1900	0.1095
C-1 NAPHTHOBENZOTHIO.	7.1000	2.1200	0.7300	0.2680
C-2 NAPHTHOBENZOTHIO.	7.9000	2.5400	0.9000	0.3321
C-3 NAPHTHOBENZOTHIO.	5.7000	1.3700	0.4600	0.1488
FLUORANTHENE	0.2500	0.0200	0.0100	0.0090
PYRENE	0.4100	0.1100	0.0300	0.0134
C-1 PYRENE	2.3000	0.6900	0.2100	0.0489
C-2 PYRENE	4.1000	1.3000	0.4400	0.1161
BENZO(a)ANTHRACENE	0.0600	0.0000	0.0100	0.0033
CHRYSENE	2.3000	0.5600	0.1900	0.0904
C-1 CHRYSENE	3.1000	0.9200	0.3200	0.1368
BENZO(b)FLUORANTHENE	0.2900	0.0800	0.0300	0.0195
BENZO(e)PYRENE	ND	ND	ND	ND
BENZO(a)PYRENE	0.6700	0.2300	0.0600	0.0265
PERYLENE	0.0000	ND	0.0100	0.0034
INDENO(1,2,3-cd)PYR.	0.0000	ND	ND	0.0031
DIBENZO(a,h)ANTHR.	0.0000	ND	ND	ND
BENZO(g,h,i)PERY.	0.1000	0.0300	ND	0.0094

All values are valid to two significant figures only.

Table I-2. Cont.

GC/MS QUANT RESULTS		
STATION:	AP-10	AP-10
SAMPLE TYPE:	-	-
DEPTH:	50-60 cm	5-15 cm
LAB ID:	B8	B24
MONTH SAMPLED:	Jan. '90	Sept.. '89
RELATED FIGURE:	-	-
COMPOUND	*	*
NAPHTHALENE	0.0008	0.0000
C-1 NAPHTHALENE	0.0000	0.0000
C-2 NAPHTHALENE	0.1500	0.0071
C-3 NAPHTHALENE	0.6900	0.0520
C-4 NAPHTHALENE	1.1000	0.3500
FLUORENE	0.0036	0.0000
C-1 FLUORENE	0.1500	0.0260
C-2 FLUORENE	0.4900	0.1600
C-3 FLUORENE	0.5000	0.2800
DIBENZOTHIOPHENE	0.0100	0.0053
C-1 DIBENZOTHIO.	0.2300	0.0640
C-2 DIBENZOTHIO.	0.7600	0.4400
C-3 DIBENZOTHIO.	0.7200	0.5310
PHENANTHRENE	0.1800	0.0070
C-1 PHENANTHRENE	0.3400	0.0980
C-2 PHENANTHRENE	0.9000	0.5200
C-3 PHENANTHRENE	0.7800	0.5800
NAPHTHOBENZOTHIO.		
C-1 NAPHTHOBENZOTHIO.		
C-2 NAPHTHOBENZOTHIO.		
C-3 NAPHTHOBENZOTHIO.		
FLUORANTHENE	0.0027	0.0000
PYRENE	0.0100	0.0110
C-1 PYRENE	0.1100	0.0810
C-2 PYRENE		
BENZO(a)ANTHRACENE	0.0000	0.0000
CHRYSENE	0.0660	0.0620
C-1 CHRYSENE	0.1400	0.1400
BENZO(b)FLUORANTHENE	0.0000	0.0075
BENZO(e)PYRENE		
BENZO(a)PYRENE		
PERYLENE		
INDENO(1,2,3-cd)PYR.		
DIBENZO(a,h)ANTHR.		
BENZO(g,h,i)PERY.	0.0049	0.0000

All values are valid to two significant figures only.
* Values given as parts per thousand in the oil.

immobile. Microbial degradation will continue, and be a function of exposed surface area of the oil, nutrient availability, and temperature. As the degradation continues, the composition of the subsurface oil will become similar to that described for the heavily weathered surface sample (Figure I-26).

I.C.4. PORE WATER CHEMISTRY

In the intertidal zone, the space between the grains of sediment (the pore or interstitial space) may be filled with water or air. The water table in the sediments on the shore rises and falls with the tide and usually is somewhat higher than the tidal level due to capillary lift of water in small pore spaces and to inflow of surface runoff water from the adjacent land and upper shore. On the falling tide, the pore water drains from the sediments lying above the tide level. Most of this drainage is subsurface and the water emerges below the tide level. However, in some cases, particularly in sandy or silty sediments (or in areas with large tidal ranges), subsurface drainage can not keep pace with the rate of decrease in tidal height and some of the water drains out onto the surface of the beach above the tide level and flows down the beach face. During periods of snow melt or rain, runoff of freshwater from land mixes with the saline interstitial water and moves with it down through the beach substrate on the falling tide.

If the intertidal sediments contain subsurface deposits of oil, the interstitial water moving with the tides through the sediments may come in contact with the oil and carry away some of it in solution or in particulate form. The amount of oil removed with the flushing of intertidal water will depend on the relative amount of surface area available for contact between the oil and water and on the degree of weathering of the oil. If the subsurface sediments are saturated with petroleum, there will be little contact between the oil and interstitial water. The oil will be relatively immobile unless removed physically by storm action or man's activities. If the concentration of oil is lower and only a fraction of the available pore space is occupied by the oil (e.g., the sediment particles have a surface coating of oil), there will be substantial surface area for contact between oil and interstitial water during each tidal cycle. Hydrocarbons may be leached continuously into the interstitial water as the interstitial water is replaced by tidal pumping and surface runoff until no leachable hydrocarbons remain. The weathered oil is somewhat less sticky than fresh crude oil, due to formation of stable oil/clay flocs (see Section II.A.5). Therefore, droplets may be dislodged by the flowing water and carried away with it.

The rate of transport of oil from the subsurface deposits depends on the amount of water that comes in contact with the oil each day and the tendency of the individual hydrocarbons remaining in the oil to move from the solid oil phase into the water phase. As the more soluble components of the oil are leached out by the flowing water, the remaining oil becomes more viscous and the rate of leaching of soluble components and dispersion of droplets into the water decreases. Thus, it can be expected that the rate of removal of oil from a subsurface oil deposit will decrease with time. Therefore, information about the composition and concentrations of petroleum hydrocarbons in the pore water is important in predicting the potential environmental fate and effects of weathered oil buried in the upper intertidal zone of some beaches in Prince William Sound.

A mathematical model was developed to predict the concentrations of PAHs (the most toxic fractions of the spilled oil) in pore water in contact with subsurface oil deposits. The model also predicts the rate at which PAHs will be leached from the deposits during tidal flushing of the intertidal zone. Validation of model predictions was provided by analysis of pore water samples from several shores where subsurface oil was present. A detailed description of the model and predicted leaching rates of PAHs from subsurface oil deposits is available from Exxon as a separate report.

In a two-phase, oil/water system, organic solutes such as PAHs will tend to become distributed between the water and oil phases according to their relative solubilities in the two phases. The distribution of the PAHs between the two phases can be expressed as a distribution or a partition coefficient. Octanol/water partition coefficients (K_{ow}) derived empirically or theoretically are frequently used to estimate the concentration of a sparingly soluble organic compound such as a PAH in water in equilibrium with a solid phase (sediment organic carbon, bulk petroleum, or the tissues of a marine organism). In the model, concentrations of different PAHs in sediment pore water in equilibrium with subsurface oil deposits were calculated based on the K_{ow} of each PAH and its concentration in a typical sample of subsurface oil. The oil samples used to calculate concentrations of PAHs in pore water included relatively lightly weathered oil similar to that which originally came ashore, moderately weathered subsurface oil collected at several locations in Prince William Sound in January 1990, and more highly weathered subsurface oil from March 1990.

It was necessary to make several assumptions in the model. All of these assumptions were conservative (e.g., they predict higher concentrations of PAHs in pore water than are likely to occur). The major assumption is that equilibrium is reached between the deposit of subsurface oil and pore water flowing over and coming into momentary contact with the oil. This could be the case if the contact time between pore water and the oil deposit is long enough, on the order of several minutes (Karickhoff, 1980). However, it is unlikely that contact between pore water and oil will reach equilibrium.

Three sets of calculations were made with the model:

- Initial-concentration-case equilibrium calculations representative of oil at high concentrations at a weathering state of the oil when it first came ashore;
- Average-case calculations representative of the average oil loading and composition found during January sampling; and
- March-case calculations using actual oil loading and composition from March samples to predict water quality.

The concentrations of PAHs in the oily sediments were measured by gas chromatography/mass spectrometry. Data were acquired for 46 samples of subsurface oily sediments chosen to be representative of the samples taken in the winter beach monitoring program. Twelve additional subsurface sediment samples, as well as pore water samples taken from the same trenches as the sediment samples, from the March 1990 survey, were also analyzed. The inputs to the model for the three test cases were as follows:

- Initial-concentration case equilibrium calculations were made using PAH concentrations determined by taking the average plus the standard deviation from the three samples with the highest measured PAH concentrations. These samples are the least weathered samples of all those that were analyzed for PAHs. Therefore, they are reasonable representations of the concentrations of PAHs in the oil that came ashore.
- Average-case equilibrium calculations were made using the average PAH concentrations from seven subsurface sediment samples taken in January 1990.
- March-cases represent equilibrium calculations using PAH concentrations and TPH concentrations in individual samples collected during March, 1990. Pore water samples were also collected at the same locations and depths as the sediment samples.

Concentrations of PAHs leaching into pore water over time were calculated based on the number of volumes of pore water that flow through the sediment during each tidal cycle. Estimates of beach flushing were made using fluid flow models that indicate that about 10 pore volumes flow through intertidal sediments of a cobble beach or a cobble beach underlain with sand during each tidal cycle. The actual flushing rate of a beach depends on several physical properties of the beach and is quite variable. Therefore, sensitivity cases were run to show the effect of flushing rates from 1 to 50 pore volumes/tidal cycle, a realistic range for different substrate types.

Initial-Concentration-Case Calculations. The leaching rates of individual PAHs were calculated over time based on the concentrations of the PAHs remaining in the subsurface oil. Values for individual PAHs were summed to produce an estimate of the leaching rate for total PAHs at three flushing rates (Figure I-33). Two tidal cycles per day were assumed for these calculations. The maximum concentration of TPH (5,000 mg/kg) was used.

In all three cases, the initial total PAH concentrations in the interstitial water were about 86 ppb. For the base case of 10 pore

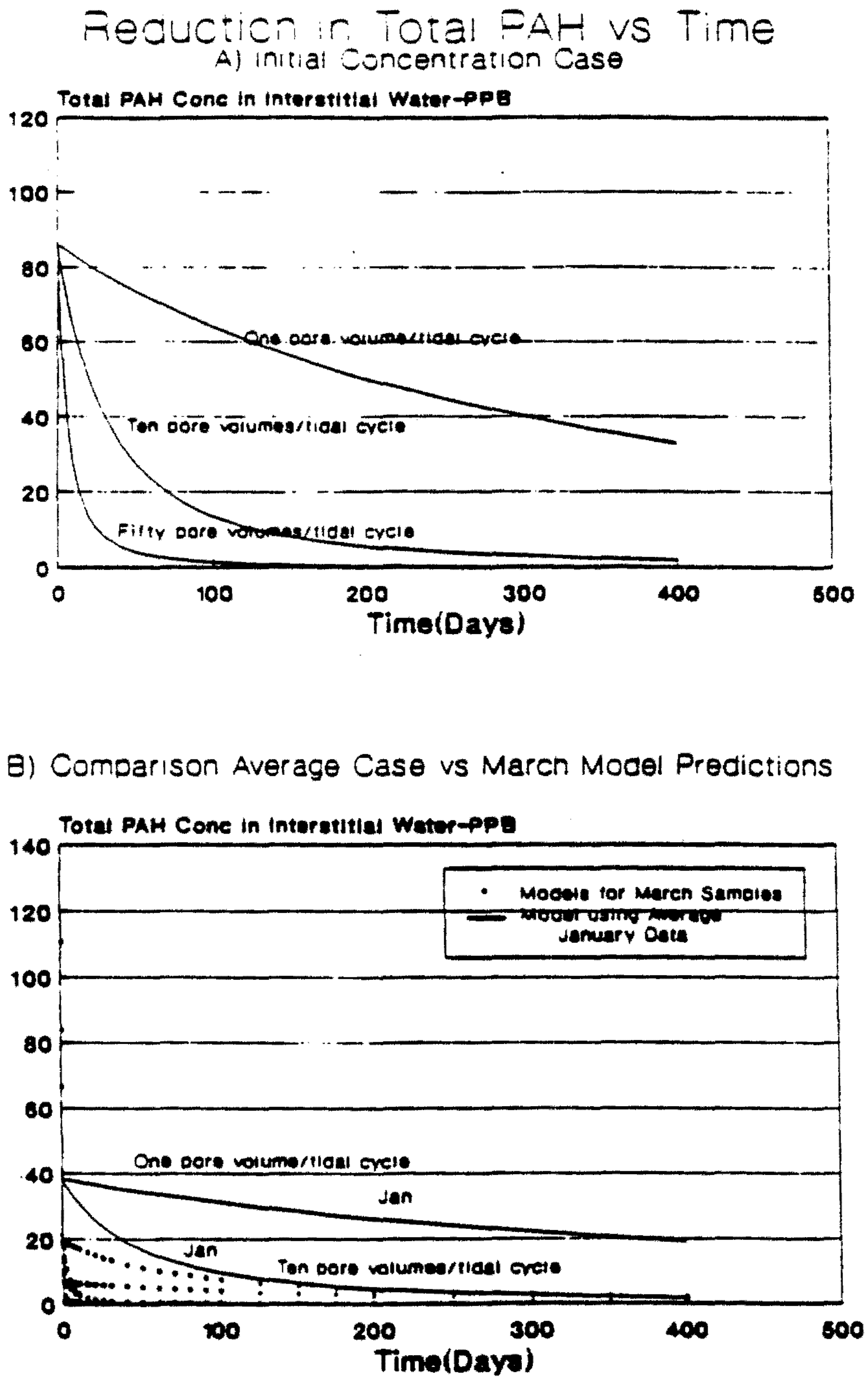

Figure I-33. Model predictions of concentrations of total PAHs in pore water in contact with intertidal deposits of subsurface oil on shores of Prince William Sound: (A) the initial concentrations case; and (B) comparison of average case and March case.

volumes/tidal cycle, the concentration drops to about 50 ppb after 20 days or 400 pore volumes of water. After 50 days, the concentration of total PAHs has dropped to about 30 ppb.

For the case of 50 pore volumes/tidal cycle, the concentration of total PAHs in pore water drops to 20 ppb within two weeks and continues to fall rapidly. On the other hand, for the case of 1 pore volume/tidal cycle, the concentration drops slowly, reaching about 65 ppb in 100 days.

Average-Case Calculation. Predicted concentrations of total PAHs for average case conditions are lower than those for the initial-case conditions (Figure I-33). The predicted initial concentration of total PAHs in pore water is about 37 ppb. The lower concentrations are due to the fact that the oil is more weathered and has lower concentrations of leachable PAHs. The rate of decrease in the concentration of total PAHs in the leach water is greater for the average case than for the initial case conditions. This is due to the lower oil loading on the subsurface sediments in January than earlier as a result of natural beach cleaning by fall and winter storms.

March-Case. The model was run using the data for the concentrations of PAHs in sediments collected in March 1990. The predicted concentrations of total PAHs in pore water are shown in Figure I-33 as dots. As expected, because of continued weathering and decreases in concentrations of oil remaining in subsurface sediments, the predicted concentrations of total PAHs in pore waters in equilibrium with the subsurface oily sediments are lower than in the January case. Predicted concentrations of total PAHs in pore water generally are below 20 ppb.

Field Observations. During the field survey in early March, pore water samples were collected at the same locations and depths as subsurface sediment samples. The results of analysis of the pore water samples for PAHs were used to validate model predictions of PAH concentrations in pore water.

The results, in general, support the premise that model predictions are conservative and that concentrations of PAHs in interstitial water in equilibrium with subsurface oil deposits are low (Table I-3). A total of eleven pairs of subsurface sediment and pore water samples were analyzed. Suspended particles or flocs were removed from the pore water samples by

Table I-3

Concentrations of Total PAHs in Pore Water and Sediments, and of TPH in Subsurface Sediments Collected From Several Oiled Shorelines in Prince William Sound in March 1990

Location	Site/Pit	PAH			TPH
		Sediment mg/kg	Water ng/L	Oil mg/kg	Sediment mg/kg
Foul Pass	AP-4/1	0.02	63.5	1621	12.4
Passage Pt.	AP-5/3	0.01	63.1	5347	2.25
	79.1*				
Latouche Bay	AP-12/1	0.07	709.9	836	82.7
Sleepy Bay	AP-13/1	32383	14293	5684	5697
Sleepy Bay	AP-13/4	0.14	124	3261	43.6
Sleepy Bay	AP-13/6	0.14	3821	2937	48.3
Latouche	AP-14/1	17899	10715	3271	5472
Latouche	AP-14/2	0.05	119.9	1693	32
NW Evans I.	AP-15/2	1.06	198.4	2684	394.2
NW Evans I.	AP-15/EB**		148.1		
S Elrington I.	AP-18/2	0.14	546	10543	13.2
Reference Site	AP-21C/1	0.0005	118.9	662	0.47
Blank		0.002	20.4		0.004
Prudhoe Bay Crude Oil				16556	

*Unfiltered sample

**Equipment blank

settling or filtration. Concentrations of PAHs in the water (presumably in solution or colloidal suspension) were uniformly low. Only three samples of pore water contained more than 1 ppb total PAHs in solution. Six of the eleven samples had concentrations below those measured in the equipment blank.

In May 1990, additional "pore water" samples were collected and analyzed for trace level PAHs by a compound-specific GC/MS method. The samples were collected at the sediment-water interface during falling tides and do not represent pore water by normal definition, in that they were diluted by the tidal water flushing through the coarse substrate. Replicate samples were collected at the middle and lower substrate. The results shown in Table I-4 suggest that few PAHs are being leached out of the beach with the action of normal tides. The concentration values ranged from below detection limits (approximately 0.005 ng/ml) to a maximum value of less than 0.020 ng/ml for any individual compound at either site sampled. These values are very low, and while not true pore water, reflect the level of potential exposure to organisms living in the intertidal zone.

Figure I-34 is a comparison of the concentrations of total PAHs in pore water predicted by the model and those actually measured in pore water samples collected in March. For the two samples with concentrations of TPH greater than 5,000 ppm (AP-13-1 and AP-14-1), the model predicts concentrations of total PAHs reasonably close to actual measured concentrations.

Where concentrations of TPH in the sediments are low, PAH concentrations in the oil tend to be relatively high, and the model tends to predict initial concentrations of total PAHs in the pore water higher than those actually measured. For such samples, the model predicts a very rapid drop in the concentration of PAHs in the pore water, as evidenced in Figure I-34 by the drop in predicted concentrations between day 0 and day 1.

Figure I-33 shows in the top series of dots the concentrations of total PAHs predicted by the model in pore water in equilibrium with the two sediment samples with the highest concentrations of TPH. Because the initial model predictions and observed concentrations are similar for these samples, it is reasonable to expect rates of decrease in the concentrations of

Table I-4. Concentration of Specific Compounds in Pore Water Collected at Candidate Beaches during May 1990.

GC/MS QUANT RESULTS					
SAMPLE LOCATION	KN405	KN405	KN405	KN405	KN405
REP. #	1	2	3	1	2
TIDAL ZONE	MI	MI	MI	LI	LI
SAMPLE TYPE	WATER	WATER	WATER	WATER	WATER
DATE SAMPLED	6/6/90	6/6/90	6/6/90	6/6/90	6/6/90
COMPOUND	(ng/ml)	(ng/ml)	(ng/ml)	(ng/ml)	(ng/ml)
NAPHTHALENE	ND	ND	ND	ND	0.011
C-1 NAPHTHALENE	ND	ND	ND	ND	0.006
C-2 NAPHTHALENE	0.008	0.012	0.012	ND	0.008
C-3 NAPHTHALENE	ND	0.005	0.005	ND	0.007
C-4 NAPHTHALENE	0.006	0.011	0.011	0.007	0.013
FLUORENE	ND	ND	ND	ND	ND
C-1 FLUORENE	ND	0.014	0.014	ND	0.010
C-2 FLUORENE	ND	0.006	0.006	ND	ND
C-3 FLUORENE	ND	0.008	0.008	ND	0.005
DIBENZOTHIOPHENE	ND	ND	ND	ND	ND
C-1 DIBENZOTHIO.	ND	0.007	0.007	ND	0.005
C-2 DIBENZOTHIO.	0.010	0.013	0.013	0.007	0.008
C-3 DIBENZOTHIO.	0.010	0.013	0.013	0.012	0.008
PHENANTHRENE	ND	ND	ND	ND	ND
C-1 PHENANTHRENE	ND	ND	ND	ND	ND
C-2 PHENANTHRENE	ND	0.006	0.006	ND	ND
C-3 PHENANTHRENE	ND	0.006	0.006	0.006	ND
NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
C-1 NAPHTHOBENZOTHIO.	ND	0.006	0.006	ND	ND
C-2 NAPHTHOBENZOTHIO.	ND	0.010	0.010	ND	ND
C-3 NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
FLUORANTHENE	ND	ND	ND	ND	ND
PYRENE	ND	ND	ND	ND	ND
C-1 PYRENE	ND	ND	ND	ND	ND
C-2 PYRENE	ND	ND	ND	ND	ND
BENZO(a)ANTHRACENE	ND	ND	ND	ND	ND
CHRYSENE	ND	ND	ND	ND	ND
C-1 CHRYSENE	ND	ND	ND	ND	ND
BENZO(b)FLUORANTHENE	ND	ND	ND	ND	ND
BENZO(e)PYRENE	ND	ND	ND	ND	ND
BENZO(a)PYRENE	ND	ND	ND	ND	ND
PERYLENE	ND	ND	ND	ND	ND
INDENO(1,2,3-cd)PYR.	ND	ND	ND	ND	ND
DIBENZO(a,h)ANTHR.	ND	ND	ND	ND	ND
BENZO(g,h,i)PERY.	ND	ND	ND	ND	ND

est. det. limit 0.005 ng/mL All values are valid to 2 significant figures only.

Table I-4. Cont.

GC/MS QUANT RESULTS					
SAMPLE LOCATION	KN405	LA18	LA18	LA18	LA18
REP. #	3	1	2	3	1
TIDAL ZONE	LI	MI	MI	MI	LI
SAMPLE TYPE	WATER	WATER	WATER	WATER	WATER
DATE SAMPLED	6/6/90	6/8/90	6/8/90	6/8/90	6/8/90
COMPOUND	(ng/ml)	(ng/ml)	(ng/ml)	(ng/ml)	(ng/ml)
NAPHTHALENE	0.013	0.011	0.005	ND	ND
C-1 NAPHTHALENE	0.007	0.007	ND	ND	ND
C-2 NAPHTHALENE	0.007	0.007	ND	ND	ND
C-3 NAPHTHALENE	ND	0.006	ND	ND	ND
C-4 NAPHTHALENE	0.011	ND	ND	0.006	0.006
FLUORENE	ND	ND	ND	ND	ND
C-1 FLUORENE	ND	ND	ND	ND	ND
C-2 FLUORENE	ND	ND	ND	ND	ND
C-3 FLUORENE	ND	ND	ND	ND	ND
DIBENZOTHIOPHENE	ND	ND	ND	ND	ND
C-1 DIBENZOTHIO.	ND	ND	ND	ND	ND
C-2 DIBENZOTHIO.	0.006	ND	ND	ND	ND
C-3 DIBENZOTHIO.	0.008	ND	ND	ND	ND
PHENANTHRENE	ND	ND	ND	ND	ND
C-1 PHENANTHRENE	ND	ND	ND	ND	ND
C-2 PHENANTHRENE	ND	ND	ND	ND	ND
C-3 PHENANTHRENE	ND	ND	ND	ND	ND
NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
C-1 NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
C-2 NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
C-3 NAPHTHOBENZOTHIO.	ND	ND	ND	ND	ND
FLUORANTHENE	ND	ND	ND	ND	ND
PYRENE	ND	ND	ND	ND	ND
C-1 PYRENE	ND	ND	ND	ND	ND
C-2 PYRENE	ND	ND	ND	ND	ND
BENZO(a)ANTHRACENE	ND	ND	ND	ND	ND
CHRYSENE	ND	ND	ND	ND	ND
C-1 CHRYSENE	ND	ND	ND	ND	ND
BENZO(b)FLUORANTHENE	ND	ND	ND	ND	ND
BENZO(e)PYRENE	ND	ND	ND	ND	ND
BENZO(a)PYRENE	ND	ND	ND	ND	ND
PERYLENE	ND	ND	ND	ND	ND
INDENO(1,2,3-cd)PYR.	ND	ND	ND	ND	ND
DIBENZO(a,h)ANTHR.	0.006	ND	ND	ND	ND
BENZO(g,h,i)PERY.	ND	ND	ND	ND	ND

est. det. limit 0.005 ng/mL

Table I-4. Cont.

GC/MS QUANT RESULTS			
SAMPLE LOCATION	LA18	LA18	EQ. BLANK
REP. #	2	3	NA
TIDAL ZONE	LI	LI	NA
SAMPLE TYPE	WATER	WATER	WATER
DATE SAMPLED	6/8/90	6/8/90	6/8/90
COMPOUND	(ng/ml)	(ng/ml)	(ng/ml)
NAPHTHALENE	ND	0.005	ND
C-1 NAPHTHALENE	ND	ND	ND
C-2 NAPHTHALENE	ND	ND	ND
C-3 NAPHTHALENE	ND	ND	ND
C-4 NAPHTHALENE	ND	0.005	ND
FLUORENE	ND	ND	ND
C-1 FLUORENE	ND	ND	0.006
C-2 FLUORENE	ND	ND	ND
C-3 FLUORENE	ND	ND	ND
DIBENZOTHIOPHENE	ND	ND	ND
C-1 DIBENZOTHIO.	ND	ND	ND
C-2 DIBENZOTHIO.	ND	ND	ND
C-3 DIBENZOTHIO.	ND	ND	ND
PHENANTHRENE	ND	ND	ND
C-1 PHENANTHRENE	ND	ND	ND
C-2 PHENANTHRENE	ND	ND	ND
C-3 PHENANTHRENE	ND	ND	ND
NAPHTHOBENZOTHIO.	ND	ND	ND
C-1 NAPHTHOBENZOTHIO.	ND	ND	ND
C-2 NAPHTHOBENZOTHIO.	ND	ND	ND
C-3 NAPHTHOBENZOTHIO.	ND	ND	ND
FLUORANTHENE	ND	ND	ND
PYRENE	ND	ND	ND
C-1 PYRENE	ND	ND	ND
C-2 PYRENE	ND	ND	ND
BENZO(a)ANTHRACENE	ND	ND	ND
CHRYSENE	ND	ND	ND
C-1 CHRYSENE	ND	ND	ND
BENZO(b)FLUORANTHENE	ND	ND	ND
BENZO(e)PYRENE	ND	ND	ND
BENZO(a)PYRENE	ND	ND	ND
PERYLENE	ND	ND	ND
INDENO(1,2,3-cd)PYR.	ND	ND	ND
DIBENZO(a,h)ANTHR.	ND	ND	ND
BENZO(g,h,i)PERY.	ND	ND	ND

est. det. limit 0.005 ng/mL

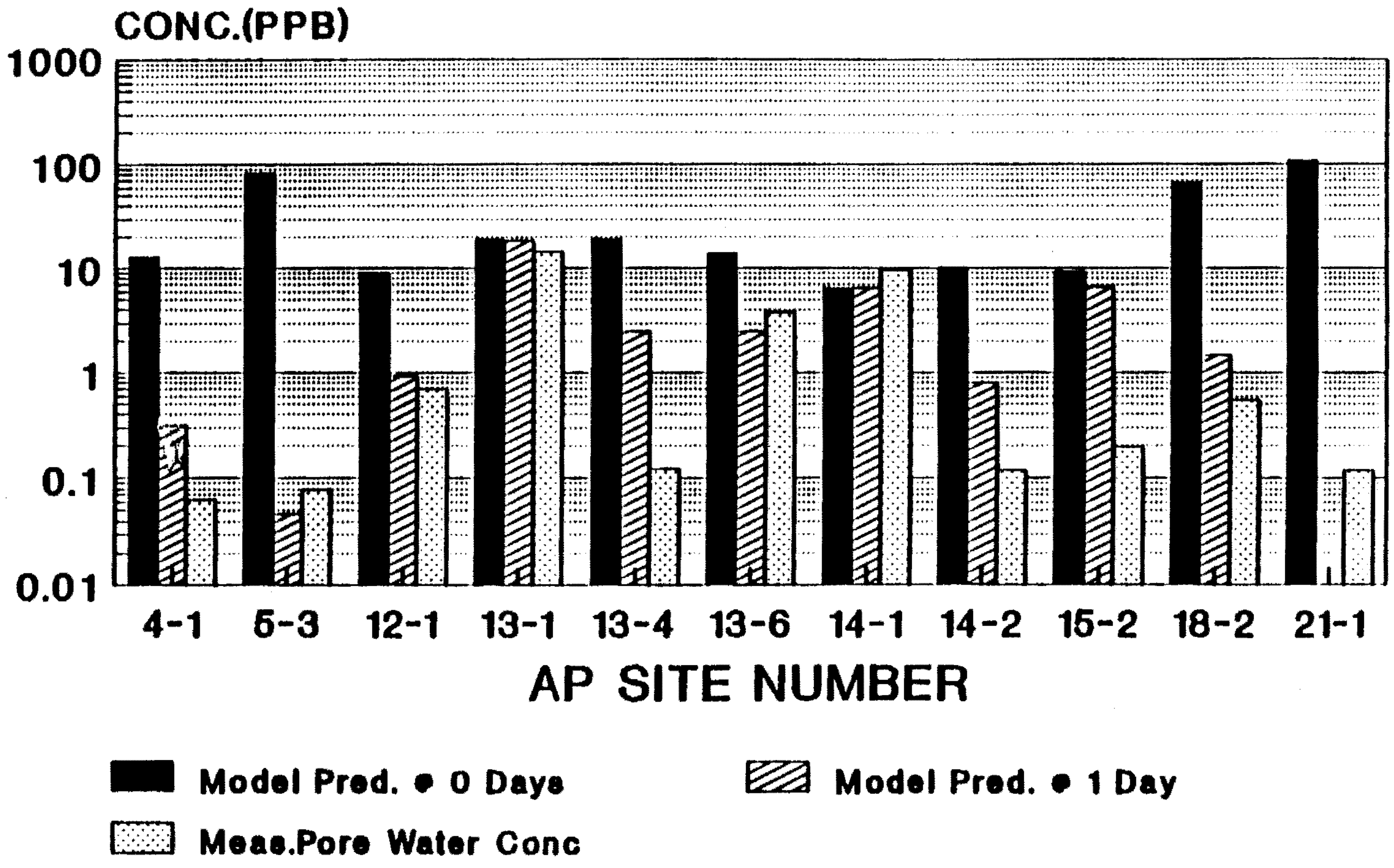

Figure I-34. Comparison of concentrations of total PAHs in pore water of sediments collected from oiled shores in Prince William Sound in March, 1990, predicted by the model and actually measured in pore water samples.

PAHs in pore water similar to these projections. The use of 10 pore volumes/tidal cycle in the model seems to predict changes over time in the concentrations of PAHs in pore water quite well. In the model, PAH concentrations in pore water approach zero in about 200 days. However, if the amount of flushing of the shore is significantly different from 10 pore volumes/tidal cycle, the predicted rate of decrease in the concentration of PAHs in pore water with time would change accordingly.

The concentrations of total PAHs in pore water samples in direct contact with oily subsurface sediments, predicted or measured, are below the concentrations of total PAHs of petroleum origin known to be acutely toxic to marine organisms. In addition, these initial concentrations will be diluted substantially as the sediment interstitial water containing PAHs mixes with clean interstitial water and as the mixed interstitial water drains from the beach and mixes with the shallow subtidal water. This dilution, which is likely to be substantial, will render the pore water completely nontoxic to marine organisms on the lower shore and in nearshore subtidal waters.

I.C.5. BEACH HYDRAULICS

Point Helen

Tidal Flushing. The sedimentology of the beach surface on this section is characterized by a coarse (pebble-cobble-boulder) and porous layer that is one or two particles thick. Water movement through this layer is unrestricted. Groundwater flow emerges through this layer in the lowest parts of the intertidal zone.

The subsurface is characterized by mixed sediments, with fine material (sands and granules) present in the interstices between the coarse material. Water movement and flow rates are restricted by the small size of the pore spaces.

The net effects of this two-flow system are that:

a) the surface armor layer is continuously washed by water (waves) during periods of tidal inundation and drys out when exposed,

b) the subsurface sediments generally remain wet,

c) in the zone of wave action, which is dictated by the tidal water level, uprush is followed immediately by infiltration, and

d) backflow following uprush, under the influence of gravity, occurs beneath the surface of the beach primarily at the interface of the surface coarse layer and the subsurface mixed sediments.

This section is therefore characterized as having an active surface zone, in which flushing takes place on a regular basis with each phase of the tidal cycle, and a less active subsurface zone with lower flushing rates.

Freshwater Runoff. No streams cross the section. There will be runoff from the backshore across and through the intertidal zone during periods of rainfall and snow melt.

Sleepy Bay

Tidal Flushing. These coarse (pebble-cobble-boulder) sediments have few fines, so that the interstitial spaces are open and water flow is unrestricted.

Freshwater Runoff. There is year-round, non-channelized freshwater runoff in the west section of the unit and there is a year-round anadromous stream within this section. There will be runoff from the backshore across and through the intertidal zone during periods of rainfall and snow melt.

I.C.6. TREATMENT HISTORIES

Point Helen (KN-405; AP-10; N-1)

This segment was surveyed by a SCAT team on 26 June 1989. The U.S. Coast Guard reported treatment began on 6 July.

Treatment:

a) Deluge header hose flood.

b) Cold water, high pressure wash.

c) Warm/hot water, moderate pressure wash.

d) Hot/steam water, high pressure wash.

e) MAXI Barge.

f) OMNI Boom.

g) Bioremediation.

Sleepy Bay (LA-18; AP-13C; N-18)

This segment was surveyed by a SCAT team on 17 June 1989. The U.S. Coast Guard reported treatment began on 22 June.

Treatment:

a) Deluge header hose flood.

b) Cold water, high pressure wash.

c) Warm/hot water, moderate pressure wash.

d) Back hoe tractor was used to move oiled sediments adjacent to the anadromous stream where they were scheduled to be washed and then returned to the stream bank. When the stream was diverted, it drained subsurface, at its seaward end. Salmon spawning season was nearing and ADF&G stopped tractor activity. Clean material from the lower portion of the intertidal zone was used to reline the stream banks.

e) Manual labor crews worked on shoreline with shovels, rakes, and chainsaws.

I.C.7. BIOLOGICAL CONDITIONS

Sources of Data and Information

There are no quantitative baseline data with which to describe the intertidal biological communities that inhabited the candidate beaches before the spill. Without these data it is difficult to estimate the characteristics of the intertidal communities that would likely reinhabit the beaches following recovery from the spill. There are data available from other sites within Prince William Sound that were surveyed before the spill (Rosenthal et al., 1982; O'Clair and Zimmerman, 1987; Feder and Bryson-Schwafel, 1988; Juday and Foster, 1990). They were used qualitatively and descriptively to provide a general portrait of the types of organisms that would be expected at these beaches.

These general data were augmented with information assembled from surveys and observations made after the spill at the candidate beaches by members of the NEBA team. Observations of the presence of oil, the geomorphology of the beaches, and presence of selected biota at the candidate sites were made in the Spring Survey Assessment Team (SSAT) surveys (4 April 1990). Members of the NEBA team visited the Sleepy Bay segments in May, early June, and late June. They visited the Point Helen segment in May. During these surveys, observations were made and recordings taken on the density of plants and animals and their vertical location on the beach.

In Sleepy Bay there were some differences between the descriptions of the biota observed in May and those observed in June by the NEBA team members. These differences are discussed below in the segment descriptions. Causes of these discrepancies are speculative: They may reflect biological changes that occurred over a period of over one month, they may reflect differences in surveying methods of the team members, or they may reflect the heterogeneity of the segments.

General Description

The candidate beaches generally consist of mixed pebbles, cobbles, and boulders, with minor components of granules and sand. The larger cobbles and boulders are sufficiently stable and large to support epibenthic species and the smaller materials surrounding and underneath the cobbles and boulders support motile and infaunal species. The occurrence and abundance of intertidal species is patchy within any single beach. The patchiness is a result of many controlling factors, including the distribution of stable cobbles and boulders, exposure to breaking waves, tidal height (vertical position on the beach), proximity to freshwater, sunlight/shading, desiccation, predation, and the composition and stability of the beach materials.

The combined effects of these factors vary from beach to beach and within any single beach. The result is that the composition of intertidal communities can differ a great deal among beaches and within the length of any beach.

Rosenthal et al. (1982) listed the most frequently occurring species for semi-protected, intertidal, mixed coarse sediments in Prince William Sound as *Monostroma / Ulva*, *Fucus distichus*, *Balanus* spp., unidentified acmaeids (limpets), and *Pycnopodia helianthoides* (sea star). Also relatively common were *Zostera marina* (eelgrass), *Desmarestia aculeata* (alga), *Echiurus echiurus* (echiurid worm), *Hemigrapsus oregonensis* (shore crab), *Pagurus hirsutiusculus* (hermit crab), *Littorina sitkana* (snail), *Mytilus edulis* (mussel), *Protothaca staminea* (clam), *Saxidomus giganteus* (clam), and *Evasteria troschelii* (sea star). Most of these species were observed on the post-spill surveys of the candidate beaches; the echiurid worms, clams, shore crabs, and sea stars generally were missing or relatively rare.

In 1986, Juday and Foster (1990) observed 96 animal species and 39 plant species in intertidal rocky areas of Green Island in lower Prince William Sound. After the spill, 37 of the animal species and six of the plant species were not observed. Fourteen animal species and two plant species were found in 1989 that had not been seen in 1986. Among the more common species observed at Green Island were *Balanus* spp., *Mytilus* spp., *Fucus* spp., littorine snails, and *Leathesia* spp. in the upper to mid-tidal zones. The authors attributed some of the apparent differences in the species that were present before and after the spill to differences in their sampling efforts and taxonomic skills.

Rosenthal et al. (1982) described the very distinct zonation of intertidal organisms in mixed coarse habitats of Prince William Sound. Species richness and total abundance of organisms increased from the high intertidal to lower intertidal zones. The species that were dominant also changed between vertical zones. In surveys performed in May 1990 for this NEBA, zonation was very distinct at Point Helen and less so in the Sleepy Bay segments.

Rosenthal et al. (1982) concluded, after examining many sites in Prince William Sound, "that the predictability of species composition is rather low" within distinct habitat types. Also, O'Clair and Zimmerman

(1987) reported relatively low similarity in species composition among 28 rocky substrate sites around the perimeter of the Gulf of Alaska. Therefore, interpolation of data from other sites with the same habitat type to the candidate beaches may have a low degree of accuracy. Many subtle differences in controlling environmental factors can have marked effects upon the species composition among areas.

As will be described below, the species observed in different visits after the spill at the candidate sites generally did not differ remarkably from those recorded by Juday and Foster (1990) and Rosenthal et al. (1982) at other sites before the spill. Despite the observations of relatively low predictability of species composition from site to site, there appears to be a rather predictable "core group" of intertidal species at most locations. This assemblage of species generally consists of one to several species of barnacles, the mussel *Mytilus edulis*, littorine snails, limpets, *Fucus*, and several species of other brown and green algae attached to rocks. Several species of polychaetes, clams, gammarid amphipods, hermit crabs, and sea stars comprise the core group of species common among or under the coarse clasts. The abundance of these organisms differs among the vertical zones of the beaches.

<u>LA-17, Sleepy Bay, Latouche Island</u>

Segment LA-17 is located to the east of the candidate site, LA-18. The substrate consists of a very heterogeneous mixture of large, angular boulders; angular, flattened boulders; cobbles; gravel; and small amounts of coarse sand and shell fragments. At the southeast end of the segment, there are rock outcrops that were not surveyed for this report. The following descriptions were based upon surveys performed in May, early June, and late June of the portions of the beach segments that consisted mainly of unconsolidated materials.

The lower tide zone included cobble which, in May, was covered with a heterogeneous algal assemblage, sparse to thick in places. *Ulva*, *Enteromorpha*, *Fucus*, and filamentous brown and green algae were dominant. By late June, the abundance of opportunistic green algae was low. Small *Fucus* and other brown algae were abundant in the lower intertidal zone wherever large rocks occurred. Patchy groups of large barnacles, many dead serpulid worms, and mussels occurred in this zone. Detached and empty mussel shells were present in May in numbers equivalent to those that were alive, but dead mussels were not observed in June. One dead *Mytilus* adult was observed in May that still had the internal soft parts intact, indicating that it had died recently. Limpets occurred in densities of up to about 20 m^2 and, in May, about 50% of the shells were dead and empty. This rate of mortality was not observed in June. Littorine snails were not observed in May, but were common in June, both on the surface of rocks and underneath rocks (often with egg masses). In late June, the abundance of these snails was still high, but the abundance of the egg masses had decreased. Young-of-the-year barnacles, mussels and limpets were common in May and/or June on stable rocks, sometimes in dense patches. Animals living among and under the boulders in May were: isopods (probably *Idotea*, 2-5/m^2 and probably *Isosphaeroma*, 20/m^2), hermit crabs (abundant, 10/m^2), mussels, and a few gunnels. In June the common animals included limpets (several species), amphipods (*Gammarus oregonensis* and others), isopods (*Idotea wosnesenskii* and others), nereid and other polychaetes, hermit crabs, and spawning gunnels. The egg masses of gunnels were not found in late May. As many as seven species of sea star and the snail *Nucella lamellosa* were observed in June, but none of these species was seen in May.

As observed in other segments and as reported by Rosenthal et al. (1982), species diversity and organism density decreased upslope in the intertidal zone. However, on steeper portions of the segment, the density and diversity of biota seemed to have been maintained higher up the shore than elsewhere, probably due to the steeper gradient and more shaded and sheltered aspect of the beach.

Much of the mid-tidal zone consisted of very large, angular black cobbles and boulders overlying pebbles, and granules/sand. Attached to the boulders was a more sparse assemblage of the algae found in the lower tide zone. In May *Littorina* occurred in patches of approximately 20-50/m^2, and about 10% of the shells were dead and empty. Young-of-the-year *Fucus* and both young and old barnacles and mussels populated this segment of beach, sometimes in dense patches. Large terebellid worms, small gammarids

($10/m^2$), and hermit crabs ($5\text{-}10/m^2$) were found among pebbles under boulders. Limpets, gunnels, and a polychaete worm (*Nereis* or *Nephthys*) were rare in May and abundant in June. A very large rock outcrop located within the segment had a relatively dense assemblage of *Fucus*, barnacles, and mussels growing on it, some of which were small, young individuals.

Much of the upper zone consisted of large boulders, oiled cobbles and pebbles, and an underlying hash made of shells and coarse sand. About 10-50% of the barnacles in the upper zone were dead in May. Most of the larger rocks had a cover of diatom scum and, occasionally, some *Fucus*. Young *Fucus* plants were observed in June. Patches of mussels and *Littorina* ($20/m^2$) were located in this region. Oligochaetes (probably an Enchytraeid) were common in the sand in May (abundances of up to $50/m^2$), but were less common in June. In June, barnacles (both recent recruits and adults), *Mytilus*, *Littorina*, amphipods, limpets, and *Pagurus* were encountered, well above the vertical height in which they are usually found on other shores

Overall, this segment has a moderately dense community of plants and animals. The species found in the segment generally are similar to those found elsewhere in Prince William Sound. Reflecting the heterogeneous geomorphology of the segment, the biota are very patchy in distribution, as is typical of the beaches of Prince William Sound. The main consistent trend was the increase in species density and diversity downslope. The upper tidal zone that remains most heavily oiled is mostly depauperate of biota, but the depauperate condition of the upper tidal zone is typical of that observed in unoiled shores throughout Prince William Sound (Rosenthal et al., 1982). The transition in biological community composition among the low-, mid-, and upper tidal zones is gradual, whereas at Point Helen, the transition between the lower and mid-tidal zones is very abrupt. Some species were noticeable in their absence in May, including predatory sea stars (*Pycnopodia helianthoides*) and whelks (*Thais*, ne *Nucella*). However, they were observed in June. An unusual population of oligochaetes (probably an Enchytraeid) has invaded the upper tidal zone sediments, presumably consuming the bacteria that are degrading the oil.

LA-18. Sleepy Bay, Latouche Island

The 4 April 1990 SSAT inventory for Site LA-18 included moderate to sparse abundances of barnacles in the mid-tide zone, mussels were dense to moderate to sparse, gastropods from dense to moderate to sparse to rare, and *Fucus* classified as dense to moderate to sparse in the mid-tide zone. Filamentous green algae were in the mid and low tide zones, and amphipods were dense under the cobble with two kinds of gastropods. Low occurrence of mortality in mussels was reported by SSAT on 4 April 1990. The biota in this segment were very patchy, reflecting the heterogeneous distribution of stable substrates.

The morphology of this segment has changed since the spill because of the meandering of an anadromous steam that bisects the segment. The lower part of the stream, including a small delta, has moved considerably since the site was first surveyed last year.

As observed in May and June of 1990, the substrate is very heterogeneous, grading from predominantly small cobbles on the stream banks to large boulders and rock outcrops in the western part of the segment. There is a small sand/silt beach at the low tide line in the easternmost part of the segment where it borders with segment LA-17. Generally, there was very little fine-grained sand and gravel in the middle and upper tidal zone. However, in some locations, fine-grained sediments were abundant below the surface cobbles. Much of the fine sediments were plate-like, probably derived from natural pulverization of the shale that was an abundant component of the cobbles and boulders on the beach surface.

The biota observed in LA-18 in May 1990 consisted mainly of the same organisms observed in LA-17. The biota were most sparse in the upper tidal zone, and rapidly increased in diversity and density downslope. Organisms that usually occurred in patches in the middle tidal zone often occurred in continuous dense populations in the lower tidal heights. An eelgrass bed occurs in the subtidal zone. The incidence of dead animals observed in LA-18 in May was somewhat lower than that observed in LA-17. One moribund *Protothaca* clam was observed in May upon the beach surface. Many dead, necrotic fronds of detached *Fucus* were found among the rocks.

In May there were many species of brown and green algae, limpets, barnacles, mussels, and serpulid worms on the rocks with isopods, hermit crabs, mussels, and a few clams and gunnels under the rocks in both the mid- and lower tidal zones. The mid-tidal zone community consisted mainly of green and brown algae (including *Ulva* and *Fucus*), *Littorina* spp., barnacles, mussels, and a few limpets on the rocks and isopods; gammarid amphipods; and terebellid worms and gunnels under the rocks. Their abundance generally increased downslope. Egg masses and adults of *Littorina* were evident in May and very abundant in June. The density of egg masses decreased late in June. Less common taxa included some bryozoans, polychaetes, and hermit crabs.

No *Nucella* or sea stars were observed in May in association with small rocks, cobbles, and gravel in the eastern portion of the segment that was surveyed. One sea star was observed in early June. In late June, sea stars (particularly *Pisaster ochraceus*) were fairly abundant upon or near large boulders and rock outcrops in the western part of LA-18. In addition, a few *Pycnopodia* were observed under overhanging rock outcrops in the lower intertidal zone. *Nucella* were observed also in June, again mainly in association with large rock outcrops in the eastern portion of the segment.

Generally, the plants and animals observed in June were similar to those seen in May in all vertical zones. A few small amphipods were observed under rocks and some barnacle settlement was evident on the larger more stable boulders. In locations where oil was present in patches on the surface, no biota, except oligochaetes, were present under or among the surface cobbles. However, large rocks that had patches of tar also supported mixed age groups of barnacles. Many of the barnacles covered by the tar were dead, but an almost equal number were alive. Other animals observed in June in the middle and lower tidal zones included hermit crabs, amphipods, limpets, isopods, gunnels (some with eggs in early, but not late June), *Nereis*, at least one other species of polychaete, and bryozoans. Large Nereid worms and a few other taxa of polychaetes were encountered frequently under rocks in the lower shore where the underlying substrate was fine-grained. Very small amphipods, limpets, and littorine snails were more abundant on and under rocks in the middle and lower intertidal zones in late June than in early June.

In June serpulid worms (*Spirorbis* sp.) were dense, and mussels were patchy and sparse on and among the rocks and dense in crevices of large rock outcrops. About 1% of the mussels were dead and the shells were empty. *Ulva* and filamentous green algae were present on stable rocks.

Many of the brown algae on the lower shore and in the shallow subtidal zones were smaller than usual and the distal parts of the fronds of many of the plants were bleached white. However, these damaged plants were accompanied by abundant young plants and newly recruited littorine snails, amphipods, mussels, and barnacles.

In May the pebbles and small cobbles in the middle and upper zones near the mouth of the creek were relatively depauperate. They generally were inhabited by only a thin film of a filamentous green alga (probably *Monostroma*) and some *Littorina*.

The biota were sparse in the upper tidal zone, consisting mainly of diatoms, a few mussels, barnacles, and *Littorina* on the cobble surfaces and relatively abundant oligochaetes and a few amphipods under the cobbles. The depauperate nature of the upper intertidal zone is typical of that encountered throughout Prince William Sound

Overall, the biological community of this segment was very patchy, differing considerably in both density and diversity along the segment and across vertical horizons within the segment. The upper tidal zones were more depauperate than the lower zones. Parts of the lower tidal zone were relatively densely populated by a number of marine plants and animals. Some species, including predator sea stars and snails, were either absent or present in very low density among unconsolidated materials in May, but they were sparse to common in abundance near or on large rock outcrops in June. A relatively dense population of oligochaetes had invaded the upper tidal zone sediments, presumably consuming the bacteria that are degrading the oil.

Sleepy Bay: A Summary

Without the benefit of pre-spill, baseline information, it is difficult to estimate accurately the degree to which the biota of Sleepy Bay have recovered from the effects of the spill and subsequent cleanup efforts. The composition of the "normal," pre-spill intertidal community is unknown.

The intertidal biota of Sleepy Bay, however, are not dramatically different than the communities previously described for similar habitats elsewhere in Prince William Sound. The group of species that would be expected to be dominant generally were present, often abundant, in Sleepy Bay. The numbers of species found generally approximated the numbers of species observed elsewhere in the Sound. The patchiness and heterogeneity both along and across the beaches were as described for other beaches. The differences in biota observed in May and June are not unusual: They could be attributable to the natural variability in the communities over time and space, to differences in the survey methods of the investigators, to differences in the habitat types that were surveyed, or they could reflect stages in the recovery of the biota in the beaches.

Mortalities among mussels, barnacles, limpets, snails and macroalgae that were observed in May could be attributable to a number of factors. A battery of natural factors, such as cold winter weather, freshwater runoff in the spring, disease, predation, and old age, could account for some percentage of the mortalities. The oil and subsequent cleanup efforts probably contributed to some of the mortalities along with these natural factors. The relatively persistent hard shells of animals such as mussels, limpets, and barnacles would remain for many months or years following the death of the organisms. A few bivalves that were either moribund or dead and still contained their internal soft parts in May suggest that some animals were dying in the spring, but the cause of these deaths is unknown and the number of animals observed dying was extremely small. Some adult mussels that survived the spill had deformities in shell produced some time ago along the posterior fringes of their shells, but more recent accretions of shell material appeared normal.

The majority of the Sleepy Bay beaches were inhabited by intertidal organisms in May and June. Only a very few isolated areas in the upper tidal zone where surface oil remained in high concentrations were still azoic. In some areas where surface oil under large rocks was relatively highly concentrated, very few infaunal organisms existed or only the infaunal oligochaetes were present. The presence of abundant populations of the oligochaetes is unusual and probably a result of the spill.

The presence of large adult invertebrates in the spring indicates that many animals, particularly those attached to large rocks and those in the lower tidal zone, survived the effects of the spill and cleanup. Large adult barnacles and mussels were often attached to large rocks.

In addition, there is considerable evidence that the biota of the Sleepy Bay beaches are recovering. The recolonization of the beaches by all of the major macroscopic taxa is in progress as evidenced by the presence of young individuals and/or egg masses of some species. It appeared that barnacles recolonized some of the rocks as early as last fall and very young barnacles were common in most areas in May and June. Young-of-the-year mussels, littorine snails, limpets, *Fucus*, and other plants were common in most areas. Young littorine snails were abundant late in June in areas where they were absent or sparse in May. Highly mobile predatory snails and sea stars were migrating across the beaches from lower zones, along the beaches from adjacent areas or from the large rocks that were not heavily impacted.

The presence in the middle and lower tidal zones of several species of invertebrates (amphipods, isopods, young molluscs) generally considered to be sensitive to the effects of oil suggests that the residual subsurface oil in these zones is not having a marked effect upon these organisms. The biological communities in the lower shore are relatively dense and diverse and include many of these sensitive species.

However, in the upper tidal zone that was likely most depauperate before the spill and was most heavily impacted by the spill, some signs of the effects of residual oil remain. Amphipods generally were missing or rare in patches that remained oiled. Only oligochaetes occurred abundantly in some oiled patches and, in a few very highly oiled patches, there were no living organisms evident. Whether these effects are attributable to only the surface oil or to both surface oil and subsurface oil residues is unknown.

Overall, the intertidal biological community of Sleepy Bay is relatively healthy and clearly showing signs of recovery. Many of the species that would be expected to inhabit the area are present. All of the major trophic levels are represented. The expected time required for full biological recovery is estimated in Section II.A.2.

KN-405, Point Helen, Knight Island

The substrate along this segment is much more homogeneous than that encountered in Sleepy Bay. Large rounded cobbles and boulders are very common and occur consistently along the segment. The biota of this segment are also much more homogeneous than those observed in Sleepy Bay. The most striking feature of the intertidal community in this segment is the *Alaria* assemblage in the lower tidal zone.

The lower tide zone consisted of large, weathered, rounded boulders and cobble overlying gravel and pebbles. The intertidal community in this zone is dominated by a very dense, complex algal mat, consisting mainly of the brown alga *Alaria marginata* (at the -2.0-foot tide level). *Porphyra*, *Enteromorpha*, and *Ulva* also occur in this mat, but were much less abundant than *Alaria*. Barnacles and mussels were present, but not abundant (S. Stoker, April 1990). Some species other than *Alaria marginata* gradually diminished in abundance up to mid-tidal levels, forming a thin carpet along with colonial diatoms. The transition between the cobbles and boulders with *Alaria* and those without *Alaria* is very distinct. Under the *Alaria* and boulders, gammarid amphipods, caprellid amphipods, cheilostome bryozoan colonies, and hermit crabs were abundant. Rare in numbers were oligochaetes, polychaetes, and the bryozoan *Bugula* sp. Many sea stars (*Pycnopodia helianthoides*), but no whelks (*Nucella*) were observed. Offshore, some harbor seals, a sea otter, and a bald eagle were observed.

The mid-tide zone was made up of large, round boulders with very little epifauna. Limpets, *Littorina*, *Enteromorpha*, and diatoms made up the sparse coverage. Under rocks, the gravel and sand matrix supported a few gammarids, gunnels, cheilostome bryozoans, and *Pycnopodia*. Egg masses from *Littorina* were apparent.

The sediments of the upper tide zone grade from small cobble/boulders to a gravelly storm berm. The beach materials are smaller material than they are in the mid-tidal zone (see Section I.C.2). There is a gradual transition between the biota of the mid-tidal zone and that of the upper zone on this beach. Oligochaetes and a few nemerteans within the coarse sand and pebbles are the only animals found in upper part of this zone. On the large monolithic rock within the segment, there was a sparse population of barnacles.

In summary, Point Helen is an exposed beach consisting of an armor of cobble and boulders, with an underlying layer of finer material. Sand is less common here than at Sleepy Bay. The beach is relatively broad, and there is a distinct *Alaria*/ algae assemblage starting abruptly at approximately the -2.0-foot tide level. The upper intertidal zone does not have observable epifauna, and the infauna consists largely of oligochaetes and some nemerteans. *Littorina* snails are numerous lower on the beach in the mid-tidal zone, and were reproducing in May 1990. Many egg masses were located on the undersurfaces of large cobbles or boulders. In the mid- and lower zones, new barnacle spat were numerous, and a film of green algae covered the tops of some of the larger rocks and boulders. No hermit crabs, *Nucella*, or sea stars were present in the middle and upper tidal zones. The sea stars normally would not be expected in the middle and upper zones in order to avoid desiccation at low tides. Hermit crabs and sea stars frequently occurred in the lower tidal zone among the dense algae.

Because of the lack of pre-spill baseline information, it is not possible to estimate the degree of progress being made in the recolonization of this beach segment. The very dense populations of plants in the lower tidal zone would suggest that this community was not affected by the spill. In contrast, the middle and upper zones remain very depauperate. It is unknown if this depauperate condition of the middle tidal zone is "normal."

I.C.8. HUMAN USES

The human use data for the segments analyzed for rock washing is based on information collected through public meetings specifically conducted to gather public uses, public use surveys, and input from public interest groups, residents, and user groups interested in Prince William

Sound. Site specific "head counts" of the number of subsistence use, recreation, and tourism visits are not available. The following descriptions of specific human uses for the study segments are a compilation on information from the Alaska Department of Natural Resources, Prince William Sound Area Plan, "Recreation and Tourism Element," "Fish and Wildlife Element" and "Cultural Resources Element" documents dated June 1987. Referenced also is the Prince William Sound Conservation Alliance "Inventory of Shorelines of Significant Importance to Recreation and Tourism Users" dated April 1990. Other information on cultural and subsistence uses within the specific segments was submitted by the village of Chenega (Gail Evanoff).

KN-405, Point Helen, Knight Island

Recreational uses of this segment are kayaking, camping, charter operators, scenic coastline, excellent fishing, hunting, and wildlife viewing. The Prince William Sound Conservation Alliance "Inventory of Shorelines of Significant Importance to Recreation and Tourism Users" ranks this segment as high use. This area is also used as a concentrated purse seine/gill net area. Subsistence activities include hunting, fishing, fish drying and smoking, and seal hunting. The residents of Chenega Village have strong historic cultural ties to the area of Point Helen.

LA-18, 19, Sleepy Bay, Latouche Island

Recreational uses of the segments are camping, and moderate hunting and sportfishing. The Prince William Sound Conservation Alliance "Inventory of Shorelines of Significant Importance to Recreation and Tourism Users" ranks this segment as moderate use. Historically, the people of Chenega have subsisted heavily off the beaches of Sleepy Bay. Gathering of gumboots, kelp, mussels, eggs, sea cucumbers, shellfish, and hunting of seal and birds were main subsistence activities as well as camping to dry and smoke fish.

II. ENVIRONMENTAL CONSIDERATIONS

II.A. BASE CASE: APPROVED 1990 TREATMENTS AND NATURAL CLEANSING

II.A.1 TRENDS IN OIL RESIDUE CONTENT OF SUBSURFACE SEDIMENT SAMPLES FROM PRINCE WILLIAM SOUND

Description of Database

A summary of the database of subsurface sediment samples from Prince William Sound collected during Exxon's winter shoreline monitoring program is provided in Table II-1, which lists monthly averages for each of 18 monitoring sites included in the winter program (counting Sleepy Bay central, east, and west separately). The "holes" in the table are indicative of gaps in the monthly monitoring schedule. No samples were taken in February.

While the monthly entries in Table II-1 show a generally downward trend, the direct comparison of monthly averages can be misleading, since a different subset of monitoring sites was visited each month. However,a valid comparison is provided by the last two columns of the table, where the "first" and "last" entries have been repeated for each site. Statistics for these two columns are given at the bottom of Table II-1, where the average, standard deviation, number of entries, and the estimated standard deviation of the average, are shown. The reduction in the averages from the first observations (about 4,500 ppm) to those of the last observations (about 1,400 ppm) is about 70%. Furthermore, these averages are large compared to their estimated standard deviations (about 1,600 and 800 ppm, respectively), which ensures that the observed trend is statistically significant.

These trends were obtained using all 250 samples in the database for sediment samples taken during the winter monitoring program and analyzed at Exxon's Corporate Research Laboratory in Clinton, New Jersey, with the following exceptions:

1) Samples of surface sediments (typically 0-5 or 0-10cm deep)

2) Bulk oil samples (e.g., oil film taken from a pit)

Table II-1

Summary of TPH Data for All Prince William Sound Sediment Samples From Exxon's Winter Monitoring Program
Monthly Averages of Subsurface Samples by Site

Site	Energy Level	Location	No.	SEP ppm	OCT ppm	NOV ppm	DEC ppm	JAN ppm	MAR ppm	First ppm	Last ppm
AP-1	High	NE Smith Island	20	1455	3072	2175	889		741	1455	741
AP-2	High	NW Smith Island	13	6017	5265		687	944	6017	6017	944
AP-3	High	Little Smith Island	3			62				62	62
AP-4	Low	Foul Pass	14	2247	144		29	459	45	2247	45
AP-5	Moderate	Passage Point	13		1024		31		118	1024	118
AP-6	Low	East Herring Bay	6	208	48			85	60	208	60
AP-7	Moderate	East Green Island	17	247	79			13	3	247	3
AP-8	High	West Green Island	16	20839	1879	7413		1056	14319	20839	14319
AP-10	High	South Point Helen	10	2368	2914			705		2368	705
AP-11	High	NE Latouche A	8	774	336				260	774	260
AP-12	High	NE Latouche B	26	20073	16589	2700	4899	6502	592	20073	592
AP-13C	High	Sleepy Bay Central	14		10792		3683		734	10792	734
AP-13E	High	Sleepy Bay East	10				2195		1275	2195	1275
AP-13W	High	Sleepy Bay West	16			14442		2198	2686	14442	2686
AP-14	High	NW Latouche Island	19	512			304	215	2427	512	2427
AP-16	Moderate	East Elrington Is.	16	174	58			68	41	174	41
AP-17	Moderate	North Elrington	12	78	115		459	131		78	131
AP-18	Low	South Elrington	17	312	279		81	315	43	312	43

Total Samples	250	
Average	4657	1399
Standard Dev.	6782	3227
Count	18	18
Std.Dev. of Avg.	1645	783
Fist/Last Ratio	3.33	
% Reduction	70	

3) Composite samples (representing several sampling locations)
4) Samples of unknown or uncertain origin

A detailed inspection of this database reveals that, while certain "principal" transects were sampled repeatedly, additional samples were often taken from other transects as well. Also, different portions of the principal transects were sampled at different times, as dictated by timing (daylight) considerations and tide windows. A more consistent dataset is obtained by selecting only samples from the principal transects and by omitting samples from the lower intertidal zone which was sampled less regularly (and which, generally, contained little oil). The transects selected for the "transect database" are those transects that were sampled at least once during both the first half (September-November) and the second half (December-March) of the program.

One other data point was omitted from the more limited transect database: Bag #24 at Site AP-8. This sample, taken in March, came from the top of a fine sand that was found under about 1 foot of coarse gravel in the storm berm. Some gravel was included in the sample, but only the sand was analyzed. The field notes indicate that oil had accumulated on top of the fine sand by gravity drainage. The TPH content of the sand was found to be about 41,800 ppm (4.1% by weight); however, being limited to a thin layer, this value was clearly not representative of the general subsurface condition at the site. Other subsurface samples taken in the upper intertidal had less than 1,000 ppm.

Transect Summaries

Table II-2 presents the data for the principal transects in summary form. This reduced, but internally consistent database consists of 162 samples from 18 transects. It contains a total of 65 entries; thus, the database reflects an average of 65/18 = 3.6 visits per transect, and an average of 162/65 = 2.5 samples per entry.

The downward trend of the TPH data in Table II-2 is again illustrated by the last two columns where the "first" and "last: observations are repeated for each of the 18 transects. The average, standard deviation, and the estimated standard deviation of the average, are given below each of these columns. The average of the last observations is again much lower

Table II-2

Summary of TPH Data for Selected Transect Prince William Sound Subsurface Sediment Samples From Exxon's Winter Monitoring Program

Monthly Averages for Principal Transects - Excluding Samples from Lower Intertidal

SITE	E	Z T	DIST		No (m)	SEP ppm	OCT ppm	NOV ppm	DEC ppm	JAN ppm	MAR ppm	First ppm	Last ppm
AP-1	H	Tr.7	5 to 27	Average	13		4541	2662	1173		906	4541	906
AP-2	H	Tr.3	11 to 22	Average	6	10614	6830		307			10614	307
AP-2	H	Tr.4	10 to 19	Average	4	1419			24	1233		1419	1233
AP-4	L	Tr.6	2 to 16	Average	12	2247				459	37	2247	37
AP-6	L	Tr.2	9 to 12	Average	2		144			90		48	90
AP-6	L	Tr.4	7 to 12	Average	4	208	48			80	60	208	60
AP-7	M	Tr.6	6 to 20	Average	10	247	55			16	0	247	0
AP-8	H	Tr.8	5 to 20	Average	14	20839	2730	7413		1056	598	20839	598
AP-10	H	Tr.6	16 to 29	Average	8	2368	4342			742		2368	742
AP-12	H	Tr.7	5 to 30	Average	16	20073		3357	5872	823	884	20073	884
AP-13C	H	Tr.17 8-21	8 to 21	11	10792		10792		4602		1094	10792	1094
AP-13W	H	Tr.24 15-24	15 to 24	10	14442			14442		3618	1848	14442	1848
AP-14	H	Tr.2	5 to 12	Average	4	353			55			353	55
AP-14	H	Tr.4	5 to 16	Average	10	824			469	304	2505	824	2505
AP-16	M	Tr.5	2 to 25	Average	13	174	70			65	45	174	45
AP-17	M	Tr.4	2 to 15	Average	8	36	172		600	13		36	13
AP-18	L	Tr.4	6 to 14	Average	7	432			204	60	38	432	38
AP-18	L	Tr.8	7 to 16	Average	10	191	279	20	20	47	48	191	48
Total Samples	162												
											Average	4992	583
											Standard Dev.	6916	703
											Count	18	18
											Std Dev. of Avg	1677	171
											Fist/Last Ratio		8.6
											% Reduction		88

than the average of the first observations (about 580 ppm versus 5,000 ppm, an 88% reduction).

It should be noted that these estimates are biased on the conservative side by the fact that not all transects were sampled in September and March. In fact, the average time span between first and last sampling for the 19 transects is about five months; hence, the reduction of TPH values observed during the six-month winter program reflects only about five months of natural cleansing.

Trend Analysis

Table II-3 shows the results of an exponential regression (by determining a least-squares linear fit to the logarithms of the monthly averages) for the transect data shown in Table II-2. To facilitate the analysis, the gaps in the data matrix were filled in by geometric interpolation; i.e., each interpolated value is equal to the square root of the product of its two neighbors. The entire column for the month of February was added in this fashion. Missing data points in September and March were added first by assuming a level trend. This preserves the first and last data points for each transect and tends to bias the analysis towards underestimating the rate of natural cleansing.

Summary statistics for the complete (interpolated) data table, including monthly averages, standard deviations, and estimated standard deviations for the averages, are shown in the lower part of Table II-3. The last row of each table shows the exponential regression fit to the monthly averages. It was obtained by fitting a straight line to the logarithms (also shown) of the monthly averages. The exponential trend line and monthly averages are plotted in Figure II-1. The indicated rate of natural cleansing is about 32% per month.

A breakdown by energy level is shown in the bottom plot in Figure II-1, where monthly averages and regression lines are plotted separately for subsurface sediment samples from ten transects on high-energy sites and eight transects from low/moderate energy sites (see Table II-1 for the classification of monitoring sites by energy level). The number of individual samples contained in these subsets is 96 and 66, respectively.

Table II-3

Regression Analysis for TPH Data from Principal Transects
Note: Data Gaps Filled by Constant Extrapolation and Geometric Interpolation

SITE	E	Z T	DIST		No	SEP (m)	OCT ppm	NOV ppm	DEC ppm	JAN ppm	FEB ppm	MAR ppm	First ppm	Last ppm
AP-1	H	Tr.7	5 to 27	Average	13	4541	4541	2662	1173	1076	987	906	4541	906
AP-2	H	Tr.3	11 to 22	Average	6	10614	6830	1448	307	307	307	307	10614	307
AP-2	H	Tr.4	10 to 19	Average	4	1419	1370	1323	1277	1233	1233	1233	1419	1233
AP-4	L	Tr.6	2 to 16	Average	12	2247	144	59	24	459	130	37	2247	37
AP-6	L	Tr.2	9 to 12	Average	2	48	48	59	73	90	90	90	48	90
AP-6	L	Tr.4	7 to 12	Average	4	208	164	129	102	80	69	60	208	60
AP-7	M	Tr.6	6 to 20	Average	10	247	55	37	25	16	4	0	247	0
AP-8	H	Tr.8	5 to 20	Average	14	20839	2730	7413	2798	1056	794	598	20839	598
AP-10	H	Tr.6	16 to 29	Average	8	2368	4342	2409	1337	742	742	742	2368	742
AP-12	H	Tr.7	5 to 30	Average	16	20073	8209	3357	5872	823	853	884	20073	884
AP-13C H	Tr.17 8-21	Average	8 to 21	10792	10792	10792	10792	7047	4602	2851	1766	1094	10792	1094
AP-13W H	Tr.24 15-24	Average	15 to 24	14442	14442	14442	14442	14442	7229	3618	2586	1848	14442	1848
AP-14	H	Tr.2	5 to 12	Average	4	353	191	103	55	55	55	55	353	55
AP-14	H	Tr.4	5 to 16	Average	10	824	683	566	469	304	873	2505	824	2505
AP-16	M	Tr.5	2 to 25	Average	13	174	70	68	67	65	54	45	174	45
AP-17	M	Tr.4	2 to 15	Average	8	36	172	322	600	13	13	13	36	13
AP-18	L	Tr.4	6 to 14	Average	7	432	336	262	204	60	48	38	432	38
AP-18	L	Tr.8	7 to 16	Average	10	191	279	75	20	47	47	48	191	48
				Average	4992	3078	2321	1457	717	592	583	4992	583	
				Standard Dev.	6916	4204	3688	2148	984	699	703	6916	703	
				Count	18	18	18	18	18	18	18	18	18	
				Std.Dev. of Avg	1677	1020	894	521	239	170	171	1677	171	
				Log of Avg.	3.70	3.49	3.37	3.16	2.86	2.77	2.77			
				Regression	4637	3140	2127	1440	975	661	447			
				Rate of Cleansing (%/month):	32									

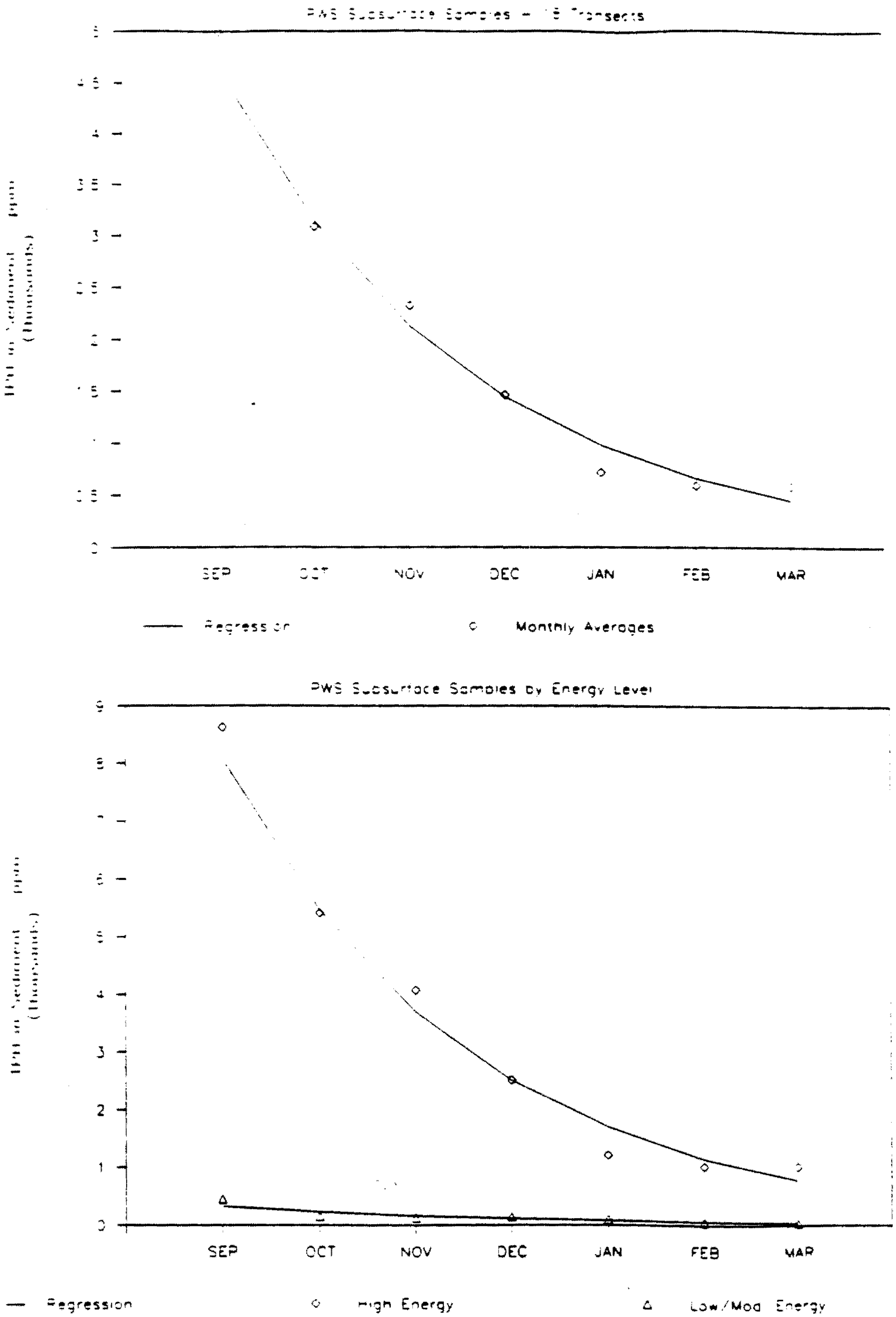

Figure II-1. TPH concentrations in subsurface sediments at 18 Exxon transects in Prince William Sound from September 1989 to March 1990, for all transects and divided into different energy levels.

Conclusion

Subsurface sediment samples from principal transects that were sampled repeatedly during Exxon's winter shoreline monitoring program in Prince William Sound show a strong downward trend in residual oil content. The data indicate that nearly 90% of the oil residue present on these transects in September was removed by natural cleansing during the course of the fall and winter. The estimated rate of reduction in TPH values was about 30% per month during the six-month period from September 1989 to March 1990.

The trend of natural cleansing is most pronounced in subsurface samples from the ten transects on high energy sites. The average TPH content in samples from these transects decreased from more than 8,000 ppm to about 1,000 ppm during the course of the winter. The residual oil content in subsurface samples from eight transects on lower energy sites started out much lower and averaged less than 100 ppm in March.

II.A.2 TRENDS IN NATURAL REMOVAL RATES OF SUBSURFACE OIL

Introduction

Natural cleansing rates for subsurface oil on specific shorelines impacted by the EXXON VALDEZ spill are problematic to predict because of the wide variability in substrate grain size, degree of exposure to wave and tidal energy, depth of penetration of the oil, and weathering of the oil which changes its properties over time. All of these factors contribute in varying amounts to determine the fate of subsurface oil at any one locality. However, it is possible to predict the persistence of subsurface oil based on studies of previous spills, as well as what has been learned from field observations and laboratory studies on the behavior, composition, and changes in the amount of oil in subsurface sediments in Prince William Sound over time.

There have been two recent reviews of the persistence of oil and recovery rates following oil spills in cold water environments. Marshall and Gundlach (1990) conducted a review of the literature focusing on the actions and environmental recovery of past spills which were similar in setting, oil type, and expected types of impacts to the EXXON VALDEZ spill, for NOAA. Baker et al. (1990) conducted a similar review, focusing on the natural recovery of cold water marine environments after an oil spill and including resource-specific analyses, for Exxon. These two reports are good sources upon which to base a synthesis of the oil persistence trends on shorelines that are potential candidates for rock washing in Prince William Sound. The reader is referred to these documents for a detailed review of oil-spill case histories.

Predictions for Alaskan Shorelines

In attempting to use previous case histories to predict the persistence and toxicity of subsurface oil in Prince William Sound, there are several factors which are specific to Prince William Sound and Prudhoe Bay crude oil which either speed up or slow down known processes.

Those factors that speed oil removal rates are:

1) Flocculation is a natural process whereby very fine-grained mineral sediments interact with oil residues on sediments and seawater to form a solids-stabilized emulsion which adheres less strongly to the sediments. The emulsion is formed when polar components in the weathered oil residue are attracted to the electrically charged surfaces of fine mineral particles such as clays and silts in the presence of salt water. The formation of these emulsions have been observed in the field and studied by Exxon in laboratory experiments (Bragg et al., 1990). Field observations in Prince William Sound over the winter/spring indicate that the subsurface oil in the upper 10-30cm is less sticky, brown in color, and readily smears off sediments, whereas deeper oil is still very tacky, more black in color, and overall fresher in appearance. Obviously, flocculation rate or effectiveness is a function of the volume or thickness of oil on individual grains and in pore spaces. The importance of emulsion formation is that it provides an additional mechanism for removal of oil from subsurface sediments. Also, flocculation may reduce the potential for formation of asphalt pavement by subsurface oil residues. Many of the case histories cited in the two reports note that the longest-term persistence of oil was in the form of asphalt pavements which contained relatively undegraded oil. However, flocculation has not been quantified as to its effectiveness under field conditions.

2) Biodegradation has been shown to be progressing at rapid rates in Prince William Sound because of the existence of naturally occurring hydrocarbon degrading bacteria. Although the degradation rates are slower with depth, removal of oil by microbial processes is an important mechanism that will speed the removal of subsurface oil at the EXXON VALDEZ site in comparison with past spills.

3) The extent of removal of surface oil during the 1989 treatment. In many of the case histories, the formation of surface pavements led to persistence of both surface and subsurface oil. The formation of pavements in Prince William Sound was significantly reduced by the extensive use of cold- to hot-water flushing of the shoreline in 1989. The SSAT data show that asphalt pavements are generally small and isolated, and they are being removed manually in 1990.

Those factors which slow subsurface removal rates include:

1) Relatively low wave energies relative to grain size of shoreline sediments. In Prince William Sound, the sediments are disproportionately coarse grained in comparison to the annual distribution of wave energy. That is, much of the shoreline sediments are residual rather than depositional. Large, storm-generated waves occur intermittently, therefore making it difficult to predict when sediments are likely to be reworked to the depths of residual oil. Thus, physical removal of deep subsurface oil by abrasion during sediment erosion and deposition cycles is not likely to occur quickly. In effect, we have an unusual condition whereby coarse-grained, porous sediments occur in an area which is not reworked to significant depths by frequently occurring storms. Therefore, most case histories are not directly applicable.

2) The initially high degree of contamination of some of the shorelines. Many of the natural removal processes, such as biodegradation, flocculation, etc., have rates which are a function of the loading of oil. Heavy loadings lead to greater retention times. Many of the shoreline areas under consideration for rock washing are those areas which were most heavily contaminated.

With these factors in mind, an attempt was made to predict the persistence of subsurface oil in different exposure settings. Reports of previous spills seldom specifically mention subsurface oil; rather, residual oil is referred to as tar, asphalt pavements, or crusts. For example, at the

ARROW spill in Chedabucto Bay, which is one of the best studied analogs for the sheltered regions of Prince William Sound, Vandermeulen (1977) reports that, "...tar stranded along the mid-water line on high- and medium-energy beaches has a self-cleaning half-life of around one and a half to two years. However, this half-life is increased by a factor of at least 10 where the Exposure Index drops such as with oil stranded on low-energy shores..." Figure II-2 shows summary curves on self-cleaning rates by wave exposure from Vandermeulen (1977). At the METULA site, asphalt pavements were found at sheltered localities 6.5 years (Gundlach et al., 1982) and 12.5 years (Owens et al., 1987) after the spill. At the Baffin Island Oil Spill (BIOS) project, long-term monitoring of an experimentally oiled beach showed that less than 10% of the original stranded oil remained after 18 months of open water (six years), based on a consistent database (Humphrey et al., 1990). This study represented a worst-case scenario for a low permeability beach in a sea-ice, cold climate, low-energy environment.

In Prince William Sound, natural removal has effectively cleaned the upper 10-30cm of sediments in areas of moderate to high wave energy. Although the data are highly variable, there have been reductions in the deep subsurface oil levels as well. Whereas there are not yet enough data to draw long-term curves for self-cleansing rates for the subsurface oil which remains as of 1990, the times required for removal of oil from all shorelines to background levels are predicted as follows:

Area	Residence Time for Oil Buried in Gravel Beaches
1) Sheltered parts of Prince William Sound	10+ years
2) Sheltered outer Kenai	3-5 years
3) Exposed parts of Prince William Sound	2-4 years
4) Exposed outer Kenai	1-2 years

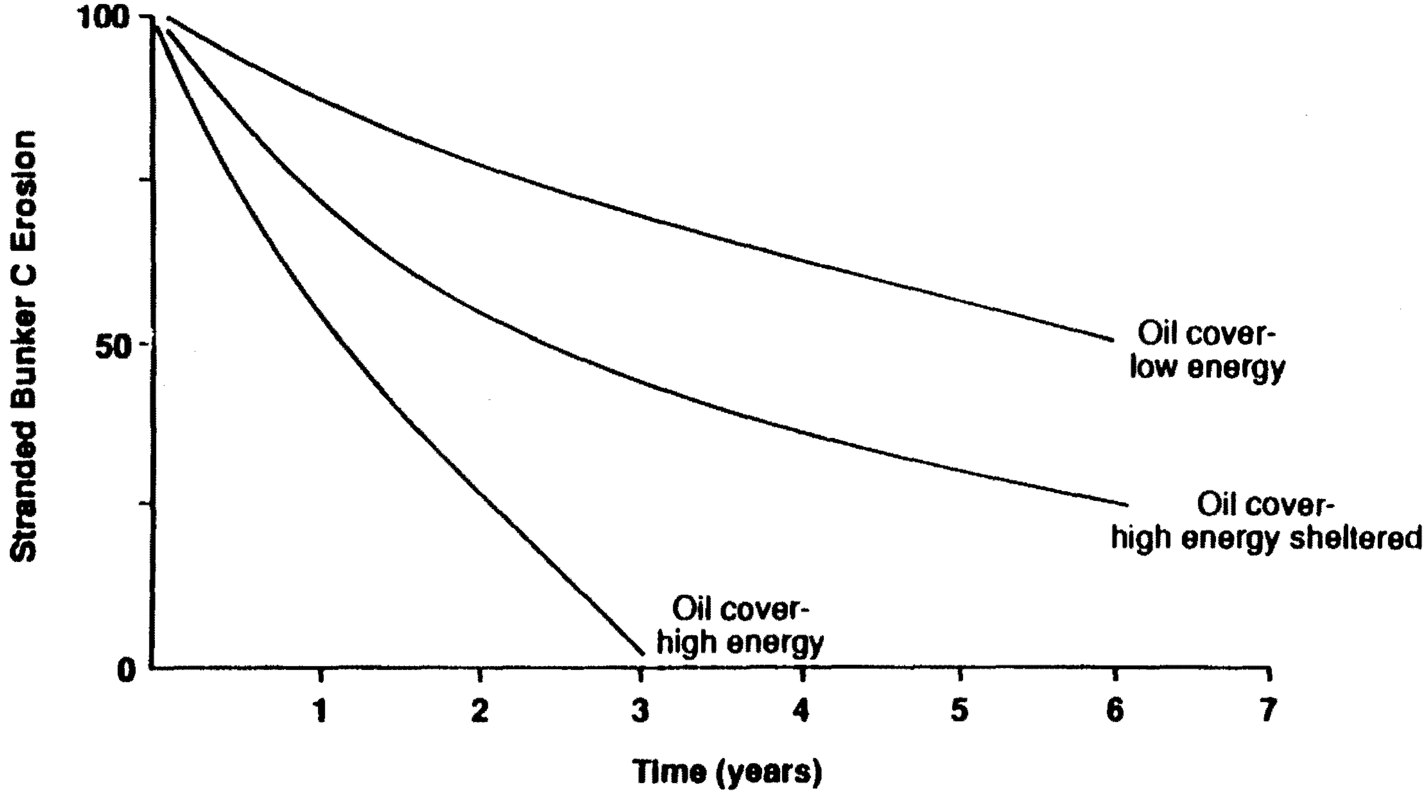

Figure II-2. Summary of oil spill self-cleaning processes (modified from Vandermeulen, 1977).

These estimates are based on the assumption that no other treatment is attempted for removal of subsurface oil, such as berm relocation.

Exposure of Subsurface Oil by Wave Action

One of the processes of natural cleansing is the physical abrasion of the oil during sediment reworking by storm waves. There is concern that the subsurface oil will be exposed during shoreline erosion, thus extending the period of impairment of use. Although there are limited data upon which to draw conclusions, the following observations made during the winter monitoring program are applicable.

The set-aside on northeastern Latouche Island, LA-15, is a good example of how subsurface oil might be exposed because it was never treated. This site was monitored monthly by NOAA from September 1989 until February 1990 and again in May 1990. By November 1989, the surface oil had changed from 100% of a heavy coating on the cobbles to a spotty, thin stain. After November 1989, scattered cobbles that had 100% of a heavy coat were observed along the profile; these oiled cobbles had been reworked from a deeper zone into the surface sediment layer. This process occurred gradually with the passage of storms. It is expected that, during major storms, there could be a pulse of oiled cobbles mixed into the surface sediments. As observed at LA-15, because the exposure occurs during storms, these oiled cobbles are rapidly reworked and diluted. With time, the available source of oiled clasts in the subsurface will be diminished, and exposure will decrease. Only the larger clasts are expected to remain oiled after exposure; the smaller sand, granules, and pebbles will be cleaned by abrasion during the storm event. At the two candidate beaches, these processes have already cleaned most of the oil from the surface sediments even at Sleepy Bay which is more sheltered but has finer-grained sediments.

It is not anticipated that subsurface oil exposed during storms will include extensive asphalt pavements. Many of the subsurface oil deposits are not concentrated enough to form pavements; the oil occurs mostly as coatings on clasts. Pavements are mostly surface deposits, although pavements at the high tide zone could be covered by recent deposition of clean sediments. The beaches in Prince William Sound are not subject to large-scale erosion/ deposition events during normal storms. Therefore, major exposures of subsurface oil would occur only during major storms.

Reapplication of Fertilizers

The reapplication of nutrient fertilizers during the Summer 1990 cleanup is generally expected to enhance the natural degradation of both surface and subsurface stranded oil. Oxidative biodegradation is a key process in the natural removal of petroleum in the environment. Microorganisms (e.g., bacteria, fungus, etc.) utilize the oil as a carbon source (food) in conjunction with oxygen and nutrients to produce carbon dioxide, water, and additional biomass. The target goal is to accomplish in one year what would be three to four years of natural biodegradation. To achieve this level of enhancement, the fertilizer needs to be reapplied every four to six weeks to maintain an adequate nutrient levels to keep the microbial population at an optima growth rate. In addition to nutrient availability, temperature and micro-nutrient availability must be at growth maxims to achieve the targeted rate of enhancement.

The two major fertilizer products being applied are Inipol and Customblen. Inipol is being used for surface treatments only while Customblen is targeted for use for both surface and subsurface treatments often in conjunction with Inipol. The actual enhancement to subsurface degradation has not been well quantified due to variability of field measurements between the treated and non-treated control stations from the Summer of 1989 studies. Visual observations from field studies as well as laboratory studies indicate that enhanced degradation is occurring through the use of bioremediation.

A detailed study to test the relative benefits of fertilization application is currently being conducted in Prince William Sound. The study will focus on the stimulation of biodegradation achieved by the addition of fertilizers, the toxicity associated with the application of fertilizers, and the potential eutrophication affect from the use of fertilizers. Preliminary results show evidence of increased mineralization rates on treated beaches of up to three times those on untreated beaches. Toxicity testing of water samples collected over the application area during the first tidal flooding showed no toxicity to mysids. The results of this study will be used to decide if fertilizer reapplication will take place.

II. A. 3. EFFECTS OF SUBSURFACE OIL RESIDUES AND PROJECTIONS OF RECOLONIZATION

Introduction

In this section, estimates are made of the probable biological effects of applying no rock washing operations to the subsurface oil and projections are made of the recolonization rate should no rock washing take place at the candidate beaches. These estimates and projections were made by surveying the existing conditions at the candidate segments; reporting observations of oil-related effects as of May 1990; evaluating the relationships between biological effects and concentrations of oil in beach materials; evaluating oil spill effects and recolonization rates at other previous spills; and using existing information to project recolonization rates.

The estimates of continuing effects and of the recolonization rates are clearly speculative, since there is no way at this point to empirically test these effects. Also, the high degree of variability in the density and kinds of organisms that inhabit Prince William Sound beaches precludes the determination of absolute estimates of effects and recolonization. All of the following projections must be recognized as best professional estimates, based upon past experience.

Estimated Effects of Oil Remaining in the Beach: Sleepy Bay Segments LA-17/18/19

Most of the intertidal biota in Sleepy Bay live either upon rock or cobble/boulder surfaces or under them to a maximum depth of 5 to 10cm. Therefore, they generally are not in direct contact with subsurface oil that is 10cm deep or greater. Only large, burrowing or tube-forming clams or echiurid worms would have the potential for contacting oil 10cm deep or greater. No large clams or echiurid worms were observed in Sleepy Bay and it is unknown if they occurred there before the spill.

In places where subsurface oil residues are covered by clean surface sediments, the effects of the oil upon intertidal organisms and recolonization would likely be minimal, unless the clean surface sediments were removed by erosion. In places where subsurface oil is contiguous with surface oil, it is impossible to distinguish the effects of the surface residues from those of the subsurface sediments. In a moderately exposed pebble/cobble beach, such as that in Sleepy Bay, sediments are continually shifting, thereby alternately exposing and covering different areas. Interstitial water is continually transporting oil from place to place within the beach. Because of these dynamic processes, subsurface oil that is contiguous with surface oil may pose a longer term inhibition to recolonization than the surface deposits alone. Also, since surface oil would be treated along with subsurface oil where they occurred together, it seems obvious to consider the effects of both with regard to estimating biological effects and recolonization.

Some of the limpets, mussels, barnacles, and littorinid snails observed in May 1990 were dead in Sleepy Bay middle and lower tidal zones. About 10 to 50% of the shells of these animals were empty (Section I.C.7), but remained unbroken. In addition, there were many detached dead dead fronds of *Fucus* and coralline algae among the rocks in the upper tidal zone. The cause of these deaths is unknown. However, it is likely that some proportion of the plants and animals had been killed by the oil some time before the May survey. Most of the delicate shells of *Littorina*, mussels, and limpets that may have been killed by the onset of cold winter conditions late in the fall of 1989 likely would have not survived the winter unbroken and likely would have not remained within the intertidal zone. One moribund clam and one dead mussel with their internal soft parts intact also were observed in May, suggesting that they had died very recently.

Top predators, such as sea stars and *Nucella*, were relatively sparse, as compared to information reported by Rosenthal et al. (1982) for other beaches in Prince William Sound. In May, these animals were absent and in June they were found mainly near or upon very large rocks or rock outcrops from which the oil was quickly removed. Animals that are generally recognized as being sensitive to the effects of oil, such as amphipods and isopods, were generally absent in patches of the upper tidal zone that had high concentrations of oil, but they were often abundant in most areas of the middle and lower tidal zones.

Collectively, the weight of these observations suggest that oil residues that remained in parts of Sleepy Bay in the spring of 1990 were adversely affecting at least some of the organisms there. These continuing adverse effects clearly are relatively minor in severity and extent.

However, there were encouraging signs of biological recovery, especially in the lower tidal zone. There were many young-of-year mussels, limpets, littorinid snails, nemerteans, *Fucus*, other algae, and barnacles. Egg masses from spawning littorinid snails were abundant. There was an abundance of oligochaete worms in very oily sand and gravel in the high tidal zone, presumably consuming bacteria that have been degrading the oil. Often they were accompanied by small predatory nemerteans and, occasionally, by young littorinid snails and mussels. There were occasional nereid worms, terebellid worms, and gunnels under boulders in the mid-and lower tidal zones. There were gammarid amphipods under boulders in the mid-and lower tidal zones where algal debris had accumulated, along with a few hermit crabs among the boulders. Biological populations observed in June were generally more dense and diverse than those observed in May.

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It appears that the epibenthic biota of the lower tidal zone were not as severely impacted by the oil as the epibenthic and infaunal biota of the upper tidal zone. Little or no oil remained in the lower tidal zone in May 1990.

The eelgrass bed in the subtidal zone would be highly susceptible to the effects of the oil (as was observed at the IRISH STARDUST spill in British Columbia), but appears to have survived the spill. The mid-tide zone had a fairly representative community of organisms (as compared to the descriptions of Rosenthal et al., 1982) and the remaining oil, if any, was buried deep in the beach, and, therefore, isolated from the plants and animals above it.

Therefore, it appears that the projected biological effects upon intertidal biota of not treating the beaches may be expressed mainly in the upper tidal zone. This zone normally is not inhabited by a very rich community (Rosenthal et al., 1982) due to desiccation and lack of food. The continuing effects to animals in this zone may be attributable more to suffocation by thick deposits of oil and lack of oxygen than to toxicity. Much of the subsurface oil occurs as shiny, black coatings upon sand, gravel, and pebbles. Much of the low molecular weight fraction of the oil that is most toxic has been removed (see section I.C.3.). Organisms were found living in high intertidal zone beach samples with up to 16,000 ppm total petroleum hydrocarbons, suggesting that the remaining oil is not very acutely toxic. Based upon the projections forecast in Section II.A.1, subsurface oil residues may remain in the beaches of Prince William Sound for up to 10 years or more. The remaining surface oil in the upper tidal zone will likely dissipate more quickly.

Continued effects upon the biota of the mid-tidal zone and lower zone are not expected to be very severe, since it appears that these zones were not heavily impacted at the outset. Relatively abundant populations of barnacles, mussels, brown, red and green algae were apparent in May 1990. There is no reason to expect that these organisms would be adversely affected by the oil remaining in the intertidal zone. Top predators, such as *Thais* and sea stars, however, were missing from unconsolidated materials in May 1990 but were observed in June 1990, suggesting that they are recolonizing the beach or are highly mobile and able to migrate around the beach over large areas. There is no reason to suspect that the lower tidal zone eelgrass bed would be affected in the future by the remaining subsurface oil in the intertidal zone. In investigations following other oil spills, permanent damage to intertidal communities usually was not observed and recovery of the communities to a status approximating pre-spill conditions was complete in one to six years (Gundlach et al., 1982; Straughan, 1976; Green et al., 1974; Mancini et al., 1989; Nelson-Smith, 1977). However, in a recent report, Dauvin and Gentil (1990) concluded that several species of crustaceans that had been abundant in coarse beaches before the AMOCO CADIZ spill had not yet recolonized the same beaches ten years later, and therefore, biological recovery was in complete.

If the oil is not removed by rock washing, effects upon organisms in the water column offshore from the beaches are expected to be minor relative to the effects experienced during the initial phases of the spill. Sheens bleeding from the beach in Sleepy Bay could adversely affect the salmon smolts migrating out of the local stream in the spring or the epibenthic zooplankton that these fish depend upon. But, these effects probably would be minor and short-term due to the weathered nature of the oil, the short exposure period, and dilution by the large water mass in Sleepy Bay and adjoining Prince William Sound.

Projected Effects of Oil Remaining in the Beach: Point Helen

The lower tidal zone at Point Helen is dominated by a luxuriant assemblage of large algae, including *Alaria marginata*. There are many sea stars, crabs, gunnels, gammarid amphipods, caprellid amphipods, bryozoans, sponges, and other sensitive species in this assemblage. It does not appear that the oil spill has had a lingering severe effect upon the biota of this zone.

The mid-tidal zone is relatively depauperate. Without baseline, pre-spill data it is not possible to know if this condition existed before the spill. The large boulders in this zone support very few organisms. The gravel/sand matrix under the boulders also supports a relatively depauperate community. Again, it is impossible to know if this condition existed before the spill.

The upper tidal zone has even fewer organisms than the mid-tidal zone, and, again, it is unknown if this condition existed before the spill. The data summarized by Rosenthal et al. (1982) suggest that the upper tidal zone of mixed coarse beaches generally is depauperate. A few littorinid snails and oligochaetes were found in this zone at Point Helen.

Continuing effects of the oil, if left without rock washing, are expected to be minimal or undetectable in the lower tidal zone. It is unlikely that the upper tidal zone normally supports a rich community due to exposure to desiccation, mobility of the beach sediments, and a lack of food among the coarse gravel. Therefore, effects in that zone would be of limited importance since a relatively small number of organisms live there. Projected effects in the mid-tidal zone are difficult to estimate. Since the grade is steep, the boulders are rounded and offer relatively poor shelter, there is a very small amount of fine sand or mud beneath the boulders and the beach is relatively exposed to waves; it is unlikely that many organisms normally live in this zone that would be effected by the remaining oil.

Projected Recovery Rates

In any oil spill, one of the major con rolling factors in the recolonization rate of beach sediments not treated by methods such as rock washing is the rate of loss of the toxic components of the oil. In Prince William Sound, the remaining oil has weathered to a considerable extent and many of the toxic components have disappeared or have been severely reduced in concentration (see Section I.C.3).

At Point Helen, most of the problematic oil exists in subsurface lenses or layers below the relatively cleaner surficial material. These layers of oil are often deeper than the zone that would be inhabitable by intertidal organisms. Therefore, the data from the quantification of oil residues in surficial sediments may overestimate the concentrations to which prospective colonists would be exposed.

An attempt was made to estimate the points in time at which recolonization and full biological recovery would be possible. The estimates were based upon a projection of the rate of loss of oil from the candidate beaches, estimates of the petroleum concentrations in sediments that equal the lower thresholds of adverse biological effects, and projections of the rates of recolonization once the oil concentrations passed below the effects thresholds. The projections were compared with observations at Point Helen and Sleepy Bay of the oil residue concentrations associated with azoic conditions versus those concentrations in which it appeared that recolonization had begun. In addition, these estimates were compared with observations of recovery in case studies at previous oil spills.

In the winter surveys conducted by NOAA and others at several of the candidate beaches, total oil residue concentrations were determined. Mass spectral analyses were performed with 110 samples (surface and subsurface) during the NOAA winter survey. The data from these oil residue analyses are summarized above in Section I.C.3. The concentrations of total oil and of selected hydrocarbons in the samples were compared with the concentrations known to be associated with biological or toxicological effects in soft-bottom sediments. No toxicological data are available for many of the substituted, high molecular weight hydrocarbons commonly found in the weathered Prince William Sound oil. Data are available from Long and Morgan (1990) for some of the aromatic hydrocarbons, however, they generally do not constitute a large proportion of the total weathered oil in the Prince William Sound beaches.

Comparisons of Hydrocarbon Concentrations with Effects Thresholds

The likelihood of continuing toxic effects of the oil that remains in the beaches can be estimated by comparing the hydrocarbon concentrations known to be associated with biological effects and the concentrations observed in the candidate beaches. If the ambient oil concentrations are below the toxic thresholds, it is reasonable to assume that continued effects of the oil would be minimal or not expected.

Apparently, there are no data from spiked sediment toxicity tests performed with weathered Prudhoe Bay or north slope crude oil to use as guidance for establishing what concentrations are significantly toxic to marine organisms. Given the lack of these data, information assembled (Long and Morgan, 1990) from other types of studies were used as an estimate of the hydrocarbon concentration commonly associated with adverse effects. These data were supplemented with the matching observations made in May 1990 by NOAA and chemical characterization of samples collected in February and May 1990.

Long and Morgan (1990) reviewed the data from many different studies in which chemical concentrations in fine-grained sediments were associated with measures of adverse biological effects and, based upon these data, offered some guidelines for use in the evaluation of sediment chemistry data. Included among the analyses that were evaluated were total polynuclear aromatic hydrocarbons and many individual aromatic hydrocarbons. The data evaluated by Long and Morgan (1990) are not strictly comparable to the data available for Prince William Sound. The Prince William Sound beaches are largely mixtures of pebbles, cobbles, and boulders with very few fine-grained particles, whereas the data evaluated by Long and Morgan (1990) often were from harbors and bays with muddy sediments. The weathered residues of the oil spilled in Prince William Sound have relatively few low molecular weight aromatic hydrocarbons and few pyrolytic hydrocarbons, whereas the examples reviewed by Long and Morgan (1990) often included relatively high proportions of these compounds. Nevertheless, by using the data base assembled by Long and Morgan (1990), the range in concentrations of hydrocarbons commonly associated with toxic effects can be estimated and compared with the Prince William Sound data on a qualitative basis.

There are no data available from the literature that identify the toxicity threshold of sediment-bound Prudhoe Bay or north slope crude oil. However, the concentrations of individual petroleum constituents can be compared to the effects thresholds with the caveats listed above in mind.

One of the major components of the weathered oil in Prince William Sound is phenanthrene. At concentrations between 0.1 and about 0.2 ppm, no effects attributable to phenanthrene were observed in most studies (Long and Morgan, 1989). At concentrations of 0.3 ppm or greater, effects were almost always observed or predicted. In a bioassay in which clean sediments were spiked with phenanthrene, an LC50 of 3.68 ppm was observed among amphipods. Long and Morgan (1990) suggested that 260 ppb (about 0.3 ppm) phenanthrene may be an overall effects threshold. Rounding 0.3 ppm to 0.5 ppm, toxic effects in sediments generally have been associated with phenanthrene concentrations of about 0.5 ppm or greater. Twenty of the 110 samples from the NOAA winter studies program analyzed by gas chromatograph had phenanthrene concentrations of 0.5 ppm or greater. Neither of the two samples collected in the winter NOAA surveys at Point Helen or at Sleepy Bay exceeded 0.3 ppm. Azoic conditions were found in two samples from Sleepy Bay that had 33 and 1 ppm phenanthrene and 12 samples populated by some organisms had 1 ppm or less phenanthrene. Two of the May 1990 samples from Sleepy Bay analyzed exceeded 0.3 ppm (Table I-2).

Toxic effects generally have been associated with naphthalene concentrations of 0.5 ppm or greater in sediments (Long and Morgan, 1990). Two samples from Sleepy Bay that were azoic had 0.01 and 0.1 ppm naphthalene, compared with 12 samples that had organisms living in them that had 0 to 0.001 ppm naphthalene. Out of 110 samples collected in the NOAA winter surveys, 99 had naphthalene concentrations of 0.01 ppm or less, far below the apparent toxicological thresholds. Three of the samples had 0.110 to 0.2 ppm naphthalene, the highest concentrations observed. None of the samples collected in Sleepy Bay or at Point Helen exceeded 0.2 ppm (Table I-2).

Toxic effects in sediments generally have been associated with fluoranthene concentrations of 1.0 ppm or greater (Long and Morgan, 1990). Effects have not been observed in association with concentrations of about 0.3 ppm or less. Two samples from Sleepy Bay that were azoic had 1.4 and 0.02 ppm fluoranthene, compared to 12 samples that had 0.05 ppm or less and had some organisms living in them. Out of the 110 samples analyzed in the NOAA winter survey, all but one had 0.240 ppm fluoranthene or less. None of the samples collected in Sleepy Bay or at Point

Helen exceeded 0.5 ppm. Only one of the May 1990 samples from Sleepy Bay exceeded 1.0 ppm (Table I-2).

Toxic effects in sediments generally have been associated with pyrene concentrations of about 1.0 ppm or greater (Long and Morgan, 1990). Two samples from Sleepy Bay that were azoic had 2.0 and 0.1 ppm, as compared to 12 samples that had living organisms in them that had 0.06 ppm or less. All but one of the 110 samples analyzed in the NOAA winter surveys had pyrene concentrations of 0.420 or less. None of the samples collected in Sleepy Bay or at Point Helen exceeded 1.0 ppm. Only one of the May 1990 samples from Sleepy Bay exceeded 1.0 ppm (Table I-2).

Toxic effects in sediments generally have been associated with chrysene concentrations of about 0.9 ppm or greater (Long and Morgan, 1990). Two samples from Sleepy Bay that were azoic had 13 and 0.5 ppm chrysene, as compared to 12 samples that had organisms living in them and 0.4 ppm or less. Eight of the 110 samples analyzed in the NOAA winter surveys had chrysene concentrations that exceeded 1.0 ppm. None of the samples collected in Sleepy Bay or at Point Helen exceeded 0.5 ppm. Only one of the May 1990 samples from Sleepy Bay exceeded 0.9 ppm (Table I-2).

From these comparisons of the selected, individual, hydrocarbon concentrations that have been associated with biological effects and the concentrations of these compounds that remain in the Prince William Sound beaches, it appears that relatively few of the samples have concentrations that would be expected to be toxic. This conclusion must be tempered by the fact that there are no toxicological data for many of the hydrocarbons in the weathered Prince William Sound oil associated with sediments. These hydrocarbons, many of which are substituted compounds, may or may not be toxic (they probably are not very toxic).

These chemical data represent the amount of oil adhering to pebble, cobble, and sand particles sampled from intertidal beaches. Since the oil was often adhered to the beach material in a coating, these concentrations may not accurately represent the amount of oil to which potential colonists would be exposed. The concentration data probably overestimate the amount of oil to which organisms are actually exposed. The dissolved, liquid, and semi-solid phases of the oil probably represent a larger threat to the health and viability of colonists. However, data are not available for these phases of oil in the Prince William Sound beaches. Therefore, the estimates of recolonization rates and recovery must necessarily be based upon an evaluation of the thresholds in effects expected for the oil concentrations on these beach materials.

The rate of recolonization of heavily oiled beaches is very difficult to predict on a site-specific basis, since many complex factors will influence the process. Some of these factors include the natural small-scale patchiness in the texture, slope, and exposure of the beach; the abundance of surviving organisms; the rate of loss of the oil components; and the season during which the concentration of the oil becomes sufficiently low to allow survival of colonists. The net effect of all of these factors would be accelerate or inhibit recolonization. The degree of influence of any one of these factors relative to the others would differ on a case-by-case basis.

Table II-4 includes a summary of case histories on biological recovery rates following oil spills. More detailed discussions of these case histories are included in Appendix A. These case histories were used to make estimates of recovery rates for the No Further Treatment consideration.

In the preparation of the projected recovery rates, it was assumed that the most resistant species would either survive the oil spill and/or they would recolonize the beaches first. Also, it was assumed that the most sensitive species would successfully recolonize the beaches later in the process. In addition, it was assumed that the successional stages in recovery that would be expected at a disturbed unoiled beach would also apply to an oiled beach, once the toxicity of the oil was abated. Finally, it was assumed that the epifaunal species that would be expected to recolonize large stable boulders and rocks would colonize them more quickly that the ambulatory and infaunal species would colonize the interstices of the beach materials. This assumption is based upon the expectation that the removal and detoxification of oil would be more rapid on solid substrates than in the spaces among beach materials.

High-Energy Beaches (e.g., some portions of KN-405)

Some Prince William Sound beaches with a relatively high degree of exposure to wave action were heavily oiled by the initial advances of the oil spill. In some cases, oil penetrated well beneath surface layers at these beaches.

Table II-4

Summary of Biological Recovery Rates for Various Types of Disturbance

NAME	DISRUPTION	BEACH TYPE	DEF. OF RECOVERY	RATE OF RECOVERY	REFERENCE
(1)METULA spill, Strait of Magellan	Arabian crude & Bunker C crude oil	Mixed sand/gravel beaches; some mud sand tideflats	Species richness & abundance	5 Months-initial damage investigated, 2 yrs-not significant; 5.5 yrs.-very little; 6.5 yrs.-new vegetation	Marshall & Gundlach, 1990 Straughan, 1976 Gundlach et al., 1982
(2)Arrow Oil Spill Chedabucto Bay, Nova Scotia	Bunker C oil	Rocky outcrops, eroding till cliffs, gravel, mixed sediment beaches	Self-cleaning bio-recovery; species abundance; biomass measurements; numbers of certain species	After 6 years: more species/biomass at control sites than at oiled sites, clams reduced greatly	Marshall & Gundlach, 1990 Thomas, 1978 Vandermeulen, 1977
(3)General M.C. Meigs oil spill, Washington	Navy Special fuel oil	Intertidal shallow rock shelf & margin of Wreck Cove; coarse sand beaches	Visual/assessments of organisms compared to unoild sites, chem. anal. of tissue hydrocarbons recovery by natural processes	General decline in abundance with exceptions; 3 to 5 years	Clark et al., 1978
(4)Irish Stardust spill 1973	1000 Second Fuel Oil; 200 tons	Mixed habitats in embayment	Return of pre-site dominant speies	Minor biological recovery appeared to be underway approx. 1 yr. after spill	Green et al., 1974
(5)US Air Force Fuel Depot Spill 1971	JP4 jet fuel & No.2 heating oil; 14 tons	Intertidal zone Long Island Cove	Sediment analysis of hydrocarbons as well as clam tissue analysis--absence or low levels would be recovery; mortality assessments	6 yrs. shows slow recovery & animals only found in well-drained upper intertidal zone; clams also had significant levels of hydrocarbons in tissues	Mayo et al., 1978
(6)Irini Oil Spill, Sweden 1970	Medium & heavy fuel oil 1000 tons	Bay habitat in Stockholm Archipelago	Recovery determined by survival and health of species showing normal development	Littoral community recovered during period of 1971-1976 6 years	Marshall & Gundlach, 1990

Recovery at oiled beaches of this type will be subject to the same general considerations important at other impacted beaches of Prince William Sound, including the rate of reduction in levels of petroleum hydrocarbons, interactions among intertidal populations after the spill, and the occurrence and relative abundances of constituent species. The greater degree of wave and weather-related exposure at these locations has three major implications for assessment of recovery for the intertidal community impacted by oiling. First, intertidal portions of exposed coastlines are generally less productive than more sheltered shorelines of similar physical character, as it is more difficult for populations of some species to establish themselves and survive on open coasts where heavy wave action frequently mobilizes the beach sediment. As a result, the more fragile representatives of the Prince William Sound intertidal zone may be absent from these more exposed beaches. Second, the greater degree of exposure will facilitate more rapid weathering of oil remaining on or in the substrate and significantly reduce the role of the oil as a potential toxicant. Third, the dynamic and less stable nature of exposed shorelines makes evaluation of "recovery" difficult, as variability is more pronounced and comparison to "normal" conditions more complicated.

Similar to the situation observed by Gundlach et al. (1981) after the AMOCO CADIZ spill, oiled portions of exposed shoreline in Prince William Sound may be rapidly recolonized by algal species such as *Fucus*. The acute toxicity of the oil that initially came ashore, as well as cleanup techniques employed at many of the impacted beaches, may have removed a large portion of the herbivorous components of the intertidal community, such as littorine snails. This selective exclusion would enable algae to quickly spread, possibly recolonizing an area to a greater degree of cover than had existed prior to the spill. However, as was found by Southward and Southward (1978) following the TORREY CANYON spill and cleanup, recovery of the herbivore populations resulted in a return to conditions that could be termed normal, or near-normal.

Because the presence and toxicity of oil at exposed beaches can be expected to be decreased due to accelerated weathering, biological recovery rates at these locations are likely to be fairly comparable to those for recolonization of new or denuded substrate (see subsequent discussion in section II.B.2). Total time for recovery to conditions approximating a pre-spill community would also be expected to be similar to that observed for other spills, or three to five years. However, the relatively lower diversity and abundance in the exposed shoreline intertidal zones suggests a less complex recolonization process, and may result in a slightly reduced time of two to four years. By these estimates, it is possible that by 1991/1992, exposed beaches may have, to a large extent, recovered. Recent observations at sites of this type suggest that recovery is well under way and may be more rapid than that at other beaches.

Moderate-Energy Beaches (e.g., portions of LA-18)

Based upon the available information, it appears that recolonization is predicted to occur relatively quickly in the stable oiled beaches in which loss and weathering of the oil is proceeding and in which an abundant intertidal community is expected. These beaches are undergoing weathering and loss of oil through natural processes and/or with the help of surface washing and bioremediation. They are beaches that are sufficiently stable to allow the successful attachment and survival of plants and animals. The beach materials are not mobile, do not roll in the waves and offer refuge for potential colonists. Indeed, the recolonization process appears to have begun in these beaches as of May 1990.

The recolonization of these beaches by intertidal species with intermittent and infrequent recruitment is impossible to predict; these species may not reappear for five to 30 years. O'Clair and Zimmerman (1986) observed that some large clams were very slow in recolonizing some beaches in Alaska following the nuclear bomb tests. Therefore, predictions of recovery are based upon the recolonization rates expected for the "core group of dominant species" that typically inhabit these habitats, as generally described by Rosenthal et al. (1982).

It is very likely that the recovery of the epibenthic fauna and flora will be quicker than that for the infauna. The plants and animals attached to the upper surfaces of large boulders and rocks often survived the initial oil spill and will proliferate in the subsequent absence of oil. Some of the sensitive echinoderms killed by the oil will not provide a predatory control of these species.

Recolonization of resistant species is predicted to proceed quickly during the spring and early summer of 1990 as the oil concentrations diminish and as the normal seasonal cycle of spawning and recruitment takes place. The colonization of the beaches likely will reach an intermediate plateau during the fall and winter of 1990. During this plateau, the community likely will be composed of a core of resistant indigenous species such as epifaunal barnacles, snails, mussels, marine algae, ambulatory snails, nemertean worms, oligochaete worms, and a few resistant crustaceans, such as shore crabs and hermit crabs, and infaunal polychaete worms. During the following spring of 1991, it is expected that some of the more sensitive species of the community, such as certain copepods, amphipods, and isopods, will be successful in recolonizing the beaches. By the following fall (1991), a community composed of many of the same species and with small-scale variability similar to that of pre-spill conditions will exist. In the subsequent years, it is expected that the most sensitive species and juveniles of the adults that colonized the beaches in the previous years and many of the top predators will inhabit the beaches in numbers similar to pre-spill conditions in the spring of the third year (1992). This pattern of spurts of recolonization followed by plateaus of relative stability probably will continue for several years.

Therefore, re-attainment of the abundances of the core group of dominant species similar to those of pre-spill conditions ("recovery") is expected to occur in about three to five years. Generally, recolonization of oiled beaches by a community more or less similar to that of pre-spill conditions has occurred in three to six years in most spills similar to the EXXON VALDEZ (Table II-4). Replacement of pre-spill dominants by other species and the very slow arrival of infrequent spawners have been recorded.

Sheltered Beaches (e.g., some portions of LA-17)

Some beaches that were relatively highly productive in the intertidal zones were heavily oiled and remain heavily oiled. The beaches remain saturated with oil in some patches, including materials at or near the beach surface. They usually have a relatively high percent of small gravel, sand, or finer material, and, therefore, have retained much of the initial loading of oil. These beaches also had a relatively productive community of mussels, barnacles and algae attached to large stable boulders; worms, snails, crabs and tidepool fishes moving around the beach; and some relatively abundant populations of sensitive crustaceans. They likely are important sources of prey species for salmon and other important resources.

In these beaches, the recovery period is predicted to be somewhat longer than in the beaches in which the oil concentrations are diminishing rapidly. The concentrations of remaining oil are patchy. In many samples, the concentrations of individual hydrocarbons were below known effects thresholds, while in other samples these concentrations equalled or exceeded the thresholds. Some successful recolonization has occurred before and during the spring of 1990, but some parts of the beach are populated only by a few of the more resistant species and some parts are azoic. These opportunist species may competitively exclude later potential colonists, and, therefore, further delay the development of a normal community. Sensitive species are not expected to colonize oiled portions of the upper tidal zones of these beaches in appreciable numbers in the first year after the spill. Additional numbers of the early opportunists may arrive in the spring of 1991, along with a few additional species. The recolonization process will likely occur in spurts each subsequent spawning season, followed by a plateau each fall/winter. Depending upon the rate of loss of the oil, the time to recovery of a community that resembled that of pre-spill conditions may be about five to eight years. Observations made at other spills with similar conditions, indicate that recovery of heavily oiled gravelly beaches required five to ten years. The time required for biological recovery was longer in fine-grained beaches than in coarse-grained beaches.

The time required for infrequent and intermittent spawners is impossible to predict; it may be five to 30 years. Fortunately, many of these species are large predators, relatively mobile, and possibly relatively resistant to the effects of oil. Therefore, they may have survived the effects of the oil spill and remain upon the oiled beaches or remain in deeper water and will invade the intertidal zones from below.

II.A.4 SHEENING—FREQUENCY, VOLUMES, PERSISTENCE

Introduction

Determining the volume of petroleum oil in sheens in Prince William Sound is a difficult problem due to observational and logistical factors. Limitations include the difficulty in describing the size of sheens (which may be very irregular in shape), and their thickness (which is indicated by the observed color of the slick). Although NOAA has published a guide relating sheen thickness to color and some standard descriptive terms, this standard is not uniformly used by all observers reporting on sheen occurrence. ADEC observations are reported in different terms than Exxon observations, making direct comparisons between these two sheen reporting systems difficult.

In addition, naturally generated slicks may be difficult to distinguish from those due to petroleum oil, and fuel spills from vessels are common. Exxon states that they have taken samples of "doubtful" sheens and have analyzed them to determine if their spectral signature indicates it is Alaska North Slope crude oil. Exxon reports sheens containing North Slope crude oil or of indeterminate source as one category, separate from sheens associated with vessels, biological activity or other identified sources. The ADEC data does not discriminate sheens in this manner.

Logistical considerations also confound the issue in that weather may limit flight and visibility conditions so that observations cannot be made on a daily basis. Further, the entire Prince William Sound may not be covered on every reconnaissance flight. These factors result in a non-uniform distribution of observation periods, with many days of no observations.

Data Sources

Two primary sources of sheen data were used in this analysis:

1) Exxon 1990 Sheen Data in Prince William Sound. This is a listing of "...the sheen sightings reported by the Exxon sheen surveillance crews during 1990 through June 15, 1990," summarized in Figure II-3.

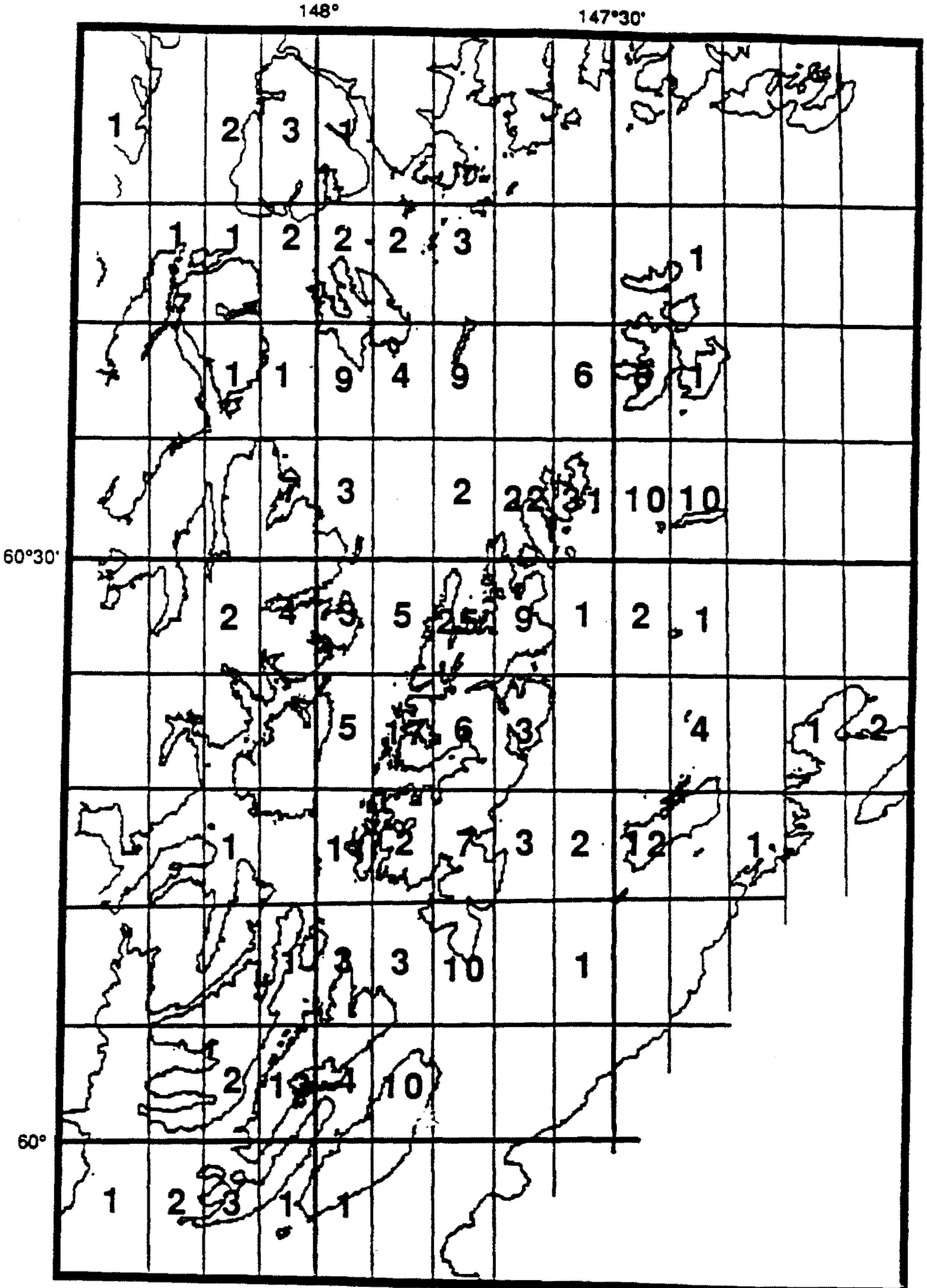

Figure II-3. Sheen sightings reported in 1990 by Exxon crews, through 2 June.

2) ADEC Oil and Equipment Tracking computer printout labeled "Sheens reported from 11/1/89 to 4/30/89," subsequently updated to 6/28/90.

Analysis

During discussions in May 1990, ADEC representatives stated their concerns that unless all the subsurface oil were removed from the beaches, there would be continuing potential to generate sheens. It was observed that in recent flights the number of sheens had been more limited than in earlier flights. It was suggested that the impending spring tides would significantly increase the sheening. An expected correlation with high-wind events was also discussed.

As a result of these discussions, the daily number of sheens reported in the ADEC data were plotted along with daily tidal range (highest high minus lowest low tide) and average daily wind speed (summarized into daily mean wind speeds from the Seal Island meteorological station) for the period 14 November 1989 through 6 March 1989 (see Figure II-4). Visual inspection of this plot indicates that there is little correlation between reported sheen frequency and either tidal phase or average daily wind speed for that period. However, there may be combinations or thresholds of wind and tide conditions that result in mobilization of oil from the shoreline. Few sheens are reported during low-wind periods.

The Exxon sheen data are listed by geographic coordinates. Figure II-3 shows that the highest density of sheen sightings was in the vicinity of Eleanor Island and the northern part of Knight Island. Of the all sheens reported in Prince William Sound, 28% were within the six tenth-degree squares centered on Disk Island. Seven percent of the sheens were reported in the vicinity of Sleepy Bay and Point Helen.

Sheen Volume Calculation

Number of sheens provides one estimate of potential risk to water-surface organisms and uses. Another measure is the volume of oil contained in the reported sheens. The volume of oil in sheens observed by Exxon and ADEC was calculated by multiplying the length and width of the sheen by a layer thickness estimated on the basis of reported sheen color.

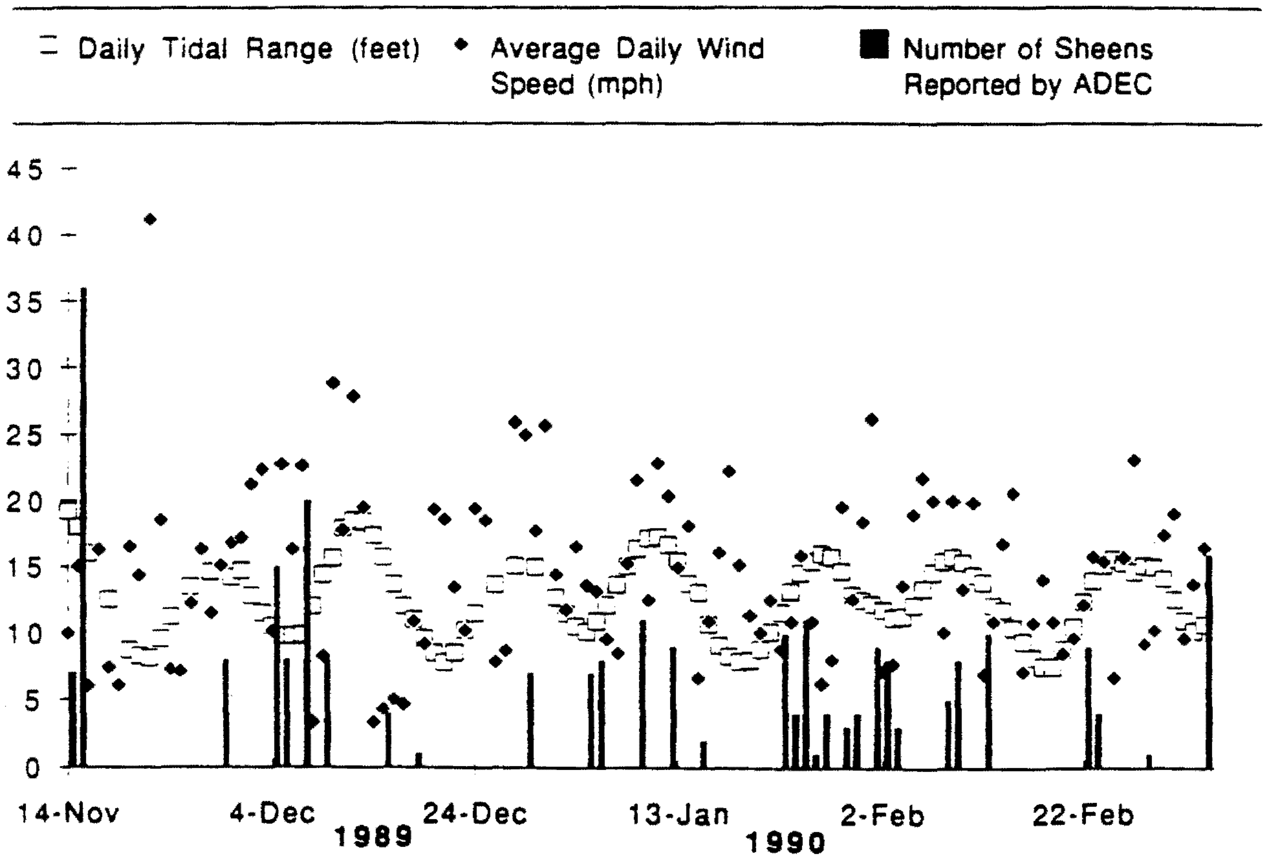

Figure II-4. Plot of average daily wind speed, tidal range, and number of sheens reported by ADEC during the four-month period beginning 14 November 1989.

The following conventions are used for relating sheen color to oil layer thickness:

ADEC*	Exxon*	Thickness (mm)
Gray	Very light sheen (transparent)	.00005
Silver	Silver sheen	.00010
Blue	First color	.00015
Rainbow	Rainbow	.0003
Copper	Dull	.001
	Yellow brown	.01
Brown	Light brown	.1
Black	Brown black	1.0

The Exxon data described the color of observed sheens in terms which are consistent with, but slightly conservative relative to, the NOAA-published Oil Spill Observation Glossary which relates color to approximate sheen thickness. The ADEC sheen color terms listed above were correlated with Exxon sheen colors because ADEC's color convention was not related to sheen thickness. Since observation flights did not occurred every day, sheen volumes were totaled on a weekly basis.

A scatter plot of the weekly totals from the Exxon data indicates that since March 1990 the volume of petroleum in sheens observed in Prince William Sound has decreased logarithmically (see Figure II-5A). Approximately every month the volume decreased by an order of magnitude. According to Exxon data, the amount of North Slope crude oil in Prince William Sound sheens was three quarts or less per week for each of the five weeks since mid-May 1990.

Analysis of the ADEC sheen data followed that of the Exxon data and the weekly values were plotted (see Figure II-5B). The trend in these data is less clearly defined, perhaps due to the inclusion of some non-North Slope crude oil sheens in the data. However, the data confirm that the volume of oil in observed sheens is relatively small.

The apparently rapid decrease of volume of North Slope crude oil in observed sheens throughout Prince William Sound suggests that, in general, the potential for further risk to water-surface resources and uses

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* Personal communication John Wilkinson, Exxon; personal communication Tim Langdon, ADEC

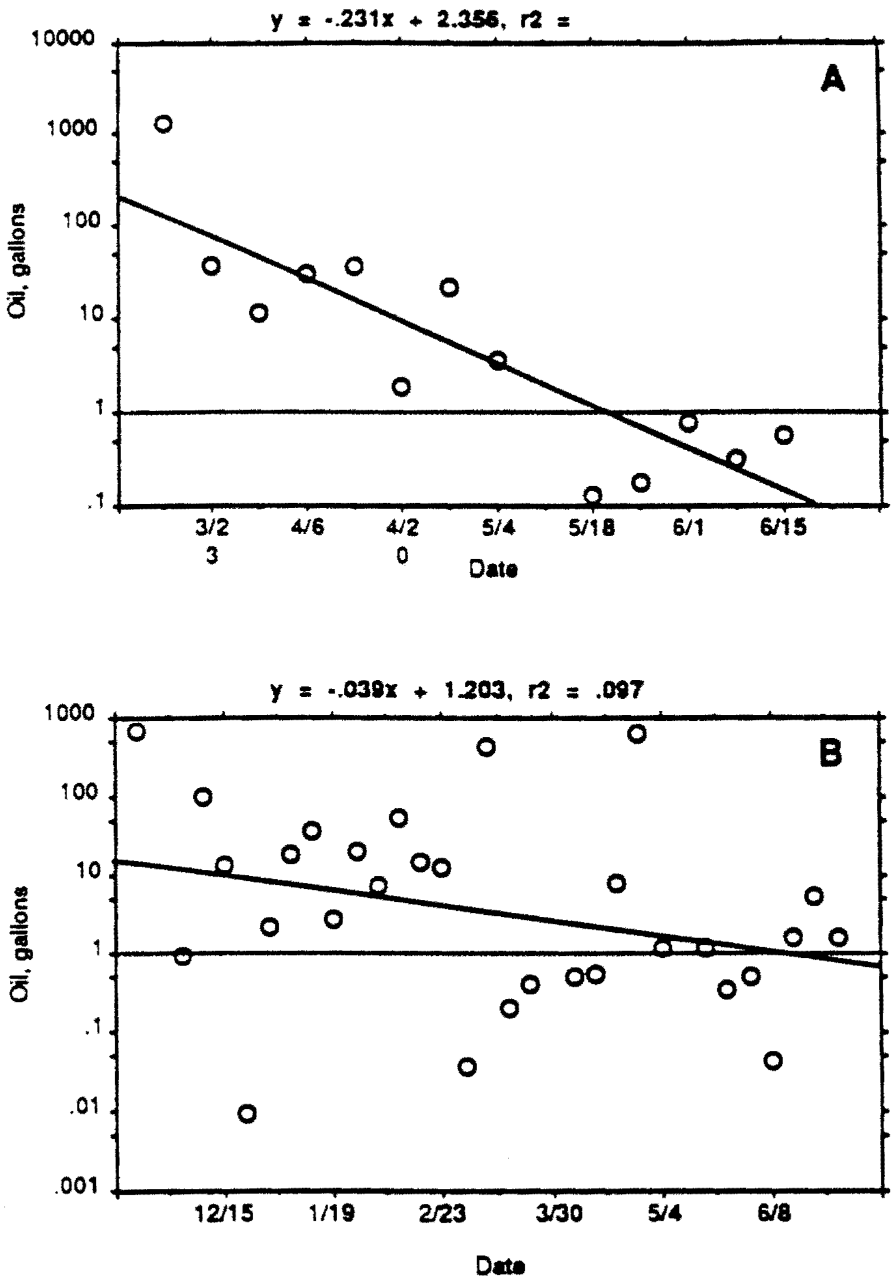


Figure II-5. A: Weekly sheen oil volumes based on Exxon data.

B: Weekly sheen oil volumes based on ADEC data. Note difference in coverage dates relative to the upper plot.

has decreased significantly over time. Both the number of sheens and, more importantly, the volume of this oil contained in sheens are currently very low. Additionally, Exxon's data show that since mid-April oil from the EXXON VALDEZ has contributed only a small portion to the sheening in the Sound (see Figure II-6).

Although intuitively one would expect that high tidal range and high winds would mobilize beach sediments and generate sheens from subsurface oil, existing data do not confirm a simple relationship. The apparent decrease in crude oil sheens can be attributed to the following factors:

1. Decrease in the amount of oil that can be mobilized from the shoreline under normal conditions.
2. Decrease in storm intensity and wave energy resulting in less reworking of upper beach face sediments.
3. Weathering of the oil and removal of the components which tend to form sheens.

Probably all three factors are contributing to the decrease in number, volume, and persistence of sheens.

<u>Release of Subsurface Oil by Storm Activity</u>

An instantaneous release of all of the subsurface oil in a beach during a major storm event would be the worst-case scenario for generation of sheens from subsurface oil. There is no way of accurately predicting when such a storm might occur. However, one did not occur in the winter of 1989, and it may be assumed that such a storm might be in the 20- to 100-year category (Miles O. Hayes, personal communication).

It is difficult to estimate the volume of oil which would be released during such an event. The total volume of oil in the beach is the maximum amount. As an example, ADEC calculated the amount of subsurface oil in Point Helen, using NOAA estimates of oil content, a shoreline length of 2.8 km, an oiled zone width of 15m, and an oiled depth of 120cm. These calculations resulted in an estimated volume of 1,832 gallons in the top 0-30cm and 19,369 gallons in the interval 30-120cm. If all of this volume of oil

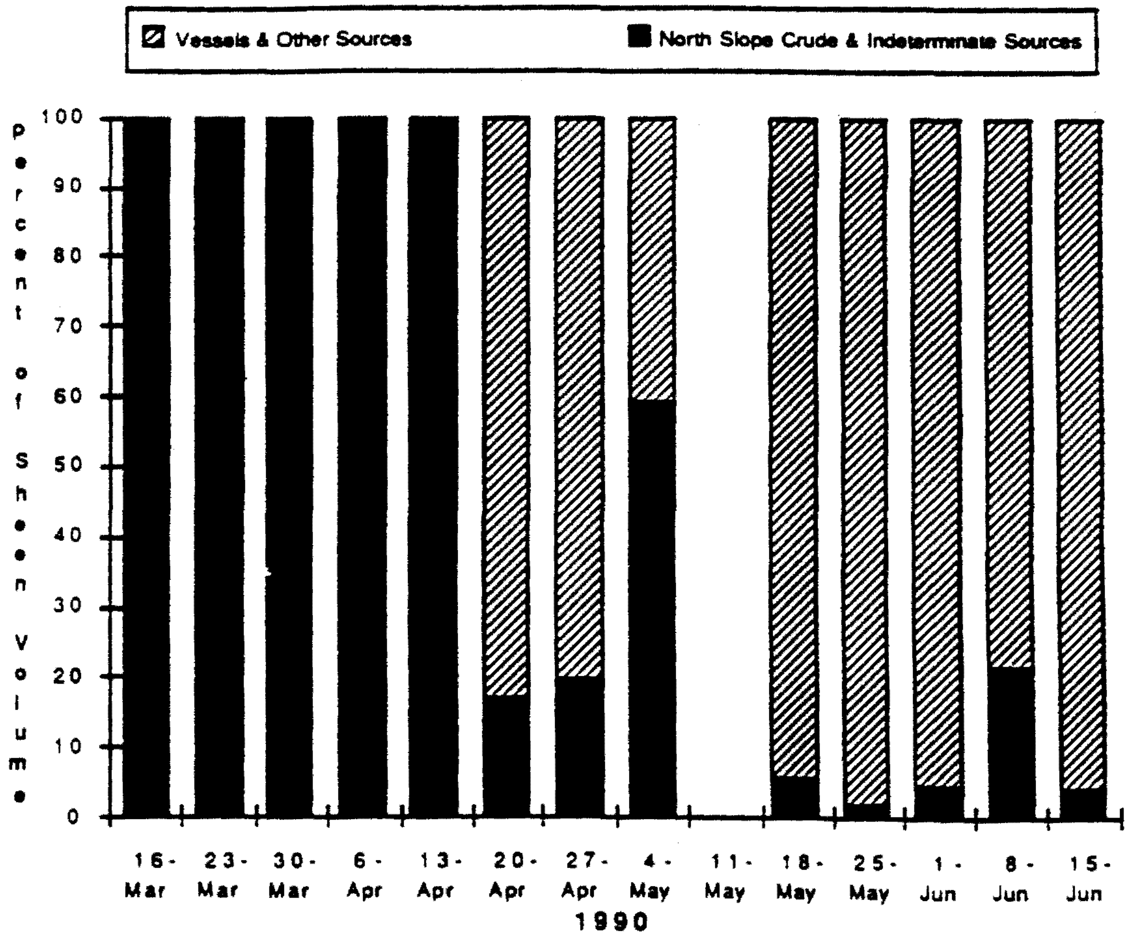


Figure II-6. Weekly sheens in Prince William Sound, as reported by Exxon.

produced a sheen with a thickness to produce the first trace of color, which is unrealistic, the size of the sheen would be 142 square miles.

There is no valid method to estimate the amount of sheen which would be produced from such a storm. After storm occurrences in 1989, oiled clasts appeared at the surface of beaches which had previously appeared clean. This indicates that not all subsurface oil brought to the surface would leave the sediments to produce sheen. Discussion in Section I.C.3. and in the following paragraphs indicates that the concentration of subsurface oil is continuing to decrease, and that the chemical nature continues to change and degrade into a substance with less potential to generate sheen. The storm waves would quickly disperse much of the oil as well. Therefore, only a fraction of the oil in the sediments disturbed by a major storm event would produce sheens. These conclusions are supported by simple field experiments in May and June 1990 at Point Helen and Sleepy Bay. Samples of oiled subsurface sediments from a number of sample pits were placed in the seawater along shore and agitated to see if sheens would be generated. Many times sheens were not evident. On the occasions when sheens were generated, they were observed to dissipate rapidly.

Composition of Sheens

A large number of qualitative sheen samples were collected during October 1989 in bays in western Prince William Sound as part of the Exxon water quality monitoring program. The results of analysis of these samples were published recently (Neff, 1990).

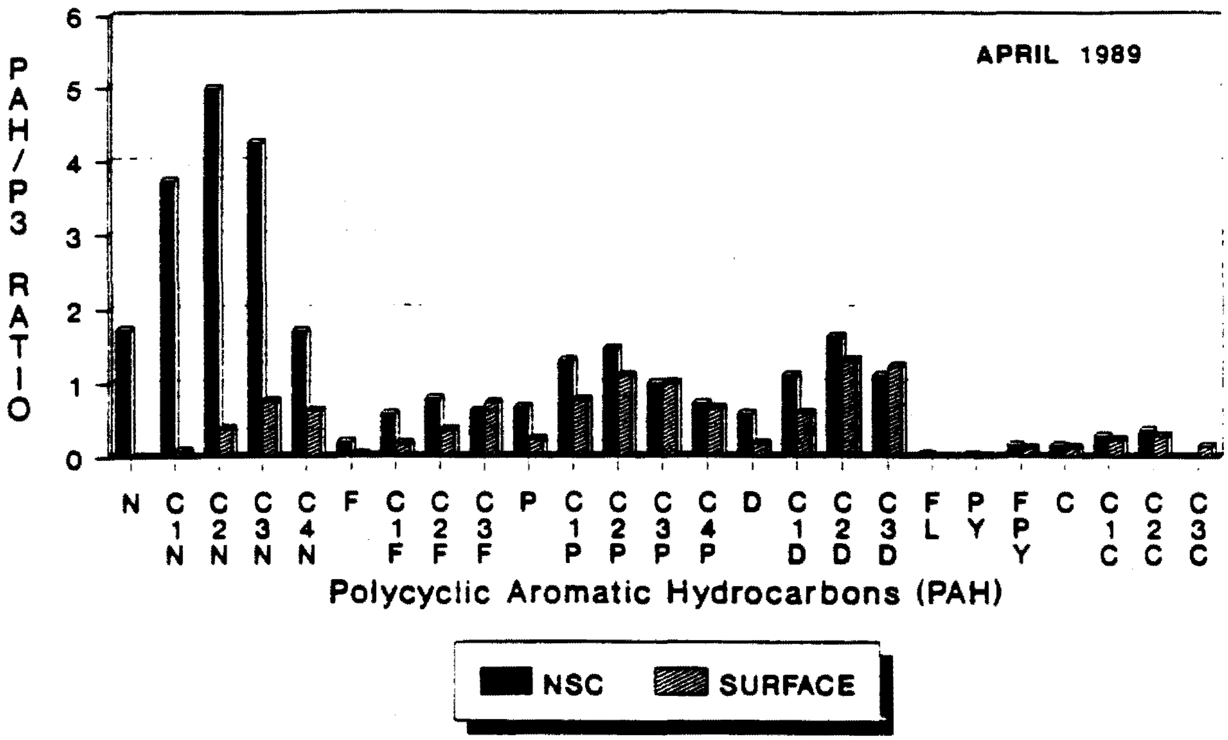
The concentration of total polycyclic aromatic hydrocarbons (PAH: the most toxic fraction in crude petroleum) in the 39 sheen samples collected in October ranged from below the detection limit (about 0.01 parts per billion: ppb) to 18.4 ppb. More than 60 percent of the samples contained less than 0.1 ppb total PAH. The sample containing 18.4 ppb was obtained from Northwest Bay, a site identified in earlier surveys as having been heavily oiled. A surface water sample collected in Northwest Bay in June 1989 contained 30 ppb total PAHs, probably derived primarily from sheen oil.

By comparison, concentrations of PAHs, measured as chrysene (a PAH compound) equivalents, in sheens collected from the open Atlantic

Ocean near Bermuda (an area of collection of abundant tar balls from heavy tanker traffic in the North Atlantic) were in the range of 0.4 to 3.04 ppb (Knap et al., 1986). These concentrations are considered typical for open ocean waters in the vicinity of heavy tanker traffic.

Careful evaluation of the PAH composition of the most concentrated sheens from bays in Prince William Sound revealed that they contained PAHs from two sources. Most of the PAHs were from highly weathered North Slope crude oil (the spilled oil). However, there were also traces of several PAHs that are found almost exclusively in PAH assemblages produced during combustion of fossil fuels. These PAHs probably were derived from engine exhaust soot. Due to weathering, nearly all the naphthalenes and most of the fluorenes and less alkylated phenanthrenes and dibenzothiophenes were depleted in the sheens compared to their concentrations in the fresh crude oil (Figure II-7). Because these are the most toxic fractions of crude oil, sheens from weathered crude oil are much less toxic to marine organisms than sheens from fresh crude oil. Some of the sheens observed in Prince William Sound during the spring of 1990 were not derived from the spilled oil but were from small discharges of diesel oil or bilge washings (often containing lube oil and hydraulic fluid) from boats. These sheens, having been derived from fresh refined oils, probably contained higher concentrations of the lower molecular weight, more toxic petroleum hydrocarbons than did the sheens of weathered crude oil.

To characterize the composition of the most recent sheens in Prince William Sound, a sample collected by NOAA in May 1990 and positively identified as spilled EXXON VALDEZ cargo oil by fluorescence was analyzed by GC/MS using a target-compound approach used to characterize the stranded oil. Figure II-8 compares the target analysis profile of fresh EXXON VALDEZ cargo oil to a sheen sample collected off of Block Island, EL-1A, in May 1990. Note that this profile comparison is limited to the concentration of target compounds only. If the GC/MS extracted ion chromatograms representing the saturate hydrocarbons (m/e 85) for the same two samples are compared as shown in Figure II-9, major differences in the composition of the aliphatic hydrocarbons are obvious. The sheening oil is much fresher in overall composition.



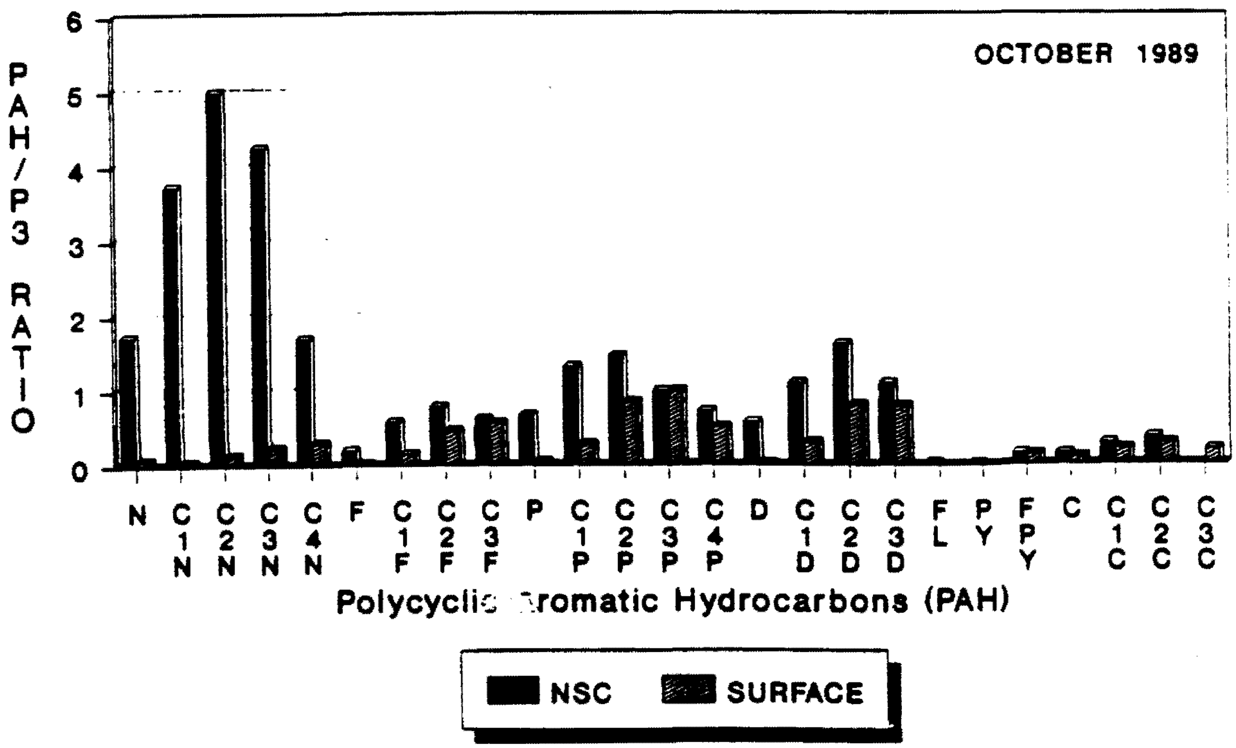


Figure II-7. PAH profiles for North Slope crude oil and sheen oil sampled in the Bay of Isles in April 1989 (upper) and October 1989 (lower).

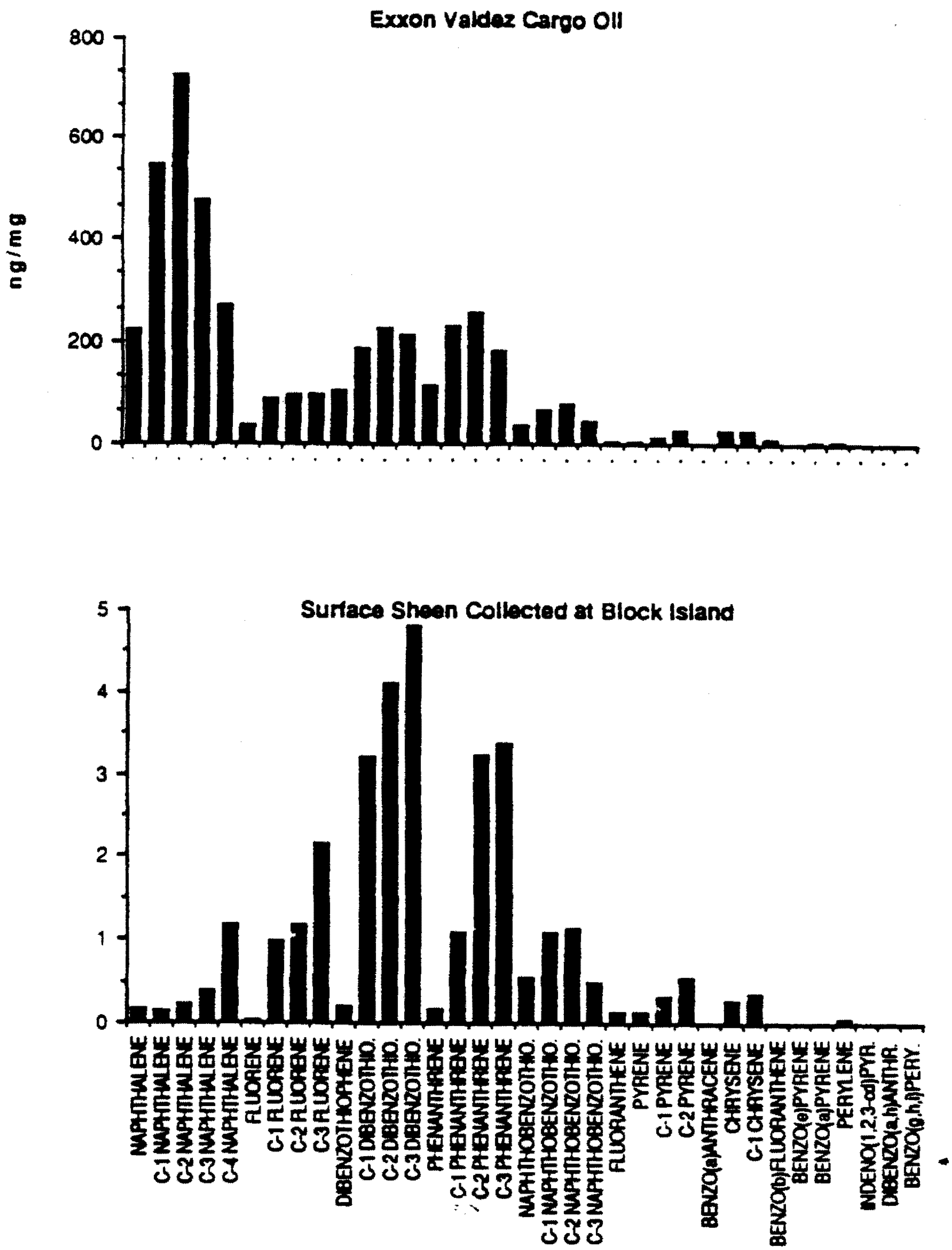


Figure II-8. Comparison of the analysis profiles of Exxon Valdez cargo oil and a sheen collected at Block Island (May 1990).

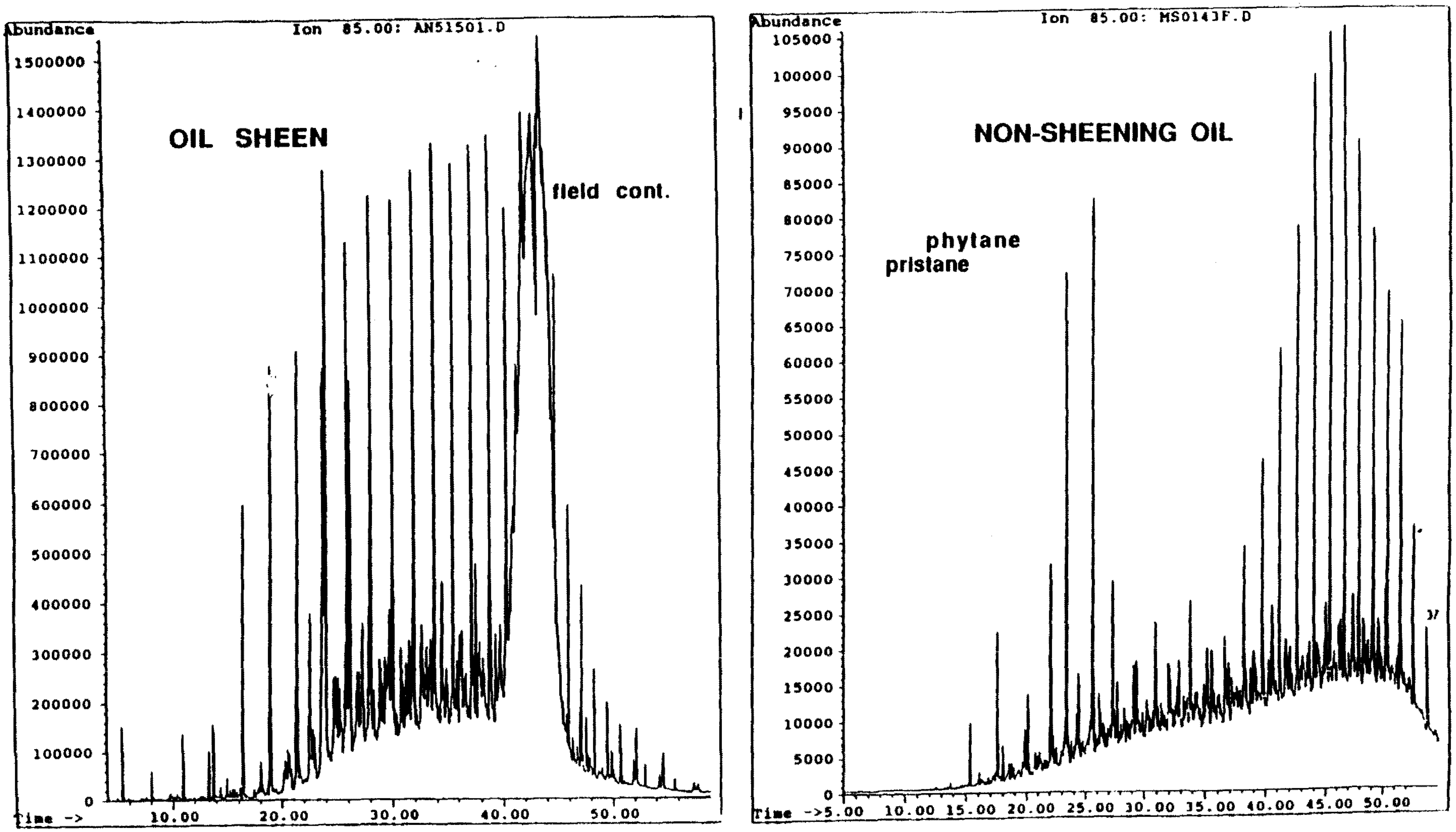


Figure II-9. Comparison of the ion chromatograms (m/e 85) for the sheen sample collected at Block Island and weathered, non-sheening subsurface oil from Sleepy Bay.

The sheen sample did not originate from fresh crude oil, but rather from stranded weathered oil trapped on the beach, which is still capable of producing sheens. When released back onto the surface of the water, the sheen's composition is continually being modified by its interaction at the air-water interface. As the weathered oil sheens, the lower molecular weight constituents are being lost to the atmosphere and water column through evaporation and dissolution. Sheens produced by weathered crude do not persist, but generally dissipate within several hundred feet from shore, probably due to dispersion into the water column.

What is the chemical composition of oil that will sheen? There is some confusion about what oil in the candidate beaches is responsible for sheens. In an attempt to characterize the oil that is a potential concern related to sheen formation, the oil types identified in section I.C.3 were characterized as to their potential to form sheens; comments about sheen formation were included in section I.C.3. These determinations were made either by comments from the samplers or by small microcosm sheen experiments.

The composition of oil that readily sheens is compared to a weathered subsurface oil sample that was identified as non-sheening in Figure II-10. The sample which sheens was collected during May 1990, at US-5 on the Barren Islands. The most obvious difference from this comparison is the presence of the lighter naphthalenes and fluorenes in the sheening oil relative to the non-sheening oil. It is concluded that, in order to sheen, weathered North Slope crude oil must have a significant amount of naphthalenes. The composition of the sheen produced during rock washer excavation operations will be source-oil dependent and be modified rapidly by weathering processes.

Effects On Biota

Weathered oil, such as mousse, tar balls, and deposits of tar and asphalt on the shore or buried in intertidal sediments, can be a source of oil sheens. Concern has been expressed that tidal and rainwater runoff containing sheen oil draining from the upper shore where most of the subsurface oil is concentrated might harm plants and animals living in the more biologically productive middle and lower intertidal zones. These concerns can be addressed from a theoretical basis or from observations of

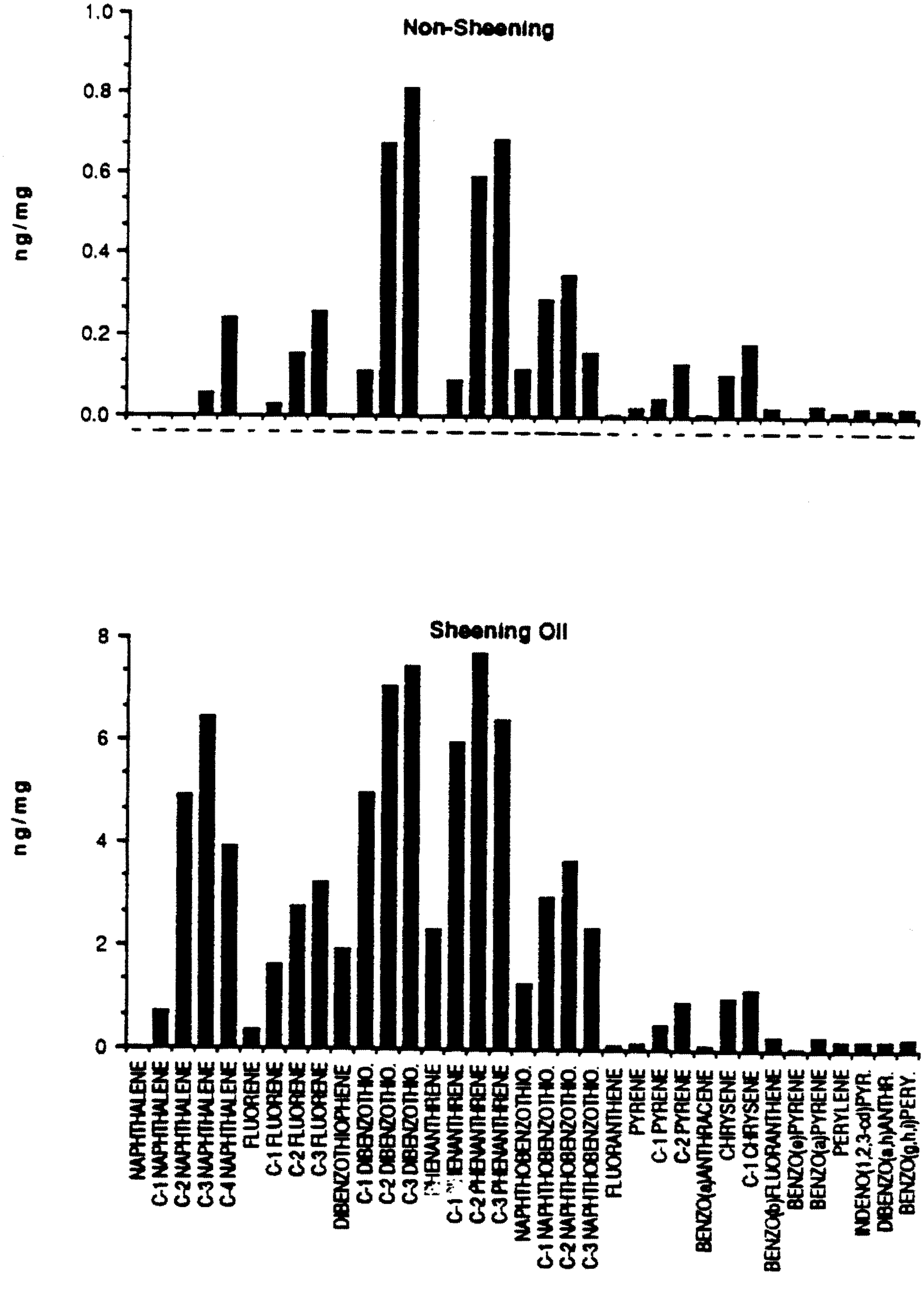


Figure II-10. Comparison of subsurface sediment from Sleepy Bay (non-sheening oil) to a sample collected at the Barren Islands that readily sheens.

the biological condition of shores in Prince William Sound that contain deposits of subsurface oil.

There is strong evidence from the scientific literature (see in particular, Southward and Southward, 1978) that recovery of a rocky shore is much faster if biological communities are only damaged and not wiped out completely by the oil spill and subsequent cleanup effort. The survivors have a strong influence on the rate and course of succession leading to reestablishment of the prespill biological community. Some, but not all, intertidal organisms were killed in the weeks immediately after the spill where the relatively fresh oil came ashore. Some organisms survived on even the most heavily oiled shores, particularly in the more productive middle and lower intertidal zones. There was little evidence of damage to biological communities in the shallow subtidal zone off oiled beaches. Subsequent shoreline treatment activities killed additional intertidal and subtidal organisms. However, die offs of intertidal organisms declined with time after the spill. This decline in biological effects probably was due to decreases in both the amount and toxicity of the oil as it weathered on or was removed from the shore.

In August, five months after the spill, even the most heavily oiled shorelines supported living biological communities. The species compositions of these communities were similar to those that would be expected on shorelines of similar types in Prince William Sound in the absence of oil. The major difference on most shorelines was a decrease in the abundance of individuals of each species and, in some cases, the absence of a few normally rare species.

Recruitment of key species of intertidal plants and animals to oiled shorelines began during the summer of 1989, continued through the winter, and is still going on, apparently at an accelerated rate, in the spring and summer of 1990, even on shores that still contain surface oil (mainly as patches of mousse or asphalt) or subsurface oil deposits. Visual inspection of the intertidal zone of shores containing subsurface oil reveals the presence of rich biological communities characteristic of the types of shores (wave energy and substrate type) being surveyed. In some places, overall abundance of plant and animal life appeared to be less than on similar but unoiled shorelines during the winter months. This apparent difference in overall biomass on the shore was much less apparent in April and May of 1990 (Sam Stoker, personal communication).

In many cases, the relative abundance of different species of plants and animals on oiled shorelines is different from that on similar but unoiled shorelines. However, the overall impression is that the intertidal communities are healthy and that ecological succession toward the prespill community structure is proceeding rapidly (Sam Stoker, personal communication).

Theoretical considerations of the behavior of subsurface deposits of weathered crude oil in coarse-grained intertidal sediments (see Sections I.C.3 and 4 on sediment oil content and pore water chemistry and Section II.A.6 on dispersion and flocculation) support the conclusion based on actual observations that biological communities are living in the intertidal zone of beaches that have remaining subsurface oil. To the extent that the remaining subsurface oil can still form sheens, the effects of these sheens on intertidal biota will depend on the duration of contact of intertidal plants and animals with the sheen hydrocarbons and the presence and relative concentrations of the potentially toxic fractions of the oil in the sheens.

When tidal water or freshwater runoff that has come in contact with subsurface oil in the upper shore drains from the intertidal zone of the beach during the falling tide, it may carry with it small amounts of oil in solution or as emulsions with clay-sized particles from the subsurface deposit. When this water emerges on the surface, petroleum hydrocarbons present in it in solution, or colloidal suspension, will have a tendency to leave the liquid phase and form a sheen on the water surface.

However, recent chemical evidence (Section I.C.3) indicates that much of subsurface oil buried in the upper intertidal storm berms on the candidate beaches is weathered to the point that it no longer readily forms a sheen. Concentrations of petroleum hydrocarbons, in particular the more toxic PAHs, in interstitial water of oiled sediments are very low (Section I.C.4). In addition, flocculation of weathered oil with clay-sized particles forms stable flocs (Section II.A.6) that probably will not readily disaggregate to form a sheen. Therefore, the likelihood that subsurface oil deposits in the upper intertidal zone of oiled shores are an important source of oil sheens is low, and will continue to decrease as the oil continues to weather.

Intertidal and shallow subtidal plants and animals could be exposed to sheen hydrocarbons in the water draining down the beach. At most, intertidal organisms will be exposed to oil sheens emanating from the shore for a brief time (about 10 to 15 minutes) twice during each tidal cycle when the organisms emerge during the falling tide and submerge during the rising tide, if sheens are actually present at these times. Under such an exposure regime, relatively high concentrations of hydrocarbons in the sheens would be required to produce acute or chronic biological effects.

The composition and relative concentrations of hydrocarbons in crude oil sheens will depend on the composition and, therefore, the degree of weathering of the oil from which the sheen emanated. It can be predicted that as the crude oil weathers and becomes depleted of the more soluble lower molecular weight hydrocarbons, the sheens derived from the oil also will become depleted of lower molecular weight hydrocarbons relative to higher molecular weight hydrocarbons. This appears to be the case. Sheens collected in April and October, 1989, from the Bay of Isles have different compositions relative to unweathered North Slope crude oil (Figure II-7). Both sheens are depleted of naphthalenes and to a lesser extent of other low molecular weight aromatics relative to the unweathered crude oil. However, the October sample is slightly more depleted than the April sample, indicating greater weathering.

Because the sheens from weathered crude oil are depleted of the more toxic fractions of the oil (the naphthalenes, phenanthrenes, and dibenzothiophenes: Neff and Anderson, 1981), they can be expected to be less toxic than the fresh crude oil or sheens derived from it. The available scientific literature indicates that weathered crude oil, and by inference sheens and water-soluble fractions produced from it, are substantially less toxic than the fresh crude oil (Guillen and Palafox, 1985; Capuzzo, 1987).

As shown in the previous section, there have been decreases in the frequency and volume of oil in sheens. Thus, the effects of subsurface oil released from the beaches likely will be of little or no consequence to marine birds or mammals if (Exxon's data are correct), a conclusion supported by the lack of any reports in 1990 of oiled birds or marine mammals in Prince William Sound.

Figure II-11 shows zones of convergence as depicted by ADEC. Concern has been expressed that sheens could congregate in convergence

Figure II-11. Zones of convergence in Prince William Sound.

zones and remain there for a length of time, increasing the potential for impacts to birds that also concentrate to feed in these zones. In June 1989, NOAA studied the potential for oiled popweed to impact the Northern Prince William Sound fishery district. Because a relatively rare storm could transport popweed (which had been exposed to heavily oiled beaches) to vulnerable areas, it was recommended that sampling be continued through that fishing season. It was also estimated that the residence time for fresh oil in convergence zones may be from 2 to 10 days.

Effects on Fisheries

The potential effects of sheens on fishery species will depend on the amount, persistence, and composition of the sheen and the mechanism of exposure of fishery species to the sheen. As discussed above, Exxon's data indicate that sheens in Prince William Sound typically are small, often containing no more than a few liters of oil, and are patchy in distribution. They usually are short-lived. Oil sheens dissipate rapidly due to natural weathering processes, including physical dispersion into the water column by breaking waves or downwelling in tidal rips (Word et al., 1986), evaporation (Stiver et al., 1989), degradation by the abundant bacteria living at the sea surface (Rambeloarisoa et al., 1984), and photooxidation by sunlight (Barth, 1984).

A great many species of organisms live on or just below the surface of the ocean. Neuston are bacteria, plants, and animals that spend all or part of their life cycles associated with the water surface (Hardy, 1990). Included among the neuston are the eggs and larvae of several species of fish and invertebrates that spend the remainder of their lives deep in the water column or on the bottom of the sea. Animals that produce highly buoyant eggs and larvae may suffer some losses if exposed to surface sheens. Research conducted in Puget Sound has shown the relatively high potential for effects in buoyant eggs exposed to sea surface microlayers. These microlayers had high concentrations of many pollutants, including aromatic hydrocarbons. The same effects could be expected in Prince William Sound where surface sheens exist. However, the magnitude of these effects would be small.

Walleye pollock and several species of soles are examples of commercially important Alaskan species of fish whose eggs and early larvae are neustonic. In sheens from Puget Sound, Washington, there was a poor correlation between concentrations of aromatic hydrocarbons in surface sheens and survival of the neustonic eggs of sand sole (Hardy et al., 1987a,b). For example, 36 percent of the fish eggs exposed to a sheen containing 8,030 ppb of total PAHs (as well as high concentrations of PCBs, pesticides, and metals) produced normal larvae. By comparison, all sheen samples from Prince William Sound analyzed to date contained 30 ppb or less of total PAHs (Neff, 1990). Some neustonic organisms seem to be able to adapt to sheens containing high concentrations of PAHs, primarily from combustion sources (Riznyk et al., 1987). Thus, based on these studies, it seems highly unlikely that sheens of weathered crude oil emanating from subsurface deposits of oil on shorelines will be toxic or produce harmful effects in plants and animals, including eggs of some fishery species, living in the neuston.

Pacific herring spawn in nearshore waters of Prince William Sound and deposit their eggs on intertidal and shallow subtidal stands of brown macroalgae. The eggs deposited on intertidal kelp could be exposed intermittently to oil sheens during successive tidal cycles. If droplets of fresh oil come in contact with and adsorb to the eggs, the eggs usually die or the embryos are malformed. However, a sheen of weathered crude oil is much less sticky than fresh crude oil or a recently-formed mousse and probably would not stick in biologically significant quantities to the eggs. In addition, the sheens are depleted of the more toxic fractions of the oil and so would be much less toxic to the eggs than fresh oil. Pearson et al. (1985) showed that herring eggs were not particularly sensitive to dispersed Prudhoe Bay crude oil. Exposure of the eggs to about 2,000 ppb of the dispersed oil for four days had no effect on the percent of eggs that hatched, the time to hatch, or on larval abnormalities. Thus, it is unlikely that the traces of toxic petroleum hydrocarbons in sheens of weathered crude oil would have significant deleterious effects on intertidal and shallow subtidal herring eggs.

The next spawn of Pacific herring in Prince William Sound will be in about 10 months. By that time the frequency of North Slope crude oil sheens on the water surface in areas where herring spawn should be substantially lower than the frequency observed in 1990, decreasing the likelihood that eggs will be exposed to crude oil sheens.

Adults of commercially important fishery species in Prince William Sound do not feed at the water surface and are unlikely to come in direct contact with petroleum hydrocarbons in surface sheens. In addition, adult fish are not particularly sensitive to oil. Effects on adult fish usually are reported at concentrations of oil hydrocarbons dissolved in the water column in the range of 1,000 to 50,000 ppb, whereas growth of young fry of some species of salmon was affected by oil concentrations in the range of 700 to 5,000 ppb (Vandermeulen and Capuzzo, 1983). The acutely lethal concentration of the water soluble fraction of Cook Inlet crude oil for pink salmon fry was about 1700 ppb (Rice et al., 1979). Sublethal effects on eggs and larvae of a few species of fish during chronic exposure to aromatic hydrocarbon concentrations of about 10 ppb were reported by Capuzzo (1987). Coho salmon were able to home successfully in seawater after exposure to 700 ppb of chemically dispersed Prudhoe Bay crude oil (Nakatani et al., 1985). Continuous exposure for 14 days of Pacific herring larvae to 300 ppb of Cook Inlet crude oil produced a significant decrease in growth rate (Carls, 1987). Ingestion of food containing 6,000 ppb oil had no effect on the larvae.

Concentrations of potentially toxic petroleum hydrocarbons measured in or directly under sheens of weathered North Slope crude oil are many times lower than concentrations required to cause acute lethal or chronic sublethal effects in juvenile and adult fish of commercial importance. Therefore, it is unlikely that the sheens coming from shores with subsurface oil deposits will cause observable harm to commercial fishery species.

There is also some concern that oil sheens could contaminate fishing nets and other fishing gear, and so taint the catch. This is unlikely. As discussed above, the amount of hydrocarbons in an oil sheen is very small per unit area of the sea surface. Therefore, the amount of contact between the sheen and a net hauled through a sheen would be very small. The sheen oil is not as sticky as fresh oil and so very little of the oil would stick to the water-wet net material. Fish coming in contact with a net that had been exposed to a sheen also would not be likely to retain much oil on their surface. Actual tainting of edible fish flesh requires that the living fish accumulate hydrocarbons from the water column or from its food. However, if surface contamination of fish from contact with tar balls or mousse was heavy, the flesh could become tainted during processing of the catch.

The Alaska Department of Fish and Game (ADF&G) has a policy of closing fishing areas if oil sheens are sighted in the area. This policy is designed to assure consumers that Alaskan seafood is uncontaminated. Thus, as a result of this policy, sheens do have a negative effect on small segments of selected fisheries, even if the sheens do not affect the fishery species themselves. As discussed above, sheens originating from deposits of subsurface weathered crude oil appear to be decreasing and are not likely to be significant in future years. Sheens are surely to be generated during excavating and they are not likely to be controlled all of the time. Therefore, if the subsurface oil deposits were left in place to weather naturally or to be treated by other methods, it is unlikely that there would be any difference in the impact on fisheries due to local closures caused by sheens in 1990.

II.A.5 HUMAN USE CONSIDERATIONS UNDER CONTINUED NATURAL CLEANSING AND APPROVED 1990 TREATMENTS

Human uses such as recreation, subsistence uses, and commercial fishing would most likely be negatively affected by remaining subsurface oil. The 1990 approved treatments that include such things as storm berm relocation and bioremediation will eliminate most of the existing living organisms in the treated beach sediments and leave remaining subsurface oil below the effective working depth of these treatments. Sheening of oil from the sediments would displace nearshore commercial fishing by gillnetters and seiners. Sheening is sufficient basis for closure of the fishery in the local area. The duration of this displacement would depend on the persistence of sheening. Recreation would be impaired whenever remaining oil is encountered by the recreationist. Bleeding of oil from the sediments, exposure of subsurface oil due to natural changes in the beach geomorphology or contact with subsurface oil in locations where they are likely to dig into the beach for the purpose of building a campfire, clam digging, or other recreational activity are likely situations that would contaminate recreational users and their equipment. Subsistence activities, especially those associated with the gathering of subsistence food from the intertidal area would continue to be curtailed due to the presence of oil. The slowed recolonization of intertidal biota and the risk of oil contact likely be reduced if sheening persists in the area.

Other activities associated with subsistence uses of these shorelines such as hunting, camping or fish drying would be negatively affected in the same manner as the recreation user if the subsistence user encounters oil.

II.A.6 ULTIMATE DISPOSITION OF SUBSURFACE OIL

Dispersion

When an oil/water mixture is subjected to turbulent mixing or wave action, droplets of oil break away from the main mass of the bulk oil and become suspended or dispersed in the water phase as small droplets. Because of the high surface tension at the oil/water interface, the droplets tend to coalesce to form larger droplets that return rapidly to the bulk oil phase. If natural or man-made surfactants are present, the surface tension is reduced and small droplets can form a relatively stable emulsion in the water phase. Subsurface oil in contact with pore water can be dispersed by normal tidal flushing if water movement and turbulence are great enough. However, as oil weathers, its viscosity increases, decreasing the tendency, and increasing the energy required, to form oil-in-water dispersions. On the other hand, some weathering processes result in the production of polar by-products of petroleum hydrocarbons, some of which have surfactant properties. In addition, many hydrocarbon-degrading bacteria produce surfactants that increase the surface area of the oil, enabling the bacteria to better metabolize it. These endogenous and biological surfactants may aid in dispersing the bulk oil into the pore water, especially during high energy situations, such as during storms.

Photooxidation and microbial degradation of the crude oil on the shores of Prince William Sound have produced polar degradation products that tend to remain with the oil. Many of the polar compounds are in the resin/asphaltene fractions of the oil and are not well characterized. However, analysis of samples of weathered North Slope crude oil from Prince William Sound by X-ray photo-electron microscopy has shown that carboxylic acids have been produced during weathering of the oil.

The relative concentrations of polar degradation products of hydrocarbons in the oil on shorelines increase as weathering of the oil occurs, indicating that the polar compounds are not readily leached from the bulk oil. Concentrations of polar organic compounds in oil samples collected on the shores of Prince William Sound have increased between March 1989 and January 1990 (Figure II-9). The increase was slow from March to June. From June to October, the concentration of polar compounds in the oil rose from about 7% to about 20% by weight of the residual oil. In the fall, the rate of increase in the concentration of polar compounds decreased again. The rate of polar organic compound production follows annual trends of intensity of solar radiation and water temperature.

These polar compounds are important because they increase the ease with which the residual oil can be dispersed into the water column by natural cleansing or human water-washing of the shore. They also have a profound influence on the interaction of the oil and the surfaces of rocks and finer sediment particles, particularly clays. The polar functional groups bind more strongly with rock or fine-particle surfaces than nonpolar functional groups do. This interaction is particularly important in the formation of flocs of oil droplets and clay particles which are removed easily from the shore by normal tidal pumping (see following section).

Flocculation

The constant movement of ice in glaciers grinds up the underlying rock to produce a fine-grained mineral dust, sometimes called glacial flour. Glacial flour is composed of micron-sized plate-like or angular mineral particles. This material is an important natural ingredient of the fine-grained marine sediments of Prince William Sound. It is an important component of the fine-grained sediment phase on many of the coarse-grained shorelines in Prince William Sound.

Research performed by Exxon has shown that these glacial flour particles interact in seawater with oil droplets to form a solids-stabilized emulsion that adheres less strongly to rocks than the bulk oil. The emulsion has the appearance under the microscope of a fluffy flocculate. It is formed when polar components in the oil residue are attracted to the electrically-charged surfaces of fine mineral particles such as clays, quartz, and silt in the presence of salt water. The process of oil flocculation will be discussed only briefly here in relation to the possible fate and effects of subsurface oil deposits on shorelines of Prince William Sound. A full report on these processes has been published (Bragg et al., 1990).

The emulsions or flocs form naturally wherever seawater, weathered oil containing high concentrations of polar substituents, and glacial flour occur together, as in subsurface deposits of oily sediments in the intertidal zone of shores in the Sound. The emulsions have a complicated and variable structure. In general, they consist of aggregates of many micron-sized droplets of weathered oil coated with mineral fines and surrounded by seawater. Flocculated clays provide the basic framework of the emulsions by weakly binding the oil droplets together. The aggregates have dimensions of from about 1 μm to about 100 μm in diameter, depending on the oil composition, mineralogy of the solid fines, amount of water agitation, and the relative concentrations of mineral fines, oil, and seawater.

The individual aggregates of the flocs are easy to wash from the rock surface because the polar bonding sites at the oil/water interface have been coated with the small mineral fines, and this reduces the ability of the oil to adhere to the larger gravel substrate. The components in the oil that cause it to stick to the gravel are polar functional groups, such as those containing hetero-atoms (N, S, O), in the high molecular weight asphaltene fraction of the oil. As discussed above, as the oil weathers, additional polar groups, particularly carboxylic acids, are produced by photooxidation and bacterial degradation. These contribute to the stickiness of the residual oil, but also contribute to stabilization of the oil/particle/seawater emulsions. As the oil becomes more polar with time, additional mineral fines bond to these polar sites, permitting the residue oil to remain as an emulsion accessible to additional weathering processes such as leaching and biodegradation, rather than becoming a more viscous asphalt-like solid on rocks, resistant to further degradation.

The emulsion also exhibits a fairly large hydrodynamic drag in flowing water because of its large surface area per unit weight. Thus, it is more easily swept away by fairly slow water currents. For example, in a series of column tests with 88 kg of oily gravel from Smith Island, almost three-fourths of the oil was removed from the gravel in less than 24 hours by flowing seawater through the column at velocities up to 19 ft/min (3.8 cm/sec), just sufficient to cause the gravel in the column to move slowly. This velocity is at least an order of magnitude less than water velocities imparted by gentle waves on the beach surface, and possibly another order of magnitude less than would be observed on the surface of high-energy beaches during a storm.

In contrast, the rate of movement of water through the interstitial spaces of intertidal sediments during normal tidal flushing is quite low. If the rate of tidal flushing of subsurface sediments is 10 pore volumes/tidal cycle, the rate of lateral movement of water through the pore spaces will be in the range of 0.05 to 0.1 cm/sec, depending on sediment porosity. In the column experiments, the minimum flow velocity required to dislodge flocs from rock surfaces is about 0.6 cm/sec. Thus, relatively little floc material is likely to be dislodged during normal tidal flushing. However, rapid and massive export of oil flocs from subtidal sediments probably will occur during storms severe enough to move sediments about.

The density of the flocculated emulsion is comparable to that of seawater. Most of the floc particles are lighter than seawater and will float on the water surface and be widely distributed by winds, waves, and currents after removal from the beach, especially since they are most likely to be removed from the substrate during storms. For those floc particles that do settle, sinking rates were estimated to be less than 1 ft/day (1.4×10^{-4} cm/sec) for particles with diameters of 1 to 10 μm and up to 45 ft/day (6×10^{-3} cm/sec) for the largest, heaviest flocs. At these settling rates, the oil/particle/seawater emulsions will be dispersed widely and diluted to very low concentrations before settling to the bottom. During settling, bacteria will continue to degrade the hydrocarbons in the flocs, further decreasing the amount of hydrocarbons reaching the bottom.

Flocs composed of weathered oil droplets and mineral particles will be much less toxic than the weathered oil in solution and probably less toxic than droplets of weathered oil, because the presence of a surface layer of mineral particles on the droplets will inhibit transfer of hydrocarbons from the droplets to organisms that might come in contact with floc particles. Because of their low toxicity and rapid dilution in the ocean, oil flocs will not be harmful to living plants and animals in Prince William Sound.

The formation of stable emulsions with clay-sized particles may help explain how some of the oil was removed naturally from the shores of Prince William Sound during the winter. It also suggests that the hydrocarbons washed from the shore were not merely redeposited at measurable concentrations in shallow subtidal waters near the shore where they might have the most adverse biological effects. Instead, they were possibly carried away and diluted and degraded to unmeasurable concentrations. This form of mobilization of subsurface oil will continue to go on wherever deposits of subsurface oil and glacial flour coexist, assuming those results from the laboratory are duplicated in the field. The rate of floc formation may increase as the oil continues to weather and the concentration in the oil of polar functional groups continues to increase.

Subtidal Deposition

Introduction. A team of ADEC divers visited each of the two rock washing candidate sites. On 7 June 1990, the team visited Sleepy Bay, Latouche Island, and on 8 June, Point Helen, Knight Island. The team constructed topographic profiles at each site and made substrate and biological observations along them. NOAA transect N-18 was continued offshore in Sleepy Bay, and NOAA transect N-1 was continued offshore at Point Helen. Subtidal profiles are presented in Figure II-12. Subtidal sediment samples were collected and released for analysis by NOAA.

Point Helen (KN-405; N-1). Table II-5 presents subtidal observations at site N-1. Changes in substrate occur where noted in the table, otherwise they remained the same to the next reported depth. In general, the substrate graded from a boulder, cobble, and sand mix at a few meters depth, to rocky gravel at greater depths, and to sandy silt at 100m. Below the mean lower low water (MLLW) level to 60m out, a dense kelp "forest," which was thick enough to make sampling of sediments difficult, covered a boulder/cobble substrate. The boulders had a coat of encrusting coralline algae and some tube worms. At approximately 60m from the MLLW mark, the kelp abruptly gave way to a predominantly sand bottom that graded abruptly to a greater proportion of fines at 100m. This bottom showed evidence of burrowing, and sea stars (*Pycnopodia*) were present as widely scattered individuals. Samples were collected at 0, 5, 10, 25, 50, and 100m water depths.

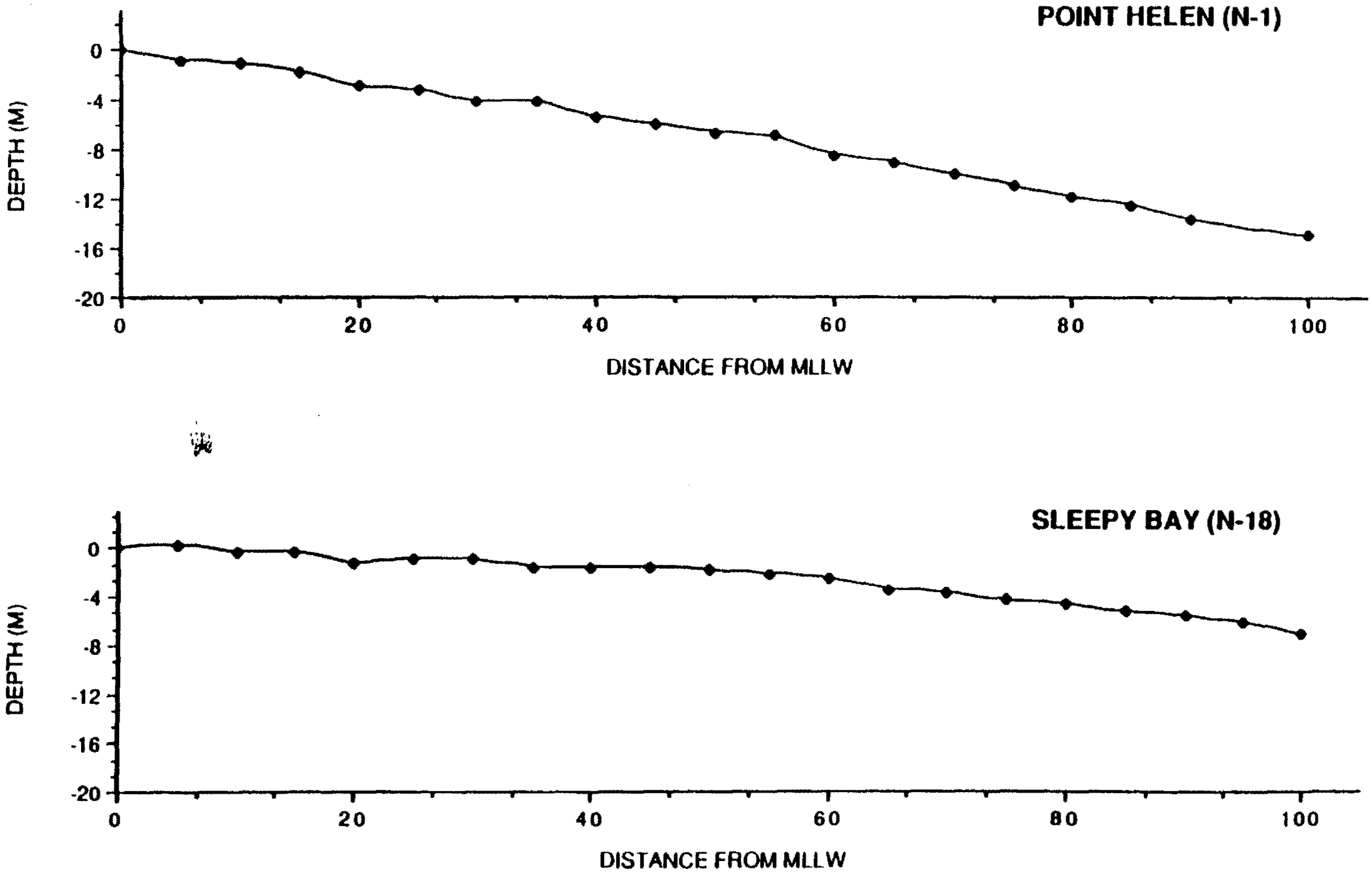


Figure II-12. Subtidal profiles off NOAA stations N-1 (upper) and N-18 (lower) in June 1990.

Table II-5

Point Helen (N-1): Dive occurred between 1616 and 1654 on 8 June 1990. Tidal height at start of dive was +8 feet. Start of 100-meter subtidal transect began at 8 fsw.

Distance From MLLW	Observations
0 meters	boulder armor with cobbles below; *Ulva* present in patches
5 meters	boulders continue, *Alaria* more common
10-15 meters	*Alaria*, 100% coverage
20 meters	laminarians begin to replace *Alaria*, som hedophyllum present, over boulders
25 meters (3 photos)	laminarians, hedophyllum, some encrusting algae on boulders
30-35 meters	less boulders, more gravel
40 meters	sand/gravel/few boulders, hedophyllum, some *Pycnopodia*
50 meters	*Laminaria*, hedophyllum, evastarias on sand/gravel substrate
55 meters	80% sand, 80% hedophyllum
60 meters	80% sand, 20% gravel, algae ends abruptly
65 meters	same substrate with diatom scum
70 meters	same substrate with shell fragments
75 meters	90% sand, some gravel, hedophyllum on large cobbles
80 meters	70% sand, 15% silt, some gravels, shell fragments
85 meters	scattered red algae, substrate grading to more fines
90 meters	more fines (fin kicks begin to resuspend silts)
100 meters	same substrate, grading to approx. 40% silt

Sleepy Bay (LA-18; N-18). Table II-6 presents subtidal observations at site N-18. Offshore, shallow depths were dominated by bedrock outcrops with sand and silt pockets; with increasing depth, outcrops diminished and silt began to dominate the bottom. In general, the subtidal community was composed of dense large brown kelps. Kelp leaves had thin white patches where spores had been released, and a light covering of spirorbid worms. White flocculent, probably organic in origin, covered much of the area from mid-transect to the end. Samples were collected at 0, 10, 25, 50, and 100m water depths.

Chemical Results of Sediment Samples. Results of chemical analysis by GC/MS of the subtidal samples collected at Sleepy Bay and Point Helen are shown in Table II-7. Extreme care in the analysis of these samples was taken because of the large fraction of biogenic hydrocarbons and their interferences . These results indicate that the level of PAH contamination is very low. Of the six samples analyzed from Point Helen, the total PAHs ranged from 0.1-8.8 ppb. At Sleepy Bay, the values ranged from 2.0 to 130 ppb. The general characteristics of the subtidal oil appear to be moderate to heavily weathered.

Higher concentrations would be expected in the more sheltered setting off Sleepy Bay. Since there does not appear to have been large-scale depositions of oiled sediments in the nearshore subtidal zone over the 1989-90 winter, when shoreline oil concentrations were high, future deposition of oiled sediments from remaining subsurface oil should be minimal.

Environmental Impacts

As described above in Sections I.C.4 and II.A.4, petroleum hydrocarbons will be washed continuously at a slow rate from deposits of subsurface oil in the intertidal zone by tidal pumping and freshwater runoff. In addition, if the subsurface sediments where the oil resides are aerobic, biodegradation by marine hydrocarbon-degrading bacteria will contribute to a decrease over time in the amount of subsurface oil remaining on the shore.

Biodegradation is the ultimate fate of all oil that is not removed from Prince William Sound by human activities, evaporation, and photooxidation. Atlas and Bronner (1981) estimated that the biodegradation The rate of rate of petroleum from the AMOCO CADIZ oil spill in intertidal

Table II-6

Sleepy Bay (N-18): The dive occurred between 2001 and 2045 on 7 June 1990. Tidal height at start of dive was approximately +4.0 feet. Start of 100 meter subtidal transect thus began at 4.0 fsw.

(Distance From MLLW)	(Observations)
0-20 meters	60% cobble, 40% pebble, w/barnacles (*B. glandula*) actively feeding (approx. 15% coverage on larger cobbles)
20 meters	light flocculent, algal growth on cobbles, *Pycnopodia helianthoides* individual present
25 meters	*Alaria* (20% coverage)
30 meters	*Alaria* and hydroids
35 meters	dense *Laminaria* begins, fully covering cobbles
45 meters	*Laminaria* continues, with white (organic) flocculent on blades
50 meters	tube worms (*Serpula* or *Crucigera*) present on larger cobbles under brown algae
55-70 meters	large cobbles continue under dense brown algal cover
70 meters	encrusting coralline algae on cobbles, spirorbids common on *Laminaria* blades
80 meters	substrate shifting to sand/cobble mix
90 meters	sand/gravel/pebble under *Laminaria* one starfish found w/tumor or undeveloped arm (bump where arm should radiate out)
100 meters	80% sand with a few boulders and silt

Table II-7. Concentration of Specific Compounds in Subtidal Sediments at the Candidate Beacnes.

GC/MS QUANT RESULTS					
SAMPLE LOCATION	Pt.Helen	Pt.Helen	Pt.Helen	Pt.Helen	Pt.Helen
SAMPLE STATION	N1	N1	N1	N1	N1
DISTANCE FROM SHORE	0M	5M	10M	25M	50M
SAMPLE TYPE	grav.	peb.	peb.	sed.	sed.
MONTH SAMPLED:	6/8/90	6/8/90	6/8/90	6/8/90	6/8/90
COMPOUND	(ng/mg)	(ng/mg)	(ng/mg)	(ng/mg)	(ng/mg)
NAPHTHALENE	ND	ND	ND	ND	ND
C-1 NAPHTHALENE	ND	ND	ND	ND	0.022
C-2 NAPHTHALENE	ND	ND	ND	0.043	0.330
C-3 NAPHTHALENE	ND	ND	ND	0.022	0.087
C-4 NAPHTHALENE	ND	ND	ND	0.024	0.110
FLUORENE	ND	ND	ND	ND	0.006
C-1 FLUORENE	ND	ND	ND	0.019	0.019
C-2 FLUORENE	ND	ND	ND	0.017	0.064
C-3 FLUORENE	ND	ND	ND	0.027	0.120
DIBENZOTHIOPHENE	ND	ND	ND	ND	ND
C-1 DIBENZOTHIO.	ND	ND	ND	ND	2.200
C-2 DIBENZOTHIO.	ND	ND	ND	0.036	0.240
C-3 DIBENZOTHIO.	ND	ND	ND	0.120	0.810
PHENANTHRENE	ND	ND	ND	0.017	0.022
C-1 PHENANTHRENE	ND	ND	ND	ND	0.018
C-2 PHENANTHRENE	ND	ND	ND	0.022	0.130
C-3 PHENANTHRENE	ND	ND	ND	0.045	0.780
NAPHTHOBENZOTHIO.	ND	ND	ND	ND	0.066
C-1 NAPHTHOBENZOTHIO.	ND	ND	ND	0.024	0.330
C-2 NAPHTHOBENZOTHIO.	ND	ND	ND	0.034	0.520
C-3 NAPHTHOBENZOTHIO.	ND	ND	ND	0.100	0.440
FLUORANTHENE	ND	ND	ND	ND	0.011
PYRENE	ND	ND	ND	ND	0.013
C-1 PYRENE	ND	ND	ND	ND	0.081
C-2 PYRENE	ND	ND	ND	0.024	2.100
BENZO(a)ANTHRACENE	ND	ND	ND	ND	0.008
CHRYSENE	ND	ND	ND	0.009	ND
C-1 CHRYSENE	ND	ND	ND	0.014	0.160
BENZO(b)FLUORANTHENE	ND	ND	ND	ND	ND
BENZO(e)PYRENE	ND	ND	ND	ND	ND
BENZO(a)PYRENE	ND	ND	ND	ND	0.041
PERYLENE	ND	ND	ND	ND	ND
INDENO(1,2,3-cd)PYR.	ND	ND	ND	ND	ND
DIBENZO(a,h)ANTHR.	ND	ND	ND	ND	ND
BENZO(g,h,i)PERY.	ND	ND	ND	ND	0.014
TOTAL=	ND	ND	ND	0.60	8.70

est. det. limit 0.01 ng/mg All values are valid to 2 significant figures only.

Table II-7. Cont.

GC/MS QUANT RESULTS					
SAMPLE LOCATION	Pt.Helen	Sleepy B.	Sleepy B.	Sleepy B.	Sleepy B.
SAMPLE STATION	N1	N18	N18	N18	N18
DISTANCE FROM SHORE	100M	0M	5M	10M	25M
SAMPLE TYPE	sed.	peb.	peb.	sed./grav.	peb.
MONTH SAMPLED:	6/8/90	6/8/90	6/8/90	6/8/90	6/8/90
COMPOUND	(ng/mg)	(ng/g)	(ng/g)	(ng/g)	(ng/g)
NAPHTHALENE	0.010	0.002	0.010	0.039	0.002
C-1 NAPHTHALENE	0.016	0.004	0.033	0.044	0.003
C-2 NAPHTHALENE	0.270	0.027	0.400	0.280	0.021
C-3 NAPHTHALENE	0.068	0.052	0.035	0.600	0.016
C-4 NAPHTHALENE	0.022	0.089	0.120	1.400	0.012
FLUORENE	0.005	0.002	0.023	ND	ND
C-1 FLUORENE	0.012	0.020	0.030	0.180	0.056
C-2 FLUORENE	0.019	0.200	0.190	1.600	0.049
C-3 FLUORENE	0.130	0.820	0.800	0.570	0.120
DIBENZOTHIOPHENE	0.003	0.007	0.017	0.034	0.004
C-1 DIBENZOTHIO.	0.840	0.096	1.300	0.890	0.035
C-2 DIBENZOTHIO.	0.097	0.007	1.100	10.000	0.170
C-3 DIBENZOTHIO.	0.310	4.500	4.100	30.000	0.190
PHENANTHRENE	0.110	0.013	0.210	0.120	0.009
C-1 PHENANTHRENE	0.023	0.072	0.110	0.550	0.025
C-2 PHENANTHRENE	0.072	0.860	0.800	6.000	0.120
C-3 PHENANTHRENE	0.210	3.400	0.230	15.000	0.210
NAPHTHOBENZOTHIO.	0.008	0.150	0.180	2.300	0.012
C-1 NAPHTHOBENZOTHIO.	0.094	3.300	2.800	13.000	0.200
C-2 NAPHTHOBENZOTHIO.	0.170	5.100	4.700	17.000	0.440
C-3 NAPHTHOBENZOTHIO.	0.340	4.600	4.200	12.000	0.650
FLUORANTHENE	0.023	0.000	0.320	0.037	ND
PYRENE	0.024	0.062	0.260	0.021	0.014
C-1 PYRENE	0.052	0.450	0.580	2.400	0.080
C-2 PYRENE	0.110	1.300	1.200	5.700	0.230
BENZO(a)ANTHRACENE	0.007	0.002	0.160	ND	0.014
CHRYSENE	0.060	0.018	1.400	0.210	0.140
C-1 CHRYSENE	0.080	1.300	2.200	7.400	0.240
BENZO(b)FLUORANTHENE	0.014	0.011	0.380	0.032	0.054
BENZO(e)PYRENE	ND	ND	ND	ND	ND
BENZO(a)PYRENE	0.016	0.016	1.100	0.087	0.100
PERYLENE	ND	ND	ND	ND	ND
INDENO(1,2,3-cd)PYR.	ND	0.002	0.058	ND	0.014
DIBENZO(a,h)ANTHR.	ND	0.010	1.200	ND	0.013
BENZO(g,h,i)PERY.	0.008	0.004	0.310	ND	0.034
TOTAL=	3.20	27.00	31.00	130.00	3.30

est. det. limit 0.01 ng/mg

Table II-7. Cont.

GC/MS QUANT RESULTS		
SAMPLE LOCATION	Sleepy B.	Sleepy B.
SAMPLE STATION	N18	N18
DISTANCE FROM SHORE	50M	100M
SAMPLE TYPE	peb./shell	peb./shell
MONTH SAMPLED:	6/8/90	6/8/90
COMPOUND	(ng/g)	(ng/g)
NAPHTHALENE	ND	0.012
C-1 NAPHTHALENE	ND	0.028
C-2 NAPHTHALENE	0.028	0.310
C-3 NAPHTHALENE	0.011	0.200
C-4 NAPHTHALENE	0.086	0.850
FLUORENE	ND	0.016
C-1 FLUORENE	ND	0.096
C-2 FLUORENE	0.027	0.820
C-3 FLUORENE	0.072	2.700
DIBENZOTHIOPHENE	0.001	0.018
C-1 DIBENZOTHIO.	0.016	1.200
C-2 DIBENZOTHIO.	0.049	4.900
C-3 DIBENZOTHIO.	0.080	0.840
PHENANTHRENE	0.007	0.055
C-1 PHENANTHRENE	0.013	0.330
C-2 PHENANTHRENE	0.073	3.600
C-3 PHENANTHRENE	0.200	6.700
NAPHTHOBENZOTHIO.	0.009	0.035
C-1 NAPHTHOBENZOTHIO.	0.190	5.200
C-2 NAPHTHOBENZOTHIO.	0.430	8.100
C-3 NAPHTHOBENZOTHIO.	0.480	6.700
FLUORANTHENE	0.009	0.028
PYRENE	0.010	0.025
C-1 PYRENE	0.044	1.100
C-2 PYRENE	0.110	2.400
BENZO(a)ANTHRACENE	ND	ND
CHRYSENE	0.064	0.110
C-1 CHRYSENE	0.120	3.700
BENZO(b)FLUORANTHENE	0.013	0.480
BENZO(e)PYRENE	ND	ND
BENZO(a)PYRENE	0.008	1.300
PERYLENE	ND	ND
INDENO(1,2,3-cd)PYR.	ND	0.075
DIBENZO(a,h)ANTHR.	0.012	ND
BENZO(g,h,i)PERY.	0.005	0.310
TOTAL=	2.10	52.00
est. det. limit 0.01 ng/mg		

sediments was 0.5 μg hydrocarbons/gram dry sediment (parts per million) per day. The rate of microbial degradation of hydrocarbons in anaerobic sediments was several orders of magnitude lower than in aerobic sediments (Atlas et al., 1981). Shiaris (1989a,b) reported that both naphthalene and phenanthrene are degraded in sediments of Boston Harbor, Massachusetts at a rate of about 0.4 μg/gram dry sediment/day. The rate of degradation of the PAHs increased as the concentration of the PAHs in the sediment increased. The rate of microbial degradation of oil in sediments usually is nutrient-limited, so if bioremediation of subsurface oil deposits is performed, degradation rates can be expected to be much higher.

Any oil that washes from the sediments onto the water surface as a sheen or into the water column as a dispersion, flocculate, or dissolved phase, also is subject to microbial degradation. Estimated rates of microbial degradations of petroleum hydrocarbons from crude oil in the water column range from 1 to 10 μl/day beneath the surface slick produced by crude oil from the AMOCO CADIZ spill (Aminot, 1981).

The buried oil itself will have no environmental effects as long as it remains buried on the shore and out of contact with living organisms. Potential effects of sheens possibly emanating from subsurface oil deposits were discussed above. However, if some of the subsurface oily sediment is mobilized from the upper shore by winter storms, it may be deposited in a location where living plants and animals might come in contact with it.

The oil coming off a shore during a storm probably will be associated primarily with clay flocs. These flocs are buoyant or neutrally buoyant and so will be carried away with the prevailing storm currents and be diluted over a wide area. The oil in the flocs, because of its stable association with clay-sized particles, probably is much less bioavailable than physically dispersed oil droplets. This low bioavailability coupled with the expected rapid dilution of the flocs in the water column will yield subsurface oil of minimal toxicity to organisms living in the water column.

Some of the oil coming off the shore may be associated with high concentrations of suspended particles (silt and sand from the shore) and be deposited with them in subtidal sediments. Studies of the Baffin Island experimental oil spill (Boehm et al., 1987) and the AMOCO CADIZ oil spill (Gundlach et al., 1983) have shown that in the few months immediately after a spill, concentrations in excess of 100 ppm of oil can be deposited in subtidal sediments if oil comes ashore and subsequently erodes from the beach. Because of substantial dilution and dispersion of the oil during remobilization and redeposition, the concentrations in subtidal sediments of oil derived from shore erosion will usually be much less than the concentrations of oil buried in the intertidal zone.

Oil bound to sediments is much less toxic than oil in solution or dispersion in the water column. This is because the sediment-bound oil has only a very limited bioavailability to marine organisms associated with the sediment (Neff, 1984). Anderson et al. (1978) exposed trays containing sediments contaminated with 700 to 6,000 ppm of Prudhoe Bay crude oil in the intertidal zone of Puget Sound. Concentrations of hydrocarbons in the sediments decreased by 21 to 85 percent in 100 days, depending on sediment grain size. Hydrocarbons were lost most rapidly from coarse sediments. No substantial inhibition of recruitment of benthic organisms was observed at these concentrations of petroleum hydrocarbons in sediments.

The discussion in Section II.A.3 summarized the PAH concentrations in fine-grained sediments which resulted in adverse impacts to benthic communities. The concentrations of PAH in subtidal sediments collected offshore Point Helen and Sleepy Bay are all below these levels of concern. With a much lower and decreasing source of oil-contaminated fines, it is unlikely that there will be any effects of weathered petroleum eroding from beaches on shallow subtidal benthic communities off oiled shorelines.

These studies show that oil, particularly weathered crude oil, in sediments is not very toxic to benthic marine organisms. Concentrations of petroleum hydrocarbons orders of magnitude higher than concentrations expected in nearshore sediments of Prince William Sound as a result of storm erosion of intertidal subsurface oil allow recruitment and normal development of benthic communities.

II.B. EXCAVATION/ROCK WASHING

II.B.1. EFFECTS ON BEACH SUBSTRATE

Introduction

Sediment transport and remobilization on gravel beaches is a very poorly understood process. Studies such as the ones by Carr (1971), Caldwell (1981), and Hattori and Suzuki (1978) give good indications of sediment transport directions and general clues concerning the wave parameters involved. However, their data cannot be used to calculate volumetric sediment discharges or even derive specific predictions of the nature of the detailed sorting and transportation processes. Fortunately, a year's worth of fairly detailed data have been collected by Exxon, ADEC, and NOAA on beach morphological changes at the two candidate sites, which can be used in a hindcast mode to predict future changes on the two beaches. Furthermore, we can call on the professional experience of the coastal geologists representing the three principal parties (Owens/Exxon; Gundlach and Pavia/ADEC; and Hayes/NOAA), who have studied the impact of several oil spills on gravel beaches.

Beach Stability

Resorting. If we make the basic assumption that the material to be washed would be dredged from approximately the upper one-third of the beaches, replaced to that area and graded to near its original configuration after washing, we can say the following regarding resorting at the two sites:

1. Point Helen

The present sorting is such that finer particles, namely smaller cobbles and pebbles, are sorted into berm accumulations at the top of the profile and the coarser cobbles and boulders form an armor on the middle and lower portions of the profile. We predict that if the washed material were replaced in approximately the same position it was taken from, the berms would reestablish themselves with more or less the same size grading after two or three medium-sized storms, probably in less than one year. The armoring process may require a few more storms, possibly covering a period of one to three years. Armor of gravel material was reestablished in two years on the inner beach of Ediz Hook at Port Angeles Harbor, Washington after disruption by cleanup of the ARCO ANCHORAGE spill (Miller, 1989).

2. Sleepy Bay

NOAA's profile data indicate that sediment on the upper and lower portions of this beach is mobilized readily. Armoring is not as well developed at Sleepy Bay as it is at Point Helen. Again, the sediment resorting process on the upper profile should occur in less than one year. Because the middle portion of the beach changes more slowly, the complete reestablishment of the original profile and sediment re-sorting may require up to three years, but would probably take place sooner.

Loss of Fines. There are no significant quantities of "fines" in the sections of either beach where the oil is buried, with the sand content being less than 10% at both sites (Sleepy Bay contains up to 15% sand plus granule). After mixing of the sediment by the washing process and replacement of it on the beach, it is conceivable that some of the sand will be transported out of the beach areas. However, because the sand exists primarily as a matrix to the already fixed framework of the beach, the impact of its potential loss on the beach's morphology should be nil.

Loss of Infauna. There is very little infauna on the macrofaunal scale in the sections of the two candidate sites where oil is buried. Micro- and meiofauna that may now exist in those sections of the beaches would obviously be eliminated during the washing process.

The sediment replaced on the beach would in all likelihood be reestablished to inhabital form (for micro- and meiofauna) within a period of one to three years at both sites. This statement is based on the conclusion that the beach morphology and sediment sorting pattern would be reestablished within that time frame.

Beach Erosion. The following quote from Carter (1988) has relevance to this issue: "both residual wave asymmetry transport and overpassing abilities, plus relative immobility, favors retention of large clasts on the beach. Secondly, large coarse grained beaches, through their enhanced roughness and permeability characteristics, are more stable in higher-energy environments." Furthermore, it has been observed at numerous localities by the authors, that during storms, coarse clasts tend to move landward, while sand moves seaward, which accounts for the omnipresent coarse-grained storm berms throughout the spill site.

The only ways we can envision serious beach erosion occurring as a result of the rock washing process follow:

1) The material removed is not replaced on the beach at or above MSL.
2) A major storm hits while the gravel is being washed, leaving the beach unprotected.
3) A significant amount (>30%) of fine material is present, which is carried away in suspension during the washing process.

As far as the two candidate sites are concerned, both are backed by bedrock, and both contain bedrock outcrops in the intertidal zone. Therefore, retreat of the beach of more than a few meters is virtually impossible at the two sites. Both beaches are coarse enough so that, if the washed material is replaced in the upper intertidal zone, it should stay there. In short, beach erosion (enhanced by rock washing) is not envisioned to be a significant problem at the candidate sites.

The Disruption of Armoring. Armoring is a sorting process that is best developed in the middle and lower portions of the beach profiles of Prince William Sound. Whereas the process of armor formation has been studied on gravel bars in rivers, the authors know of no such study on beaches. Armoring of a gravel beach in a wave tank experiment was recently reported by Petrov (1989), but no explanation for the process was given. On river bars, once the armor is formed, a process known as "structural strengthening" occurs, such that a stronger flow is required to transport the materials available (at least one-fourth greater). Consequently, disruption of armoring could conceivably create a more unstable condition on the beach regarding sediment transport. However, at the two candidate sites, armoring is not as well developed in the areas where oil is buried as it is further seaward, with the Point Helen beaches showing more advanced armoring than the beaches at Sleepy Bay. Therefore, care should be taken at Point Helen to avoid disrupting the armored lower portions of the beach, if at all possible, should gravel washing be carried out.

II.B.2. EFFECTS ON INTERTIDAL BIOTA BY EXCAVATION/ROCK WASHING OF BEACH MATERIAL AND PROJECTIONS OF RECOLONIZATION

Introduction

The impacts of field implementation of the rock-washing method on indigenous biota of candidate beaches would be expected to be significant, encompassing both direct and indirect effects. The physical components of the process that will be necessary to remove the oil remaining in the targeted beach material would involve major direct effects, while the subsequent disruption of habitat would result in a number of indirect and more subtle effects primarily associated with delayed recolonization of the treated intertidal zone.

Effects of Removal/Processing/Replacement

Intertidal marine organisms generally inhabit the surfaces of beach materials such as boulders and cobbles, the depressions underneath these materials, and the interstices between these materials (see Section I.C.7). Some species live in the coarse sand and gravel that underlies the larger cobbles and boulders, generally frequenting approximately the upper 10cm of the beach materials. In order to excavate the subsurface oil remaining on the candidate beaches, the uppermost layers of surficial material must be removed, and it is likely that organisms inhabiting these strata will be displaced.

Under normal circumstances, the upper tidal zone in Prince William Sound usually supports considerably fewer species and lower numbers of organisms than the middle and lower tidal zones on the beaches (Haven, 1971). The richest intertidal communities generally inhabit the lower tidal zone (sometimes referred to as the "*laminaria* zone") that is dominated by thick algal mats and is uncovered by only minus tides. The rock-washing operations would be expected to directly affect primarily the upper portions of the intertidal beach, as these were the most heavily oiled and contain the residual amounts that would be subject to removal. However, some secondary impacts of the process would also be anticipated in the middle and lower intertidal zones.

It is very unlikely that any organisms could survive the impacts of excavation, hotwater washing, and replacement. Most of the organisms from the excavation zone would be crushed, cooked, or buried during the operation. However, it is relevant to note that the upper tidal zone, where most of the excavation work would be focused, generally does not support a very rich biotic community.

If wheeled or tracked vehicles are used on the beach to excavate the oiled material, they would affect horizons of the beach both above and below the oiled zone. Backing and turning maneuvers would crush biota outside the targeted oiled zone and rearrange substrate comprising habitat for biota. Should backing and turning maneuvers extend into the relatively rich communities of the middle and lower tidal zones, significant intertidal resources could be damaged or destroyed. These lower tidal zone resources can include large algae, prey items for salmonids and marine mammals, bivalve molluscs, and many species of nearshore fish.

Many of the prey items that juvenile salmon depend upon as they enter saltwater are detritivores, or carnivores that prey upon detritivores. The detritivores are often epibenthic zooplankton, such as gammarid amphipods and harpacticoid copepods, that live near or on the bottom and consume the debris that results from the decomposition of plant and animal matter. If the gravel washing process removes the fine particulate materials from the beach, including the detritus accumulated under boulders and cobbles, the food supply for these detritivores will be lost. Observations made at Sleepy Bay suggest that gammarid amphipods were abundant under rocks that had accumulated algal debris and absent under rocks that had no debris. The adverse impact of the removal of organic material on a key component of the nearshore food web likely would be a fairly short-term phenomenon. The detritivore-based food web could be expected to reestablish once the beach became physically stabilized within one to three years.

Dead plant and animal materials resulting from the beach treatment activities likely will be consumed by a variety of birds, crabs, or other scavengers. This sudden source of food may attract opportunistic feeders to the treatment sites, but this congregation would be expected to be a temporary phenomenon.

Recolonization of Treated Beaches

Controlling Factors. In estimating the recolonization rate and time to recovery of the beaches to be treated with the rock washer, a number of factors must be considered. First, it was assumed that the rock washer would sterilize all of the beach materials that are treated, and, therefore, recolonization would start with effectively denuded, inorganic substrate. Second, it was assumed that the recolonization process would be unimpeded by the presence of any remaining oil after treatment by rock washing. Third, it was assumed that the beaches that would be treated with rock washing are those that were both oiled and composed primarily of pebbles and cobbles.

Two important milestones that were estimated in this analysis were the time required to reach the "equilibrium point" and the time to "recovery." The equilibrium point is the time following a disturbance at which the number of species colonizing the beach has stabilized. Recovery is a variously defined concept, which in its most rigorous definition specifies the point in time at which the pre-spill conditions are attained in terms of the number of species, the relative abundance of each species, pre-spill patchiness, and the dynamic functioning of the community (i.e., the normal proportion of autotrophs, grazers, detritivores, predators). Others have adopted a less comprehensive and more pragmatic definition of recovery as being the point in time when the community in a disturbed area exhibits variation within the range of that observed in comparable communities from undisturbed control areas (Lissner et al., 1988).

The amount of information and the accuracy of the models available to estimate the recolonization equilibrium points greatly exceed those available for estimating biological recovery. Although the concept of recovery is not an overly complex one, the task of studying and estimating recovery in natural systems is extraordinarily difficult because of the number of factors considered, the inherent variability in those systems, and the lack of baseline data to which post-disturbance conditions can be compared. Ganning et al. (1984) have commented that rarely are sufficient data available to evaluate "natural" variability in a community, and they noted that a previously existing community may have been replaced by a slightly altered community that exhibits variability within a range considered "normal," despite its different composition.

In Prince William Sound, very little quantitative information is available on the intertidal communities that inhabited gravel beaches before the oil spill. While it would be possible to generate data of this type by examining unoiled beaches, such an inventory and estimates of variability within Prince William Sound gravel beach communities do not currently exist. With these limitations in mind, the operational definition of recovery that is used here is a descriptive one: that point in time when a core group of dominant species (expected in the absence of a major disturbance such as an oil spill) has returned to generally comparable levels of abundance and community structure. The core group of dominants that is expected at the candidate beaches is based upon the lists and descriptions of Rosenthal et al. (1982). This qualitative definition is necessitated by the uncertainties discussed above and the lack of information on the intertidal zone of Prince William Sound gravel beaches that would be likely candidates for the considered treatment.

The initial stages of colonization of a sterile substrate, similar to that expected after rock washing, usually begins with a bacterial and colonial diatom community (Woods Hole Oceanographic Institution, 1952; Johansen, 1971). Subsequent colonists often depend upon these organisms for food or to modify the substrate so it is suitable for further colonization. Organic matter and fine-grained material encourage the growth of the pioneer colonizing species, and because the rock washing process will remove both organic and smaller grain-size fraction materials from the beaches and not return them after treatment, it is possible that the initial stages of recolonization will be delayed. However, it should be noted that fines will be generated by sediment handling and transport back to the shore, as well as grading, replacing some of this sediment loss

The oil that accumulated on the Prince William Sound beaches had different effects upon different species of organisms. The most resistant species survived the direct exposure to oil. However, according to data available from two locations in Herring Bay (Houghton and Erikson, unpublished), many of the organisms that initially survived the oil spill were subsequently eliminated by the effects of high temperature, high pressure beach washing. Most of the littorinid snails, green algae, brown algae, sponges, large snails and barnacles did not survive the washing. At two lower intertidal zone sites, the numbers of taxa observed before treatment were 28 and 21 and decreased to 16 and 17 after treatment, respectively. In two upper intertidal zone sites, the numbers of taxa decreased from 14 to 5 at one site and remained unchanged at 5 taxa at the other site. In the IRISH STARDUST oil spill in Alert Bay, British Columbia, numerous resistant species of algae and invertebrates survived the effects of the oil spill (Green et al., 1974). The more sensitive amphipods did not survive, but recolonized the rock and cobble beaches within one year following the spill.

The factors that would control the recolonization rate following the selection of the rock-washing process include the time at which the beach material would be resorted and restabilized, the season during which restabilization would take place, the proximity to brood stock for potential recruitment, the size of the treated area, and the geomorphology of the post-treatment beach. The net effect of all these influences determines whether recolonization is inhibited or accelerated.

An important factor in the restabilization of the beach to pre-treatment conditions is the rate at which the layer of "armor" cobbles and rocks would be reestablished. Many of the infaunal and ambulatory species find refuge from exposure and predators under and around the relatively larger, or armoring, material or in the smaller beach material sheltered by the armoring substrate. Large cobbles (some in the form of horizontal plates) generally are uppermost on these beaches and protect the gravel underlying them. Beneath the gravel, it is common to find coarse sand, small pebbles and gravel, and, occasionally, some mud. If these materials are redistributed back onto the beach randomly after the rock washing treatment, recolonization can be seriously hampered until the materials become resorted in a manner similar to that existing before the treatment.

In the ARCO ANCHORAGE spill of 1985 in the state of Washington, Miller (1989) noted that physical disturbance of beaches from intrusive cleanup activities breached the naturally occurring armor layer, but also found that within two years much of the armor layer had been reestablished. In contrast, a recent habitat restoration project in Vancouver, B.C. (J. Marliave, personal communication; Vancouver Aquarium, 1990) returned a silted intertidal area to its status as a spawning ground for several fish species only after careful selection and placement of substrate. Other intertidal mitigation projects in the same area that placed substrate in a more random fashion were completely unsuccessful in providing functional habitat. Although this study focused on the lower intertidal and subtidal zones, it demonstrated that in some habitats, the physical characteristics of a predisturbance beach may take a long time to naturally reestablish. Protected, low-energy beaches would probably require much longer time periods to return to their earlier configurations.

In the absence of a directed effort to redistribute cleaned substrate to approximate the physical structure of the pretreatment beach (which would obviously be a painstaking undertaking), resorting and restabilization of the beaches are expected to occur as a result of wave and tidal action. On many of the beaches that would be considered for excavation and washing, the larger cobbles and boulders are located on top of and provide a protective armor for the underlying finer material. After treatment, these materials would be redeposited in a random, unsorted manner. The process of resorting would involve the redistribution of the finer material, the uncovering of the larger material and the settling and restabilization of materials that would be mobilized.

The completion of this restabilization process on an exposed boulder/cobble beach, such as KN-405 (Point Helen), would likely require one year (Hayes, personal communication). It may be slower in the lower tidal zone than in the upper and mid-tidal zones, due primarily to the increased exposure to high energy dynamic forces in the upper and middle intertidal portions of the beach. Moreover, the sediments in the lower tidal zone are not expected to be physically mixed by the rock washer. The completion of the restabilization process on a more protected and less armored beach, such as that of LA-18 (Sleepy Bay), would likely be slower, possibly requiring up to three years.

Many intertidal invertebrates breed in the springtime, with timing of larval release or spawning linked either directly or indirectly to the spring bloom of phytoplankton (Carefoot, 1977). Most intertidal species in Prince William Sound spawn successfully each year in the spring and summer, although a small amount of spawning of a few species may occur year-round. Off Kodiak Island, a significant proportion of colonization of artificial substrates was found to occur between March and September (Long, 1972). If restabilization of the beaches occurs during the late summer or fall, the first major recruitment may involve one or a few opportunists late in the summer or early fall. For example, oligochaete worms have been observed in May 1990, in relatively heavily oiled beach materials in Sleepy Bay and at Point Helen. These opportunists may delay the development of a normal community if they competitively exclude other colonists the following spring. The recruitment of the "normal" mixture of species likely may not begin until the subsequent spring (beginning in mid-May or early June). Recolonization initiated during the spring would most likely result in the quickest attainment of a "normal" community.

When limpets and other grazing gastropods have been killed or removed and excluded from rocky shores, successive blooms of green and brown algae often have occurred. As a result of the competitive exclusion of other species from the community, recovery has been relatively slow—up to seven years (Nelson-Smith, 1975). The rock washing process would probably kill all gastropods in the treated portions of the beaches. The subsequent recolonization process may also involve the "green" phase of intense algal growth described by Nelson-Smith (1975).

Recolonization of a shore by animals can occur in four ways: 1) by migration of adults of mobile species from unaffected areas; 2) by direct settlement of planktonic spores or larvae dispersed by breeding organisms in unaffected areas; 3) by migration of juvenile stages of species with direct development (i.e., without planktonic larvae); and 4) rafting in of adults or their egg masses attached to floating seaweed or debris (Southward and Southward, 1978). Most intertidal invertebrates spawn by broadcasting progeny into the water column as plankton. Potential colonists for a clean beach can be produced great distances away and will drift to or actively seek suitable substrate for possible colonization. Other species do not broadcast their young into the water for dispersal, instead attaching their progeny to the rocks on the beach or carrying their young. Proximity of a clean beach to potential parents (brood stocks) becomes a very important controlling factor for these organisms, especially if the treated beach is large. Reintroduction of these species from the fringes of the cleaned beach would require that the species move onto the newly cleaned beaches. A relatively long time, several to many years, may be required for these organisms to reinvade and reinhabit all of the cleaned beach.

Some species do not routinely spawn every year, but, rather, spawn irregularly over 5- to 30-year cycles. These infrequently reproducing organisms include some predatory snails and sea stars, and some clams (D. Lees, personal communication). However, most of the species occur in either the lower intertidal or the subtidal regions that probably would not be directly impacted by a rock washing operation, although secondary effects of vessel operations and heavy machinery movements might stress the animals. Also, the sea stars and snails are motile and able to migrate from unaffected areas to affected areas.

Recolonization Projections. There are no site-specific data available from studies of beach excavations for use in estimating the recolonization rates of Prince William Sound beaches. The following projections, therefore, are based upon the observations made in other types of studies and from data collected elsewhere in northern latitudes. Table II-8 summarizes information obtained on recolonization of clean substrates following some kind of disruption, including oil spills in which dispersants or removal treatments were used. The studies listed in this table and described in more detail in Appendix A, were used to help predict recovery rates presented in this section.

Once the beach becomes stabilized and potentially recolonizable, the recolonization process would be expected to follow a pattern observed on clean materials in many studies performed throughout the world. The early colonists likely will be opportunists such as bacteria, colonial diatoms, brown and green algae, some infaunal worms, and epifaunal barnacles. The number of species will increase very quickly during the first three to six months, particularly in the spring, and will reach an equilibrium point during the following autumn beyond which few additional species will arrive. The assemblages will undergo a dormant season during the late fall and winter during which little recruitment will occur, followed by another burst of recolonization in the following spring. During the following spring, some early colonizing species will likely be replaced by late-arriving species, but the total number of species will not change to any great degree.

The rock washer will significantly deplete beaches of the detritus and other organic matter upon which many species feed. Reaccumulation of this material from phytoplankton, upland vegetation, and marine algal

Table II-8
Summary of Biological Recovery Rates for Previous Oil Spills

NAME	DISRUPTION FACTOR	BEACH TYPE	DEF. OF RECOVERY	RATE OF RECOVERY	REFERENCE
(1)Fouling panels Kodiak, Alaska	Sterile surfaces	Pressed asbestos wood panels	Equilibrium point in species colonization	Initiated in June 1969, 1970 Equilibrium point in 3 to 9 months	Long, 1972
(2)Lake Huron beach nourishment	Movement of gravel beach-crushing/grinding	Cobble/Gravel/ Sand beaches	Compare with beaches before removal	8 months-no difference between control & new habitats	Nester & Poe, 1982
(3)Puget Sound colonization curves	Sterile surfaces	Pressed asbestos wood panels	Equilibrium points for number of species	Clean sites--18-22 species at 7 months Contaminated sites--14-16 species at 7 months Both equilized at 18 months	Schoener, 1983
(4)Great Alaska Earthquake 1964	Uplift at sites	Mostly rocky outcrops/beach areas	Replacement of original species; equilibrium of species	Verrucaria/algae films--1 yr. 3 successive barnacle species replacements in 0-1-5 yrs. Mussel attachment replacements 3 algal species replacements of Fucus/Porphyra/Fucus 0-1-5 yrs.	Haven, 1971
(5)Torrey Canyon spill, England, 1976	Kuwait Crude oil 15,000 tons; toxic dispersants 10,000 tons	Intertidal Rocky shores, coves, beaches	Comparison with other areas not contaminated; species richness; wide range of sizes & age groups	Recolonization sequence; wave exposed areas Fucus dominant--5-8 yrs; other areas with residual toxicity, large scale mortality, removal of herbivores 9-10 yrs. for recovery	Southward & Southward, 1978
(6)Amchitka Atomic bomb tests 1969 & 1971	Atomic explosions uplifted 12cm & 3-110cm respectively	Bedrock & Boulder	Comparison of communities with pre-tested communities	Approximately 3 yrs. for community to contain the same species & abundance as pre-tested habitats	O'Clair & Zimmerman, 1986
(7)Recovery & Recolonization Outer Continental Shelf	Any operational disturbance due to man's activities on the environment (oil exploration in outer cont. shelf regions)	Continental Shelf region	Return to pre-disturbance state; return to the range of variation compared to control areas	None predicted; principles to consider: 1)type & size of disturbance, 2)seasonal fluctuations, 3)water movement, 4)selectivity of substratum for settlement, 5)biological interactions, 6)frequency of disturbance of substratum	Lissner, et al., 1986
(8)Amoco Cadiz spill Brittany 1978	Light Arabian & Iranian crude oil 223,000 metric tons; Bunker C 2000 tons and beach treatments	Bedrock headlands, sandy beaches, mud estuaries, mixed sand/gravel/cobble beaches, marshes	Returen to normal population densities; no pathologies; amount of catches normal; mortalities normal; equilibrium; hydrocarbons in tissue low; species numbers return	Oil removed (gravel/cobble) naturally in 1-1.5 years; Plants come back approx. in 1-2 years; animals come back approx. in 1-3 years; dominant species switching; still some imbalances & perturbances after 3 yrs	Marshall & Gundlach, 1990 Gundlach, 1981 Gundlach, 1983 Glemarec, et al., 1982 Bodin, et al., 1982
(9)Tamano Oil Spill Casco Bay, Maine, 1972	No.6 fuel oil 100,000 gal. algae removal & hot water treatment	Steep rock shelves, outcrops, intrusions, cobble beaches	Comparisons between oiled & unoild control sites	1 yr. after--mud flats-no new recruits; new shellfish at control sites; best recovery where no complete removal of seaweeds, uncropped areas, areas not cleaned with hot water, less heavily oiled lower intertidal area	Marshall & Gundlach, 1990
(10)Amazzone Oil Spill Brittany 1988	Paraffinic medium oil 150 tons	Bedrock headlands, sandy beaches, mud estuaries, mixed sand/gravel/ cobble beaches, marshes	Measurements of different temperature washes; cost of rock washing technology Biological measure of recovery?	Results: higher temp. better than cold water; water + petrolium cut & surfactant, removal good but not complete with pebbles still $50.00 per cubic meter of pebbles + transporting costs	Marshall & Gundlach, 1990

debris will dictate the colonization rates of these species, which include ambulatory species, epibenthic zooplankton, and infauna. Many of the epibenthic zooplankton are also important prey items for young salmonids. Slow recovery of these species may adversely affect the salmon from local streams.

Epifauna probably would locate stable suitable settling substrates before the infauna would find the sand and small gravel stabilized sufficiently for the latter to recolonize. The beach restabilization process involves the redistribution and winnowing of the finer material, and until this material ceases shifting, the prospective infaunal recruits would be faced with a relatively unfavorable environment.

Studies of the recolonization of new or denuded substrate following the Alaskan earthquake of 1964 (Haven, 1971), the U.S. nuclear tests on Amchitka Island between 1969 and 1971 (O'Clair and Zimmerman, 1986), and the TORREY CANYON oil spill of 1967 (Southward and Southward, 1978) all suggest that while an equilibrium point in the number of intertidal species observed may be reached in a relatively short period of time after a disturbance, return to comparable patterns of species dominance and distribution may take much longer to occur.

Haven (1971) noted an apparent change in the dominant barnacle species of the middle intertidal zone of areas uplifted during the 1964 Alaskan earthquake. Both *Balanus glandula* and *Semibalanus balanoides* had been present in the pre-earthquake intertidal zone, with *B. glandula* probably being the dominant species. Post-earthquake settlement of barnacles was almost exclusively composed of *S. balanoides*. However, five years after the earthquake, Haven found that *B. glandula* had once again become the dominant barnacle species*. Similarly, in 1965 an intertidal site that had been dominated by the brown alga *Fucus* was instead heavily covered by *Porphyra*. By 1968, *Fucus* had returned to its previous dominance.

O'Clair and Zimmerman (1986) discussed the effects of shoreline uplift caused by detonation of nuclear devices below Amchitka Island in

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*O'Clair and Zimmerman (1986) have questioned Haven's identification of the two species of barnacles, but Haven's conclusions are presented here as stated.

1969 and 1971, and found that nine months after the second event, the intertidal shoreline remained in flux. Recolonization was still taking place, and algal and invertebrate species richness was found to be lower than they were under pretest conditions. Nearly three years after the uplift, intertidal communities were similar to those existing prior to the tests. In 1985, O'Clair found that with some differences, the new communities approximated those that had existed prior to the disturbance.

Southward and Southward (1978) documented marked differences in occurrence and distribution of intertidal species, particularly algal and herbivore colonizers, over a seven year timespan along rocky shores denuded by heavy use of oil dispersants in the TORREY CANYON spill. Three years after the spill, algal species such as *Fucus* had recolonized the effectively sterilized shoreline to a considerable extent. Seven years after that observation, however, apparently after herbivore populations had recovered, the *Fucus* cover was substantially reduced.

In summary, these studies indicate that while a functioning intertidal community similar in occurrences of species may be fairly quickly reestablished in a disturbed area, closer approximation to the preexisting structure appears to take a relatively much longer time to establish itself. The literature suggests an average time to equilibrium, or a stable number of species, to be about one to two years. In contrast, the time required for recovery to a comparably structured intertidal community varied from three to five years, with longer time periods also noted. Based upon the available information, therefore, recovery to reestablishment of a community composed largely of the same dominant species present before the spill and treatment, would be expected to become established three to five years after the beach becomes resorted and stabilized. For a newly cleaned and physically disturbed pebble/cobble/boulder beach requiring about two years to restabilize, the total elapsed time to recovery would be about five to seven years.

<u>Siltation of Lower Intertidal/Subtidal Zones</u>. Siltation per se cannot occur at the two candidate sites, because there is apparently little silt in the sediments to be washed. However, it is possible that some of the sand and granule material may be transported to the lower zones after being suspended by waves following washing. The volumes of sand involved are so small and probable dispersion so great that serious mortality of epifauna

or flora as a result of sediment suspension caused by the gravel washing process is not envisioned. The possibility of complete burial of lower intertidal/subtidal zones by gravel/sand because of gravel washing also seems remote for the same reasons. Sand is already present and in motion to some extent in these areas. An intertidal sand bar was present near low tide in Sleepy Bay on 26 May 1990 (see map in Figure I-13). Therefore, the fauna are obviously adjusted to some fluctuations in suspended and bedload sand volumes at Sleepy Bay, the candidate site with the finest sediments.

Siltation effects and burial could be a problem if sediments are washed on beaches that contain significant proportions of silt (>±10%) and fine sand (>±30%). Therefore, site-specific tests should be carried out at low energy areas that contain such sediments before washing is initiated. During the summer of 1989, use of high-pressure water washing to treat cobble beaches mobilized finer-grained sediments at some sites and washed them into the lower intertidal zone. Increases of 10 to 25 percent in the amount of fine-grained sediments in the lower intertidal zone after shoreline treatment were not uncommon, according to Exxon scientists.

As discussed above, the amount of weathered crude oil entering lower intertidal sediments associated with fine sediments from the upper shore probably is not sufficient to adversely affect the resident biota through toxicity. However, it should be pointed out that excavation of sediments containing significant quantities of silt may produce a more concentrated distribution of oiled sediments in the lower intertidal and shallow subtidal zones than a storm would, because of the possible lack of waves to disperse the sediments at the time of washing.

A majority of the plants and animals living in the lower intertidal and shallow subtidal zones of cobble/boulder shores in Prince William Sound are adapted to live in or on coarse-grained substrates and are possibly not well adapted to tolerate high suspended sediment loads over an extended period of time, although plumes of suspended sediment extending out from the numerous glacial streams are common phenomena in Prince William Sound in the summer months. Thriving communities of lower intertidal and shallow subtidal organisms exist near these stream mouths. Also, the stream that bisects the Sleepy Bay candidate beach must carry in suspended sediments during periods of high runoff. Introduction of large amounts of fine-grained sediments into the lower intertidal and shallow subtidal habitats may render the substrate less suitable for habitation by the normal resident biota. This effect would be greatest on larvae settling from the plankton. However, established biota could also be affected.

Under a worst-case scenario involving the washing of silt-rich sediments, damage to the resident biota might result from outright burial by fine-grained sediments. Attached plants and animals possibly would be smothered. Motile animals, such as snails, amphipods, and starfish might also be buried, but most of them could dig themselves out of the sediment and survive. However, hard substrates on which many of the herbivores graze might be covered, diminishing the grazers' food supply. Most infauna can migrate vertically through several centimeters of freshly deposited sediment and so probably could avoid smothering if the amount of sediment deposited is not too great.

The burial and sediment texture changes associated with sediment mobilization from the upper shore and deposition in the lower intertidal and shallow subtidal zones, should it occur, probably would be of relatively short duration. A storm would be expected to remobilize the fine-grained sediments and move them offshore or back into the upper intertidal zone.

Disturbance by Vessels, Equipment, and Foot Traffic

The intertidal biota of exposed rocky shores are well adapted to the pounding of surf and the grinding of gravel by wave action. However, animals such as mussels, snails, and barnacles are readily crushed by foot traffic and movement of heavy equipment in the intertidal zone. It is reasonable to assume that in areas where the earth-moving equipment and rock washer moved, all plants and animals on the upper surfaces of rocks and most infauna in finer-grained sediments would be destroyed. This zone of destruction is likely to be larger than the area actually excavated.

The heavy equipment probably will have to be moved off the beach upon the approach of each high tide, requiring movement of large barges or landing craft up to the beach. Such large boat activities in the intertidal zone each day probably will disturb the abundant growths of large kelps and other macroalgae occupying the lower shore. However, these kelps grow rapidly, and any damage probably will be repaired within a few months.

Removal of Processed Fines

The rock washer probably will be able to clean sediments with a grain size larger than coarse sand and finer than 24 inches. Smaller-grained sediments and flocs probably will be passed to the oil/water/sediment separator device for further treatment. It may be difficult to remove sufficient oil from the finer-grained sediments to allow them to be returned to the shore. This oily sediment would be removed with the recovered oil and disposed of elsewhere.

The biological consequences of this change in the textural distribution of substrates in the middle and upper shore are uncertain. The upper and middle shores of exposed cobble/boulder beaches are not very productive biologically because of the instability of the substrate. However, in the middle intertidal zone, infauna often live in the finer-grained sediments below the surface layer of boulders and cobbles. With the destruction of their habitat, these animals would no longer be present on treated shorelines until natural geologic processes reestablish the sediment to near its original makeup.

Impact on Anadromous Fish Streams

Anadromous fish streams will benefit from the physical removal of oil if the oil threatens to leach into streambeds and affect the growth or survival of developing fish, or solidifies into asphalt pavement, thereby decreasing the amount of available spawning habitat. Even if oil is not located immediately adjacent to an anadromous stream it may still affect fish if flows occasionally cause the stream to meander laterally along the beach into an oiled area, or the movement of water, either on the surface or subsurface of the beach, causes oil to migrate into the stream. The effects of oil on these habitats and the species that use them are not well understood; therefore, a conservative approach appears to be warranted in removing the oil in the shortest time possible. However, excavation/rock washing has the potential to cause additional impacts if not carefully controlled. Potential impacts can include the removal of suitable spawning substrates, increased erosion, sedimentation of stream gravels, stream rechannelization, and altered hydrological patterns (stream flows, upwelling, and subsurface flows), all of which can render a formerly productive stream unsuitable for future use by salmon.

Air and Noise Pollution

Air and noise pollution will be an unavoidable component of any excavation/rock washing operation, although the extent to which such impacts are likely to cause environmental damages cannot be accurately predicted. It is primarily expected, however, from the use of heavy excavating equipment to remove contaminated sediments. In addition, a rock washer and associated support vessels or equipment (generators, heaters, boilers, etc.) could incrementally increase the amount of pollutants that are discharged into the air at a given site.

Conversely, noise pollution has a much greater potential to affect fish and wildlife species and human activities that occur in an area. The noise associated with an excavation/rock washing operation is likely to have little benefit, but could cause displacement of animals that normally use an area for important life functions such as birthing, foraging, nesting, or staging. This will result in a short term loss of habitat that should not be significant during non-sensitive time periods, and that support vessels and/or aircraft will access the area in a direct manner from offshore. However, human use will be affected in that recreationists and subsistence users will likely avoid an area where excavation/rock washing operations are occurring, although for reasons only partially attributable to noise generation.

Residual Oil

The performance objective for the rock washer is that, after treatment, the substrate should contain less than 800 ppm total oil. Assuming that this objective can be met with current technology, the upper shore will still contain some subsurface oil. A key question in evaluating the benefits of rock washing is whether this residual oil could produce any adverse biological effects.

As discussed above in Section II.A.5, petroleum adsorbed to sediments is less bioavailable and less toxic than oil in solution or dispersion in the water column. Several studies, discussed above, have documented that marine animals can tolerate and even recruit to sediments containing more than 800 ppm oil, particularly if the oil is weathered. In place, the subsurface oil is out of contact with any plants and animals and so does not pose a potential hazard to marine biota. However, the oiled sediments may be mobilized by storms and deposited in the lower intertidal and shallow subtidal zones, though this has not been documented to date in Prince William Sound. The concentrations of petroleum hydrocarbons in these mobilized and redeposited sediments would be lower than in the sediments before mobilization due to dilution. Even at current levels of oil contamination of subsurface sediments, the concentrations of PAHs are generally below those known to cause adverse impacts, as discussed in Section II.A.2 and 5. Therefore, it is unlikely that these residual hydrocarbons would have any adverse effects on the biota of the lower shore. Furthermore, it is probable that most of the remaining subsurface oil will have weathered to the point where it no longer produces sheens.

II.B.3. WATER QUALITY

Increased Sediment Loading

Questions concerning sediment loading in the water column and rates of sedimentation are complex, and difficult to accurately determine without extensive research and careful consideration of a number of parameters that are probably beyond the scope of this investigation and report.

The beaches at the two candidate sites are composed of coarse sediments that will readily settle out of suspension. The water would clear up within a matter of a few hours, at most, of the end of operations at these two sites.

Secondary Spills of Fuel or Process Water

During the summer of 1989, there were 147 fuel spills throughout the spill site related to the cleanup activities. The average number of spills per month was 24.5. There were 949 vessels per month involved in cleanup in 1989, or an average of one spill per month per 39 vessels. We assume that 23 vessels are required to support one rock washer unit, if the unit has a throughput capacity of 100 yd^3/hour. If four or five of these units are required to complete the necessary rock washing in the Sound within two months, we project that eight to ten fuel spills would occur in the Sound related to the rock washer operation. There are no data on the size of oil releases in the 1989 incidents.

II.B.4 IMPACTS ON HUMAN INTERESTS RELATED TO EXCAVATION/ROCK WASHING

The benefits of excavation/rock washing on fishing will be to remove the oil from the beach matrix so that it cannot foul gear, taint fisheries products, or cause health concerns either to developing fish or human consumers. In this respect, excavation/rock washing is one of the most predictable means of assuring that oil is completely removed from the environment rather than remaining as a chronic threat. The expected impacts of the excavation and cleaning process include potential for increased sheening or resuspension of oiled particulates, which then can affect fisheries openings or harvests if the oil is not adequately contained. Silt plumes from the operation may also have an adverse effect on fishing. The disturbance by vessels, equipment, and personnel associated with the rock washing process and the increased sheening and siltation would displace recreational and subsistence uses in the area for the duration of the operation. Recreational uses and some subsistence uses could resume soon after the treatment process. Other subsistence activities would follow the natural recolonization of intertidal biota.

II.B.5 IMPACTS ON CULTURAL RESOURCES RELATED TO EXCAVATION/ROCK WASHING

The rock washing technique would obliterate the interpretive value of archaeological sites and probably damage or destroy artifacts. However, this negative impact can be avoided by an archaeological inventory of the rock washing areas prior to treatment and avoiding the specific archaeological sites within those areas during the rock washing operation.

II.B.6. IMPACTS OF WASTE HANDLING AND DISPOSAL FOR EXCAVATION ROCK WASHING

Introduction

This section describes potential environmental contamination pathways associated with handling and disposal of wastes that would be generated by excavation rock washing if it were to be conducted on an actual beach. These pathways may impact offshore or coastal waters, onshore surface waters, soils, or groundwater, and are associated with handling and disposal of the three primary waste streams: removed oil, oil and sediment bearing wastewater, and oil-containing sludges. Though human health and environmental impacts cannot be quantified, types of impacts that could develop via these pathways can be identified.

The risk of environmental contact associated with landfilling of the large volumes of oil-containing sludges generated by this technology is the primary potential environmental concern related to handling and disposal of excavation rock washing wastes. However, the ever-present risk of accidental spills, leaks, and other discharges through the handling and disposal of large volumes of any waste may also present potential environmental impacts.

Removed Oil

The primary pathway for potential environmental impacts from oil removed from beaches by excavation rock washing is accidental spillage. If spilled at the beach site, the oil could contribute to sheening or could recontaminate beach sediments, with impacts ranging from those of the oil contact to those due to additional cleanup measures required. If spilled during marine transport, sheen or a plume of floating oil could be produced. If spilled onshore or at the treatment, storage, or disposal facility, soils, groundwater, or surface water could be contaminated resulting in potential impacts on vegetation or human use, including potential human use of affected surface or groundwater.

Once successfully reclaimed or recovered, no further potential impacts would be anticipated, with the possible exception of impact on receiving waters of discharges related to such treatment. With properly controlled discharges, significant impacts should not occur. On the other hand, if removed oils are landfilled, using the best available landfilling technology, the potential still exists for soil, groundwater, and surface water contamination due to seepage from the landfill if containment fails.

Wastewater (contains oil and suspended sediment)

Potential pathways for environmental impacts from oil and suspended sediment-bearing wastewaters are similar to those for the removed oil, but the potential impacts differ. Although oil content of wastewater is lower than that of the waste oil, volumes are much greater, and the suspended sediments themselves can create impacts. If spilled on the beach site, the primary impact would be creation of a suspended sediment plume and potential deposition of particulate matter, along with associated oil, on intertidal and subtidal zones. Spills occurring during marine transport could create sediment/oil plumes with a low potential for actual environmental impact. Spills occurring onshore or at the treatment, storage, or disposal facility could result in soil or groundwater contamination, or fouling of surface waters.

Once treated, little further potential for impacts exists. The wastewater should present no unusual treatment problems for a well-designed water treatment system, and properly controlled associated discharges should not create any significant impacts. However, any oil recovered from the wastewater presents the same impact potential as the oil initially removed from beach sediments.

Waste Sludge (contains oil)

The waste sludges have a higher potential for environmental impact than the removed oil or waste water due to the concentrated nature of the material, relatively large volumes, and greater handling/treating requirements. Any accidental spill at the beach site has the potential to create heavy sediment plumes with associated significant deposition of particulate matter and associated oil on intertidal and subtidal zones. If a spill occurred during marine transport, significant oil/sediment plumes could result, although, as for the wastewater, the actual impact of such a plume in open water may be low. Potential impacts of onshore spills would be similar in character to those related to spills of the wastewater, but of greater potential magnitude in surface waters (due to concentrated nature of waste) and smaller potential magnitude in low to medium permeability soil due to the high solids content. Likely land transport of this material prior to disposal poses additional opportunities for accidental spills onshore.

Since this material will likely require treatment prior to disposal, additional waste streams will likely be generated. Dewatering of the material will create additional oil and sediment containing wastewater with potential impacts similar to those for the wastewater initially generated by the process. Any additional oil removal step will result in creation of an oily waste with potential impacts similar to those for the oil initially removed from beach sediment, though the magnitude of impact has lower potential due to probable lower volumes.

Finally, it is anticipated that the bulk of the oiled sludge produced by the excavation rock washing process would be disposed by landfilling. Possible seepage from a landfill containing this oily sludge has the potential to impact nearby soil, groundwater, and surface water with a variety of potential effects on vegetation and human use, including use of the surface water or groundwater.

An additional environmental consideration is the use of landfill capacity. As current landfill capacity is taken up, new landfill capacity is created, generating new potential sources of soil, groundwater and surface water contamination.

III. SUMMARY

A technical committee was formed in May 1990 to investigate the environmental tradeoffs associated with excavating and washing oiled sediments remaining deeply buried along some sections of the Alaskan shoreline affected by the EXXON VALDEZ oil spill. The committee, comprised of Exxon, NOAA, and State of Alaska scientists, was charged with conducting a Net Environmental Benefit Analysis (NEBA) related to the advisability of excavation/washing in comparison to natural cleansing and application of treatment protocols approved by the Federal On-Scene Coordinator for use in 1990.

The NEBA committee selected two shoreline segments, Sleepy Bay on the north coast of Latouche Island and Point Helen on the southeast coast of Knight Island, as candidate locations to pursue field studies associated with the analysis.

The committee collectively outlined the range of environmental issues important to the analysis. The benefits and negative impacts of excavation and rock washing were researched in some detail, using both site-specific data from the candidate beaches and published studies on the fate and effect of oil and the impacts of physical disturbances to the shoreline. The results of these analyses can be summarized as follows.

How much oil is buried at what depths along the most severely affected sections of the coast? What are the prospects for removal by natural processes?

During the March/April 1990 shoreline assessment, 212 of the 1,134 shoreline segments surveyed had subsurface oil deposits thicker than 15cm and buried deeper than 15cm. Of 5,071 pits dug in the assessment, only 279 contained oil that had penetrated greater than 30cm, with 25 that had oil penetration greater than 60cm. The deepest occurrences of subsurface oil were found on coarse-grained beaches (boulder/cobble/pebble) along exposed shorelines.

NOAA studies at 18 stations in Prince William Sound over the six-month period from September 1989 to March 1990 indicated that, on the exposed shoreline segments most likely to be amenable to excavation, sediment reworking resulted in the removal of oil from the top 10-20 cm of surface sediments. Data taken from Sleepy Bay and Point Helen reflected a 90% oil reduction in the top 30cm, but no discernible trend in oil reduction was evident below this depth. Exxon's winter monitoring study at 18 sites in Prince William Sound concluded that the concentration of total petroleum hydrocarbons (TPH) in subsurface sediments (generally >10cm) declined by 88% over the same period.

Using various survey methodologies, the estimated amount of oil remaining in subsurface sediments at Sleepy Bay, one of the candidate sites for excavation, was:

Gallons/meter of Shoreline	No. of Samples	Source
9.9	43	NOAA Oct-Feb Survey
5.0	13	Exxon Mar Survey
6.1	11	Exxon Jun Survey

Perhaps of more importance than the quantity of subsurface oil is the rate of natural removal which may be expected over the next few years. There have been numerous studies of previous spills in which the persistence of stranded oil has been surveyed. However, there are many factors specific to Prince William Sound that might alter the extrapolation of oil persistence curves from other spills to the EXXON VALDEZ. The factors that may speed natural removal processes in Alaska are:

1) Flocculation, the process by which fine-grained sediments adhere to the subsurface oil and make the oil less sticky and more biologically available.
2) Enhanced biodegradation, both naturally occurring and stimulated by the addition of nutrients.
3) Removal of surface oil during the 1989 treatment, which limited the formation of asphalt pavements.

The factors that may slow natural removal rates in Alaska are:

1) The relatively low wave energies relative to the grain size of shoreline sediments.
2) The initial high degree of contamination of some shoreline segments.

Oil removal is expected to be most pronounced early in the recovery period; this expectation is clearly being borne out by data gathered thus far through the various monitoring programs. NOAA estimates that virtually all of the oil buried in gravel shorelines should be removed, given no further treatment, in the periods of time indicated below:

Sheltered parts of Prince William Sound	10+ years
Sheltered outer Kenai	3-5 years
Exposed parts of Prince William Sound	2-4 years
Exposed outer Kenai	1-2 years

At what rate is subsurface oil being weathered?

Samples of subsurface oil were collected by NOAA from the two NEBA study areas and analyzed by gas chromatograph/mass spectrometry (GC/MS) to determine both chemical composition and weathering trends. Most of the subsurface oil in these samples was moderately to heavily weathered and altered significantly in both physical and chemical properties from the original state. Physically, the oil had become more viscous, less sticky, and would not readily sheen. Chemically, the oil had lost many of the light- to moderate-weight polynuclear aromatic hydrocarbons (PAHs), components which are of greatest toxicity to aquatic organisms. For example, the naphthalenes and phenanthrenes had been reduced by more than a factor of 100 when compared with fresh EXXON VALDEZ oil. Analysis of the saturate fraction of this subsurface oil showed strong evidence that microbial degradation had occurred. Overall, the subsurface oil had decreased in both concentration and toxicity and showed evidence of significant microbial degradation.

Will subsurface oil become exposed by wave action?

It is likely that there will be further removal of subsurface oil by wave action and shoreline erosion; however, these processes are most likely to occur in the winter when sediment abrasion and thus natural cleaning of the surface is at a maximum. There is no general agreement as to the rate at which the oiled subsurface sediments would be exposed by these processes. Larger cobbles exposed during severe winter storms may remain with some surface oil contamination; the smaller sand, granules, and pebbles would rapidly be cleaned.

To what degree is water within the beach being contaminated by subsurface oil and thus exposing organisms to toxic hydrocarbon compounds?

Water within the beach ("pore water") comes in contact with subsurface oil as it moves through the interstitial space among oiled sediment particles. The prospect that pore water may become contaminated was investigated by the committee.

An Exxon mathematical model was used to predict the concentration of PAHs in pore water, which contacts subsurface oil deposits, and the rate at which tidal flushing would leach the water-soluble fractions of the oil. Using data on concentrations of PAHs in sediments collected in March 1990, Exxon predicted that concentrations of total PAHs in pore water would be generally below 20 ppb. These model results were confirmed by Exxon analyses of field samples of pore water collected in March at eleven sites in Prince William Sound. Only one sample contained PAH levels greater than 10 ppb and only three reflected levels greater than 1 ppb. Two of the three highest values were for samples collected from Sleepy Bay. All observed levels of pore water contamination are below concentrations known to be acutely toxic to marine organisms.

An additional twelve water samples were collected by NOAA at the sediment/water interface during falling tides from Sleepy Bay in May 1990. Total PAH concentrations in these samples were all less than 1 ppb. The rapid dilution of the pore water as it mixes with clean interstitial and shallow subtidal waters should result in PAH concentrations that pose little or no toxic hazard to marine organisms on the lower shore and in subtidal waters.

To what extent can sheens be produced by subsurface oil? If such sheens are produced, what adverse effects can be anticipated?

The committee investigated the frequency, magnitude, composition, and effect of oil sheens likely to originate from subsurface oil. Observations by ADEC and Exxon show a declining trend in both sheen frequency and volume throughout Prince William Sound. The downward trend in Exxon data is more pronounced, possibly due to the elimination of non-Prudhoe Bay crude sheens from the data set. On a weekly basis, the volume of sheens from EXXON VALDEZ oil is now well below the present level of petroleum hydrocarbons introduced to surface water of Prince William Sound from other sources (vessel traffic, combustion, natural organics, etc.).

NOAA chemical analysis indicates that the natural weathering of subsurface oil has rendered the oil both less toxic and much less capable of producing sheen. As long as the oil remains physically stable in the shoreline, sheen frequency should be minimal. A temporary increase in sheening would likely be produced either by excavation or by intense winter storms. Sheens produced by excavation could be expected to be controlled in light to moderate weather conditions. While control of storm-induced sheens is unlikely, the range and effect of such sheens would be limited by natural dispersion during the storm event. NOAA studies in 1989 indicated that only in unusual weather conditions would convergence zones have the potential to concentrate free-floating oil into popweed windrows to the extent fishing gear might be contaminated. The threat of exposure to birds feeding at these windrows is likely to be small as exemplified by the lack of reports of oiled marine birds this year.

In summary, subsurface oil is relatively stabilized and insulated from exposure to vulnerable resources through sheening. Adverse effects on birds, fish, wildlife habitat, or human uses are not expected to originate from sheens produced by subsurface oil.

What is the near- and long-term fate of subsurface oil?

Subsurface oil that persists in the shoreline will eventually be degraded by indigenous bacteria. However, prior to the complete microbial breakdown of oil buried deep within the shoreline, other processes may be important in controlling the intermediate distribution and fate of the oil.

Flocculation is one process that has been proposed by Exxon as being an important mechanism for enhanced natural removal and dispersal of subsurface oil. The flocculation process has been demonstrated in laboratory studies, and in the field some subsurface oil appears to have taken on a more weathered, smeary texture indicative of floc formation. However, the rate of flocculation and its relative importance, particularly in the removal of heavily oiled subsurface sediments, is not known.

There is some concern that the fine-grained fraction of oiled subsurface sediments may be exposed by erosion and deposited in the nearshore subtidal environment. Since there are few data available to evaluate this potential risk, a diving survey was conducted by ADEC in June 1990 to collect subtidal sediments along a transect out to 100m off both Point Helen and Sleepy Bay. Total PAHs in these subtidal sediments were 0.1-9 ppb off Point Helen and 1-130 ppb off Sleepy Bay. This limited sampling would indicate that deposition of oiled sediments, derived from either subsurface or surface sources, may be minimal.

What are the potential effects of rock excavation on the shoreline substrate?

There is great variability in sediment grain size and shoreline exposure in Prince William Sound, and therefore the effects of excavation and rock washing on the structure of the intertidal zone would vary considerably from location to location. Shoreline segments that contain high concentrations of subsurface oil are also generally exposed to moderate or high wave energy. The upper part of the intertidal zone in these locations is affected by the erosion and deposition of high-tide berms, as shown through analysis of data obtained during the Exxon, NOAA, and ADEC winter monitoring programs. The middle and lower intertidal zones are less affected, particularly if a cobble/boulder armor is present, such as on Point Helen, one of the candidate sites for rock washing.

Using data from the winter monitoring programs and case histories from other studies, it is predicted that, if the material were removed from the upper one-third of the beach and replaced in approximately the same position from which it was taken, the upper berms at Point Helen would reestablish themselves after two or three medium-sized storms. The re-armoring process in the middle and lower part of the shoreline would require a few more storms, possibly covering a period of one to three years.

For Sleepy Bay, which is more sheltered but also is composed of finer-grained sediments, the upper surface sediment resorting should occur in less than one year. However, because the middle portion of the beach changes more slowly, complete reestablishment of the original profile and sediment resorting may require up to three years, though it may take place sooner. Recolonization by intertidal organisms would be delayed during these periods of sediment restabilization.

Loss of fines is not expected to be of concern, mainly because of the relatively small percent of sand-sized materials in the upper zone on most beaches in Prince William Sound. Because the sand exists primarily as a matrix to the already fixed framework of the beach, there should be no impact of its potential loss on beach morphology or erosion. There were concerns that the loss of sand might change the hydraulics of the beach, but the magnitude or consequences of this impact are unknown.

What are the effects of subsurface oil and excavation on the recolonization by intertidal communities?

In the areas where excavation is being considered, the intertidal zone can be divided into three subzones on the basis of the distribution of biological communities. The upper intertidal zone, generally the location of the highest concentrations of subsurface oil, is normally not inhabited by a very rich biological community because of relative dryness, sediment mobility, and lack of food.

The middle intertidal zone contains a more rich and diverse biological community. Observations made this spring and summer indicate that these communities are recovering from the oil and treatment activities. Oil concentrations of the surface sediments in this zone are generally low and, as discussed earlier, toxic impacts from the discharge of groundwater through oil-contaminated sediments across this zone are not expected to be significant. Because most of the plants and animals in this zone live in the top 10-15cm, there is no pathway of exposure to subsurface oil other than oil exposed during erosional events. To the extent the oil remains buried, it poses no serious risk to intertidal communities in this zone.

The lowermost intertidal zone has the greatest biomass and species diversity. In most cases, this zone is showing evidence of recovery and only very low concentrations of oil occur in the surface sediments. The lower intertidal zone would be least impacted by the residual subsurface oil.

With the information available on the likely zone of disturbance by excavation and rock washing, it is assumed that few organisms in the upper and middle intertidal zones would survive. There may also be impacts to these zones resulting from the traffic of equipment and people on the shoreline. It is likely that these surface impacts would extend to some degree into the relatively rich lower intertidal communities.

NOAA estimates that three to five years would be reuired for biological recolonization after the shoreline stabilizes. Adding one to three years for stabilization, the total recovery period after excavation is estimated to be four to eight years. If excavation does not occur, NOAA estimates recolonization will occur in two to five years at Point Helen and three to six years post-spill at Sleepy Bay.

What would be the secondary impacts from excavation and rock washing?

Increased sediment loading in the water column would be unavoidable during excavation and replacement. Coarse material would settle out readily, although it is likely that oiled silts and clays would settle slowly and might be transported along the beach by tidal currents.

Secondary spills of fuel and process water were estimated using the spill rate of cleanup activities in 1989. Assuming that four or five rock washing units were active for a two-month period, an estimated eight to ten fuel spills would occur.

Excavation and rock washing would disturb wildlife with work activity and noise for six weeks or longer in each work area. Sources of disturbance include personnel, vessels, equipment, and aircraft. Traffic in the intertidal area may further affect the recolonization process.

Exxon estimates that excavation and rock washing at Sleepy Bay would generate 12,000 gallons of skimmed oil/water mixture, 168,000 gallons of wastewater, 500,000 gallons of sludge waste, and 300,000 pounds of air emissions. Fuel consumption is estimated at 400,000 gallons.

What would be the impacts on human use of the shoreline and nearshore waters should subsurface oil not be removed? What would be the related effects of excavation/rock washing?

The State of Alaska indicates that any of the shoreline segments which are candidates for excavation and rock washing are in areas of moderate to high human use. These areas are used throughout the year for recreation, subsistence and commercial fishing, and are of significant cultural importance. Subsurface oil has some capacity to pose an episodic threat to these uses of the shoreline and nearshore waters.

Although sheens are declining in Prince William Sound, and the relationship between sheens and subsurface oil is tenuous in non-storm conditions, commercial fishing in the immediate vicinity could be affected in the event sheening does occur. Episodic releases of subsurface oil may contaminate subsistence fisheries, interfere with shore-based commercial and recreational fishing, and impair use of the shoreline for recreation. Large amounts of subsurface oil can impact recreation and subsistence users who dig fire pits in the upper intertidal zone.

Excavation/rock washing is one of the most predictable means of assuring that oil is removed from the environment rather than remaining as a potential source for episodic exposure during winter storms. The expected impacts of the excavation and cleaning process include potential for temporary sheening or resuspension of oiled particulates, creation of silt plumes, and disturbance by equipment and personnel. These impacts can be mitigated by proper containment of sheens and timing of operations to minimize impacts to resources and users. Recreational and some subsistence uses could resume soon after the treatment process. Other subsistence activities would be delayed four to eight years until natural recolonization of intertidal biota occurred.

REFERENCES

Advanced Technology, Inc. and Continental Shelf Associates, Inc. 1990. Exxon Valdex oil spill winter study site program. Seattle: Hazardous Materials Response Branch, NOAA. 92 pp. 5 appendices.

Aminot, A. 1981. Anomalies du systeme hydrobiologique cotier apres l'echouage de l'Aoco Cadiz. Considerations quantitatives et qualitatives sur la biodegradation in situes hydrocarbures. In: Amoco Cadiz. Consequences d'une Pollution Accidentalle par les Hydrocarbures. Fates and Effects of the Oil Spill. Paris: NOAA-CNEXO Joint Scientific Commission. pp. 223-242.

Anderson, J.W., R.G. Riley, and R.M. Bean. 1978. Recruitment of benthic animals as a function of petroleum hydrocarbon concentrations in the sediment. J. Fish. Res. Bd. Can.35: 776-790.

Atlas, R.M., P.D. Boehm, and J.A. Calder. 1981. Chemical and biological weathering of oil from the Amoco Cadiz spillage within the littoral zone. Estuar. Cstl. Shelf Sci.12: 589-608.

Atlas, R.M. and A. Bronner. 1981. Microbial hydrocarbon degradation within intertidal zones impacted by the Amoco Cadiz oil spillage. In: Amoco Cadiz: Fates and Effects of the Oil Spill. Paris: NOAA-CNEXO Joint Scientific Commission. pp. 251-256.

Baker, J.M., R.B. Clark, P.F. Kingston, and R.H. Jenkins. 1990. Natural recovery of cold water marine environments after an oil spill. In: 13th Annual Arctic and Marine Oilspill Program. Preprint. 111pp.

Bakke, T., J.A. Berge, K. Naes, F. Oreld, L.O. Reiersen, and K. Bryne. 1989. Long term recolonization and chemical change in sediments contaminated with oil-based drill cuttings. In: Drilling Wastes. F.R. Englehardt, J.P. Ray, and A.H. Gillam, eds. London: Elsevier Applied Science. pp. 521-544.

Barth, T. 1984. Weathering of crude oil in natural marine environments: the concentration of polar degradation products in water under oil as measured in several field studies. Chemosphere 13: 67-86.

Blaylock, W.M. and J.P. Houghton. 1989. Infaunal recovery at Ediz Hook following the Arco Anchorage oil spill. Proceedings of the 1989 Oil Spill Conference, San Antonio, Texas, February 13-16, 1989. pp. 421-426.

Bodin, P. and D. Boucher. 1982. Evolution a moyen-terme du meiobenthos et du microphytobenthos sur quelques plages touchees par la Maree Noire de L'Amoco Cadiz. In: Ecological Study of the Amoco Cadiz Oil Spill. Paris: NOAA-CNEXO Joint Scientific Commission. pp. 245-268.

Boehm, P.D., M.S. Steinhauer, D.R. Green, B. Fowler, B. Humphrey, D.L. Fiest, and W.J. Cretney. 1987. Comparative fate of chemically dispersed and beached crude oil in subtidal sediments of the Arctic nearshore. Arctic 40: 133-148.

Bragg, J.R., S.H. Yang, and J. C. Roffall. 1990. Experimental studies of natural cleansing of oil residue from rocks in Prince William Sound by wave action. Houston: Exxon Prod. Res. Co. 12pp.

Butler, J.N., B.F. Morris, and T.D. Sleeter. 1976. The fate of petroleum in the open ocean. In: Sources, Effects, and Sinks of Hydrocarbons in the Aquatic Environment. Washgton, D.C.: American Institute of Biological Sciences. pp. 287-297.

Cairns, J. Jr., K.L. Dickson, and E.E. Herricks. 1975. Recovery and Restoration of Damaged Ecosystems. Proceedings of the International Symposium on the Recovery of Damaged Ecosystems, Blacksburg, Virginia, March 23-25, 1975. pp. 187-207.

Caldwell, N.E. 1981. Relationship between tracers and background beach material. J. Sediment. Petrol. 51: 1163-1168.

Capuzzo, J.M. 1987. Biological effects of petroleum hydrocarbons: assessments from experimental results. In: Long-Term Environmental Effects of Offshore Oil and Gas Development. D.F. Boesch and N.N. Rabalais, eds. London: Elsevier Applied Science. pp. 343-410

Carefoot, T. 1977. Pacific Seashores. Seattle: University of Washington Press. 208 pp.

Carls, M.G. 1987. Effects of dietary and water-borne oil exposure on larval Pacific herring (*Clupea harengus pallasi*). Mar.Environ. Res. 22: 253-270.

Carr, A.P. 1971. Size grading along a pebble beach, Chesil Beach, England. J. Sediment. Petrol. 39: 279-311.

Carter, R.W.G. 1988. Coastal Environments. New York: Academic Press. 617 pp.

Chan, G.L. 1975. A Study of the Effects of the San Francisco Oil Spill on Marine Life Part II: Recruitment. Proceedings of the 1975 Oil Spill Conference, San Francisco, California, March 25-27, 1975. pp. 457-461.

Chan, G.L. 1977. The five-year recruitment of marine life after the 1971 San Francisco oil spill. Proceedings of the 1977 Oil Spill Conference, New Orleans, Louisiana, March 8-10, 1977. pp. 543-545.

Clark, R.C. Jr., et.al. 1978. Observations of a cold-water intertidal community after 5 years of a low-level, persistent oil spill from the GENERAL M.C. MEIGS. J. Fish. Res. Board Can. 35: 754-765.

Clark, R.C., B.G. Patten, and E.E. DeNike. 1975. Long-term chemical and biological effects of a persistent oil spill following the grounding of the GENERAL M.C. MEIGS. Proceedings of the 1975 Oil Spill Conference, San Francisco, California, March 25-27, 1975. pp. 1479-1487.

Colwell, R.R., A.L. Mills, J.D. Walker, P. Garcia-Tello, and P.V. Campos. 1978. Microbial ecology studies of the METULA spill in the Strait of Magellan. J. Fish. Res. Board Can. 35: 573-580.

Dauvin, J.C. and F. Gentil. 1990. Conditions of the peracarid populations of subtidal communities in Northern Brittany ten years after the AMOCO CADIZ oil spill. Mar. Poll. Bull. 21(3):123-130.

Davies, J.M., J. Addy, L. Blackman, J. Blanchard, J. Ferbrache, D. Moore, H. Somerville, A. Whitehead, and T. Wilkinson. 1983. Environmental effects of oil based mud cuttings. London: Joint Working Group of UKOOA Clean Seas and Environmental Committee/Dept. of Energy/Dept. of Agriculture and Fisheries for Scotland/Ministry of Agriculture, Fisheries and Food. 24 pp.

Ganning, B., D.J. Reish, and D. Straughan. 1984. Recovery and restoration of rocky shores, sandy beaches, tidal flats, and shallow subtidal bottoms impacted by oil spills. In: Restoration of Habitats Impacted by Oil Spills. J. Cairns, Jr., and A.L. Buikema, Jr., eds. Stoneham, Massachusetts: Butterworth Publishers. pp. 7-36.

Glemarec, M. and E. Husenot. 1982. Responses des peuplements subtidaux a la perturbation cree par l'Amoco Cadiz dans la Abers Benoit et Wrac'h. In: Ecological Study of the Amoco Cadiz Oil Spill. Paris: NOAA-CNEXO Joint Scientific Commission. pp. 191-204.

Green, D.R., C. Bawden, W.J. Cretney, and C.S. Wong. 1974. The Alert Bay Oil Spill: A One-Year Study of the Recovery of a Contaminated Bay. Victoria, British Columbia: Environment Canada. Pacific Region Pacific Marine Science Report 74-9. 43 pp.

Guillen, G.J. and D. Palafax. 1985. The effects of weathered crude oil from the M/T Alvenus spill on eggs and yolk-sac larvae of red drum (*Sciaenops ocellatus*). Gulf Res. Rep.8: 15-20.

Gundlach, E.R., S. Berne, L. d'Ozouville, and J.A. Topinka. 1981. Shoreline oil two years after Amoco Cadiz: new complications from Tanio. Proceedings of the 1981 Oil Spill Conference, Atlanta, Georgia, March 2-5, 1981. pp. 525-534.

Gundlach, E.R., D. Domeracki, and L.C. Thebeau. 1982. Persistence of METULA oil in the Strait of Magellan, six and one-half years after the incident. J. Oil and Petrochemical Pollution 1 (1): 37-38.

Gundlach, E.R., P.D. Boehm, M. Marchand, R.M. Atlas, D.M. Ward, and D.A. Wolfe. 1983. Fate of Amoco Cadiz oil. Science 221: 122-129.

Hardy, J.T. In press. Air above and water below: the aquatic surface layer. Natural History Magazine.

Hardy, J.T., E.A. Crecelius, L.D. Antrim, V.L. Broadhurst, C.W. Apts, J.M. Guritsen, and T.J. Fortman. 1987a. The sea-surface microlayer of Puget Sound: Part II. Concentrations of contaminants and relation to toxicity. Mar. Environ. Res. 23:251-272.

Hardy, J.T., S. Kiesser, L. Antrim, A. Stubin, R. Kocan, and J. Strand. 1987b. The sea-surface microlayer of Puget Sound: Part I. Toxic effects on fish eggs and larvae. Mar. Environ. Res. 23: 227-249.

Hattori, M. and T. Suzuki. 1978. Field experiment on beach gravel transport. Proceedings of the 16th Conference on Coastal Engineering, Hamburg, Germany. pp. 1688-1704.

Haven, S.B. 1971. Effects of land-level changes on intertidal invertebrates, with discussion of postearthquake ecological succession. In: The Great Alaska Earthquake of 1964. Washington, D.C.: National Academy of Sciences. pp. 82-126.

Houghton, J. and D. Erikson. 1989. Biological data from Herring Bay Omni-Barge test, July 1989. Valdez, Alaska: Dames & Moore. Obtained through Freedom of Information Act request (unpublished).

Humphrey, B., G. Sergy, and E.H. Owens. 1990. Stranded oil persistence in cold climates. Proceedings of the 1990 Arctic and Marine Oil Polluton Conference, Calgary, Canada, June 6-8, 1990.

Johansen, H.W. 1971. Effects of elevation changes on benthic algae in Prince William Sound. In: The Great Alaska Earthquake of 1964. Washington, D.C.: National Academy of Sciences. pp. 35-68.

Juday, G.P. and N.R. Foster. 1990. A preliminary look at effects of the Exxon Valdez oil spill on Green Island Research Natural area. Agroborealis 22 (1): 10-17.

Karickhoff, S.W. 1980. Sorption kinetics of hydrophobic pollutants in natural sediments. In: Contaminants and Sediments. R.A. Baker, ed. Ann Arbor, Michigan: Ann Arbor Science Publishers. pp. 193-205.

Kingston, P.F. 1987. Field effects of platform discharges on benthic macrofauna. Phil. Trans. R. Soc. Lond.316: 545-565.

Knap, A.H., K.A. Burns, R. Dawson, M. Ehrhardt, and K.H. Palmork. 1986. Dissolved/dispersed hydrocarbons, tarballs and surface microlayer: experiences from an IOC/UNEP workshop in Bermuda, December, 1984. Mar. Pollut. Bull. 17: 313-319.

Lee, R.F. 1981. Mixed function oxygenase (MFO) in marine invertebrate. Mar. Biol. Lett. 2: 87-105.

Lissner, A., G. Taghon, D. Diener, S. Schroeter, and J. Dixon. 1988. Review of recovery and recolonization of hard substrate communities of the outer continental shelf. Anchorage: U.S. Department of the Interior, Minerals Management Service. Final report MMS 88-0034. 127 pp.

Long, E.R. 1972. Studies of marine fouling and boring off Kodiak Island, Alaska. Marine Biology 14: 52-57.

Long and Morgan. 1990. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. Seattle: Pacific Office, NOAA. NOAA Tech. Memo. NOS OMA-52. 175pp. and appendices.

Mair, J.M., I. Matheson, and J.F. Appelbee. 1987. Offshore macrobenthic recovery in the Murchison Field following the termination of drill-cuttings discharges. Mar. Pollut. Bull. 18: 628-634.

Mancini, E.R., J. Lindstedt-Siva, and D. W. Chamberlain. 1989. Environmental impacts of the 1985 Arco Anchorage oil spill: 1988 conclusions. Proceedings of the 1989 Oil Spill Conference, February 13-16, 1989, San Antonio, Texas. pp. 459-462.

Marshall, M. and E. Gundlach. 1990. The Physical Persistence of Spilled Oil: An Analysis of Oil Spills Previous to EXXON VALDEZ. Seattle: Office of Oceanography and Marine Assessment NOAA. 242pp.

Mayo. D.W., D.S. Page, J. Cooley, E. Sorenson, F. Bradley, E.S. Gilfillan, and S.A. Hanson. 1978. Weathering characteristics of petroleum hydrocarbons deposited in fine clay marine sediments, Searsport, Maine. J. Fish. Res. Board Can. 35:552-562.

Miller, J.A. 1989. Physical and chemical recovery of intertidal and shallow subtidal sediments impacted by the Arco Anchorage oil spill, Ediz Hook, Washington. Proceedings of the 1989 Oil Spill Conference, February 13-16, 1989, San Antonio, Texas. pp. 487-491.

Nakatani, R.E., E.O. Salo, A.E. Nevissi, R.P. Whitman, B.P. Shyder, and S.P. Kaluzny. 1985. Effect of Prudhoe Bay crude oil on the homing of coho salmon in marine waters. Washington, D.C.: American Petroleum Institute. Publication No. 4411.

Neff, J.M. 1984. Bioavailability of sediment-bound contaminants to marine animals. Fes. Z. Anal. Chem. 139: 67-74.

Neff, J.M. 1988. Bioaccumulation and Biomagnification of Chemicals from Oil Well Drilling and Production Wastes in Marine Food Webs: A Review. Washington, D.C.: American Petroleum Institute. 67 pp.

Neff, J.M. 1990. Water Quality in Prince William Sound. Duxbury, Massachusetts: Battelle Duxbury Operations. 31 pp.

Neff, J.M. and J.W. Anderson. 1981. Responses of Marine Animals to Petroleum and Specific Petroleum Hydrocarbons. New York: Halsted Press. 177 pp.

Nelson-Smith, A. 1975. Recovery of some British rocky seashores from oil spills and cleanup operations. In: Recovery and Restoration of Damaged Ecosystems. J. Cairns, Jr., ed. Charlottesville, Virginia: University of Virginia Press. pp. 191-207.

Nester, R.T. and T.P. Poe. 1982. Effects of beach nourishment on the nearshore environment in Lake Huron at Lexington Harbor (Michigan). Fort Belvoir, Virginia: US. Army Corps of Engineers. Coastal Engineering Research Center Misc. Rpt. No. 82-13. 56 pp.

O'Clair, C.E. and S.T. Zimmerman. 1987. Biogeography and ecology of intertidal and shallow subtidal communities. In: The Gulf of Alaska Physical Environment and Biolgocial Resources. D.W. Hood and S.T. Zimmerman, eds. Anchorage: Alaska Office, NOAA. pp. 305-344.

Owens, E.H., J.R. Harper, W. Robson, and P.D. Boehm. 1987. Fate and persistence of crude oil stranded on a sheltered beach. Arctic 40:109-123.

Pearson, W.D., D.L. Woodruff, S.L. Kiesser, G.W. Fellingham, and R.A. Elston. 1985. Oil effects on spawning behavior and reproduction in Pacific herring (*Clupea harengus pallasi*). Washington, D.C.: American Petroleum Institute. Publication No. 4411.

Petrov, V.A. 1989. The differentiation of material on gravel beaches. Oceanology 29(2):208-212.

Rambeloarisoa, E., J.F. Rontani, G. Giusti, Z. Duvnjak, and J.C. Bertrand. 1984. Degradation of crude oil by a mixed population of bacteria isolated from sea-surface foams. Mar. Biol. 83: 69-81.

Rice, S.D., D.A. Moles, T.L. Taylor, and J.F. Karinen. 1979. Sensitivity of 30 Alaskan marine species to Cook Inlet crude oil and No. 2 fuel oil.

Proceedings of the 1979 Oil Spill Conference, Los Angeles, California, March 19-22, 1979. pp. 549-554.

Rice, S.D., D.A. Moles, J.F. Karinen, S. Korn, M.G. Carls, C.C. Brodersen, J.A. Gharrett, and M.M. Babcock. 1985. Effects of petroleum hydrocarbons on Alaskan aquatic organisms: a comprehensive review of all oil-effects research on Alaskan fish and invertebrates conducted by the Auke Bay Laboratory, 1970-1981. In: Final Reports of Principal Investigators. Outer Continental Shelf Environmental Assessment Program. Vol. 29. Anchorage: OCSEAP Office, NOAA. pp. 311-427

Riznyk, R.Z., J.T. Hardy, W. Pearson, and L. Jabs. 1987. Short-term effects of polynuclear aromatic hydrocarbons on sea-surface microlayer phytoneuston. Bull. Environ. Contam. Toxicol. 38: 1037-1043.

Rosenthal, R.J., D.C. Lees, and D.J. Maiero. 1982. Description of Prince William Sound shoreline habitats associated with biological communities. Juneau: NOAA. 58 pp.

Schoener, A. 1983. Colonization rates and processes as an index of pollution severity. Seattle: Office of Marine Pollution Assessment, NOAA. NOAA Tech. Memo. OMPA-27. 37 pp.

Schoener, A., E.R. Long, and J.R. DePalma. 1978. Geographic variation in artificial island colonization curves. Ecology 59(2): 367-382.

Sergy, G.A. 1985. The Baffin Island Oil Spill (BIOS) Project: A Summary. Proceedings of the 1985 Oil Spill Conference, Los Angeles, California, February 25-28, 1985. pp. 571-575.

Shiaris, M.P. 1989a. Phenanthrene mineralization along a natural salinity gradient in an urban estuary, Boston Harbor, Massachusetts. Microb. Ecol. 18: 135-146.

Shiaris, M.P. 1989b. Seasonal biotransformation of naphthalene, phenanthrene, and benzo[a]pyrene in surficial estuarine sediments. Appl. Environ. Microbiol. June 1989: 1391-1399.

Southward, A.J. and E.C. Southward. 1978. Recolonization of rocky shores in Cornwall after use of toxic dispersants to clean up the Torrey Canyon spill. J. Fish. Res. Bd. Can. 35: 682-706.

Spies, R.B., D.D. Hardin, and J.P. Toal. 1988. Organic enrichment or toxicity? A comparison of the effects of kelp and crude oil in sediments on the colonization and growth of benthic infauna. J. Exp. Mar. Biol. Ecol. 124: 261-282.

Stegeman, J.J. 1981. Polynuclear aromatic hydrocarbons and their metabolism in the marine environment. In: Polycyclic Aromatic

Hydrocarbons and Cancer. Volume 3. H.V. Gelboin and P.O.P. Ts'o, eds. New York: Academic Press. pp. 1-60.
Stiver, W. W.Y. Shiu, and D. Mackay. 1989. Evaporation times and rates of specific hydrocarbons in oil spills. Environ. Sci. Technol. 23: 101-105.

Straughan, D. 1976. Biological survey of intertidal areas in the Straits of Magellan in January 1975, five months after the METULA oil spill. In: Fate and Effects of Petroleum Hydrocarbons in Marine Ecosystems and Organisms. Wolfe, D.A., J.W. Anderson, K.B. Button, D.C. Malins, T. Roubal, and U. Varanasi, eds. Permagon Press, Oxford. pp. 247-260.

Thomas, M.L.H. 1978. Comparison of oiled and unoiled intertidal communities in Chedabucto Bay, Nova Scotia. J. Fish. Res. Board Can. 35: 707-716.

Vancouver Aquarium. 1990. Environment Youth Corps continues work on Stanley Park shoreline. Sea Pen 34(2):5.

Vandermeulen, J.H. 1977. The Chedabucto Bay Spill, ARROW 1970.. Oceanus 20(4). pp. 32-39.

Vandermeulen, J.H. and J.M. Capuzzo. 1983. Understanding sublethal pollutant effects in the marine environment. In: Ocean Waste Management: Policy and Strategies. M.A. Champ and M. Trainor, eds. Kingston, Rhode Island: University of Rhode Island. Paper No. 9.

Wong, W.J., D.R. Green, and C.A. Bawden. 1978. Long-ferm fate of a heavy fuel oil in a spill-contaminated B.C. coastal bay. J. Fish. Res. Board Can. 35: 521-527.

Woods Hole Oceanographic Institution. 1952. Marine Fouling and its Prevention. Annapolis, Maryland: U.S. Naval Institute. 388 pp.

Word, J.Q., L.S. Word, J.N. McElroy, and R.M. Thom. 1986. The surface microlayer: review of literature and evaluation of potential effects of dredge activities in Puget Sound. Seattle: U.S. Army Corps of Engineers. 134 pp.

APPENDIX A: CASE HISTORIES

A variety of case histories were reviewed in which biological recovery was monitored after oil spill events, or in which unpopulated materials were introduced into the marine environment and recolonization rates monitored. It is recognized that differences, sometimes substantial in nature, exist between the case histories presented here and the situation in Prince William Sound. However, in the absence of other more directly applicable data, these provide an information base from which projections of impacts and recovery might be derived. The studies described below are also summarized in accompanying tables.

No further treatment case histories

1. The IRISH STARDUST Spill

On 24 January 1973, the freighter IRISH STARDUST grounded on Haddington Reef in British Columbia. Two fuel tanks were ruptured, and approximately 200 tons of heavy "1,000 second" fuel oil spilled into Broughton Strait. Major cleanup operations were undertaken in most of the oiled areas, but one of the more contaminated bays was sufficiently isolated that it could be left undisturbed for scientific study. The recovery of this untreated bay was tracked for a period of one year, with both chemical and biological observations recorded and reported in Green et al. (1974).

Although the study period was relatively short for assessing recovery, it was noted that only those species that had been in direct contact with the oil appeared to have been adversely affected (e.g., limpets, periwinkles, isopods, rockweed, marsh grass). No species was completely eliminated, and there were indications of recolonization. The authors conluded that within the limitations of the study, it appeared there would be no permanent effects on the biological community.

The major mechanism for altering the chemistry of the oil was found to be bacterial. The asphalt-like covering that remained after bacteria degraded the paraffin portion of the oil was apparently more susceptible to physical weathering, and was slowly disappearing. Approximately 90-95% of the oil was estimated to have been removed by natural processes, and recovery from the relatively minor biological damage was judged to be underway.

Cretney et al. (1978) tracked the chemical fate of the spilled oil for four years following the grounding. Six visits were made to the site over the period. The composition of the oil was first altered by loss of lower molecular weight fractions through dissolution and evaporation. In the first year, biodegradation almost completely removed n-alkanes. By the fourth year, pristane and phytane were nearly completely gone. The most resistant components appeared to be the non-n-alkane compounds in the nC_{28-36} range.

2. METULA Spill: Microbiological Studies

The oil tanker METULA spilled approximately 51,500 tons of Arabian crude oil and 2,000 tons of Bunker C oil in the Strait of Magellan between Patagonia and Tierra Del Fuego on 9 August 1974. There was no cleanup performed, and this site serves as a natural laboratory to monitor the long-term persistence of oil. The location of the spill has some similarities to the EXXON VALDEZ spill.

Some of these similarities are:

- temperature
- ice cover
- some beach sites (i.e., mixed sand and gravel, deposited by glacial action)
- climate
- exposure of beaches
- and fetch and wave conditions.

Because of these characteristics, the two oil spills can be used as comparison models to some degree (Marshall and Gundlach, 1990).

The oil spill was studied by Baker and reported little damage from oil in the months immediately following the spill, but Straughan indicated reduction in distribution and abundance of species due to the presence of oil five months after the incident (Straughan, 1976). During this time, the inventory of cobble/gravel/sand type beaches had none to one or two species living at these kind of sites.

Six and one-half years after the METULA spill, the sites again were inventoried. It was found that on cobble and gravel type beaches only algae and mussels attached to gravel The investigators indicated that areas previously impacted, but now free of oil, were found to have a tremendous repopulation by mussels (Gundlach et.al., 1982).

Colwell et al. (1978) studied the microbiology of the bacteria associated with biodegradation of the oil that had been left along the shorelines of the Straits of Magellan. They arrived at five conclusions:

(1) The METULA petroleum contamination selectively stimulated certain bacterial species and so induced major shifts in the population structure.
(2) Temperature did not appear to be a limiting factor for biodegradation. In fact, there were some indications that cold-tolerant bacteria were more efficiently metabolizing the crude oil at 3°C than at 22°C.
(3) The biodegradation process in the marine environment is a slow process, possibly limited by nutrients and by physical access to degradable compounds in aggregations or tarballs.
(4) Weathering can be a significant removal process in concert with microbial degradation. However, neither mechanism is effective if oil is deposited and buried, or forms asphalt layers or tarballs.
(5) The weathering and microbial action may contribute significantly to formation of polar compounds.

3. U.S. Air Force Fuel Depot Spill, Searsport, Maine

In early March 1971, a pipeline ruptured at the U.S. Air Force Fuel Depot adjacent to the western shore of Long Cove in Searsport, Maine. At least 14 tons of a mixture of JP4 jet fuel and No. 2 heating oil was spilled, and reached the intertidal zone of Long Cove on March 16. Mayo et al. (1978) reported that in 1974, the loss of the standing stock of resident softshell clam, *Mya arenaria*, due to the spill was estimated to be greater than 85%. Chemical analysis of sediments showed substantial contamination with light to medium refined hydrocarbons.

By 1976, recovery of the *Mya* population was found to be taking place, but at a very slow rate. Animals were found only in well-drained portions of the upper intertidal zone. Comparison to chemical analyses of sediments showed that clams were found in only one area with petroleum hydrocarbon concentrations exceeding 100 ppm. In addition, very few clams were observed to be living in sediments with residues in the range of 50-100 ppm (six sites). The remaining 44 sites were populated with clams but had hydrocarbon residue concentrations at 49 ppm or less.

4. ARCO ANCHORAGE Spill

The tanker ARCO ANCHORAGE ran aground in Port Angeles Harbor, Washington, on 21 December 1985. Approximately 239,000 gallons of Alaska crude oil were spilled, and the the south-facing shore of Ediz Hook was impacted by both the spill and the subsequent cleanup activities. The latter resulted in physical disturbance of the beach between February and early April, 1986. Data collected as part of the Ediz Hook monitoring program indicated that more than 74% of the residual crude oil in sediments was removed by physical agitation during the beach reclamation process. Blaylock and Houghton (1989) asserted that the reclamation efforts undoubtedly decreased the retention time of oil in the sediments from that expected based on natural weathering processes alone, especially for deeper material.

A study (Word et al., 1987) cited by Blaylock and Houghton (1989) estimated that it would take 18.5 months for the intertidal beach to reach background hydrocarbon levels, and nearly four years for complete biological recovery. Blaylock and Houghton concluded that recovery at agitated transects likely occurred much more rapidly than in the absence of beach agitation. In particular, they found that recolonization by bivalves would probably not have occurred as quickly without the rapid reduction in sediment hydrocarbon concentrations that followed sediment agitation. The authors suggested that sediment chemistry and the biological conditions relative to infaunal biomass, density, and diversity indicated that 1988 conditions were similar to prespill conditions.

Mancini et al. (1989) affirmed these observations, noting that while it was not possible to conclude that biological recovery was "complete," available data clearly indicated it was well underway.

Miller (1989) examined physical and chemical recovery of intertidal and subtidal sediments from the ARCO ANCHORAGE spill, and noted that residual oil concentrations in Ediz Hook sediments declined steadily between March 1986 and July 1987. He also described the "armoring" of the pre-agitation beach by a surficial layer of cobbles on which barnacle encrustation was evident. This indicated that the cobbles were moved infrequently. The armor layer was breached by beach reclamation activities, suface cobbles becoming mixed with deeper sediments during beach agitation. The agitation equipment also left ridges in the beach surface roughly parallel to the shoreline. Miller commented that photographic documentation indicated little residual evidence of the beach ridges in May 1988, and much of the armor layer had been reestablished by this time (within two years of the beach disturbance).

5. TORREY CANYON Spill

Nelson-Smith (1977) discussed the recovery of British shorelines that had been affected by oil spills, including those that had been impacted by both oil from the TORREY CANYON and the ill-advised use of dispersants during the cleanup operations. Although noting that the coast of Cornwall was not rigorously resurveyed during the later stages of recovery, Nelson-Smith casually observed that within two to three years of the spill, the less seriously disturbed areas had regained their normal appearance, although some less common inhabitants were still rare or missing. The worst sites were superficially indistinguishable from normal after six to seven years with the exception of a higher density of limpets.

More generally, Nelson-Smith tentatively concluded that the rate of recovery from oil spills and damaging cleanup treatments depended very largely on the extent to which the numbers of limpets have been reduced and the speed with which they can reestablish themselves densely enough to control the resulting bloom of large algae. This, in turn, depends not only on the nature of the oil spilled and any cleansing treatments, but also on the extent of the damage and location of the affected shore with respect to sources of adult or larval animals able to recolonize the damaged area. The time scale of the process depends on how "recovery" is defined: if this is accepted as meaning a reinstatement of the previously dominant community, even if some minor members of the ecosystem may still be absent or fewer, then the oiling or cleaning incident may be less serious than a climatic aberration such as a hard freeze. Nelson-Smith noted that while recovery of intertidal communities from the TORREY CANYON incident seemed to have recovered after seven years, even impeded by destructive cleanup techniques, those animals most affected by a severe cold spell showed little sign of regaining their previous status on the worst-affected shores after 12 years.

6. Ecological Monitoring in Port Valdez, Alaska

Cowell and Monk (1979) discussed baseline surveys in the Port Valdez region of Prince William Sound and noted the technical and scientific problems associated with ecological monitoring there. In particular, they discussed the lack of understanding of the processes governing the Alaskan rocky shore ecosystem and the paucity of data on the natural stresses controlling temporal and spatial variation in populations. In addition, taxonomic difficulties, particularly in littoral macro-algae, compounded survey problems. Although the sheltered environment of Port Valdez differs from that encountered in central and southern Prince William Sound, many if not most, of the intertidal species are the same, and many if not most of the problems encountered by Cowell and Monk are the same as those that complicate an assessment of the proposed rock-washing activities.

Cowell and Monk found a distinct gradation in size of the limpet Collisella pelta with location in the intertidal. That is, a clear increase in the size of the limpets was observed with increasing height on the shore, and it was evident that very young limpets occurred only on the lower shore. Three possible explanations for this observed trend were given:

(1) Recruitment may occur over a large part of the shore, but only individuals which have settled on the lower shore may survive and older animals migrate upward.
(2) Recruitment may be limited to the lower shore and be followed by slow migration up the shore over a number of years.
(3) Annual limpet migration could occur, in which the winters are spent in the subtidal and migration into the intertidal could occur during the summer. The observed size distribution would be explained if the distance travelled by the limpets was in direct proportion to their size.

Colwell and Monk did not successfully elucidate the physical and biotic factors controlling the intertidal ecology near Port Valdez. Some links to salinity and exposure were suggested but data were limited. Unfortunately, examination of shores geographically more distant from Valdez was intended but not accomplished. This information would have been more relevant to the rock-washing evalutation.

7. Baffin Island Oil Spill (BIOS) Project

The BIOS Project was a multidisciplinary program of research on arctic marine oil spill fate, effects, and countermeasures. It was sponsored by governmental and industry agencies from four nations and took place over a four-year period between 1980 and 1984 on the northern end of Baffin Island, Northwest Territories, Canada. The major components of the study involved the controlled releases of oil to two complementary beaches, where the consequences of treatment with dispersants were compared to no treatment.

Sergy (1985) discussed results of the BIOS study. He commented that despite the significant accumulation of hydrocarbons and aside from acute effects, the overall magnitude of impact on subtidal (3-10m water depth, therefore well below the anticipated zone of major impact in rock-washing) was that, over two years following the releases, the beached oil (as well as the dispersed oil) did not cause any large-scale mortality of benthic biota or any significant change in benthic infaunal community structure. Relatively minor effects were noted in only a few of the species examined (temporary reduction in abundance, some reproductive impacts, changes in length-weight relationships).

8. San Francisco Bay Oil Spill

On 18 January 1971, two Standard Oil tankers collided under the Golden Gate Bridge, spilling 840,000 gallons of Bunker C oil into San Francisco Bay coastal waters. Pre-oil and post-oil transect data were compared by Chan (1975), and it was estimated that 4.2 to 7.5 million marine invertebrates, mostly barnacles, were smothered by the spill. In subsequent observations from 1972 to 1974, the sample counts of invertebrates had equalled or surpassed pre-oil transect levels. No long term effects of the spill were noted in any of the marine species.

In 1974, less than 5% of the oil spilled was estimated to be remaining on rocky surfaces. Good recruitment of marine organisms at oiled transects was observed, particularly for barnacles, mussels, periwinkles, and limpets. Decreases were seen in some reef organisms such as crabs. In a discussion of monitoring results five years after the spill, Chan (1977) noted some unusual population lows for a few intertidal species, but he attributed these to natural physical phenomena such as large waves. Chan called the overall health of marine life in the intertidal good. Species such as barnacles, limpets, mussels, periwinkles, starfish, turban snails, and shore crabs all showed steady population recruitment in the five years after the 1971 spill.

9. <u>ARROW Spill at Chedabucto Bay, 1970</u>

On 4 February 1970, the tanker ARROW grounded on rocks and spilled approximately 10,000 tons of Bunker C oil in Chedabucto Bay, Nova Scotia. The topography of this region consists of rocky outcrops, eroding till cliffs, and gravel and mixed sediment beaches. Six years after the spill, Thomas (1978) attempted to determine the effects of the spill on intertidal communities, but was confronted with the common problem of a lack of information on normal (pre-spill) community structure and species abundances. This was compounded by the fact that the spill occurred under winter conditions and in the presence of sea ice. Thomas found that on rocky shorelines, although there were no significant differences in intertidal distribution of species between oiled and control sites, a consistent difference in species diversity was observed. It was noted that this is typical of pollution-stressed communities, but also that it was interesting that the difference persisted six years after the spill event, even in areas that had little evidence of oil remaining. Six of the ten most common species were more abundant at control sites, and biomass of algae was found to exceed that at oiled sites by a factor of three.

Cleanup of the shoreline in the ARROW spill included manual pickup and use of heavy equipment to remove oiled sediments. In some cases, clean gravel was brought in to replace oiled material that was removed (Marshall and Gundlach, 1990). Thomas (1978) commented that it appeared that cleanup on rocky shores did not speed recovery and in fact may have hindered it, while that in finer-grained environments was less clear in its long-term impacts. It was noted that in the latter areas, if techniques could be improved to be less intrusive, then cleanup should help to minimize oil spill impacts.

10. GENERAL M. C. MEIGS Oil Spill, 1972

In January 1972, the troopship GENERAL M. C. MEIGS broke loose under tow and came ashore on the northwest coast of Washington. Navy Special fuel oil was spilled on a rich intertidal community. The ship continued to release oil to this habitat for more than five years, first in the form of fluid which came ashore on the shallow rock shelf and intertidal margins, and later in the form of discrete globules, floating into Wreck Cove where it became incorporated into the coarse sand beach (Clark et al., 1978).

The initial survey of the plants and animals in 1972, just after the spill, revealed no major damage or recent extensive mortalities in the animal populations, with the exception of sea urchins (*Strongylocetrotus purpuratus*). The latter lost spines from their aboral surface, and shortly after the spill some individuals had lost nearly all their spines. There were also some indication of direct mortality. Macrophytic algae in the affected area lost fronds and showed signs of bleaching, which were taken as signs of damage. By analyzing the tissues of mussels and algal species, it was noted that there was moderate uptake of hydrocarbons which occurred approximately nine months to one year after the spill. In 1973 there was a decline of the abundance of live barnacles, mussels, and colonial anemones. Sea urchin pathologies continued to be observed, as was bleaching in algal species. Brown algae became the dominant form over the mid-intertidal zone. Four years later, in 1977, the abundance of barnacles increased, and the other species of plants and animals continued to maintain their lower level of population numbers. No species completely disappeared, nor were abundances extremely low. The other algal types replaced the brown algae as dominant in the region. Hydrocarbons in the tissues were still found at measurable concentrations, probably due to recontamination by tar balls.

The definition of recovery in this case involved visual investigation and assessments of animal and plant communities, compared to sites in the region that were not contaminated with oil.

Overall, there was a general decline in the abundance of organisms, except in the barnacle population, over a five-year period of time. Sea urchins suffered up to three years (1975), but seemed to recover afterwards. This study indicated that there are species that are more sensitive than others to oil, and that the abundances were lowered after exposure, but the community reached a point of equilibrium after three to five years.

11. NESTUCCA spill, coasts of Washington and British Columbia

The NESTUCCA spilled Bunker C oil on the coasts of Washington and British Columbia. Monitoring for the National Park Service along the Olympic Peninsula has been conducted.

The Environmental Protection Service (EPS) also monitored recovery along the southwest coast of Vancouver Island where heavy oiling and various treatment technologies were used. On Vancouver Island, eelgrass beds were impacted and mortalities among other organisms were observed. One one beach an attempt was made to dig up the beach material and burn the oil residue. This operation was not successful since the material would not burn. One year later, the biota in the treated beach appeared to be similar to that of adjacent beaches that were not treated (L. Harding, EPS, personal communication).

12. IRINI Oil Spill, Stockholm Archipelago, Sweden, 1970

Medium and heavy fuel oils were spilled from the tanker IRINI in October 1970. About 1000 tons of this material drifted onto the shores, and an estimated 400 tons went into a small bay where it killed most of the littoral fauna. Much of the oil was collected or dispersed by booms, mechanically, or through the use of solvents. Oil was still visible under the surfaces of sandy beaches and stones. The oil degraded much more rapidly on rocky shores than in the sandy beaches. Notini (1978) reported recovery of the littoral community during the period 1971-1978. No oil effects were noted on macroalgae in 1971. Littoral fauna recovered more slowly, but had returned by 1976 (Marshall and Gundlach, 1990). The recovery period for this habitat, then, was about one to six years.

Rock washing case histories

1. The Alaskan Earthquake, 1964

On 27 March 1964, an earthquake of unusual severity struck the Prince William Sound region of Alaska. The earthquake, measuring between 8.3 and 8.6 on the Richter scale and lasting for over three minutes, caused extensive damage over a wide area and substantially changed the character of the shoreline in Prince William Sound. Both vertical and horizontal elevations in landmasses and the seafloor occurred. As a result, zones of distinct tidal communities were displaced and new zones were defined. Studies of the recovery of shoreline communities in Prince William Sound could offer general insights into mechanisms of ecological succession relevant to assessment of intertidal recovery after rock washing.

Haven (1971) surveyed Prince William Sound in 1965 (fifteen months after the earthquake), and observed that although recovery of shoreline biota was widespread, the intertidal areas had not yet settled into a climax community. As evidence of this, he cited (1) the lack of post-earthquake growth of the lichen *Verrucaria*, which normally defines the upper limit of the intertidal zone; (2) in areas of maximum uplift, the dominance of the red alga *Porphyra* in zones normally dominated by *Fucus*; (3) the reversal in dominance of two *Balanus* barnacle species in certain intertidal areas; and (4) the attachment of new mussel recruitment to algae rather than hard substrate.

Haven (1971) returned to the area in 1968 (four and one half years after the earthquake), and found that recovery to preearthquake community structure was much further advanced. It appeared that *Verrucaria* was establishing itself in the new upper intertidal zone, dominance of algae and barnacles had reverted to the previous conditions, and mussels were attached primarily to rocky substrates or barnacles. Haven concluded that with a few exceptions, the intertidal had returned to essentially its pre-earthquake condition in about four and one-half years.

2. Underground Nuclear Testing in the Aleutian Islands

Two nuclear underground tests were detonated on the Aleutian island of Amchitka on 2 October 1969, and on 6 November 1971. O'Clair and Zimmerman (1986) visited the island and studied changes there over a nearly fourteen year period. Portions of the rocky intertidal shoreline were uplifted 12cm by the first explosion, and 3-110cm by the second. Although most of the affected shoreline was bedrock or boulder in character, some intertidal species observed were the same as those encountered in much of Prince William Sound (e.g., *Fucus distichus*, *Mytilus edulis*, *Balanus glandula*). O'Clair and Zimmerman (1986) found that nine months after the second detonation event, the intertidal shoreline remained in flux, with recolonization still taking place and invertebrate and algal species richness decreased in the upper intertidal zone. Although intertidal communities similar to pretest assemblages were observed 33 months after the uplift of the shoreline, upper limits of the intertidal zone were below those that had existed previously. O'Clair and Zimmerman suggested that one reason for this may have been the gradual sporophytic reproductive process of the dominant macrophytes.

O'Clair returned to Amchitka Island in 1985, nearly fourteen years after the last test. He found that with some differences, the new communities approximated the biota that had existed prior to the disturbance. However, the upper extension of the intertidal, which in 1974 had appeared to be lower than the original limit, remained at essentially the same level.

It was concluded that in regions of uplift like that experienced on Amchitka Island, it normally takes three years before community development within the new uplifted community approximates the same species composition and relative abundances that had existed previously.

3. Review of Recovery and Recolonization of Hard Substrate Communities of the Outer Continental Shelf

This review was prepared for the Minerals Management Service and was published as Lissner et al. (1988). Although it was intended for use in evaluation of California outer continental shelf leasing areas, it contains a good summary of ecological theories of recovery and recolonization. As such, it provides insights into processes that would be operative in any marine environment, including that of gravel beaches in Prince William Sound that are under consideration for rock-washing.

Six factors were listed as influencing the recovery/recolonization process in marine ecosystems:

1) the type and size of the disturbed patch
2) seasonal fluctuations in abundances of propagules
3) water movement, including boundary layer flow, currents, and storm surge
4) selectivity of propagules for substratum characteristics during settlement
5) biological interactions occurring among the organisms colonizing patches
6) frequency of disturbances of the substratum

Some discussion was devoted to defining recovery/recolonization from an operational perspective, and this is relevant for the rock-washing assessment as well. In the extreme case, complete recovery from a disturbance could imply that a community has returned to its pre-disturbance state for all parameters such as species composition, species diversity, abundance of organisms, and age structure of populations. However, it was noted that many recent studies have defined benthic communities as consisting of a mosaic of patches varying in size, age, species composition, and disturbance, and a less rigorous definition was said to be both more appropriate and more practical. Lissner et al. (1988) used the definition of recovery as being complete when the variation in a disturbed area has returned to a range of variation observed within undisturbed control areas.

4. Colonization Curves for Clean Artificial Substrates

Long (1972) reported results of fouling accumulation studies at three sites off Kodiak Island, Alaska at depths of 5, 15, and 30 meters. Clean pressed asbestos panels were submerged in June of 1969 and 1970. Schoener et al. (1978) compared the rates of colonization of hard artificial substrates at numerous sites in tropical, subtropical, temperate and boreal latitudes, including the sites off Kodiak Island. The colonization curves were steeper (more species arrived quicker) in tropical and subtropical areas than in more northerly sites.

The substrates deployed off Kodiak Island in June quickly were colonized by four to five species within two months. Barnacles, mollusks, tubeworms, bryozoans, and tunicates were among the dominant organisms inventoried. The number of colonizing species gradually increased to about seven or eight species after eight months (February) and remained constant thereafter to the termination of the experiments (11 months). From these data it appears that initial colonization is very quick in the spring/summer, results in a community consisting of relatively few species, continues at a more subdued rate in the fall and winter and undergoes a small spurt in the following late winter/early spring. The equilibrium points were reached in 3 to 9 months. On panels that were deployed for just one month, most recruitment occurred in April through September, with very little to no recruitment in November through March.

It should be noted, however, that this study involved organisms most likely to colonize a continually submerged substrate. As a result, the flora and fauna were more typical of an Alaskan subtidal environment that is constantly emersed, instead of the intertidal portions of beaches most susceptible to impacts from a rock washing operation.

5. <u>The TORREY CANYON Spill/Dispersant Treatment</u>

The unique characteristic of this spill was the widespread use of oil dispersants. The spill consisted of 14,000 tons of Kuwaiti crude oil which was stranded along 150 km of the coast of West Cornwall, England in March 1967. The oil was treated with 10,000 tons of toxic dispersants during the cleaning operations. A combination of oil and dispersants killed off most animals and plants in heavily treated areas, but in less treated areas, some of the organisms survived. This produced a situation analogous to the results of rock washing, since it is presumed that rock washing would destroy the organisms that exist on and around the beach material being treated. However, in the TORREY CANYON spill, residues of dispersants remained after cleanup, and a slower rate of recolonization in certain areas was attributed to them. Moreover, most of the sites in this study consisted of rocky shores rather than the unconsolidated beaches characteristic of the candidate sites in Prince William Sound.

The definition of recovery in this study was related to three aspects: (1) comparison with other areas not affected by oil or dispersants, (2) species richness, and (3) the wide range of sizes and age groups of the organisms being inventoried.

Studies of the process of recolonization were an important part of the research occurring because of the TORREY CANYON spill. As a result of this research a greater understanding was gained of the colonization sequence that occurs when a natural marine site is profoundly disturbed. Following the oil spill and the indiscriminate use of dispersants, the sequence of colonization events were found to be:

(1) delayed settlement of organisms
(2) appearance of diatoms and filamentous algae
(3) rapid greening by Enteromorpha
(4) Fucus growth and die-off of barnacles
(5) settlement of limpets and other grazers
(6) loss of brown algae
(7) reduction in abundance of limpets and resettlement of barnacles
(8) the return of limpets and barnacles.

When the dominant *Fucus* growth was not heavy, the entire sequence took approximately five to eight years. When *Fucus* was a more prevalent component of the intertidal flora, the process was completed in approximately nine to ten years.

It was also found that some species may disappear completely from the intertidal communities in which they had existed prior to such disasters. This was the case with one species of hermit crab in a particular region impacted by this spill and the ensuing cleanup (Southward and Southward, 1978).

6. Lake Huron Beach Nourishment

In this project, beach material (cobble/gravel/coarse sand) was removed from one beach and added to another nearby beach in Lake Huron (Nester and Poe, 1982). The excavation and augmentation took place in October 1980. Samples were analyzed for biological community composition before and after the project. The abundance of oligochaete worms and chironomid insect larvae in shallow nearshore samples collected in June 1981, eight months after the project, was not significantly different from the abundance of these groups in the June 1980, sampling period before the project. The data suggest that biological recovery by these two groups was complete within eight months. It should be pointed out that the death of the organisms was most likely caused by the grinding and crushing of the beach materials on the organisms, rather than a chemical alteration. Some species may have survived the transport ordeal and initiated the recolonization processes in the new beach locations.

7. Puget Sound Colonization Curves

Colonization curves were determined for artificial substrates (panels) that were deployed in different sites in Puget Sound (Schoener, 1983). Some sites were known to be relatively highly contaminated with toxic chemicals and others were known to be relatively clean. Replicate clean substrates were deployed near the bottom at each site and colonization was monitored monthly for 18 months using a non-destructive approach. The colonization curves for the contaminated and clean sites differed. At the clean sites, the equilibrium points consisting of about 18 to 22 species, and were reached within about seven months. At the contaminated sites, equilibrium points of about 14 to 16 species were reached within seven months. Within 18 months, the numbers of species at all sites were similar (about 15 species) and the distinction between clean and contaminated sites disappeared. From this study, it appears that some initial delay in colonization of hard surfaces, such as boulders, would be expected if toxicants remained in the environment as compared to relatively clean environments.

8. AMOCO CADIZ Oil Spill 1978/Beach Treatment

In 1978, the AMOCO CADIZ spilled 223,000 metric tons of light Arabian and light Iranian crude oil, and 2,000 tons of Bunker C oil off the coast of Brittany, France. Large scale clean-up operations took place in most localities, using booms, tractors, honey wagons, front-end loaders, high pressure flushing, movement of oiled gravel/cobble into the surf zone, and hot water wands. The kind of shorelines affected included bedrock headlands, sandy beaches, mud estuaries, seagrass beds, mixed sand and gravel and cobble beaches, and marshes. Some of the sites are similar to the Alaska rock washing sites. The Brittany coast resembles the coasts of Washington and Oregon in climate, and is similar to portions of the Alaskan coast in wave action and tides. Information from several sources is compiled here and will be related as closely as possible to the Prince

William Sound spill situation. The oil has persisted for a long period of time, and still may be found on the beach in some areas.

There were multiple definitions of recovery for this spill. These included: the return of populations to normal densities; the absence of reproductive pathologies and fin necroses in fish; normal catch numbers of commercially important species (crabs, lobsters, fish, etc.); decline of mortality rates and levels of hydrocarbons in the tissues of certain key species to low or acceptable levels; and the return of species thought to have disappeared due to the oil/cleanup impact (amphipods and copepods, and clams). Equilibrium of the communities was a final measure of recovery in these habitats (Marshall and Gundlach, 1990).

On gravel/cobble beaches, it was reported that the oil was removed by a combination of clean-up procedures and natural processes within 1 to 1.5 years after the spill. Certain beaches were impacted again by another spill, the TANIO (7 March 1980), which obviously complicated assessment of recovery. With that spill, recovery occurred within approximately one to two years for plants (Gundlach et al., 1981), and within one to three years for animals (Gundlach et al., 1983). Finfish returned to normal population densities in one year, and flatfish in three years, showing no pathologies in reproduction and fin necroses. Crabs returned in one year, oysters in one to three years, and the plants *Laminaria*, *Fucus*, and *Ascophyllum* in one to two years after the spill. The chronologic sequence (Glemarec and Husenot, 1982) leading to equilibrium covered three years, but certain imbalances and perturbances persisted longer (Bodin and Boucher, 1982).

9. TAMANO Oil Spill and Treatment, Casco Bay, Maine, 1972

On 22 July 1972, the TAMANO spilled 100,00 gallons of No. 6 fuel oil at the enterence of Casco Bay, Maine. The areas shoreline included steep rock shelves, outcrops, and intrusions. This area is a major recreational site, and significant amounts of seafood (lobsters, shellfish, and seaweeds) are harvested from its waters.

Seaweed beds were cropped, but holdfasts were left and hot water under pressure was used to clean rocks. Booms were used to contain oil and sorbents picked up refloated oil. Oil penetrated the beaches into the underlying sand.

Comparisons between oiled and unoiled sites were studied one year after the spill to determine the effects of the contamination and recovery. Intertidal mudflat fauna were completely lost at one site, and contamination of clams and sediments were recorded at less affected areas. Seaweeds, barnacles, and snails were killed and lobsters showed increased hydrocarbon burdens. One year later, the sites were studied again and it was found that mudflats showed no evidence of new recruits. In contrast, new sets of shellfish did occur at control sites. Seaweeds did best where oiled fronds had not been removed, but still were in reduced in abundance in cropped areas. Hot water cleaned areas were slower to recolonize, but in the lower intertidal rock surfaces and tidal pools, life was more diverse (Marshall and Gundlach, 1990).

10. AMAZZONE Oil Spill/Hot Water Washing, Brittany, 1988

About 1,500 tons of paraffinic medium fuel oil from the AMAZZONE impacted the coast. The beaches that were of greatest concern were the cobble/pebble types. A large-scale washing plant was moved in for testing, which consisted of water washes of varying temperatures with the use of petroleum cut and surfactants. The Brittany coast is much like the coasts of Washington and Oregon in climate, and like Alaska in wave action and tides. The kind of beach material along this coast includes bedrock headlands, sandy beaches, mud estuaries, seagrass beds, mixed sand/gravel/cobble beaches, and marshes. Some of the sites were similar to the candidate Alaska rock washing sites.

Although equipment limitations restricted the highest water temperature to 17°C, the results of the washing tests indicated that higher temperatures were better than lower temperatures in oil removal. However, the oil was still present on the pebbles even though the efficacy of the cleaning technique was rated good. It was concluded that modifications of technology were needed (Marshall and Gundlach, 1990).

11. Restoration of Intertidal Beach Habitat near Stanley Park, Vancouver, B.C.

Under the direction of Dr. Jeff Marliave of the Vancouver Aquarium, a portion of the intertidal shoreline near Stanley Park in Vancouver, B.C. was reconstructed to correct a problem with siltation that had effectively eliminated natural spawning habitat for sculpins and blennies. Observed results of the rehabilitation project were summarized in Sea Pen (1990), with discussions in scientific literature in preparation.

Over 70 tons of rock and gravel were carefully placed on the beach to approximate preexisting and preferred habitat. A section of shoreline 10 meters wide was restored. Fish, shrimp, and crab were observed to move into the new habitat, and wintertime low-tide surveys indicated that the restored area was being utilized by cockscomb prickleback (*Anoplarchus purpurescens*) and crescent gunnel (*Pholis laeta*) for spawning. The padded sculpin (*Artedius fenestralis*) also commonly spawned at the site. Comparisons to observations made by Marliave prior to the episode of siltation showed that the spawning populations included both the same species that had previously used the area, as well as related but new species.

While demonstrating the feasibility of habitat restoration in the intertidal zone, the Stanley Park project also demonstrated the necessity for careful placement of substrate during beach rehabilitation. A federally mandated mitigation project in outer Vancouver Harbor that emphasized substrate selection and placement to a much lesser degree than Marliave's project was monitored as part of the latter, and failed to show any fish reproduction at all despite its location adjacent to spawning areas.

NATURAL RECOVERY OF COLD WATER MARINE ENVIRONMENTS AFTER AN OIL SPILL

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CONTENTS

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.*

Different from US law

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.*

what about most sensitive species

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

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

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

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

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

Natural Fluctuations

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

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

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

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

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

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

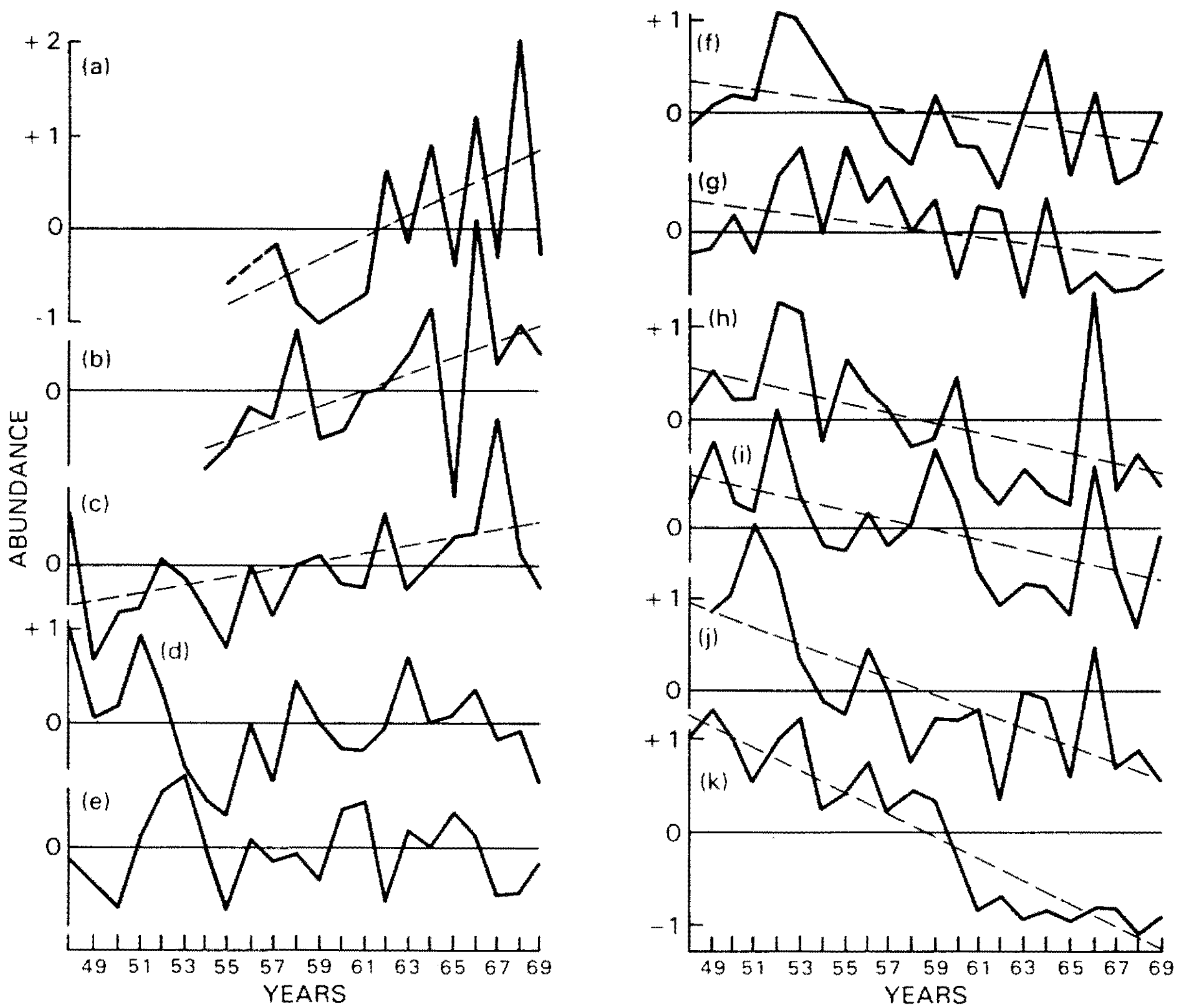


Figure 1: Changes in the abundance of zooplankton species in the North Atlantic. The organisms are: (a) *Pleuromamma borealis*; (b) *Euchaeta norvegica*; (c) *Acartia clausi*, (d) *Temora longicornis*; (e) *Clione lamacina*; (f) *Calanus helgolandicus* and *C finmarchicus* stages V and VI; (g) *Metridia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

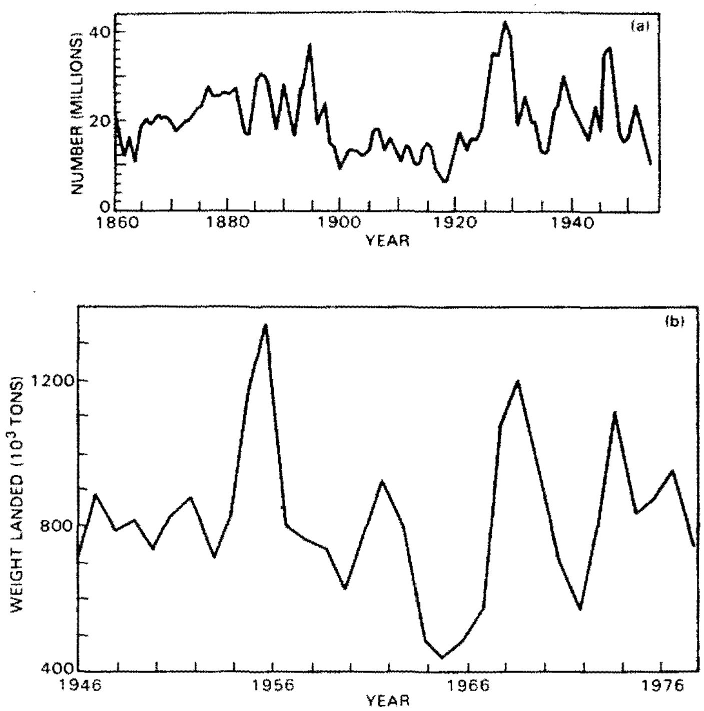


Figure 2: Landing of cod (a) in the Lofoten, Norway, fishery and (b) in the whole northeast Atlantic fishery (after Jones, 1982).

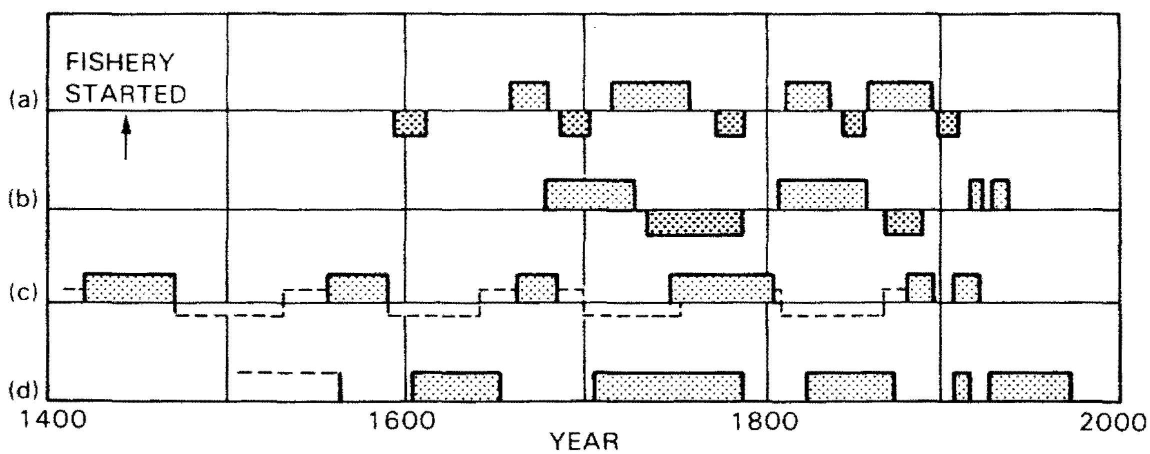
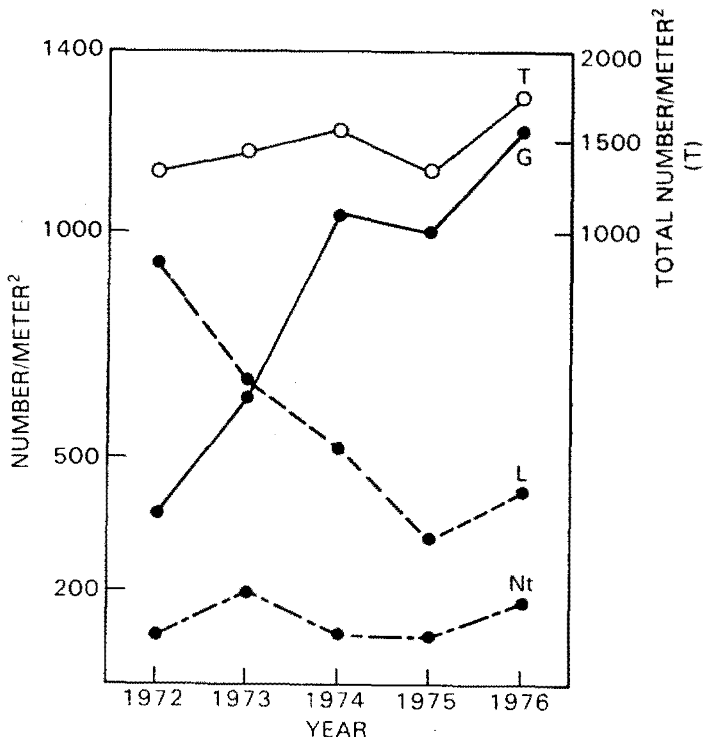


Figure 3: Periods of good and bad years (above and below line, respectively) for (a) Hokkaido herring; (b) Japanese sardine; (c) Bohuslan herring; (d) Atlantoscandian herring. (Dashed lines are speculative.) (After Jones, 1982.)



Gaining species 1972–6 (G)

Species	Gain/meter²
Paraonis gracilis	238
Prionospio malmgreni	144
Myriochele oculata	129
Thyasira flexuosa	115
Owenia fusiformis	104
Magelona minuta	100
Tharyx multibranchiis	27

Losing species 1972–6 (L)

Species	Loss/meter²
Spiophanes bombyx	219
Chaetozone setosa	129
Ampelisca tenuicornis	112
Amphiura filiformis	50
Nephtys hombergi	31
Scoloplos armiger	26
Rhodine gracilior	24
Ammotrypane aulogaster	17

Neutral species 1972–6 (Nt)

Species
Ampharete finmarchica
Glycera rouxi
Goniada maculata

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

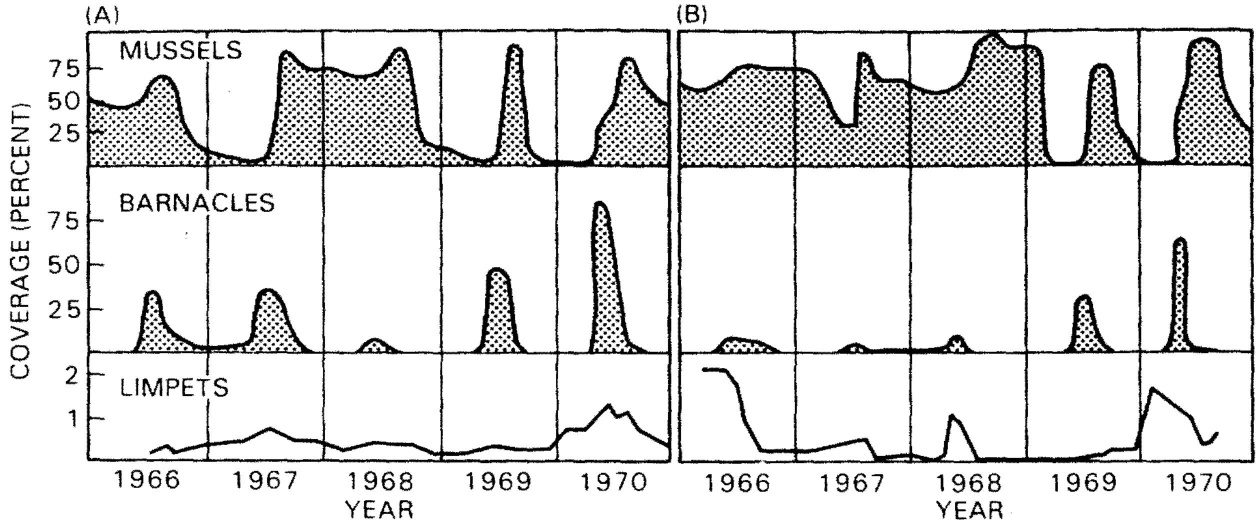


Figure 5: Fluctuations in the percentage cover on rocks by mussels, barnacles and limpets at two sites (A and B) on the north Yorkshire coast, 1966–1970 (after Lewis, 1972).

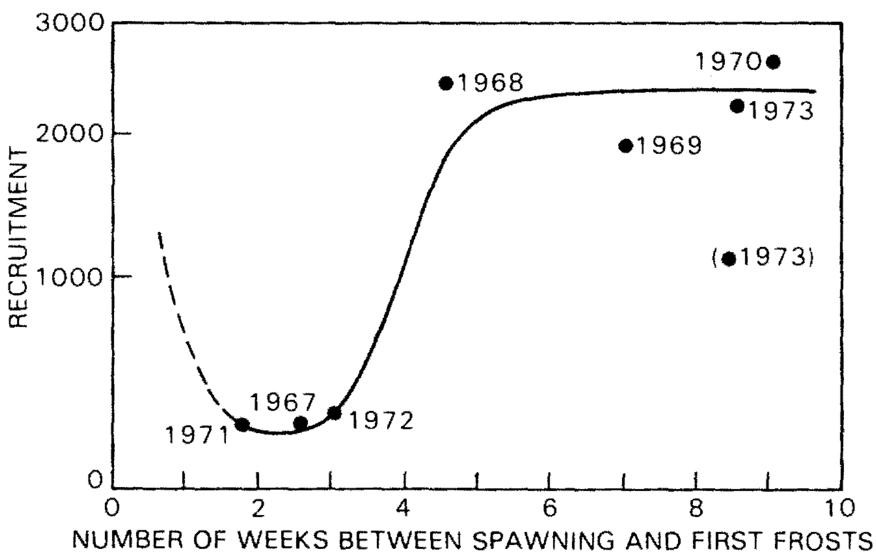


Figure 6: Relation between recruitment of limpets, *Patella vulgata*, and the duration of the frost-free period after spawning for the years 1967–1973. In 1973 there were two successional spawnings. (After Bowman and Lewis, 1977.)

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

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

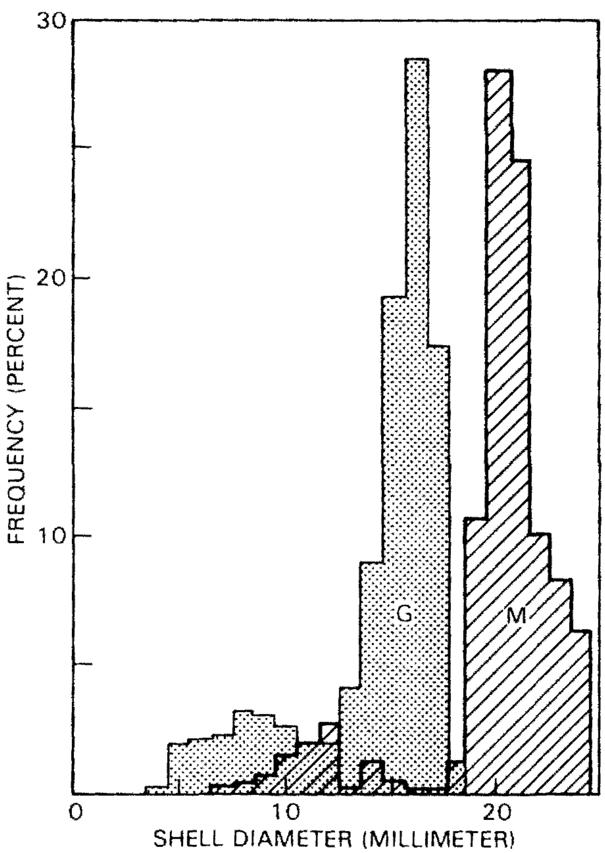


Figure 7: Size distribution of the intertidal gastropods *Gibbula* (G) and *Monodonta* (M) in a population on the Pembrokeshire coast, October 1981 (after Lewis, 1982).

A Working Definition of Ecological Recovery

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

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

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

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

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

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

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

COLD WATER ECOSYSTEMS

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

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

The reader is referred to Appendix A for further details.

PHYSICAL AND CHEMICAL BEHAVIOUR OF SPILLED OIL

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

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

The reader is referred to Appendix B for further details.

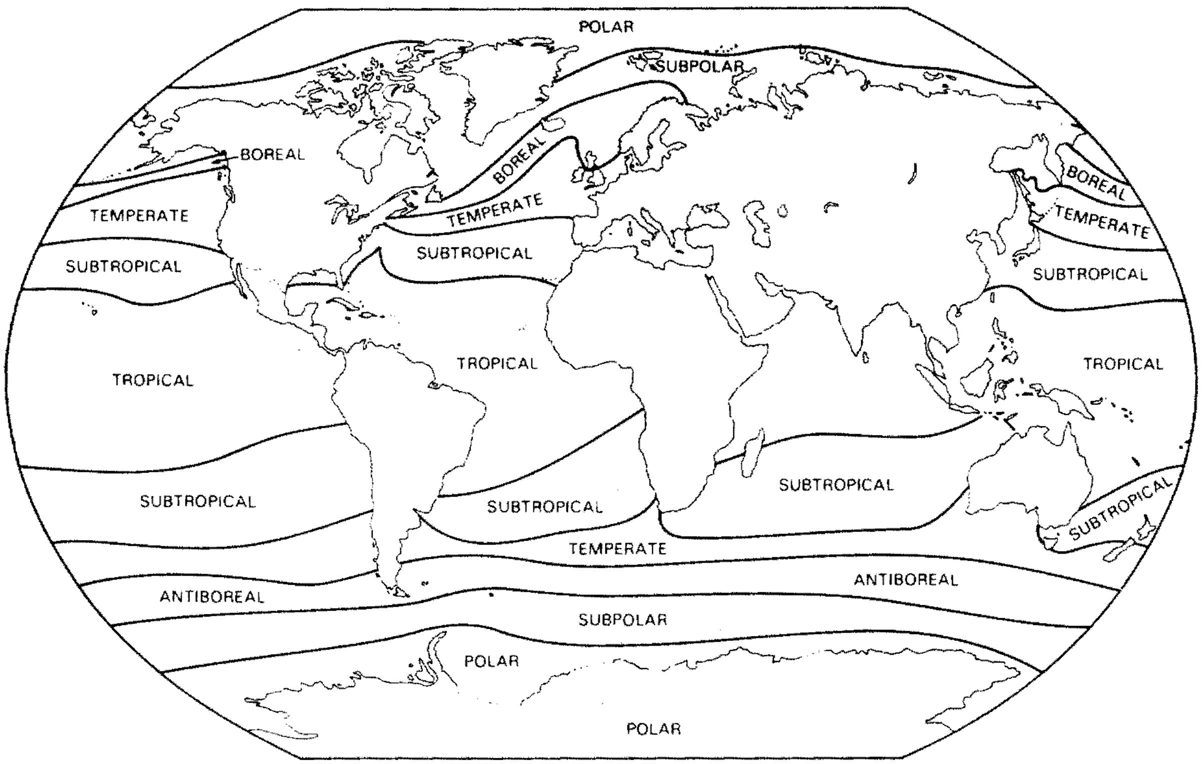


Figure 8: Map of the world showing the distribution of temperate to polar waters (Bogdanov, 1963).

NATURAL CLEANING

PHYSICAL REMOVAL BY NATURAL PROCESSES

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

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

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

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

Factors affecting residence times include

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

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

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

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

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

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

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

CHEMICAL AND BIOLOGICAL DEGRADATION

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

Chemical Degradation

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

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

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

Biological Degradation

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

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

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

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

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

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

- Limited circulation of interstitial water in fine-grained sediments

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

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

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

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

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

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

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

NATURAL RECOVERY

BIRDS

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

Losses of Seabirds from Oil

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

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

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

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

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

Other Causes of Mass Seabird Mortality

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

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

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

Population Effects

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

not mur

Murres

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

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

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

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

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

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

Cleaning and Rehabilitation

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

Polar Conditions

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

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

FISHERIES

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

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

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

Interference

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

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

Tainting

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

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

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

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

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

Sublethal Effects

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

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

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

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

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

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

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

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

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

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

Experience in Oil Spills

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

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

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

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

SEA MAMMALS

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

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

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

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

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

Effects of Oil

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

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

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

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

Avoidance of Oiled Waters

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

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

Casualties in Oil Spills

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

PELAGIC ENVIRONMENT

Plankton

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

BENTHIC ENVIRONMENT

Littoral Zone (Intertidal Zone)

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

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

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

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

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

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

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

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

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

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

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

Sublittoral Zone (Subtidal Zone)

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

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

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

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

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

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

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

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

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

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

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

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

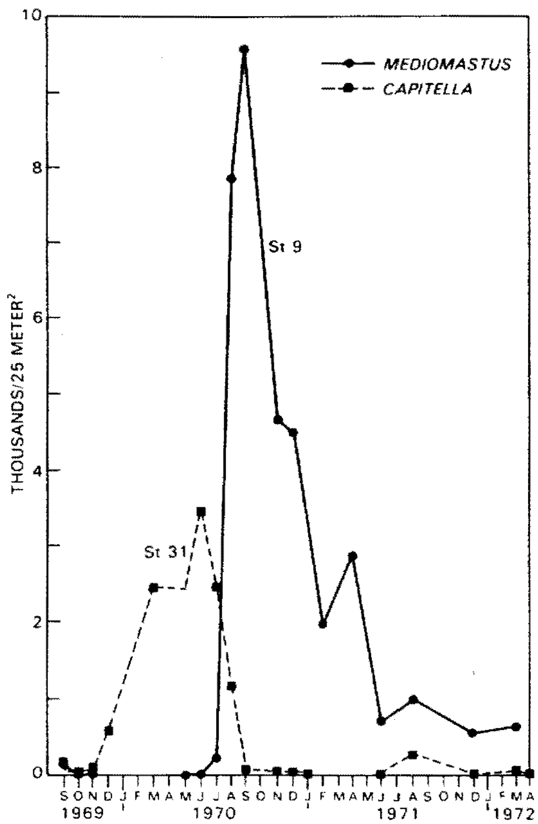


Figure 9: Succession of *Capitella* and *Mediomastus* nearshore (St 31) and offshore (St 9) after the *Falmouth* oil spill in Wild Harbor, Buzzards Bay, Mass. (Sanders *et al.*, 1980).

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

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

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

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

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

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

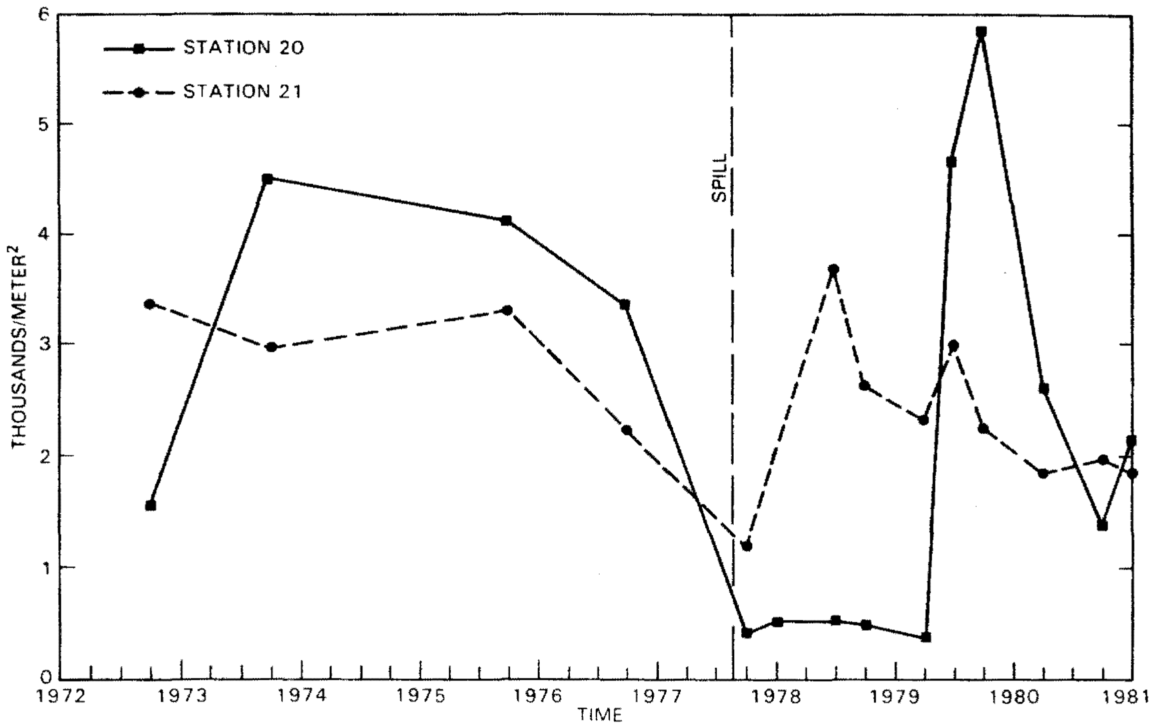


Figure 10: Changes in the faunal abundance at two localities affected by the *Tsesis* oil spill (Elmgren *et al.*, 1983).

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

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

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

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

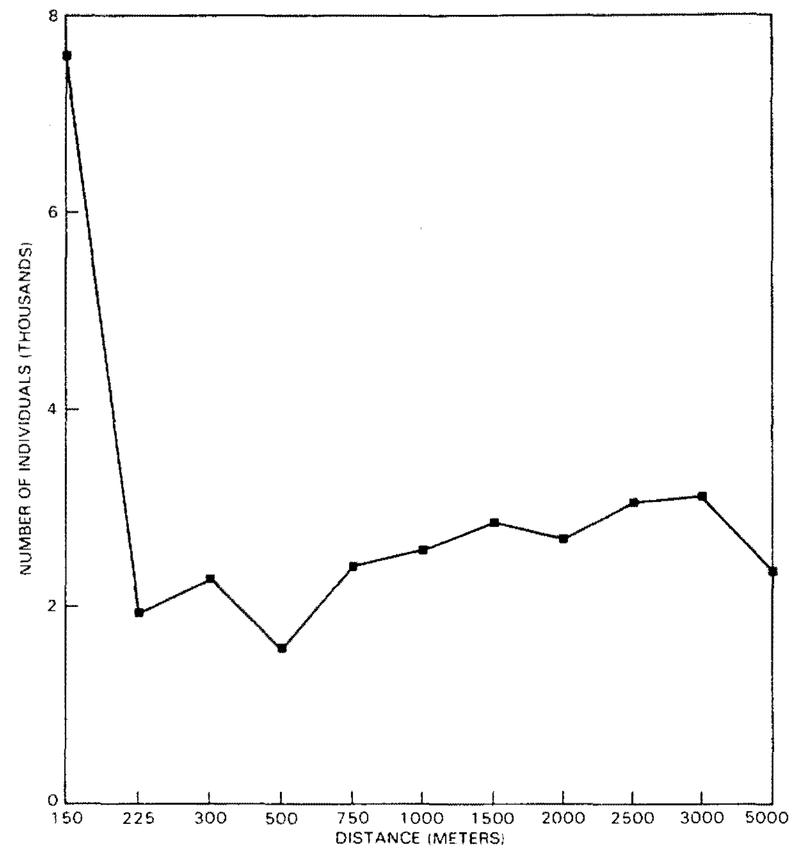


Figure 11: Changes in the number of individuals with increasing distance from the Statfjord Bravo production platform, 1984 (Matheson *et al.*, 1986).

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

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

ACKNOWLEDGMENTS

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TABLE 1

Examples of Background Concentrations* of Hydrocarbons in Water

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

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

TABLE 2

Examples of Background Concentrations* of Hydrocarbons in Sediments

Location	Hydrocarbon Type and Concentration	Reference
Baffin Bay and eastern Canadian Arctic	Extractable residues 1–41 ppm. Solvent $CC1_4$.	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.	Larsen *et al.* (1986)
Boston Harbour	14 PAH compounds, total concentrations 483–718,364 ppb (dry weight). The highest concentrations from the inner harbour almost an order of magnitude higher than previously reported concentrations from a range of urban estuaries.	Shiaris and Jambard-Sweet (1986)
Narragansett Bay	Alkanes and aromatics. 50–120 ppm south, 130–440 ppm central, 500–700 ppm north.	Farrington and Quinn (1973)
UK marine sediments	THC's (Ekofisk crude oil equivalents) 0.27–340 ppm, with the highest concentration being in the entrance to the Mersey.	Law (1981)
English channel	THC's (Ekofisk crude oil equivalents) dry mass, 0.3–5.60 ppm. Highest in Southampton Water.	Fileman and Law (1988)
Milford Haven and Daucleddau, SW Wales (subtidal)	THC's (ppm) commonly 100–300 up to 700. Contaminants concentrated in areas of fine sediment (high in organic and clay content).	Little and McLaren (1989)
Milford Haven (Angle Bay intertidal), SW Wales	THC's (ppm dry weight) in fine sand (0–5 cm layer), 18–70.	Howard *et al.* (1989)
Danish marine sediments	Alkanes and aromatics. 5–390 ppm Danish coast, 46–1800 ppm Copenhagen Sound.	Jensen (1981)
Southern Baltic Sea	THC's (Ekofisk crude oil equivalents) dry mass, 4.0–140 ppm, with the highest concentration occurring in the Gotland and Gdansk basins.	Law and Andrulewicz (1983)
Southern Baltic Sea	C15-C32 n-alkanes, 626–23,040 ppm benzo(a)pyrene, 2–49 ppm, with the highest concentrations occurring in Gdansk and Pomerania Bays.	Lamparczyk *et al.* (1988)

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

TABLE 3

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

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

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

TABLE 4

Residence Times of Oil on Shores

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

TABLE 4 Continued

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

TABLE 4 Continued

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

TABLE 4 Continued

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

TABLE 4 Continued

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

TABLE 5

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

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

TABLE 6

Microbial Degradation

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

TABLE 6 Continued

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

TABLE 7

Recovery of Littoral Benthos

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

TABLE 7 Continued

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

TABLE 7 Continued

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

TABLE 7 Continued

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

TABLE 7 Continued

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

REFERENCES

Abbot, B. C. and Straughan, D. M. (1969). Biological and oceanographic effects of oil spillage in the Santa Barbara Channel following the 1969 blowout—6 month progress report. *Mar. Pollut. Bull.*, 16, 6–11.

Addy, J. M., Levell, D. and Hartley, J. P. (1978). Biological monitoring of sediments in Ekofisk oilfield. In *Proceedings of a Conference on Assessment of Ecological Impacts of Spills*, 514–19. Washington, D.C.: American Institute of Biological Sciences.

Allen, A. A., Schlueter, R. S. and Mikolaj, P. G. (1970). Natural oil seepage at Coal Oil Point, Santa Barbara, California. *Science*, 170, 974–77.

Anderson, J. W., Dixit, D. B., Ward, G. S. and Foster, R. S. (1977). Effects of petroleum hydrocarbons on the rate of heart beat and hatching success of estuarine fish embryos. In *Physiological Responses of Marine Biota to Pollutants* (eds. F. J. Vernberg, A. Calabrese, F. P. Thurberg and W. B. Vernberg), 241–58. New York: Academic Press.

Anderson, J. W., Riley, R. G. and Bean, R. M. (1978). Recruitment of benthic animals as a function of petroleum hydrocarbon concentrations in the sediment. *J. Fish. Res. Board Can.*, 35, 679–80.

Anon. (1970). Report of the Task Force—Operation Oil (clean-up of the *Arrow* oil spill in Chedabucto Bay). Ottawa: Canadian Ministry of Transport, 164 pp.

Arhelger, S. D., Robertson, B. R. and Button, D. K. (1977). Arctic hydrocarbon biodegradation. In *Fate and Effects of Petroleum Hydrocarbons in Marine Ecosystems and Organisms* (ed. D. Wolfe), 270–75. New York: Pergamon Press.

Atlas, R. M. (1985). Effects of hydrocarbons on micro-organisms and petroleum biodegradation in Arctic ecosystems. In *Petroleum Effects in the Arctic Environment* (ed. F. R. Engelhardt), 63–99. London and New York: Elsevier Applied Science Publishers.

Atlas, R. M., Roubel, G., Bronner, A. and Haines, J. (1980). Microbial degradation of hydrocarbons in mousse from Ixtoc I. In *Proceedings of Conference on Researcher/Pierce* Ixtoc I *Cruises*, 1–24. National Oceanographic and Atmospheric Administration, Atlantic Oceanographic and Meteorological Laboratories, Miami, Florida.

Atwood, D. K., Benjamin, J. A. and Farrington, J. W. (1980). The mission of the September 1979 Researcher/Pierce Ixtoc I cruise and the physical situation encountered. In *Proceedings of a Symposium, Preliminary Results from the September 1979 Researcher/Pierce* Ixtoc I *Cruise*, 1–16. Washington, D.C.: Office of Marine Pollution Assessment (NOAA).

Baca, B. J., Lankford, T. E. and Gundlach, E. R. (1987). Recovery of Brittany coastal marshes in the eight years following the *Amoco Cadiz* incident. In *Proceedings of the 1987 Oil Spill Conference*, 459–64. API Publ. No. 4452. Washington, D.C.: American Petroleum Institute.

Bailey, E. P. and Davenport, G. H. (1972). Die-off of common murres on the Alaska peninsula and Unimak Island. *Condor*, 74, 215–19.

Baker, J. M. (1971). Seasonal effects of oil pollution on salt marsh vegetation. *Oikos*, 22, 106–10.

Baker, J. M., Bayley, J. A., Howells, S. E., Oldham, J. and Wilson, M. (1989). Oil in wetlands. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 37–59. Chichester and New York: John Wiley and Sons.

Baker, J. M., Campodonico, I., Guzman, L., Texera, J. J., Texera, B., Venegas, C. and Sanhueza, A. (1976). An oil spill in the Straits of Magellan. In *Marine Ecology and Oil Pollution* (ed. J. M. Baker), 441–71. New York: John Wiley and Sons (Halstead Press).

Baker, J. M., Crothers, J. H., Little, D. I., Oldham, J. H. and Wilson, C. M. (1984). Comparison of the fate and ecological effects of dispersed and nondispersed oil in a variety of intertidal habitats. In *Oil Spill Chemical Dispersants: Research, Experience and Recommendations*, STP 840 (ed. T. E. Allen), 239–79. Philadelphia: American Society for Testing and Materials.

Ballantine, W. J. (1961). A biologically-defined exposure scale for the comparative description of rocky shores. *Field Studies*, 1, 1–19.

Balouet, G. and Baudin-Laurencin, F. (1980). Penetrating ulcers of mullet and possible relationships with petroleum. In *ICES* [International Council for the Exploration of the Sea] *Spec. Mtg. Diseases of Commercially Important Fish and Shellfish*. Doc. No. 44. Copenhagen: International Council for Exploration of the Seas.

Bayne, B. L., Widdows, J., Moore, M. N., Salkeld, P., Worral, C. M. and Donkin, P. (1982). Some ecological consequences of the physiological and biochemical effects of petroleum compounds on marine molluscs. *Phil. Trans. R. Soc. Lond.*, B, 297, 219–39.

Bergmann, G. (1961). The migrating populations of long-tailed duck (*Clangula hyemalis*) and common scoter (*Melanitta nigra*) in the spring 1960. *Suomen Riista*, 14, 69–74.

Berman, M. S. and Heinle, D. R. (1980). Modification of the feeding behaviour of marine copepods by sub-lethal concentrations of water accommodated fuel oil. *Mar. Biol.*, 56, 59–64.

Birkhead, T. R., Lloyd, C. and Corkhill, P. (1973). Oiled seabirds successfully cleaning their plumage. *Brit. Birds*, 66, 535–37.

Blackman, R. A. A. and Law, R. J. (1980a). The *Eleni V* oil spill: Fate and effects of the oil over the first twelve months; Part 1, Oil in water and sediments. *Mar. Pollut. Bull.*, 11, 199–204.

Blackman, R. A. A. and Law, R. J. (1980b). The *Eleni V* oil spill: Fate and effects of the oil over the first twelve months; Part 2, Biological effects. *Mar. Pollut. Bull.*, 11, 217–20.

Blackman, R. A. A. and Law, R. J. (1981). The *Eleni V* oil spill: Return to normal conditions. *Mar. Pollut. Bull.*, 12, 126–30.

Blaylock, W. M. and Houghton, J. P. (1989). Infaunal recovery at Ediz Hook following the *Arco Anchorage* oil spill. In *Proceedings of the 1989 Oil Spill Conference*, 421–26. API Publ. No. 4479. Washington, D.C.: American Petroleum Institute.

Blokker, P. C. (1964). Spreading and evaporation of petroleum products on water. In *Proceedings of the 4th International Harbour Conference*, 911–19. Antwerp, Belgium.

Blumer, M., Ehrhardt, M. and Jones, J. H. (1973). The environmental fate of stranded crude oil. *Deep Sea Res.*, 20, 239–59.

Boehm, P. D., Barak, J. E., Fiest, D. L. and Elskus, A. A. (1982). A chemical investigation of the transport and fate of petroleum hydrocarbons in littoral and benthic environments: The *Tsesis* oil spill. *Mar. Env. Res.*, 6, 157–88.

Boehm, P. D. and Fiest, D. L. (1980). Aspects of the transport of petroleum hydrocarbons to the offshore benthos during the Ixtoc I blowout in the Bay of Campeche. In *Proceedings of a Symposium on Preliminary Results from the September 1979 Researcher/Pierce Ixtoc I Cruise*, 207–36. Washington, D.C.: Office of Marine Pollution Assessment (NOAA).

Boehm, P. D., Fiest, D. L. and Elskus, A. (1980). Comparative weathering patterns of hydrocarbons from the *Amoco Cadiz* oil spill observed at a variety of coastal environments. In *Proceedings of the International Symposium,* Amoco Cadiz *Fates and Effects of the Oil Spill*, 159–73. Paris: Le Centre National pour l'Exploitation des Oceans.

Bogdanov, D. V. (1963). Map of the natural zones of the ocean. *Deep Sea Res.*, 10, 520–23.

Bonsdorff, E. (1981). The *Antonio Gramsci* oil spill: Impact on the littoral and benthic ecosystems. *Mar. Pollut. Bull.*, 12, 301–05.

Bourne, W. R. P. (1971). Atlantic puffin decline. In *Smithsonian Inst. Center for Short-lived Phenomena Ann. Rep.*, 83–91.

Bourne, W. R. P. (1976). Seabirds and oil pollution. In *Marine Pollution* (ed. R. Johnston), 403–502. London: Academic Press.

Bowman, R. S. (1978). Dounreay oil spill. *Mar. Pollut. Bull.*, 9, 269–73.

Bowman, R. S. and Lewis, J. R. (1977). Annual fluctuations in the recruitment of *Patella vulgata*. *J. Mar. Biol. Ass. UK*, 57, 793–815.

Braithwaite, L. (1983). Effects of oil on the feeding mechanism of the bowhead whale. Bureau of Land Management, Rept. No. MMS/AK/ESU—83/013. Washington, D.C.: U. S. Govt. Printing Office, 51 pp.

Brandal, P. O., Grahl-Nielsen, D., Neppelberg, T., Palmork, K. H., Westrham, K. and Wilhelmsen, S. (1976). *Oil tainting of fish, a laboratory test of salmon and saithe*. CM 1976/E:33. Copenhagen: International Council for the Exploration of the Sea.

Broman, D., Colmsjo, A., Ganning, B., Naf, C., Zebuhr, Y. and Ostman, C. (1987). 'Fingerprinting' petroleum hydrocarbons in bottom sediment, plankton, and sediment trap collected seston. *Mar. Pollut. Bull.*, 18, 380–88.

Brown, R. G. B. (1982). Birds, oil and the Canadian environment. In *Oil and Dispersants in Canadian Seas—Research Appraisal and Recommendations* (eds. J. B. Sprague, J. H. Vandermeulen and P. G. Wells), 105–12. Rept. No. EPS 3-EC-82-2. Ottawa: Environmental Protection Service.

Brownell, R. and Le Boeuf, B. (1971). California sea lion mortality: Natural or artifact. In *Biological and Oceanographical Survey of the Santa Barbara Channel Oil Spill, 1969–1970* (ed. D. Straughan). Sea Grant Publ. 2, 1, 287–306. Los Angeles: Univ. of Southern California.

Buchanan, J. B., Sheader, M. and Kingston, P. F. (1978). Sources of variability in the benthic macrofauna off the south Northumberland coast, 1971–6. *J. Mar. Biol. Assoc. UK*, 58, 191–210.

Burns, K. A. and Teal, J. M. (1979). The West Falmouth oil spill: Hydrocarbons in the salt marsh ecosystem. *Estuar. Coast. Mar. Sci.*, 8, 349–60.

Burwood, R. and Speers, G. C. (1974). Photo-oxidation as a factor in the environmental dispersal of crude oil. *Estuar. Coast. Mar. Sci.*, 2, 117–35.

Cabioch, L., Dauvin, J. C., Mora Bermudez, J. and Rodriguez Babio, C. (1980). Effets de la maree noire de l'*Amoco Cadiz* sur le benthos sublittoral du nord de la Bretagne. *Helgolander Meeresunters*, 33, 192–208.

Calder, J. A. and Boehm, P. D. (1981). The chemistry of *Amoco Cadiz* oil in Aber Wrac'h. In *Proceedings of the International Symposium,* Amoco Cadiz: *Fates and Effects of the Oil Spill,* 149–58. Paris: Le Centre National pour l'Exploitation des Oceans.

Calder, J. A., Lake, J. and Laseter, J. (1978). Chemical composition of selected environmental and petroleum samples from the *Amoco Cadiz* oil spill. In *The* Amoco Cadiz *Oil Spill,* 21–84. Rept. No. NOAA—78080901. Washington, D.C.: U. S. Govt. Printing Office.

Chassé, C. (1978). The ecological impact on and near shores by the *Amoco Cadiz* oil spill. *Mar. Pollut. Bull.,* 9, 298–301.

Clark, R. B. (1978). Oiled seabird rescue and conservation. *J. Fish. Res. Board Can.*, 35, 675–78.

Clark, R. B. (1982). The impact of oil pollution on marine populations, communities and ecosystems. *Phil. Trans. R. Soc. Lond.*, B, 297, 433–43.

Clark, R. B. (1984). Impact of oil pollution on seabirds. *Env. Pollut.*, A, 33, 1–22.

Clark, R. B. (1987). Summary and conclusions: Environmental effects of North Sea oil and gas development. *Phil. Trans. R. Soc. Lond.*, B, 316, 669–77.

Clark, R. B. (1989). *Marine Pollution* (2nd ed.). Oxford: Clarendon Press.

Clark, R. C., Jr., Patten, B. G. and De Nike, E. E. (1978). Observations of a cold-water intertidal community after five years of a low-level persistent oil spill from the *General M. C. Meigs*. *J. Fish. Res. Board Can.*, 35, 754–65.

Clarke, A. (1983). Life in cold water, the physiological ecology of polar marine ectotherms. *Oceangr. Mar. Biol. Ann. Rev.*, 21, 341–453.

Clifton, H. E., Kvenvolden, K. A. and Rapp, J. B. (1984). Spilled oil and infaunal activity—Modification of burrowing behaviour and redistribution of oil. *Mar. Env. Res.*, 11, 111–36.

CNEXO (1981). Amoco Cadiz: *Fates and Effects of the Oil Spill.* Paris: Le Centre National pour l'Exploitation des Oceans.

Cole, T. J. (1978). Preliminary ecological-genetic comparison between unperturbed and oil-impacted *Urosalpinx cinerea* (Procobranchia: Gastropoda) populations: Hobska Point (Woods Hole) and Wild Harbour (West Falmouth), Massachusetts. *J. Fish. Res. Board Can.*, 35, 624–29.

Colebrook, J. M. (1979). Continuous plankton records: Monitoring the plankton of the North Atlantic and North Sea. In *Monitoring the Marine Environment* (ed. D. Nichols). Symposium of the Institute of Biology, 24, 87–102. London: Institute of Biology.

Colwell, R. R., Mills, A. L., Walker, J. D., Garcia-Tello, P. and Campos-P, V. (1978). Microbial ecology studies of the *Metula* spill in the Straits of Magellan. *J. Fish. Res. Board Can.*, 35, 573–80.

Conan, G. (1982). The long-term effects of the *Amoco Cadiz* oil spill. *Phil. Trans. R. Soc. Lond.*, B, 297, 323–33.

CONCAWE (1979). *The Environmental Impact of Refinery Effluents*, IV-25-IV-33, Conservation of Clean Air and Water—Europe Report 5/79, Den Haag.

Cormack, D. (1983). *Response to Oil and Chemical Marine Pollution.* London and New York: Applied Science Publishers.

Cormack, D. and Nicholls, J. A. (1978). Investigation of factors affecting the fate of North Sea oils discharged at sea. Part 2: The Ekofisk blowout, April/May 1977. Report No. LR 272 (OP), Warren Spring Laboratory, Stevenage, UK.

Cormack, D., Nicholls, J. A. and Lynch, B. (1978). Investigation of factors affecting the fate of North Sea oils discharged at sea. Part 1: Ekofisk crude oil, July 1975–February 1978. Report No. LR273 (OP), Warren Spring Laboratory, Stevenage, UK.

Corner, E. D. S., Harris, R. P., Kilvington, C. C. and O'Hara, S. C. M. (1976). Petroleum hydrocarbons in the marine food web: Short-term experiments on the fate of naphthalene in *Calanus*. *J. Mar. Biol. Ass. UK*, 56, 121–33.

Costa, D. P. and Kooyman, G. L. (1980). Effects of oil contamination in the sea otter, *Enhydra lutris*, 10, 65–107. Outer Continental Shelf Environmental Assessment Program, Final Report, Research Unit No. 71. National Oceanic and Atmospheric Administration. Washington, D.C.: U.S. Dept of Commerce.

Cretney, W. J., Wong, C. S., Green, D. R. and Bawden, C. A. (1978). Long-term fate of a heavy fuel oil in a spill-contaminated B. C. coastal bay. *J. Fish. Res. Board Can.*, 35, 521–27.

Cross, W. E. and Thomson, D. H. (1981). Macrobenthos—1980 Study Results (BIOS) Baffin Island Oil Spill Working Report 80-3. Environmental Protection Service. Ottawa: Environment Canada, 81 pp.

Cross, W. E. and Thomson, D. H. (1982). Macrobenthos—1981 Study Results (BIOS) Baffin Island Oil Spill Working Report 81-3. Environmental Protection Service. Ottawa: Environment Canada, 105 pp.

Crothers, J. H. (1983). Field experiments on the effects of crude oil and dispersant on the common animals and plants of rocky sea shores. *Mar. Env. Res.*, 8, 215–39.

Croxall, J. P. (1977). The effects of oil on seabirds. *Rapp. P.-V. Reun., Cons. Int. Explor. Mer*, 171, 191–95.

Croxall, J. P. and Prince, P. (1980). Food, feeding ecology and ecological segregation of seabirds of South Georgia. *Biol. J. Linn. Soc.*, 14, 103–31.

Cushing, D. H. (1953). Studies on plankton communities. *J. Cons. Perm. Int. Explor. Mer*, 19, 3–22.

Dauvin, J. C. and F. Gentil (in press). Conditions of the paracarid populations of subtidal communities in northern Brittany ten years after the *Amoco Cadiz* oil spill. *Mar. Pollut. Bull.*

Davenport, J. (1982). Oil and planktonic ecosystems. *Phil. Trans. R. Soc. Lond.*, B, 297, 369–84.

Davenport, J., Lonning, S. and Saethre, L. J. (1979). The effects of Ekofisk oil extract upon oxygen uptake in eggs and larvae of the cod. *Astarte*, 12, 31–34.

Davies, J. M., Addy, J. M., Blackman, R. A., Blanchard, J. R., Ferbrache, J. E., Moore, D. C., Somerville, H. J., Whitehead, A. and Wilkinson, T. (1984). Environmental effects of the use of oil based drilling muds in the North Sea. *Mar. Pollut. Bull.*, 15, 363–70.

Davies, J. M., Baird, I. E., Massie, L. C., Hay, S. J. and Ward, A. P. (1980). Some effects of oil-derived hydrocarbons on a pelagic food web from observations in an enclosed ecosystem and a consideration of their implications for monitoring. *Rapp. P.-V. Reun. Cons. Int. Explor. Mer*, 179, 201–11.

Davis, J. E. and Anderson, S. S. (1976). Effects of oil pollution on breeding grey seals. *Mar. Pollut. Bull.*, 7, 115–18.

Davis, P. H. and Spies, R. B. (1980). Infaunal benthos of a national petroleum seep: Study of community structure. *Mar. Biol.*, 59, 31–41.

Desaunay, Y. (1981). Evolution des stocks de poissons plats dans la zone contaminee par l'*Amoco Cadiz*. In *Colloque sur Les Consequences de la Pollution Accidentale par les Hydrocarbures de l'* Amoco Cadiz, 727–35. Paris: Le Centre National pour l'Exploitation des Oceans.

Dicks, B. (1976). Offshore biological monitoring. In *Marine Ecology and Oil Pollution* (ed. J. M. Baker), 325–440. New York: John Wiley and Sons (Halstead Press).

Dicks, B. and Levell, D. (1989). Refinery-effluent discharges into Milford Haven and Southampton Water. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 287–316. Chichester and New York: John Wiley and Sons.

Dunbar, M. J. (1968). *Ecological Development of Polar Regions*. Englewood Cliffs, N.J.: Prentice Hall, 119 pp.

Dunnet, G. M. (1982). Oil pollution and seabird populations. *Phil. Trans. R. Soc. Lond.*, B, 297, 413–27.

EEC [European Economic Community] (1986). Emulsification of hydrocarbons spilled at sea. Shell Research B V Report for Commission of European Communities. Available from Directorate General XI, European Economic Community, Brussels.

Ekman, S. (1967). *Zoogeography of the Sea*. London: Sedgewick and Jackson, 417 pp.

Ellis, D. V. (1960). Marine faunal benthos in Arctic North America. Arctic Institute of North America, Technical Paper No. 5, 53 pp.

Elmgren, R., Hansson, S., Larsson, U., Sundelin, B. and Boehm, P. D. (1983). The *Tsesis* oil spill: Acute and long-term impact on the benthos. *Mar. Biol.*, 73, 51–65.

Engelhardt, F. R. (1981). Oil pollution in polar bears: Exposure and clinical effects. In *Proceedings, Fourth Arctic Marine Oil Spill Program Technical Seminar*, 139–79. Ottawa: Environmental Protection Service.

Eurell, J. A. and Haensley, W. E. (1981). The effects of exposure to water-soluble fractions of crude oil on selected histochemical parameters of the Atlantic croaker, *Micropogon undulatus*. *J. Fish. Dis.*, 4, 187–194.

Fairhall, D. and Jordan, P. (1980). *Black Tide Rising: The Wreck of the* Amoco Cadiz. London: André Deutsch Ltd., 250 pp.

Farrington, J. W., Albaiges, J., Burns, K. A., Dunn, B. P., Eaton, P., Laseter, J. L., Porter, P. L. and Wise, S. (1980). Fossil fuels. In *The International Mussel Watch*, 7–77. Washington, D.C.: National Academy of Sciences.

Farrington, J. W., Davis, A. C., Frew, N. M. and Knap, A. (1988). ICES/IOC intercomparison exercise on the determination of petroleum hydrocarbons in biological tissues (mussel homogenate). *Mar. Pollut. Bull.*, 19, 372–80.

Farrington, J. W. and Quinn, J. G. (1973). Petroleum hydrocarbons in Narragansett Bay. 1. Survey of hydrocarbons in sediments and clams. *Estuar. Coast. Mar. Sci.*, 1, 71–79.

Fay, J. A. (1969). Spread of oil slicks on a calm sea. In *Oil in the Sea* (ed. D. P. Hoult), 53–64. New York: Plenum Press.

Fiest, D. L. and Boehm, P. D. (1980). Subsurface distributions of petroleum from an offshore well blowout, Bay of Campeche. In *Proceedings of a Symposium on Preliminary Results from the September 1979 Researcher/ Pierce Ixtoc I Cruise*, 169–85. Washington, D.C.: Office of Marine Pollution Assessment (NOAA).

Fileman, T. W. and Law, R. J. (1988). Hydrocarbon concentrations in sediments and water from the English Channel. *Mar. Pollut. Bull.*, 19, 390–93.

Fogelqvist, E., Lagerkvist, S. and Lindroth, P. (1982). Petroleum hydrocarbons in the Arctic Ocean surface water. *Mar. Pollut. Bull.*, 13, 211–13.

Ford, R. G., Wiens, J. A., Heinemann D. and Hunt, G. L. (1982). Modelling the sensitivity of colonially breeding marine birds to oil spills: Guillemot and kittiwake populations on the Pribilof islands, Bering Sea. *J. Appl. Ecol.*, 19, 1–31.

Foster, G. D. and Wright, D. A. (1988). Unsubstituted polynuclear aromatic hydrocarbons in sediments, clams, and clam worms from Chesapeake Bay. *Mar. Pollut. Bull.*, 19, 459–65.

Fricke, A. H., Hennig, H. F-K. O. and Orren, M. J. (1981). Relationship between oil pollution and psammolittoral meiofauna density of two South African beaches. *Mar. Env. Res.*, 5, 59–77.

Fusey, P. and Oudot, J. (1984). Relative influence of physical removal and biodegradation in the depuration of petroleum-contaminated seashore sediments. *Mar. Pollut. Bull.*, 15, 136–41.

Gardner, G. R. (1975). Chemically induced lesions in estuarine and marine teleosts. In *The Pathology of Fishes* (eds. W. Ribelin and G. Migaki), 657–93. Madison: Univ. of Wisconsin Press.

Geraci, J. R. and Smith, T. G. (1976). Direct and indirect effects of oil on ringed seals, *Phoca hispida*, of the Beaufort Sea. *J. Fish. Res. Board Can.*, 33, 1976–84.

Geraci, J. R. and St. Aubin, D. J. (1982). Study of the effects of oil on cetaceans. Contract AA-551-CT9-29, Rept. No. BLM/YL/SR-82/01. Washington, D.C.: U.S. Dept. of the Interior, Bureau of Land Management, 274 pp.

Geraci, J. R., St. Aubin, D. J. and Reisman, R. J. (1983). Bottlenose dolphins, *Tursiops truncatus*, can detect oil. *Can. J. Fish. Aquat. Sci.*, 40.

GESAMP (1977). Impact of oil on the marine environment. U.N. Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP). *Rep. Stud. GESAMP*, No. 6. Rome: Food and Agriculture Organization of the United Nations, 250 pp.

Gilfillan, E. S. and Vandermeulen, J. H. (1978). Alterations in growth and physiology of soft-shell clams, *Mya arenaria*, chronically oiled with Bunker C from Chedabucto Bay, Nova Scotia, 1970–1976, *J. Fish. Res. Board Can.*, 35, 630–36.

Gillam, A. H., O'Carroll, K. and Wardell, J. N. (1986). Biodegradation of oil adhering to drill cuttings. In *Proceedings of Conference: Oil Based Drilling Fluids. Cleaning and Environmental Effects of Oil Contaminated Drill Cuttings*, 123–36. Oslo: Statens forurensningstilsyn.

Glemarec, M. and Hussenot, E. (1981). Definition d'une succession ecologique en milieu meuble abnormalement enrichi en matieres organiques à la suite de la catastrophe de l'*Amoco Cadiz*: Fate and Effects of the Oil Spill, 499–512. Paris: Le Centre National pour l'Exploitation des Oceans.

Glover, R. S., Robinson, G. A. and Colebrook, J. M. (1972). Plankton in the North Atlantic—an example of the problems of analysing variability in the environment. In *Marine Pollution and Sea Life* (ed. M. Ruivo), 439–45. London: Fishing News (Books) Ltd.

Goodale, D. R., Hyman, M. A. M. and Winn, H. E. (1981). Cetacena response in association with *Regal Sword* oil spill. In *A Characterisation of Marine Mammals and Turtles in the Mid and Atlantic Areas of the U.S. Outer Continental Shelf* (eds. R. K. Edel, M. A. Hyman and M. F. Tyrell), Chapter XI, pp. XI, XI-15. Cetacean Turtle Assessment Program Annual Report 1979. Kingston: Univ. of Rhode Island.

Gordon, D. C., Jr., Dale, J. and Keizer, P. D. (1978). Importance of sediment working by the deposit-feeding polychaete *Arenicola marina* on the weathering rate of sediment-bound oil. *J. Fish. Res. Board Can.*, 35, 591–603.

Gordon, D. C. and Prouse, N. J. (1973). The effects of three oils on marine phytoplankton photosynthesis. *Mar. Biol.*, 22, 329–33.

Grahl-Nielsen, O. (1978). The Ekofisk Bravo blowout. Petroleum hydrocarbons in the sea. In *Proceedings of the Conference on Assessment of Ecological Impacts of Oil Spills*, 477–99. Washington, D.C.: American Institute of Biological Sciences.

Grainger, R. J. R., Duggan, C., Minchin, D. and O'Sullivan, D. (1980). Fisheries related investigations in Bantry Bay following the *Betelgeuse* disaster. CM 1980/E:54. Copenhagen: International Council for the Exploration of the Sea.

Grau, C. R., Roudybush, T., Dobbs, J. and Wathen, J. (1977). Altered yolk structure and reduced hatchability of eggs from birds fed single doses of petroleum oils. *Science*, 195, 779–81.

Greenwood, J. J. D., Donally, R. C., Feare, C. J., Gordon, N. J. and Waterston, G. (1971). A massive wreck of oiled birds: Northeast Britain, Winter 1970. *Scott. Birds*, 6, 235–50.

Grose, P. L. and Mattson, J. S. (1977). The *Argo Merchant* oil spill: A preliminary scientific report. NOAA/EPA Special Report, No. NOAA-79030901. Springfield, Va.: Natl. Technical Information Services, 133 pp.

Gross, M. G. (1987). *Oceanography: A View of the Earth* (4th ed.). Englewood Cliffs, N.J.: Prentice-Hall International Editions, 406 pp.

Grzybowski, J., Halkiewicz, J., Lamparczyk, H. and Radecki, A. (1987). Aliphatic and polycyclic aromatic hydrocarbons in the southern Baltic Sea. *Mar. Pollut. Bull.*, 18, 247–48.

Gulliksen, B. and Taasen, J. P. (1982). Effect of an oil spill in Spitzbergen in 1978. *Mar. Pollut. Bull.*, 13, 96–98.

Gundlach, E. R. (1987). Oil-holding capacities and removal co-efficients for different shoreline types to computer simulate spills in coastal waters. In *Proceedings of the 1987 Oil Spill Conference*, 451–57. API Rept. No. 4452. Washington, D.C.: American Petroleum Institute.

Gundlach, E. R., Berne, S., D'Ozouville, L. and Topinka, J. A. (1981). Shoreline oil two years after *Amoco Cadiz*: New complications from *Tanio*. In *Proceedings of the 1981 Oil Spill Conference*, 525–34. API Rept. No. 4334. Washington, D.C.: American Petroleum Institute.

Gundlach, E. R. and Hayes, M. O. (1978). Vulnerability of coastal environments to oil spill impacts. *Mar. Tech. Soc. Jour.*, 12, 18–27.

Gundlach, E. R. and Hayes, M. O. (1982). The oil spill environmental sensitivity index applied to the Alaskan coast. *Proceedings of the 5th AMOP* [Arctic Marine Oilspill Program] *Technical Seminar*, 311–23. Ottawa: Environmental Protection Service.

Gundlach, E. R., Kana, T. W. and Boehm, P. D. (1985). Modelling spilled oil partitioning in nearshore and surface zone areas. In *Proceedings of the 1985 Oil Spill Conference*, 379–83. API Rept. No. 4385. Washington, D.C.: American Petroleum Institute.

Guzman, L. and Campodonico, I. (1981). Studies after the *Metula* oil spill in the Straits of Magellan. In *Proceedings of the Petromar 80 Conference*, 363–76. London: Graham and Trotman.

Haines, J. R. and Atlas, R. M. (1982). In situ microbial degradation of Prudhoe Bay crude oil in Beaufort Sea sediments. *Mar. Env. Res.*, 7, 91–102.

Hampson, G. G. and Sanders, H. L. (1969). Local oil spill. *Oceanus*, 15, 8–11.

Hann, R. W. (1977). Fate of oil from the supertanker *Metula*. In *Proceedings of the 1977 Oil Spill Conference*, 465–68. API Rept. No. 4284. Washington, D.C.: American Petroleum Institute.

Hann, R. W. (1988). Prevention and control of oil and hazardous-material spills. In *Hazardous and Toxic Materials: Safe Handling and Disposal* (2nd ed.) (ed. H. H. Fawcett), 320–323. Chichester and New York: John Wiley and Sons.

Harper, J. R., Miskulin, G. A., Green, D. R., Hope, D. and Vandermeulen, J. H. (1985). Experiments on the fate of oil in low energy marine environments. In *Proceedings of the 8th AMOP* [Arctic Marine Oil Spill Program] *Technical Seminar*, 383–89. Ottawa: Environmental Protection Service.

Harris, M. P. and Murray, S. (1981). Monitoring of puffin numbers at Scottish colonies. *Bird Study*, 28, 15–20.

Hawkes, J. W. (1980). The effect of xenobiotics on fish tissues: Morphological studies. *Proc. Fedn. Am. Socs. Exp. Biol.*, 39, 3230–36.

Hayes, M. O., Gundlach, E. R. and D'Ozouville, L. (1979). Role of dynamic coastal processes in the impact and dispersal of the *Amoco Cadiz* oil spill (March 1978) Brittany, France. In *Proceedings of the 1979 Oil Spill Conference*, 193–98. API Rept. No. 4308. Washington, D.C.: American Petroleum Institute.

Hedgpeth, J. W. (1957). *Treatise on Marine Ecology and Paleoecology; Vol. 1, Ecology*. Boulder, Colo.: Geological Society of America.

Hess, W. N. (ed.) (1978). The *Amoco Cadiz* oil spill: A preliminary scientific report. NOAA/EPA Special Report NTIS PB-285-805. Washington, D.C.: U.S. Dept. of Commerce, 355 pp.

Hinga, K. R., Pilson, M. E. Q. and Almquist, G. (1986). The degradation of 7,12-dimethylbenz(a)anthracene in an enclosed marine ecosystem. *Mar. Env. Res.*, 18, 79–91.

Hollinger, J. P. and Mennella, R. A. (1973). Oil spills: Measurements of their distributions and volumes by multifrequency microwave radiometry. *Science*, 181, 54–56.

Hope Jones, P., Monnat, J.-Y., Cadbury, C. J. and Stowe, T. J. (1978). Birds oiled during the *Amoco Cadiz* incident—an interim report. *Mar. Pollut. Bull.*, 9, 307–10.

Hopner, Th., Harder, H., Kiesewetter, K. and Tegelkamp, B. (1987). Hydrocarbon biodegradation in marine sediments: A biochemical approach. In *Fate and Effects of Oil in Marine Ecosystems* (eds. J. Kuiper and W. J. Van den Brink), 41–55. Dordrecht: Martinus Nijhoff.

Hovland, M. and Judd, A. J. (1988). *Seabed Pockmarks and Seepages Impact on Geology, Biology and the Marine Environment*. London, Dordrecht and Boston: Graham and Trotman.

Hovland, M. and Thomsen, E. (1989). Hydrocarbon-based communities in the North Sea? *Sarsia*, 74, 29–42.

Howard, S., Baker, J. M. and Hiscock, K. (1989). The effects of oil and dispersants on seagrasses in Milford Haven. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 61–96. Chichester and New York: John Wiley and Sons.

Howells, S. E., Dodd, N. H. and Turner, C. (1989). The effects of oil and dispersants on seagrasses in Milford Haven. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 135–67. Chichester and New York: John Wiley and Sons.

Hurst, R. J., Oritsland, N. A. and Watts, P. D. (1982). Metabolic and temperature responses of polar bears to crude oil. In *Land and Water Issues Related to Energy Development* (ed. P. J. Rand), 263–80. Ann Arbor: Ann Arbor Science Publishers, Inc.

Jensen, K. (1981). Levels of hydrocarbons in mussels *Mytilus edulis* and surface sediments from Danish coastal areas. *Bull. Env. Contam. Toxicol.*, 26, 202–06.

Johannessen, K. I. (1976). Effects of sea water extract of Ekofisk oil on the hatching success of Barents Sea capelin. CM 1976/E:29. Copenhagen: International Council for the Exploration of the Sea.

Johansson, S., Larsson, U. and Boehm, P. (1980). The *Tsesis* oil spill: Impact on the pelagic ecosystem. *Mar. Pollut. Bull.*, 11, 284–93.

Johnson, J. H., Brooks, P. W., Aldridge, A. K. and Rowland, S. J. (1978). Presence and sources of oil in the sediment and benthic community surrounding the Ekofisk field after the blowout at Bravo. In *Proceedings of a Conference on Assessment of Ecological Impacts of Oil Spills*, 488–513. Washington, D.C.: American Institute of Biological Sciences.

Johnston, A. C., Larsen, P. F., Gadbois, D. F. and Humason, A. W. (1985). The distribution of polycyclic aromatic hydrocarbons in the surficial sediments of Penobscot Bay (Maine, U.S.A.) in relation to possible sources and to other sites worldwide. *Mar. Env. Res.*, 15, 1–16.

Johnston, R. (1977). What North Sea oil might cost fisheries. *Rapp. P.-V. Reun. Cons. Int. Explor. Mer*, 171, 212–23.

Jones, R. (1982). Population fluctuations and recruitment in marine populations. *Phil. Trans. R. Soc. Lond.*, B, 297, 353–68.

Jordan, R. E. and Payne, J. R. (1980). *Fate and Weathering of Petroleum Spills in the Marine Environment: A Literature Review and Synopsis*. Ann Arbor: Ann Arbor Science Publishers, Inc.

Karrick, N. L. (1977). Alterations in petroleum resulting from physico-chemical and microbiological factors. In *Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms* (ed. D. C. Malins), 1, 225–29. New York: Academic Press.

Keizer, P. D., Ahern, T. P., Dale, J. and Vandermeulen, J. H. (1978). Residues of Bunker C oil in Chedabucto Bay, Nova Scotia, six years after the *Arrow* spill. *J. Fish. Res. Board Can.*, 35, 528–35.

Kennicutt, M. C., Brooks, J. M., Bidigare, R. R., Fay, R., Wade, T. and Mcdonald, T. J. (1985). Vent type taxa in a hydrocarbon seep region on the Louisiana slope. *Nature*, 317, 351–53.

Kiceniuk, J. W. and Williams, U. P. (1987). Sediment hydrocarbon contamination at the site of the Baie Verte, Newfoundland, oil spill. *Mar. Pollut. Bull.*, 18, 270–74.

King, J. E. (1964). Seals of the world. In *Trustees of the British Museum* (Nat. Hist.), 124–25. London: British Museum of National History.

King, P. J. (1977). An assessment of the potential carcinogenic hazard of petroleum hydrocarbons in the marine environment. In *Petroleum Hydrocarbons in the Marine Environment* (eds. A. D. McIntyre and K. J. Whittle). *Rapp. P.-V. Reun. Cons. Int. Explor. Mer*, 171, 202–11.

Kingston, P. F. (1987). Field effects of platform discharges on benthic macro-fauna. *Phil. Trans. R. Soc. Lond.*, B, 316, 545–65.

Knap, A. H., Williams, P. J. Le B., and Tylor, I. (1979). Contribution of volatile petroleum hydrocarbons to the organic carbon budget of an estuary. *Nature*, 279, 517–19.

Knap, A. H., Williams, P. J. Le B., and Lysiak, E. (1982). Petroleum hydrocarbons in sediments of Southampton water estuary. *Mar. Env. Res.*, 7, 235–49.

Koblentz-Mishke, O. J., Volkovisny, V. V. and Kabanova, J. G. (1970). Plankton primary production of the world ocean. In *Scientific Exploration of the South Pacific* (ed. W. S. Wooster), 183–93. Washington, D.C.: National Academy of Science.

Koons, C. B. (1987). The source of petroleum in seawater. *Energy Progress*, 7, 204–07.

Kooyman, G. L., Davis, R. W. and Castellini, M. A. (1977). Thermal conductance of immersed pinniped and sea otter pelts before and after oiling with Prudhoe Bay crude. In *Fate and Effects of Petroleum Hydrocarbons in Marine Organisms* (ed. D. A. Wolfe), 151–57. New York: Pergamon Press.

Korringa, P. (1968). Biological consequences of marine pollution with special reference to North Sea fisheries. *Helgol. Wiss. Meeresunters.*, 17, 126–40.

Krebs, C. T. and Burns, K. A. (1978). Long-term effects of an oil spill on populations of the salt-marsh crab *Uca pugnax*. *J. Fish. Res. Board Can.*, 35, 648–49.

Kress, S. W. (1977). Establishing Atlantic puffins at a former breeding site. In *Endangered Birds: Management Techniques for Preserving Threatened Species* (ed. S. A. Temple), 373–77. Madison: Univ. of Wisconsin.

Kuhnhold, W. W. (1972). The influence of crude oils on fish fry. In *Marine Pollution and Sea Life* (ed. M. Ruivo), 315–18. London: Fishing News (Books) Ltd.

Kuhnhold, W. W. (1978). Impact of the *Argo Merchant* oil spill on macrobenthic and pelagic organisms. *AIBS Conf. Assessments of Ecological Impacts of Oil Spills*, 153–79. Washington, D.C.: American Institute of Biological Sciences.

Lamparczyk, H., Ochocka, R. J., Gryzbowski, J., Halkiewicz, J. and Radecki, A. (1988). Parameters related to pollution by n-alkanes and PAH in Baltic water and sediments. *Mar. Pollut. Bull.*, 19, 222–26.

Larsen, P. F., Gadbois, D. F. and Johnson, A. C. (1986). Polycyclic aromatic hydrocarbons in Gulf of Maine sediments: Distributions and mode of transport. *Mar. Env. Res.*, 18, 231–44.

Laubier, L. (1978). The *Amoco Cadiz* oil spill—lines of study and early observations. *Mar. Pollut. Bull.*, 9, 285–87.

Law, R. J. (1981). Hydrocarbon concentrations in water and sediments from UK marine waters, determined by fluorescence spectroscopy. *Mar. Pollut. Bull.*, 12, 153–57.

Law, R. J. and Andrulewicz, E. (1983). Hydrocarbons in water, sediment and mussels from the southern Baltic Sea. *Mar. Pollut. Bull.*, 14, 289–93.

Law, R. J. and Fileman, T. W. (1985). The distribution of hydrocarbons in surficial sediments from the central North Sea. *Mar. Pollut. Bull.* 16, 335–37.

Law, R. J., Marchand, J., Dahlmann, G. and Fileman, T. W. (1987). Results of two bilateral comparisons of the determination of hydrocarbon concentrations in coastal seawater by fluorescence spectroscopy. *Mar. Pollut. Bull.*, 18, 486–89.

Le Boeuf, B. (1971). Oil contamination and elephant seal mortality: A "negative" finding. In *Biological and Oceanographical Survey of the Santa Barbara Channel Oil Spill* (ed. D. Straughan), 277–86. Los Angeles: Univ. of Southern California.

Lee, R. F. (1975). Fate of petroleum hydrocarbons in marine zooplankton. In *Proceedings, 1975 Conference on Prevention and Control of Oil Pollution*, 549–53. Washington, D.C.: American Petroleum Institute.

Lee, R. F. and Takahashi, M. (1977). The fate and effect of petroleum in controlled ecosystem enclosures. *Rapp. P.-V. Reun. Cons. Int. Explor. Mer*, 171, 150–56.

Lee, R. F., Takahashi, M., Beers, J. R., Thomas, W. H., Sieber, D. L. R., Koeller, P. and Green, D. R. (1977). Controlled ecosystems: Their use in the study of the effect of petroleum hydrocarbons on plankton. In ***Physiological Responses of Marine Biota to Pollutants*** (eds. F. J. Vernberg, A. Calabrese, F. P. Thurberg and W. B. Vernberg), 323–42. New York: Academic Press.

Lee, W. Y. and Nicol, J. A. C. (1977). The effects of the water soluble fractions of No. 2 fuel oil on the survival and behaviour of coastal and oceanic zooplankton. *Env. Pollut.* 12, 279–92.

Lemmetyinen, R. (1966). Damage to waterfowl in the Baltic area caused by waste oil. *Suomen Riista*, 19, 63–71.

Leppakoski, E. (1973). Effects of an oil spill in the northern Baltic. *Mar. Pollut. Bull.*, 4, 93–94.

Leslie, F. H. (1966). The intrinsic rate of increase and overlap of successive generations in a population of guillemots (*Uria aalge*). *J. Anim. Ecol.*, 35, 291–301.

Levy, E. M. (1981). Background levels of petroleum residues in Baffin Bay and the eastern Canadian Arctic: Role of natural seepage. In *Proceedings of the Petromar 1980 Conference*, 345–62. London: Graham and Trotman.

Lewis, J. R. (1964). *The Ecology of Rocky Shores*. London: The English Universities Press Ltd., 323 pp.

Lewis, J. R. (1972). Problems and approaches to baseline studies of coastal communities. In *Marine Pollution and Sea Life* (ed. M. Ruivo), 401–04. London: Fishing News (Books) Ltd.

Lewis, J. R. (1982). The composition and functioning of benthic ecosystems in relation to the assessment of long-term effects of oil pollution. *Phil. Trans. R. Soc. Lond.*, B, 297, 257–67.

Linden, O., Elmgren, R. and Boehm, P. (1979). The *Tsesis* oil spill: Its impact on the coastal ecosystems of the Baltic Sea. *Ambio*, 8, 244–53.

Little, D. I. (1987). The physical fate of weathered crude and emulsified fuel oils as a function of intertidal sedimentology. In *Fate and Effects of Oil in Marine Ecosystems* (eds. J. Kuiper and W. J. Van den Brink), 3–18. Dordrecht: Martinus Nijhoff.

Little, D. I., Howells, S. E., Abbiss, T. P. and Rostron, D. (1987). Some factors affecting the fate of estuarine sediment hydrocarbons and trace metals in Milford Haven 1978–1982. In *Pollutant Transport and Fate in Ecosystems* (eds. P. J. Coughtrey, M. H. Martin and M. H. Unsworth), 55–87. Oxford and London: Blackwell.

Little, D. I. and McLaren, P. (1989). Sediment and contaminant transport in Milford Haven. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 203–34. Chichester and New York: John Wiley and Sons.

Little, D. I. and Scales, D. L. (1987). The persistence of oil stranded on sediment shorelines. In *Proceedings of the 1987 Oil Spill Conference*, 433–38. API Publ. No. 4452. Washington, D.C.: American Petroleum Institute.

Long, B. F. N., Vandermeulen, J. H. and Ahern, T. P. (1981). The evolution of stranded oil within sandy beaches. In *Proceedings of the 1981 Oil Spill Conference*, 519–24. API Rept. No. 4334. Washington, D.C.: American Petroleum Institute.

Longwell, A. C. (1978). Field and laboratory measurements of stress response at the chromosome and cell levels in planktonic fish eggs and the oil problem. In *In the Wake of the* Argo Merchant, 116–25. Kingston: Univ. of Rhode Island.

Mackay, D. and McAuliffe, C. D. (1989). Fate of hydrocarbons discharged at sea. *Oil and Chemical Pollut.*, 5, 1–20.

Mackie, P. R., Hardy, R., Whittle, K. J., Bruce, C. and McGill, A. S. (1980). The tissue hydrocarbon burden in mussels from various sites around the Scottish coast. In *Polynuclear Aromatic Hydrocarbons: Chemistry and Biological Effects* (eds. A. Bjorseth and A. J. Dennis), 379–93. Columbus (Ohio): Battelle Press.

Mair, J. McD., Matheson, I. and Appelbee, J. F. (1987). Offshore macrobenthic recovery in the Murchison field following the termination of drill cuttings discharge. *Mar. Pollut. Bull.* 18, 628–34.

Malins, D. C. (1981). The chemical and biological degradation of petroleum: A foremost challenge for the analytical chemist. In *Proceedings of the Petromar 1980 Conference*, 319–43. London: Graham and Trotman.

Mancini, E. R., Lindstedt-Siva, J. and Chamberlain, D. W. (1989). Environmental impacts of the 1985 *Arco Anchorage* oil spill: 1988 conclusions. In *Proceedings of the 1989 Oil Spill Conference*, 459–62. API Publ. No. 4479. Washington, D.C.: American Petroleum Institute.

Massie, L. C., Ward, A. P. and Davies, J. M. (1985a). The effects of oil exploration and production in the northern North Sea: Part 1—the level of hydrocarbons in water and sediments in selected areas, 1978–1981. *Mar. Env. Res.*, 15, 165–213.

Massie, L. C., Ward, A. P. and Davies, J. M. (1985b). The effects of oil exploration and production in the northern North Sea: Part 2—microbial biodegradation of hydrocarbons in water and sediments, 1978–1981. *Mar. Env. Res.*, 15, 235–62.

Matheson, I., Kingston, P. F., Johnston, C. S. and Gibson, M. J. (1986). Statfjord field environmental study. In *Proceedings of Conference on Oil-based Drilling Fluids—Cleaning and Environmental Effects of Oil Contaminated Drilling Cuttings*, 3–16. Oslo: Statens forurensningstilsyn.

Mattsson, J. and Lehtinen, C. (1985). Increased levels of petroleum hydrocarbons in the surface sediments of Swedish coastal waters. *Mar. Pollut. Bull.*, 16, 390–95.

Mayo, D. W., Page, D. S., Cooley, J., Sorenson, E., Bradley, F., Gilfillan, E. S. and Hanson, S. A. (1978). Weathering characteristics of petroleum hydrocarbons deposited in fine clay marine sediments, Searsport, Maine. *J. Fish. Res. Board Can.*, 35, 552–62.

McAuliffe, C. D. (1986). Organism exposure to volatile hydrocarbons from untreated and chemically dispersed crude oils in field and laboratory. In *Proceedings of the Ninth Arctic Marine Oil Spill Program Technical Seminar*, 497–526. Ottawa: Environment Canada.

McCain, B. B., Hodgins, H. O., Gronlund, W. D., Hawkes, J. W., Brown, D. W. and Myers, M. S. (1978). Bioavailability of crude oil from experimentally oiled sediments to English sole (*Parophyrys vetulus*), and pathological consequences. *J. Fish. Res. Board Can.*, 35, 657–64.

McIntyre, A. D. (1982). Oil pollution and fisheries. *Phil. Trans. R. Soc. Lond.*, B, 297, 401–11.

McIntyre, A. D., Bayne, B. L., Rosenthal, H. and White, I. C. (eds.) (1978). On the feasibility of effects monitoring. *Coop. Res. Rep. Cons. Int. Explor. Mer*, No. 75, 42 pp.

McLaren, P. (1984). The *Whytecliff* oil spill, British Columbia: Sediment trends and oil movement on a beach. *Spill Technology Newsletter*, 9, 11–19.

McLaren, P. (1985). Behaviour of diesel fuel on a high energy beach. *Mar. Pollut. Bull.*, 16, 191–96.

McLaren, P. and Little, D. I. (1987). The effects of sediment transport on contaminant dispersal: An example from Milford Haven. *Mar. Pollut. Bull.*, 18, 586–94.

Mead, C. and Baillie, S. (1981). Seabirds and oil: The worst winter. *Nature*, 292, 10–11.

Michael, A. D., Van Raalte, C. R. and Brown, L. S. (1975). Long-term effects of an oil spill at West Falmouth, Massachusetts. In *1975 Conference on Prevention and Control of Oil Pollution: Proceedings*. Washington, D.C.: American Petroleum Institute.

Mille, G., Chen, Y. Y., Guisti, G., Dou, H., Vacelet, E., Le Campion-Alsumard, T. and Plante-Cuny, M-R (1984). Evolution des hydrocarbures, des peuplements bacteriens et microphytiques dans les sediments des marais maritimes de L'Ile Grande polues par L'*Amoco Cadiz*: 1, Evolution des hydrocarbures et des bacteries. *Mar. Env. Res.*, 11, 213–32.

Mills, S. (1981). Graveyard of the puffin. *New Scient.*, 91, 10–13.

Minchew, C. D. and Yarbrough, J. D. (1977). The occurrence of fin rot in mullet (*Mugil cephalus*) associated with crude oil contamination of an estuarine-pond ecosystem. *J. Fish. Biol.*, 10, 319–23.

Mix, M. C. and Schaffer, R. L. (1983a). Concentrations of unsubstituted polynuclear aromatic hydrocarbons in bay mussels (*Mytilus edulis*) from Oregon, USA. *Mar. Env. Res.*, 9, 193–209.

Mix, M. C. and Schaffer, R. L. (1983b). Concentrations of unsubstituted polycyclic aromatic hydrocarbons in softshell clams from Coos Bay, Oregon, USA. *Mar. Pollut. Bull.*, 14, 94–97.

Moore, M. N., Pipe, R. K. and Farrar, S. V. (1982). Lysosomal and microsomal responses to environmental factors in *Littorina littorea* from Sullom Voe. *Mar. Pollut. Bull.*, 13, 340–45.

Morris, R. (1970). Alaska peninsula oil spill. Event No. 36–70. Annual Report, 154–57. Washington, D.C.: Smithsonian Institute, Center for Short-lived Phenomena.

Morzer-Bruyns, M. F. and Tanis, J. J. C. (1968). The impact of oil pollution on sea birds in Europe. In *Proceedings of International Conference: Oil Pollution of the Sea*, (ed. P. Barclay-Smith), 67–74. British Advisory Committee on Oil Pollution of the Sea, Italian National Commission on Oil Pollution of the Sea, and Nordic Union for Prevention of Oil Pollution of the Sea.

Mottershead, D. N. (1981). The persistence of oil pollution on a rocky shore. *Applied Geography*, 1, 297–304.

Myers, E. P. and Gunnerson, C. G. (1976). *Hydrocarbons in the Ocean.* Washington, D.C.: U. S. Dept. of Commerce (NOAA), 42 pp.

Nelson-Smith, A. (1970). The problem of oil pollution in the sea. *Adv. Mar. Biol. Ann. Rev.*, 8, 215–306.

NERC (1971). The sea bird wreck in the Irish Sea, Autumn 1969. *NERC Publ.*, Ser. C, No. 4. London: Natural Environment Research Council.

NERC (1977). Report of a working group on ecological research on seabirds. *NERC Publ.*, Ser. C, No. 18. London: Natural Environment Research Council.

Notini, M. (1978). Long-term effects of an oil spill on *Fucus* macrofauna in a small Baltic bay. *J. Fish. Res. Board Can.*, 35, 745–53.

Notini, M. (1980). Impact of oil on the littoral ecosystem. In *The* Tsesis *Oil Spill* (eds. J. J. Kineman, R. Elmgran and S. Hansson), 129–45. Washington, D.C.: U.S. Dept. of Commerce (NOAA).

NRC [Natl. Research Council] (1985). *Oil in the Sea: Inputs, Fates and Effects.* Washington, D.C.: National Academy Press, 601 pp.

Olive, P. J. W., Garwood, P. R., Bentley, M. G. and Wright, N. (1981). Reproductive success, relative abundance and population structure of two species of *Nephtys* in an esturine beach. *Mar. Biol.*, 63, 189–96.

Oritsland, N. A., Engelhardt, F. R., Juck, F. A., Hurst, R. and Watts, P. O. (1981). Effect of crude oil on polar bears. Department of Indian Affairs and Northern Development Canada, Publ. QS-8283-020—EE-AI. Canada Catalog No. R71-19/24-1981E, 268 pp.

Owens, E. H. (1978). Mechanical dispersal of oil stranded in the littoral zone. *J. Fish. Res. Board Can.*, 35, 563–72.

Owens, E. H. (1984). Variability in estimates of oil contamination in the intertidal zone of a gravel beach. *Mar. Pollut. Bull.*, 15, 412–16.

Owens, E. H. (1985). Factors affecting the persistence of stranded oil on low energy coasts. In *Proceedings of the 1985 Oil Spill Conference*, 359–65. API Publ. No. 4385. Washington, D.C.: American Petroleum Institute.

Owens, E. H. (1987). Estimating and quantifying oil contamination on the shoreline. *Mar. Pollut. Bull.*, 18, 110–18.

Owens, E. H., Foget, C. R. and Robson, W. (1982). Arctic shoreline countermeasures—initial results and implications from the BIOS experiment. In *Proceedings of the 5th AMOP* [Arctic Marine Oilspill Program] *Technical Seminar*, 349–62. Ontario: Environmental Protection Service.

Owens, E. H., Foget, C. R. and Robson, W. (1984). Experimental use of dispersants for spill countermeasures on Arctic beaches. In *Oil Spill Chemical Dispersants: Research, Experience and Recommendations* (ed. T. E. Allen), 324–27. Philadelphia: American Society for Testing and Materials.

Owens, E. H., Humphrey, B. and Sergy, G. A. (1987). The migration of stranded oil from a sheltered beach into adjacent nearshore subtidal sediments. In *Fate and Effects of Oil in Marine Ecosystems* (eds. J. Kuiper and W. J. Van den Brink), 231–41. Dordrecht: Martinus Nijhoff.

Owens, E. H., Humphrey, B., Hope, D., Robson, W. and Harper, J. R. (1987). The fate of stranded oil four years after an experimental spill on a sheltered gravel beach. In *Proceedings of the 1987 Oil Spill Conference*, 473–78. API Publ. No. 4452. Washington, D.C.: American Petroleum Institute.

Owens, E. H., Robson, W. and Humphrey, B. (1987a). Observations from a site visit to the *Metula* spill twelve years after the incident. *Spill Technology Newsletter*, 12, 83–96.

Owens, E. H., Robson, W. and Humphrey, B. (1987b). Observations and initial results from a site visit to the *Metula* spill 12-1/2 years after the incident. In *Proceedings of the Tenth Annual Arctic Marine Oil Spill Program Technical Seminar*, 21–41. Ottawa: Environmental Protection Service.

Page, D. S., Foster, J. C., Fickett, P. M. and Gilfillan, E. S. (1988). Identification of petroleum sources in an area impacted by the *Amoco Cadiz* oil spill. *Mar. Pollut. Bull.*, 19, 107–15.

Page, D. S., Foster, J. C., Fickett, P. M. and Gilfillan, E. S. (1989). Long-term weathering of *Amoco Cadiz* oil in soft intertidal sediments. In *Proceedings of the 1989 Oil Spill Conference*, 401–05. API Publ. No. 4479. Washington, D.C.: American Petroleum Institute.

Page, D. S., Foster, J. C. and Gilfillan, E. S. (1987). Kinetics of aromatic hydrocarbon depuration by oysters impacted by the *Amoco Cadiz* oil spill. In *Fate and Effects of Oil in Marine Ecosystems* (eds. J. Kuiper and W. J. Van den Brink), 243–51. Dordrecht: Martinus Nijhoff.

Page, D. S., Foster, J. C., Hotham, J. R., Pendergast, E., Hebert, S., Gonzalez, L., Gilfillan, E. S., Hanson, S. A., Gerber, R. P. and Vallas, D. (1983). Long-term fate of dispersed and undispersed crude oil in two nearshore test spills. In *Proceedings of the 1983 Oil Spill Conference*, 465–71. API Publ. No. 4356. Washington, D.C.: American Petroleum Institute.

Parker, C. A., Freegarde, M. and Hatchard, C. G. (1971). The effect of some chemical and biological factors on the degradation of crude oil at sea. In *Water Pollution by Oil* (ed. P. Hepple), 237–44. London: Institute of Petroleum (Amsterdam: Elsevier).

Parsons, J., Spry, J. and Austin, T. (1980). Preliminary observations on the effect of Bunker C fuel oil on seals on the Scotian shelf. In *Scientific Studies During the Kurdistan Tanker Incident: Proceedings of a Workshop* (ed. J. H. Vandermeulen), 193–202. Report Series BI-R-80-3. Dartmouth, Nova Scotia: Bedford Institute of Oceanography.

PAU [Programme Analysis Unit] (1973). *The Environmental and Financial Consequences of Oil Pollution from Ships*. UK submission to the IMCO International Marine Pollution Conference, 1973. Harwell: UKAE.

Payne, J. R., Flynn, N. W., Mankiewicz, P. J. and Smith, G. S. (1980). Surface evaporation/dissolution partitioning of lower-molecular-weight aromatic hydrocarbons in a down-plume transect from the Ixtoc I wellhead. In *Proceedings of a Symposium, Preliminary Results from the September 1979 Researcher/Pierce Ixtoc I Cruise*. Washington, D.C.: Office of Marine Pollution Assessment (NOAA).

Payne, J. R., McNabb, G. D., Jr., Hachmeister, L. E., Kirkstein, B. E., and Clayton, J. R. (1987). Development of a predictive model for the weathering of oil in the presence of sea ice. Final report prepared for NOAA, Outer Continental Shelf Environment Assessment Program. Washington, D.C.: U. S. Dept. of Commerce, 314 pp.

Pearson, T. H. and Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceangr. Mar. Biol. Ann. Rev.*, 16, 299–311.

Percy, J. A. (1976). Responses of Arctic marine crustaceans to crude oil and oil tainted food. *Env. Pollut.*, 10, 155–62.

Percy, J. A. (1977). Responses of Arctic marine benthic crustaceans to sediments contaminated with crude oil. *Env. Pollut.* 13, 1–10.

Percy, J. A. (1981). Benthic and intertidal organisms. Oil and Dispersants in Canadian Seas. Research Appraisal and Recommendations (eds. J. B. Sprague, J. H. Vandermeulen and P. G. Wells), 87–104. *Economic and Technical Review Report,* EPS-3-EC-82-2. Ottawa: Environment Canada.

Petersen, H. K. (1978). Fate and effect of Bunker C oil spilled by the *USNS Potomac* in Melville Bay, Greenland, 1977. In *Proceedings, Conference on Assessment of Ecological Impacts of Oil Spills*, 332–43. Washington, D.C.: American Institute of Biological Sciences.

Picard-Berube, M. and Cossa, D. (1983). Teneurs en benzo 3/4-pyrene chez *Mytilus edulis* L. de l'estuaire et du golfe du Saint-Laurent. *Mar. Env. Res.*, 10, 63–71.

Pratt, S. D. (1978). Interactions between petroleum and benthic fauna at the *Argo Merchant* spill site. In *In the Wake of the* Argo Merchant (eds. M. P. Wilson, J. G. Quinn and K. Sherman), 131–36. Center for Ocean Management Studies. Kingston: Univ. of Rhode Island.

Prouse, N. J. and Gordon, D. C. (1976). Interaction between the deposit feeding polychaete *Arenicola nonina* and oiled sediment. In *Sources, Effects, and Sinks of Hydrocarbons in the Aquatic Environment*, 407–22. Washington, D.C.: American Institute of Biological Sciences.

RCEP (1981). *Oil Pollution of the Sea.* Royal Commission on Environmental Pollution 8th Report. London: H. M. Stationery Office.

Ridgway, R. H. (1972). Homeostasis in the aquatic environment. In *Mammals of the Sea—Biology and Medicine* (ed. S. H. Ridgway). Springfield, Ill.: Charles C Thomas Publ.

Riley, R. G., Thomas, B. L., Anderson, J. W. and Bean, R. M. (1980). Changes in the volatile hydrocarbon content of Prudhoe Bay crude oil treated under different simulated weathering conditions. *Mar. Env. Res.*, 4, 109–19.

Rowland, S. J. and Volkman, J. K. (1982). Biogenic and pollutant aliphatic hydrocarbons in *Mytilus edulis* from the North Sea. *Mar. Env. Res.*, 7, 117–30.

RSPB (1979). *Marine Oil Pollution and Birds.* Sandy, Beds., UK: Royal Society for the Protection of Birds.

Sabo, D. J. and Stegeman, J. J. (1977). Some metabolic effects of petroleum hydrocarbons in marine fish. In *Physiological Responses of Marine Biota to Pollutants* (eds. F. J. Vernberg, A. Calabrese, F. P. Thurberg and W. B. Vernberg), 279–87. New York: Academic Press.

Samuels, W. B. and Lanfear, K. J. (1982). Simulations of seabird damage and recovery from oil spills in the northern Gulf of Alaska. *J. Env. Mgmt.*, 15, 169–82.

Sanders, H. L. (1968). Marine benthic diversity: A comparative study. *Am. Nat.*, 102, 243–82.

Sanders, H. L. (1978). *Florida* oil spill impact on the Buzzards Bay benthic fauna, West Falmouth. *J. Fish. Res. Board Can.*, 35, 717–30.

Sanders, H. L., Grassle, J. F. and Hampson, G. R. (1972). The West Falmouth oil spill. I. Biology. *Technical Report, WHOI-72-20.* Woods Hole, Mass.: Woods Hole Oceanographic Institute, 23 pp.

Sanders, H. L., Grassle, J. F., Hampson, G. R., Morse, L. S., Price-Garner, S. and Jones, C. C. (1980). Anatomy of an oil spill: Long-term effects from the grounding of the barge *Florida* off West Falmouth, Massachusetts. *J. Mar. Res.* 38, 265–380.

Seip, K. L. (1984). The *Amoco Cadiz* oil spill—at a glance. *Mar. Pollut. Bull.*, 15, 218–20.

Sergy, G. A. (1985). The Baffin Island oil spill (BIOS) project: A summary. *Proceedings of the 1985 Oil Spill Conference*, 571–75. API Publ. No. 4385. Washington, D.C.: American Petroleum Institute.

Shiaris, M. P. and Jambard-Sweet, D. (1986). Polycyclic aromatic hydrocarbons in surficial sediments of Boston Harbour, Massachusetts, USA. *Mar. Pollut. Bull.*, 17, 469–72.

Simpson, J. G. and W. G. Gilmartin (1970). An investigation of elephant seal and sea lion mortality on San Miguel Island. *Bioscience*, 20, 289.

Sindermann, C. J. (1982). Implications of oil pollution in production of disease in marine organisms. *Phil. Trans. R. Soc. Lond.*, B, 297, 385–99.

Smith, J. E. (1970). Torrey Canyon *Pollution and Marine Life*. London: Cambridge Univ. Press.

Smith, T. G. and Geraci, J. R. (1975). The effect of contact and ingestion of crude oil on ringed seals of the Beaufort Sea. Beaufort Sea Project Technical Report 5. Victoria, B.C.: Department of the Environment, 67 pp.

Southern, H. N., Carrick, R. and Potter, W. G. (1965). The natural history of a population of guillemots (*Uria aalge*). *J. Anim. Ecol.*, 34, 649–65.

Southward, A. J. (1982). An ecologist's view of the implications of the observed physiological and biochemical effects of petroleum compounds on marine organisms and ecosystems. *Phil. Trans. R. Soc. Lond.*, B, 297, 241–55.

Southward, A. J. and Southward, E. C. (1978). Recolonization of rocky shores in Cornwall after use of toxic dispersants to clean up the *Torrey Canyon* spill. *J. Fish. Res. Board Can.*, 35, 682–706.

Spies, R. B. and Davis, P. H. (1979). The infaunal benthos of a natural oil seep in the Santa Barbara Channel. *Mar. Biol.* 50, 227–37.

Spies, R. B., Davis, P. H. and Stuermer, D. H. (1980). Ecology of a submarine petroleum seep off the California coast. In *Environmental Pollution:* I. Hydrocarbons (ed. R. Geger), 208–63. Amsterdam: Elsevier.

Spies, R. B. and Des Marais, D. J. (1983). Natural isotope study of trophic enrichment of marine benthic communities by petroleum seepage. *Mar. Biol.*, 73, 67–71.

Spies, R. B., Hardin, D. D. and Toal, J. P. (1988). Organic enrichment or toxicity? A comparison of the effects of kelp and crude oil in sediments on the colonization and growth of benthic fauna. *J. Exp. Mar. Ecol.*, 124, 261–82.

Spooner, M. F. (1967). Biological effects of the *Torrey Canyon* disaster. *J. Devon Trust Nat. Conserv. Suppl. 1967*, 12–19.

Staveland, J. T. (1979). Effects on hatching in *Littorina littorea* after an oil spill. *Mar. Pollut. Bull.*, 10, 255–58.

Steele, J. H. (1979). The uses of experimental ecosystems. *Phil. Trans. R. Soc. Lond.*, B, 286, 583–95.

Stewart, J. E. and Marks, L. J. (1978). Distribution and abundance of hydrocarbon-utilising bacteria in sediments of Chedabucto Bay, Nova Scotia, in 1976. *J. Fish. Res. Board Can.*, 35, 581–84.

Strain, P. M. (1986). The persistence and mobility of a light crude oil in a sandy beach. *Mar. Env. Res.*, 19, 49–76.

Straughan, D. (1972). Biological effects of oil pollution in the Santa Barbara Channel. In *Marine Pollution and Sea Life* (ed. M. Ruivo), 355–59. London: Fishing News (Books) Ltd.

Straughan, D. (1976). Sublethal effects of natural chronic exposure to petroleum in the marine environment. API Publ. No. 4280. Washington, D.C.: American Petroleum Institute, 120 pp.

Straughan, D. (1978). Biological studies of the *Metula* oil spill. In *Proceedings of the Conference on Assessment of Ecological Impacts of Oil Spills,* 365–77. Washington, D.C.: American Institute of Biological Sciences.

Straughan, D. (1979). Variability in chemical exposure of marine organisms to petroleum. In *Symposium on Chemistry and Economics of Ocean Resources*, 467–75. Publ. of Chemical Marketing and Economics Div., American Chemical Society, Staten Island, NY.

Sveum, P. and Sendstad, E. (1985). Oil polluted seaweeds in the Arctic. In *Proceedings of the 8th AMOP* [Arctic Marine Oilspill Program] *Technical Seminar*, 59–77. Ottawa: Environmental Protection Service.

Thomas, M. L. H. (1978). Comparison of oiled and unoiled intertidal communities in Chedabucto Bay, Nova Scotia. *J. Fish. Res. Board Can.* 35, 707–16.

Thorson, G. (1950). Reproductive and larval ecology of marine bottom invertebrates. *Biol. Rev.*, 25, 1–45.

Thorson, G. (1957). Bottom communities (sub-littoral and shallow shelf). *Mem. Geol. Soc. Am.*, 67, 461–534.

Tilseth, S., Solberg, T. S. and Westrheim, K. (1981). Sublethal effects of the water-soluble fraction of Ekofisk crude oil on the early larval stages of cod. CM 1981/E:52. Copenhagen: International Council for the Exploration of the Sea.

Tsouk, E., Amir, S. and Goldsmith, V. (1985). Natural self-cleaning of oil polluted beaches by waves. *Mar. Pollut. Bull.*, 16, 11–19.

Tuck, L. M. (1961). *The Murres.* Ottawa: Canadian Wildlife Service, 260 pp.

Vandermeulen, J. H. (1982). Some conclusions regarding long-term biological effects of some major oil spills. *Phil. Trans. R. Soc. Lond.*, B, 297, 335–51.

Vandermeulen, J. H., Keizer, P. D. and Penrose, W. R. (1977). Persistence of non-alkane components of Bunker C oil in beach sediments of Chedabucto Bay, and lack of their metabolism by molluscs. In *Proceedings of the 1977 Oil Spill Conference*, 469–73. API Rept. No. 4284. Washington, D.C.: American Petroleum Institute.

Vandermeulen, J. H., Long, B. F. N. and Ahern, T. P. (1981). Bioavailability of stranded *Amoco Cadiz* oil as a function of environmental self-cleaning: April 1978–January 1979. In *Proceedings of the International Symposium:* Amoco Cadiz *Fates and Effects of the Oil Spill*, 585–97. Paris: Le Centre National pour l'Exploitation des Oceans.

Walker, J. D., Cofone, L., Jr. and Cooney, J. J. (1973). Microbial petroleum degradation: The role of *Cladosporium resinae*. In *Proceedings of the 1973 Conference on the Prevention and Control of Oil Spills*, 821–25. Washington, D.C.: American Petroleum Institute.

Wallhauser, K. H. (1967). Olabbauende mikroorganismen in natur und tecknik. *Helgolander wiss. Meeresunters*, 16, 320–35.

Warner, R. F. (1969). Experimental effects of oil pollution in Canada. An evaluation of problems and research needs. Miscellaneous Report 645. Ottawa: Canadian Wildlife Service.

Wells, P. G. and Percy, J. A. (1985). Effects of oil on Arctic invertebrates. In *Petroleum Effects in the Arctic Environment* (ed. F. R. Engelhardt). London and New York: Elsevier Applied Science Publishers.

Westwood, S. S. C., Dunnet, G. M. and Hiscock, K. (1989). Monitoring the Sullom Voe terminal. In *Ecological Impacts of the Oil Industry* (ed. B. Dicks), 261–85. Chichester and New York: John Wiley and Sons.

Whittle, K. J. (1978). Tainting of marine fish and shellfish with reference to the Mediterranean Sea. In *Data Profiles for Chemicals for the Evaluation of their Hazards to the Environment of the Mediterranean Sea*, 3, 89–108. Geneva: U.N.: Environment Programme.

Whittle, K. J., Hardy, R., Mackie, P. R. and McGill, A. S. (1982). A quantitative assessment of the sources and fate of petroleum compounds in the marine environment. *Phil. Trans. R. Soc. Lond.*, B, 297, 193–218.

Widdows, J., Donkin, P., Salkeld, P. N. and Evans, S. V. (1987). Measurement of scope for growth and tissue hydrocarbon concentrations of mussels (*Mytilus edulis*) at sites in the vicinity of the Sullom Voe oil terminal—a case study. In *Fate and Effects of Oil in Marine Ecosystems* (eds. J. Kuiper and W. J. Van den Brink), 269–77. Dordrecht: Martinus Nijhoff.

Williams, T. D. (1978). Chemical immobilisation, baseline hematological parameters and oil contamination in the sea otter. Rep. No. MMC-77/06, Marine Mammal Commission, 27 pp.

Wilson, D. G. and Mackay, D. (1986). The behaviour of oil in freezing situations. In *Proceedings of the Ninth Arctic Marine Oil Spill Program, Technical Seminar 1986*, 51–63. Ottawa: Environment Canada.

Wilson, D., Poon, Y. S. and Mackay, D. (1985). An exploratory study of the buoyancy behaviour of weathered oil in water. Ottawa: Environment Canada, EE Series Report.

Wise, S. A., Chester, S. N., Guenther, F. R., Hertz, H. S., Hilpert, L. P., May, W. E. and Parris, R. M. (1980). Interlaboratory comparison of determinations of trace level hydrocarbons in mussels. *Analyt. Chem.*, 52, 1828–33.

Wolfe, D. A. (1987). Interactions of spilled oil with suspended materials and sediments in aquatic systems. In *Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems* (eds. K. L. Dickson, A. W. Maki and W. A. Brungs), 299–316. New York and Oxford: Pergamon Press.

APPENDIX A

COLD WATER ECOSYSTEMS

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

PHYSICAL CONDITIONS

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

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

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

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

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

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

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

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

(from Ekman, 1967)

FAUNA AND FLORA

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

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

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

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

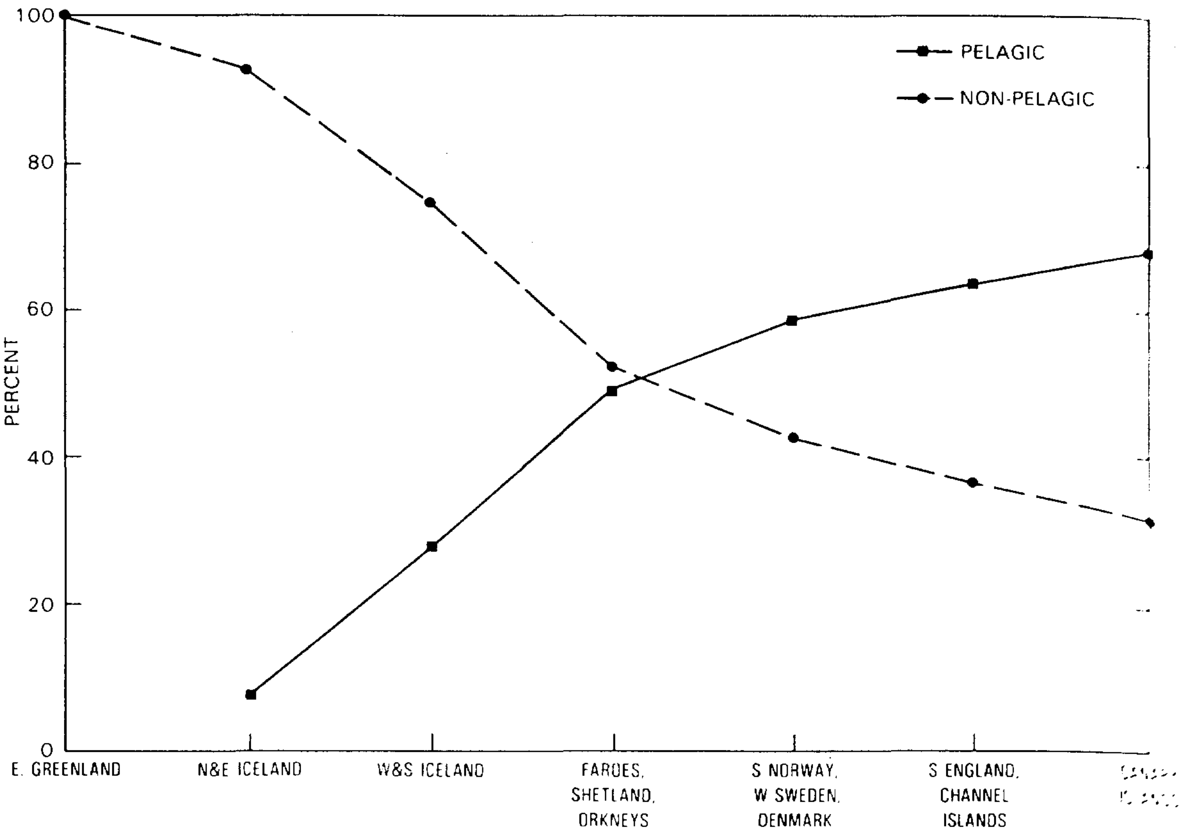

Figure A1: The relations between a pelagic/non-pelagic larval development in prosobranchs throughout the North Atlantic (Thorson, 1950).

APPENDIX B

PHYSICAL AND CHEMICAL BEHAVIOUR OF SPILLED OIL

BEHAVIOUR AT SEA

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

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

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

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

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

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

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

Spreading

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

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

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

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

Evaporation

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

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

The rate of evaporation is dependent upon

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

- Wind speed
- Sea state conditions
- Oil composition

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

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

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

Sinking

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

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

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

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

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

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

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

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

Dissolution

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

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

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

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

Natural Dispersion Into Water Column

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

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

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

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

Emulsification

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

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

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

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

Biodegradation and Photolysis

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

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

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

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

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

BEHAVIOUR IN ICE

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

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

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

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

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

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

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

BEHAVIOUR NEAR SHORES

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

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

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

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

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

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

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

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

- Retention of oil in the sediments was proportional to the loading thickness, up to a limit beyond which the retention increased very little. This limiting thickness for oil loading was thought to be a function of the sediment size.
- The retention was proportional to the sediment emergence time. Intertidal sediments which are exposed longer have greater oil retention times.
- The retention was inversely proportional to the mud content of the sediments. Even small amounts of mud may significantly reduce oil retention in sediments.
- Sediment size and composition were thought to be the most important factors which influence oil retention.

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

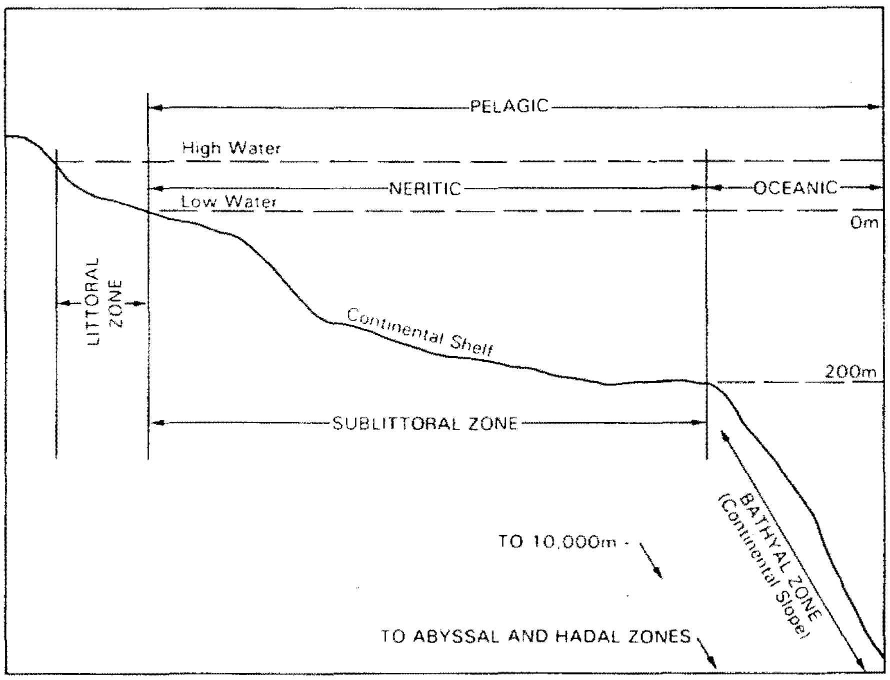

Figure B1: Divisions of the marine environment (modified after Hedgpeth, 1957).

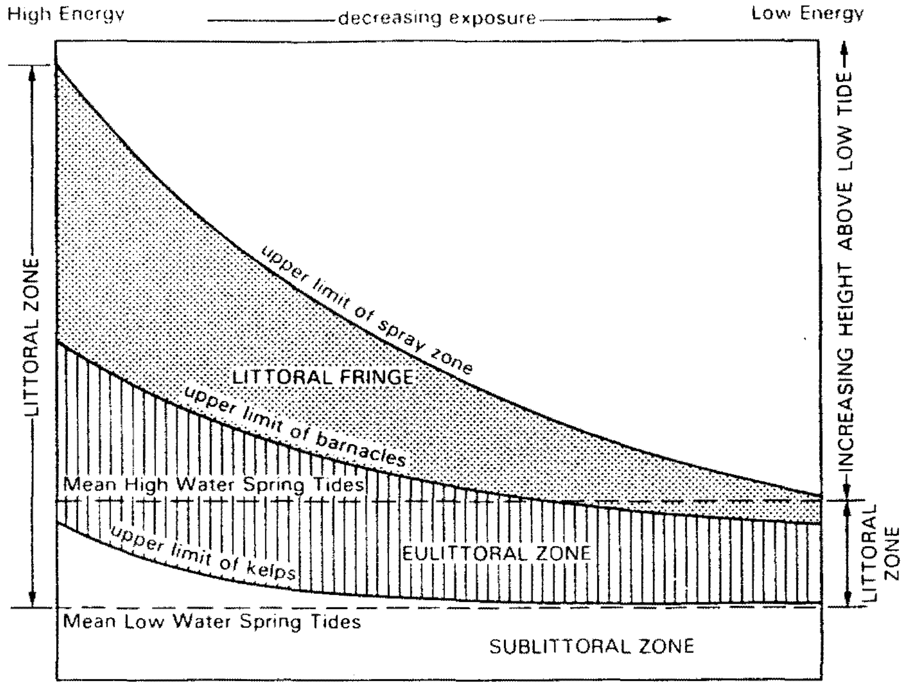

Figure B2: Effect of exposure on the width of the littoral zone of a rocky shore (modified after Lewis, 1964).

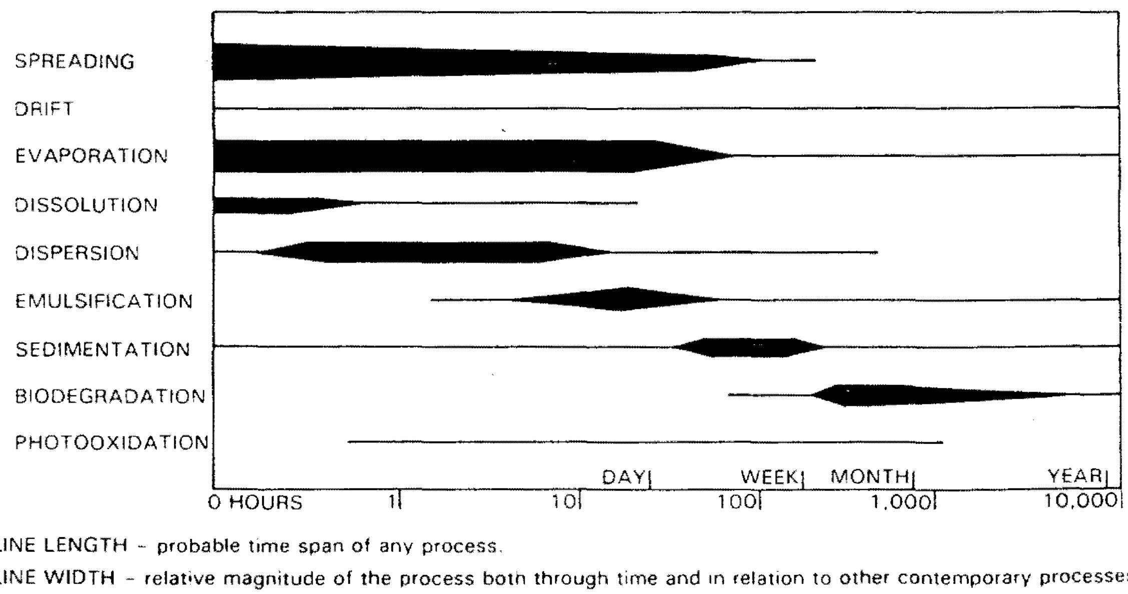

Figure B3: Time span and relative magnitude of processes acting on spilled oil (modified after Whittle *et al.*, 1982).

APPENDIX C

EFFECTS OF OIL ON BIRDS

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

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

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

Birds at Risk

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

Alcids (auks)

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

Diving Sea Ducks

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

Grebes and Loons

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

Pelican, Gannet, Tern

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

Other Birds

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

BIOGRAPHICAL SKETCHES OF THE AUTHORS

DR. JENIFER M. BAKER

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

DR. ROBERT B. CLARK

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

DR. ROWLEY H. JENKINS

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

DR. PAUL F. KINGSTON

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

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

GLOSSARY

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

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

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

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

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

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