Assessment of Injuries to Prince William Sound Killer Whales

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Killer whales, *Orcinus orca*, occur in all oceans of the world. Population estimates, based on photo-identification studies, are available for the inland waters of Washington State, British Columbia, southeast Alaska, and Prince William Sound. Each area contains approximately 100 to 250 whales.

Two forms of killer whales co-exist in Prince William Sound which are known as residents and transients (Heise et al., 1992). The resident type is characterized by forming matrilineal groups within the pod. Resident pods have consistent membership overtime and have low birth and death rates (Olesiuk et al., 1990).

Birth rates are based upon the observation of new calves within a pod. Mortality rates, however, are based on the lack of observation of a known individual within a pod. The social structure of resident killer whales is such that an animal not observed for more than one year is considered dead. The social organization of transient killer whales is not well understood and the dynamic nature of these pods makes determination of their rates of mortality difficult.

The reported loss of 14 individual whales from the resident AB pod (which numbered 36 whales in 1988) for the years 1989 through 1991 (Matkin et al., this symposium) is unprecedented. Several possible explanations for the missing whales were examined.

The missing 14 animals could have been an artifact of the survey protocol. This problem was evaluated by looking at the potential for error in the photo-identification process and the bias in survey coverage. The number of animals present in Prince William Sound pods during summer surveys in 1989-91 was obtained through detailed examination of the photographic database of individual animals. Presence or absence of members of each pod were evaluated by comparing photographs taken during the 3-year study period to previous years.

Results of the comparisons verified the absence of 14 whales in AB pod. To evaluate whether or not a mistake was made during the identification process (for example, was a whale present but mis-identified) four independent tests were conducted. Animals were recorded as being present or absent each time the pod was encountered. The results showed that earlier identifications were correct and that 14 whales were in fact missing.

Another possible bias that could have resulted in the 14 whales not being seen and photographed was the amount of effort put forth to establish presence or absence of individuals in the pod. The overall effort (miles surveyed) conducted during 1989-1991 resulted in the greatest amount of effort to date in Prince William Sound. The number of times each pod was seen in 1989, 1990, and 1991 seasons exceeded that reported for earlier studies. The amount of effort and the number of times each pod was encountered was more than adequate for locating and identifying the presence of individual animals.

We next considered the possibility that individual whales may have moved out of the Prince William Sound area and were not available to be photographed during these studies. Although considerable searching effort took place in southeast Alaska, the missing whales were not encountered. Unfortunately, minimal effort was expended near Kodiak Island and the waters adjacent to Prince William Sound to locate the missing whales during the 1989, 1990, and 1991 seasons.

However, in 1992 researchers from the National Marine Mammal Laboratory conducted killer whale photo-identification studies from Kodiak Island to Seward, Alaska. The AB pod was not seen during these investigations. An examination of the 20-year killer whale database from British Columbia and Puget Sound, Washington, indicated that no resident killer whale consistently missing during repeated encounters had ever returned to its pod or appeared in another pod.

The possibility that the missing whales have moved out of the area is not supported by our knowledge of the social structure and behavior of resident killer whales. Based upon the historical life history information, it is likely that the missing resident whales are dead and have not moved off to other areas.

However, a perturbation as severe as the *Exxon Valdez* oil spill and its direct impact on cetaceans has never before been investigated. It is therefore possible that a major catastrophe such as the *Exxon Valdez* oil spill could have affected killer whales in ways never described before. This possibility, although highly unlikely, should not be disregarded.

The most reasonable explanation for the disappearance of the 14 whales is that they are dead. However, the cause(s) of their death remain unclear. Natural mortality is certainly plausible, but unlikely. This species is characterized by a low birth and death rate (less than 2.2% per year or less; Olesiuk et al., 1990). The mortality rate for AB pod calculated for the 1989 season with the loss of seven whales was 19.4%. Six additional whales were reported missing from AB pod resulting in a 20.7% mortality rate for the 1989/90 season. In 1991, one more whale was noted as missing from AB pod (mortality rate of 4.3%).

These rates for the 1989 and 1990 season are significantly higher than would be expected from natural causes. It is unlikely that natural mortality would account for more than 1-3 animals, and not the loss of 14 whales over a 3-year period as observed.

Examination of other causes to explain the mortality of the 14 missing whales are complicated by the past history of AB pod. This pod was involved in interactions with the Prince William Sound sablefish longline fishery in the mid 1980's. In 1985, we received reports of killer whales being shot at by fishermen. Several of the animals showed evidence of bullet wounds. In 1985, three whales were reported missing. In 1986, three additional whales were gone. In 1987 and 1988, this pod lost two more individuals. The loss of at least some of these 8 whales was attributed to shooting (although never confirmed). These whales were not seen again after the year they were first identified as missing.

It is possible that the 14 whales reported missing during the 1989 through 1991 season could have been shot. However, this is unlikely because (1) longline fishing was closed between the time when all whales were accounted for (September 1988) and the time when the first seven whales were first determined missing (March 1989), (2) there were no reports of shootings and, (3) no new bullet wounds have been observed on individuals of AB pod since 1986.

The remaining cause of death considered was the effect of the oil spill. Six different killer whale pods were observed transiting directly through oil (light sheen) but only AB pod suffered losses. The loss of the first seven animals from AB pod could have been through direct contact with the oil, such as from inhalation of toxic volatile gases or ingestion. The loss of the six additional whales one year later is more difficult to explain from oil effects, but might have been associated with residual effects or from indirect effects (e.g., eating contaminated prey).

None of the missing whales were found stranded, although killer whales typically sink upon death. Four carcasses (only one whale could be identified and it was not from AB pod) were found during the three-year period (1989-1991). This stranding rate is high compared to other geographical areas, and from previous stranding rates from the Prince William Sound region. However, this may simply have been an artifact of increased effort after the spill. Blubber samples and scrapings from the stomach lining from the stranded whales were analyzed for hydrocarbons. There was no indication of oil contamination in these tissues and cause of death could not be determined. Caution, however, must be used when interpreting these results since the carcasses were old when found and decomposition decreases the viability of the tissue samples for hydrocarbon analysis.

In conclusion, the cause(s) of the deaths of 14 killer whales from AB pod is unknown. We are confident that (1) whales have not been mis-identified, (2) adequate effort was made in Prince William to locate the missing animals, and (3) the number of encounters was sufficient to evaluate the presence or absence of an individual whale. The current life history information available on killer whales precludes the possibility that the whales moved elsewhere.

Therefore, we assume that the whales are dead from either, or a combination of, natural causes; a result of interactions with fisheries; or, for the *Exxon Valdez* oil spill. The highest mortality rate ever reported in the literature for North Pacific resident killer whales occurred in 1989 and 1990, coinciding with the *Exxon Valdez* oil spill. There is a strong correlation between the loss of the 14 whales and the *Exxon Valdez* oil spill, but there is no clear cause and effect relationship.

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Humpback Whale Abundance and Distribution in Prince William Sound

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Humpback whales, *Megaptera novaeangliae*, number about 10,000 animals worldwide, of which perhaps 1,500 occur in the North Pacific (Baker and Herman, 1987). They are currently listed as an endangered species under the U. S. Endangered Species Act of 1973. During winter the North Pacific population occurs principally off Hawaii and Mexico. During summer they range widely across the North Pacific with concentrations off Kodiak Island, in Prince William Sound and in southeast Alaska. Prince William Sound is considered a major feeding area for the North Pacific stock of humpback whales.

Identification of individual whales is possible through use of ventral fluke coloration patterns and natural marks. Researchers conducting photo-identification studies in Prince William Sound and southeast Alaska had collected a substantial photographic database on humpback whales prior to the spill.

Based on this historical database, at least 50 individual animals were known to occur annually in Prince William Sound and approximately 300 whales from southeast Alaska. Not all identified whales are seen each year in each area, however, site fidelity has been reported. Whales seen in Prince William Sound are generally not seen elsewhere, and whales seen in southeast Alaska are not seen in Prince William Sound. Over the years, however, three whales have been seen in both Prince William Sound and southeast Alaska.

The objectives of this study were to (1) count and individually identify humpback whales in Prince William Sound and southeast Alaska; (2) test the hypothesis that humpback whale distribution and abundance within Prince William Sound and adjacent waters is similar to that reported for years prior to 1989; and (3) test the hypothesis that humpback whale natality and mortality in Prince William Sound has not changed since the oil spill.

During 1989 and 1990, photographs of individual humpback whales occurring in Prince William Sound were collected from May to September each year to address objectives one and three. To determine if significant changes occurred in whale distribution (objective two), concurrent photo-identification work was carried out in 1989 in southeast Alaska.

Photographic analysis of Prince William Sound humpbacks documented 59 individual whales during the 1989 season. In southeast Alaska, 552 individual whales were identified. During 1990, 66 individual whales were documented in Prince William Sound. The 1989 and 1990 counts represent the largest number of humpback whales ever photographed in Prince William Sound. The effect of increased observer effort during these two seasons, however, must be considered. In particular, the high levels of effort during these two years have clearly increased the number of individual humpback whales identified relative to numbers prior to 1989.

The distribution of whales in Prince William Sound during the 1989 season was compared to the known distribution prior to the oil spill. Distributional data prior to 1989 noted concentrations of whales in the Lower Knight Island Passage area. In 1989, fewer whales were observed in this area. The reasons for this are unclear, but may be related to increased vessel and aircraft traffic in1989, or may simply be caused from natural fluctuations in prey. In 1990, whales were again abundant in the Lower Knight Island Passage area, similar to thier distribuiton prior to 1989. Despite considerable effort in 1989, humpback whales known to occur in Prince William Sound were not observed in southeast Alaska.

No apparent shift in distribution within Prince William Sound was noted in 1990; whales were again abundant in the Lower Knight Island passage area, similar to their distribution in 1988. Despite considerable effort in 1989, humpback whales known to occur in Prince William Sound were not observed in southeast Alaska.

The combined average annual reproductive rate for 1980 through 1988 for Prince William Sound humpback whales was 9.4%. In 1989, the reproductive rate was 6.3%; in 1990 it was 10.8%. Seven out of the eight females present with calves in 1990 had been photographed in Prince William Sound in 1989, and thus these whales were pregnant at the time of the spill. No reports of dead stranded humpback whales occurred within Alaskan waters during this two-year period.

From the available data, it does not appear that the *Exxon Valdez* oil spill had any measureable impact on the North Pacific humpback whale population.

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Occupational Exposures from Oil Mist During the *Exxon Valdez* Spill Cleanup

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Crude oil cleanup during the *Exxon Valdez* spill relied heavily on high pressure water and steam (Exxon, 1989) which generated an oil mist. Monitoring records document an average oil mist exposure 12 times in excess of permissible exposure limits.

The National Institute for Occupational Safety and Health (NIOSH) reported 1,811 worker's compensation claims in 1989 related to the *Exxon Valdez* oil spill (Gorman et al., 1991). The leading non-physical injury reported was respiratory system damage. Inhalation of oil mist is well recognized as a cause of occupational respiratory damage (Lancet, 1990; Robertson et al., 1988).

Prior evaluations of the 15,000 *Exxon Valdez* cleanup workers occupational exposure to airborne contaminates stated that, in general, exposures were a fraction of permissible exposure limits (PEL), (Gorman et al., 1991; Wade, 1990). However, oil mist measurements were not mentioned by Wade and NIOSH conducted limited oil mist testing. This is the first independent review of *Exxon Valdez* oil spill cleanup worker exposure records.

The objective of this study is an evaluation of the health and safety implications of using hot water and steam at high pressure and elevated temperature to clean crude-oil-contaminated beaches.

Under contract to Exxon, Med-Tox Associates collected over 6,000 air samples in 1989 from *Exxon Valdez* oil spill cleanup workers. The Alaska Health Project obtained Exxon and Med-Tox exposure data, health and safety records, and laboratory procedure manuals. This information was then compared to the literature.

The data collected from workers revealed that the average exposure exceeded the NIOSH limit by 12-fold. The maximum overexposure of 400 times the PEL was found on a "hot wash beach." Average exposures for other chemicals were below NIOSH recommended PEL. However, maximum exposures were significantly greater than NIOSH limits; that is, total PNAs 170 times greater than the limit, benzene 160 times greater, hydrogen sulfide 40 times greater, butoxyethanol eight times greater, and carbon monoxide six times greater. NIOSH limits were adjusted for the increased length of working day, as recommended by Exxon publications (see "Extended Work Days" below), but were not adhered to.

Another issue of particular concern is the fact PEL are developed on a chemical-by-chemical basis and Exxon did not take into account multiple simultaneous exposures with synergistic potential. Finally, the upper 5% of exposures, in every case listed below, exceeded NIOSH limits.

Three serious problems are evident with Exxon's laboratory procedures and data interpretation regarding oil mist monitoring records: standard reference material, applicability of PEL, and extended work days.

Standard Reference Material

The standard reference material for oil mist PEL is "mineral oil" (NIOSHa, 1990). Mineral oil is a highly purified product designed for non toxicity and freedom of irritation to humans and use in the preparation of pharmaceuticals (ASTM, 1989). Oil spill cleanup workers were exposed to Prudhoe Bay crude oil (PBCO). PBCO consists mostly of aliphatic and aromatic components and smaller amounts of heterocycles and asphaltenes. The aliphatic fraction is dominated by n-alkanes containing 11 to 40 carbon atoms and isoprenoid hydrocarbons. The aromatic components consist of a series of parent and alkylated naphthalenes, phenanthrenes, fluorenes, biphenyls, chrysenes, and benzanthracenes (Rahimtula, 1987).

In addition to hydrocarbons crude oil contains sulfur compounds such as thiols, sulfides, disulfides, and thiophenes. The higher boiling point sulfur compounds are thiocyclo-, thiobicyclo-, thiotricycloalkanes, complex thiophenes, and benzothiophenes. Basic nitrogen compounds found are pyridines and quinolines while nonbasic nitrogen compounds include pyrroles, indoles, and carbazoles. Oxygen compounds found include ketones and phenols with alkane and cycloalkane acids in the higher boiling point fraction (Costantinides and Arich, 1967).

Nickel and vanadium occur primarily as complexes such as porphyrins and over 30 metals commonly occur in crude oil. Other substances are introduced into crude oil during the process of drilling, pumping, preparing and transportation (IARC, 1989). Although mineral oil may be derived from crude oil, the refining process selectively removes a specific hydrocarbon fraction leaving most of the components mentioned above in other residues.

Based on the differences between mineral oil and PBCO two things should have been done regarding calibration standards and PEL. Laboratory equipment should have been calibrated using PBCO as the standard. A review of laboratory procedures and quality control documentation did not find evidence that PBCO was used as the standard for which to measure PBCO derived oil mist (Pristas, 1989). Substantial bias likely exists in the data because of inappropriate use of standard reference materials. The problem of inappropriate standards and infrared spectrophotometric quantification of crude oil is well documented (Baugh and Lovegreen, 1989).

Applicability of PEL

No corrections were applied to PEL for the elevated toxicity of crude oil compared to mineral oil. Crude oil is a carcinogen, neoplastigen and tumorigen when applied to the skin. Inhalation of vapor or particulates can cause aspiration pneumonia (Sax, 1989). A material safety data sheet for crude oil recommends a PEL of 0.2 mg/m^3 (Lyondell, 1990); 25 times lower than the 5.0 mg/m^3 PEL selected as relevant by Exxon. NIOSH recommends a PEL of 0.1 mg/m^3 (NIOSHb, 1990); 50 times lower than Exxon's.

Extended Work Days

Finally, the working conditions during the *Exxon Valdez* cleanup were not 8 hour days with weekends off. More typically, workers were on the job in excess of 12 hours a day, seven days a week and some for months without a break. Exxon recognized more than ten years ago that the PEL for airborne toxicants were probably inappropriate without modification for unusual work shifts. A simple linear equation was proposed by Exxon as a first step toward health and safety concerns (Exxon, 1986). However these considerations were not taken into account for the extremely long shifts of *Exxon Valdez* spill cleanup workers. If we apply Exxon's model to NIOSH oil mist PEL, the acceptable limit should be reduced by a factor of at least 2.1 (84 vs. 40 hour week). Using the PEL of 0.1 mg/m^3 and a factor of 2.1 yields a PEL of 0.05 mg/m^3.

The average worker was exposed to 12 times more oil mist than what NIOSH standards permit. Some exposures were 400 times higher than PEL. Whether or not an individual worker's health problem was caused by over exposures during the *Exxon Valdez* cleanup can only be determined on a case-by-case basis. Based on the information summarized above, further research is needed regarding medical histories of exposed workers to protect future generations when selecting cleanup technologies at other spills.

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Boat-Based Surveys of Sea Otters (*Enhydra lutris*) in Prince William Sound, Alaska.

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When the *T/V Exxon Valdez* ran aground on Bligh Reef on March 24, 1989, the resulting spill of 11 million gallons of crude oil into Prince William Sound resulted in the death and injury of more than a thousand of sea otters (*Enhydra lutris*). As part of the Natural Resources Damage Assessment effort, the U.S. Fish and Wildlife Service conducted boat-based population surveys of marine birds and sea otters in Prince William Sound between June, 1989 and July 1991. Based in part on similar surveys conducted during the summers of 1984 and 1985, the purpose of this work was to estimate post-spill sea otter abundance in order to determine initial injury to the population, and monitor continuing injury or recovery.

The study area consisted primarily of the waters of Prince William Sound, Alaska. The study area was divided into three distinct strata: shoreline, coastal, and pelagic. The shoreline stratum was based on shoreline transects surveyed by Irons, Nysewander and Trapp (1988) during the summers of 1984 and 1985, and was defined as the 200 m-wide strip immediately adjacent to the coastline. Within the Prince William Sound study area, 742 shoreline transects were defined with a total area of 822.3 km^2.

Waters outside the shoreline stratum were divided into sampling "blocks" based on a 5-minute latitude/longitude grid system. These blocks were then stratified into two categories: coastal and pelagic. The coastal stratum was comprised of those blocks that are immediately adjacent to 1 km or more of shoreline, while the pelagic stratum was comprised of those blocks that are adjacent to less than 1 km of shoreline. This classification scheme resulted in the creation of 207 coastal and 86 pelagic blocks, with total areas of 4,524 km^2 and 3,637 km^2, respectively. Within each block, a number of 200 m-wide strip transects (usually two) were systematically placed and sampled.

Watercraft used in this survey were 25' Boston Whalers, with three crew members serving equally as operator and observers. Shoreline transects were surveyed from 100 m offshore at a cruising speed of 5-10 knots. One observer scanned the water from the vessel up to and including the shoreline, while another observer scanned the water from the vessel seaward an additional 100 m. Coastal and pelagic transects were surveyed at a slightly faster cruising speed of 10-15 knots, with each observer scanning the water from the trackline of the boat outward 100 m. In addition, the watercraft operator assisted with observations of animals directly ahead of the vessel. While the vessel was in motion, all marine mammals and birds sighted were recorded on standardized data sheets.

As stated earlier, the shoreline stratum was based on a set of transects originally surveyed during the summers of 1984 and 1985 (Irons et al. 1988). Over the course of two field seasons, virtually all of the available shoreline habitats were surveyed (708 out of the possible 742 transects). These data served as the baseline for comparison with post-spill surveys.

Post-spill surveys were conducted in June, July, and August of 1989, March, June, July, and August of 1990, and March and July of 1991. Approximately three weeks were needed to complete each replicate of the survey. Post-spill surveys were initially conducted during the summer of 1989 as a random sample of approximately 25% of available shoreline transects and the original coastal and pelagic blocks. Only the shoreline stratum was sampled during June 1989. All three strata were sampled in each of the remaining surveys. Once the initial random sample of transects and blocks was chosen, each successive survey replicated the same sampling units.

Classification of sampling units as oiled or unoiled was based on Alaska Department of Environmental Conservation overflight data collected at the time of the spill (ADEC 1989). Aerial observations were used to create a GIS coverage depicting the movement of oil over the surface of the water. Since sea otters are highly mobile animals, otters inhabiting areas adjacent to the path of the oil could have encountered the slick during their normal movement patterns. Given this fact, coupled with an inherent uncertainty as to the exact geographical extent of the surface oiling, a buffer zone of 5 km was added to the ADEC overflight data coverage to represent an area within which otters might have been affected by oil. Shoreline transects, and coastal and pelagic and offshore blocks with any area located within 5 km of surface oil were classified as oiled.

Sea otter density and abundance estimates for each survey strata were calculated using ratio estimator techniques (Cochran 1977).

In the unoiled area, otter densities in the shoreline stratum increased 13.5% between the pre-spill surveys of Irons et al. (1988) and the summer 1989 surveys. Otter densities in the oiled shoreline stratum declined approximately 34.6% during the same period. Surveys conducted in the summer of 1990 show further declines in shoreline sea otter density within the oiled area. However, otter density in unoiled areas also exhibited a decline during the same period. Otter density within the oiled area did not appear to have changed between July 1990 and July 1991. With the exception of the July 1990 survey, otter densities in the oiled area were consistently lower than those in the unoiled area, which is in contrast to the pre-spill pattern.

In the unoiled area, there was considerable overlap between pre- and post-spill shoreline population estimates. There was no overlap between pre- and post-spill estimates in the oiled area for this stratum. This lack of overlap between estimates within the oiled area suggests that the post-spill population was significantly lower that the pre-spill level.

Although sea otter densities were lower in coastal and pelagic strata, given their larger total areas, these strata contained a considerable number of otters. In some instances, these strata accounted for over 50% of the total estimated population. Post-spill population estimates from this study range from a high of 8,242 (± 2,280) in July 1989 to a low of 4,399 (± 948) in March 1991.

In July 1990, it was the decision of the Management Team that data from the various damage assessment studies should be brought together in an attempt to quantify initial injury to the Prince William Sound sea otter population. In a cooperative effort, results of studies on sighting probability, carcass recovery rates, and the age structure of the recovered carcasses were synthesized with these survey data to calculate an estimate of the initial first-year injury (Garrott, Eberhardt, and Burn 1992). This exercise produced a loss estimate of approximately 2,800 sea otters for Prince William Sound (Garrott et al. 1992). Population trends in the oiled area suggest that additional losses may have occurred beyond the first year, but have not been quantified at this time.

Although estimates of shoreline sea otter density within the oiled area fell well below their pre-spill values, it is important to note that a substantial fraction of the population survived the spill and its aftermath. One reason for this may have been the presence of small bays and coves that remained relatively oil-free (Garrott et al. 1992). In the southwest portion of the Sound for example, Bainbridge Passage was heavily oiled. Yet during each of the three surveys conducted in June, July, and August 1989, we observed 30-40 otters concentrated in an apparently unoiled cove on the southern side of the Passage. These unoiled refuges were scattered throughout the spill zone, and may have provided a haven for otters.

The long-term effects of the spill on sea otters in the western portion of Prince William Sound are unknown. Two key factors that will determine those long-term effects on sea otters will be the impact of the spill on the populations of sea otter prey items (primarily mussels and clams), and continued exposure of sea otters to hydrocarbons through their prey. Either one or both of these factors could have an impact on the recovery of the sea otter population within the oiled area of the Sound.

As a means of estimating the Prince William Sound sea otter population, this survey suffered from a lack of precision. Most of this variability in the estimates came from the coastal and pelagic strata. Although 25% of the blocks within these strata were sampled, only 10% of the area within the blocks themselves were surveyed. The net result of this design was that every otter sighted in the coastal and pelagic strata equated to roughly 50 otters in the final estimate.

In order to monitor population trends with respect to recovery, I believe that the shoreline stratum is the best means of judging the status of the population. The majority of otters sighted during this survey were observed on shoreline transects, making this stratum a good index of population size. It is also the only area for which pre-spill data exist. One potential criterion for estimating when the sea otter population is fully recovered, is that point when sea otter densities within the shoreline stratum increase to pre-spill levels.

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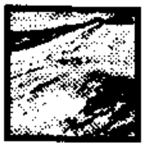
Hydrocarbons in Mussels and Subtidal Sediments: Graphical Presentation of Hydrocarbon Analysis Data With Geographic Map Data



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Oil movements on the surface were tracked during the spill, but there were large unknowns as to what the subsurface fauna were exposed to. Several studies sampled mussels and subtidal sediments for hydrocarbon analysis in an attempt to characterize the oil exposure at sites and habitats appropriate to individual projects. This project attempts to map oil exposures geographically and temporally, using samples collected by many projects.

Maps displaying the distribution and persistence of *Exxon Valdez* crude oil through presentation of sediment and of mussel tissue hydrocarbon analysis data on shoreline oiling maps will be presented, together with an interpretation of the hydrocarbon data symbols used.

A map for each project that had sediments or mussel tissues analyzed will be available for public inspection, with at least one map for each project containing all the hydrocarbon data generated for each year of the project. This poster session will focus on the data manipulation and quality assurance steps from the raw data generated by the chemistry laboratories through the GIS mapping software.

Management of Natural Resource Damage Assessment Samples and Analytical Data

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Within 48 hours after the grounding of the *Exxon Valdez*, samples were being collected by State of Alaska and Federal Trustee Agencies (Alaska Departments of Fish and Game, Environmental Conservation and Natural Resources; U.S. Department of Agriculture, Forest Service; U. S. Department of Commerce, National Oceanic and Atmospheric Administration; and U. S. Department of Interior, Fish and Wildlife Service) to document the exposure of natural resources to the spilled oil and to provide a basis for the determination of the effects of the oil. During the course of the Natural Resource Damage Assessment, over 36,000 samples of water, biota and sediment were collected from Prince William Sound and the Gulf of Alaska to meet these objectives. An additional 1,500 experimental samples were generated in various laboratory experiments.

A cooperative project between the U.S. Fish and Wildlife Services (F&WS) and the National Oceanic and Atmospheric Administration (NOAA), Natural Resources Damage Assessment (NRDA) Project Technical Services #1, was responsible for (1) archiving and tracking of these samples; (2) analysis of the samples, including selection of samples for analysis, and the development and implementation of an analytical quality assurance plan which defined criteria for the quality and acceptability of the data; and (3) management of the analytical data. A relational database (PWSOIL) was used to carry out these tasks, i.e. to maintain and manipulate all data and information related to the collection and analysis of samples for petroleum hydrocarbons.

PWSOIL is based on the design used by NOAA's National Status and Trends Program, modified for similarity to other databases either already in use or being planned by F&WS and the U.S. Environmental Protection Agency at the time of the grounding of the *Exxon Valdez*. The original design was then modified in an iterative fashion, however, to meet newly defined needs and objectives. All changes to the design of PWSOIL and to the data and information maintained by this database were preceded - and often initiated - by discussion with the users, the Project Leaders. PWSOIL is supported by hard copies of all chain-of-custody records and all developed analytical data, which themselves are kept under chain-of-custody procedures. A complete description of the design, including definitions of the variables and instructions for users, and implementation of PWSOIL is documented in Manen et al.

PWSOIL is focused on the unique identification number assigned to each sample by the database manager. Associated with this number are the data and information which describe and identify the sample, e.g. where the sample was collected (place name and latitude/longitude); who collected it; when it was collected; how it was collected; the sample identification number assigned by the collector; what kind of sample it is (sediment, water or tissue); if a tissue sample, what kind of tissue and from what plant or animal; if a sediment or water sample, the depth at which it was collected; and whether or not it has been analyzed. All of these variables are provided by the collector or Project Leader as part of the chain-of-custody record. The maintenance of these variables in PWSOIL allows the sorting of samples by location, project, species, etc.

Over 12,000 of the NRDA samples have been analyzed. The majority of the samples were analyzed for petroleum hydrocarbons (73 parameters) by GC/MS at Texas A&M University; smaller, specialized sample sets were analyzed at NOAA's Auke Bay Laboratory (ABL) and NOAA's Northwest Fisheries Center (NWFC). Semi-qualitative, non-compound specific information was developed for some sediment samples as well as the concentrations of petroleum metabolites in the bile of fish, birds, terrestrial and marine mammals by UVF, a technique to determine the presence of petroleum hydrocarbons. These analyses were performed by Texas A&M University and NOAA's NWFC. All data resulting from these analyses, as well as (1) calculated indices and parameters; pristane/phytane ratio, carbon preference index, saturated hydrocarbon weathering ratio, and sums of the alkanes, aromatics and hydrocarbons; (2) supporting data; grain size, total organic carbon, percent moisture and surrogate recoveries and (3) quality control data; the results of the analysis of blanks, standard reference materials and in-house control materials are maintained by PWSOIL in a batch fashion associated with the sample identification numbers.

The use of PWSOIL to maintain and manipulate the analytical data in program-defined reporting formats and file structures allows the use of data across projects and analytical laboratories. PWSOIL has supported data analysis for all individual NRDA Projects which collected samples for hydrocarbon analysis; forms the basis of the secondary database developed by NRDA Project Subtidal #8, and has been used by NRDA Project Technical Services #3 to develop GIS (mapping) products and by Exxon in continuing litigation. To facilitate access to this database and these data; the database (PWSOIL), Users' Manual, and supporting analytical and quality assurance documentation have been made available as a stand-alone CD-ROM (Compact Disc-Read Only Memory). This CD uses non-proprietary software developed by U.S. Geological Survey and is the result of a cooperative project between NOAA and the Geological Survey.

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Pre-spill and Post-spill Concentrations of Hydrocarbons in Sediments and Mussels in Prince William Sound

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This study provides comparison of petroleum hydrocarbon concentrations in mussels and sediments before and after the *Exxon Valdez* oil spill. Within several days of the spill in 1989, mussels (*Mytilus trossulus*) and sediments at six historically (1977-1981) established intertidal baseline sites in Prince William Sound were resampled. Additionally, six sites were established along the spill trajectory before oiling, and sampled both before and after oiling to measure changes in petroleum hydrocarbon levels in sediments and mussels. Sampling continued in 1990 and 1991.

Both mussel and sediment transects were 30 m long and usually set parallel to the water line. The sediment transect was generally down slope from the mussel bed in finer grain material. Triplicate (30 each) samples of mussels were taken along the transect line; and triplicate samples (each a composite of 10 subsamples) were collected of sediment. All samples were placed in hydrocarbon-free jars and frozen according to established protocol.

There were no detectable polynuclear aromatic hydrocarbons (PAHs) in mussels sampled prior to the *Exxon Valdez* oil spill (1977-1980) and levels in sediments at 4 of these sites were low, generally under 20 ng/g dry weight. The pattern of low hydrocarbon levels in both mussels and sediments continued after 1989 at sites not impacted by the spill.

Sleepy Bay, a heavily oiled site, had PAH concentrations in sediments nearly 100 times historical levels (established for other sites in the Sound) in May, 1989 (939±404 standard errorng/g dry weight —an amount approximating 6% *Exxon Valdez* crude oil). Sediment PAH's at this site declined to 168±17 ng/g in 1990 and 42±4 in 1991. Two other oiled sites (Bay of Isles and Elrington Island) showed increases (10-20 fold) of aromatic hydrocarbons in sediments in 1989, decreases in August, 1989, and increases in April, 1990. By August 1990, PAHs in sediments from Elrington Island had decreased to values similar to unoiled sites, but Bay of Isles sediments remained elevated over unimpacted sites. Differences in exposure to wave action probably accounts for these variations in recovery. Both Sleepy Bay and the Elrington Island site are quite exposed to wind and wave action while the Bay of Isles site at the southern tip of the South Arm is quite protected from inclement weather.

PAH's in sediments from most of the other sites, Bligh Island, Naked Island, Olsen Bay, Siwash Bay, and Perry Island, were detected at levels not elevated from historical concentrations.

Mussels from Sleepy Bay, the South Arm of the Bay of Isles, and the Fox Farm on Elrington Island all showed high PAH concentrations in 1989 (up to 143,000±13,900 ng/g dry weight). These levels had decreased to 174±27.0 - 21,700±1,500 ng/g in 1990 and 166±16.0 - 5,960±1,100 ng/g in 1991. Mussels from Naked Island and Crab Bay in Sawmill

Bay showed elevated PAH concentrations (1,950±135 - 6,990±247 ng/g) in April and May of 1989. PAH's in mussels from Olsen Bay, Bligh Island, Barnes Cove and Siwash Bay were usually detected only sporadically at concentrations near detection limits (about 10 ng/g dry weight for individual PAH's); naphthalene and substituted naphthalenes were the most frequently detected PAH's at these sites. Mussels from the latter 5 sites had between 60.0±5.00 and 243±27.0 ng/g during 1990 and 1991.

Maps created by GIS systems detailing extent and concentrations of PAHs will be displayed.

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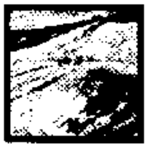

Surface Modeling of Floating Oil, The 1989 *Exxon Valdez* Oil Spill, Prince William Sound, Alaska

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Shortly after the *Exxon Valdez* ran aground on Bligh Reef in the early morning of March 24, 1989, response teams were formed to collect information on oiling, to protect the priority areas, and to begin clean up. Due to adverse weather, difficult logistics, and the size of the spill, observational information collected during this time on surface oiling was limited. Unlike shoreline oiling, surface oiling cannot be surveyed at a later time.

Through a series of modeling efforts by National Oceanic & Atmospheric Administration—Hazardous Materials Response Branch (NOAA) and the Natural Resource Damage Assessment—Technical Services 3 (TS3), surface oiling could be estimated for general purposes. NOAA has designed a trajectory hindcast model, called the On-Scene Spill Model (OSSM), which estimates the flow of oil based on wind and current patterns.

Technical Services 3 is an inter-agency group composed of geographic information systems (GIS), and technical staff from the Alaska Department of Natural Resources and the U. S. Fish and Wildlife Service. TS3 used the OSSM results in a geographic surface model, called Triangular Irregular Network (TIN). As a result of the OSSM and TIN models, a series of maps were produced illustrating the general flow over time and the relative concentration of the oil.

This poster explains and illustrates what information and advanced techniques were used to create a modeled surface oiling map. In addition, the first 2 weeks of the spill will be represented for Prince William Sound.

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Integration of Shoreline Oiling Data Sets

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Since the *Exxon Valdez* oil spill disaster of March 1989, much attention has been focused on the beaches of southcentral Alaska. Multiple federal, state, and local agencies collected information for both the response effort and the damage assessment process. Because of the number of agencies involved and the urgency of data collection, much of this information was captured using a variety of digital mapping techniques and tools. As a result, many agencies had their own digital interpretations of the coastline.

When analysis began, many of the damage assessment and restoration teams needed to know the extent of the oiling based on a specific geographic criteria, such as, land ownership (state, federal or private) or environmental sensitivity. However, before this type of spatial analysis could be done, an integrated data set containing the selection criteria and shoreline oiling needed to be created.

Data Integration

After reviewing different software packages, our findings were that shoreline data integration was difficult at best, with any software. Initially, highly interactive attribute transfer procedures were used, but they were very time consuming and subject to operator error. This type of manual processing would not suffice for an exercise of this dimension.

We looked for an automated method for producing large-scale shoreline integration across multiple data sets with available software tools. We found that in using available software, all the shoreline data sets would have to be coincident in their shape and contain segments of equal length. Even with the best control practices available, it would be difficult to ensure such extreme levels of accuracy across multiple data sets. Because of the large geographical extent, multi-agency participation, and the variation between the data sets, we found no automated process existed to provide multiple data set integration with a high degree of geographical accuracy. The only other option was to design a process that would allow large scale data set integration, while providing a high degree of accuracy.

Methodology

While direct, large scale line-on-line integration was not available, tools do exist for large scale polygonal integration, such as polygon-to-point, polygon-to-polygon, and polygon-to-line. Technical Services Study No. 3 developed a process to spatially convert selected linear (shoreline) data sets into polygon data sets so that currently available tools could be used for data set integration.

The specialized software that was developed used a three-fold process to convert the shoreline data set into a polygon data set. First, a raw polygon data set was created whose width was tailored to coincide with the shoreline data set and reflect the differences in spatial resolution between the various shoreline data sets that were to be integrated. Second, adjacent polygon boundaries were created to delineate the changes in the original shoreline data set. Polygon label points were also created to match the original line data set. When the shoreline attributes are transferred to the polygon label points, the polygon data set is spatially equal to the shoreline data set from which it was made (McMahon, et al 1991).

Third, once the polygon data set was completed, attributes were integrated, or transferred, onto a common shoreline using polygonal integration algorithms (ESRI 1990). The transferred attributes spatially match the original shoreline data set.

Final Verification

Plots were used for a spatial check by visually comparing both the original shoreline data set and the new integrated data set for accuracy. A quantitative analysis between the original data set and the integrated data set showed an average difference shoreline oiling totals of less than 1%. Spacial analysis showed maximum shoreline deviation of between 2 and 3 meters in less than 3% of the shoreline. Maps of integrated data sets were also reviewed for accuracy by field investigators. At the standard scale that these maps were produced, these differences would not be seen.

This process started with the integration of two major coastal themes: shoreline oiling and environmental sensitivity. Many of the NRDA investigators needed to focus specifically on areas of high environmental sensitivity. Maps and statistical summaries were produced of specific sites where habitat protection and analysis was of concern. These results were also used to provide NRDA investigators and peer reviewers with estimates of total oiled shoreline by shoreline sensitivity.

Because some government agencies focused primarily on oiling to federal or state lands only, we combined land status into the integrated data set to provide ownership damage assessment.

One of the more interesting views of the combined data set was in the form of shoreline oiling change. By looking at the difference in shoreline oiling between various survey years, changes in shoreline oiling become apparent. Some areas indicate less oiling from year to year, possibly due to mechanical and/or natural cleaning. While other beach segments show an increased amount of shoreline oiling, possibly due to the re-floating and beaching of oil between surveys. This is a very interesting look into beach dynamics.

Data Sources

The 1989 oiling data was delivered from the oil spill response staff of the Alaska Department of Environmental Conservation (ADEC 1989; 1990a; 1990b; 1990c). The spring 1990 shoreline survey (SSAT) was digitized by Exxon from multi-agency field reports. Environmental sensitivity index maps were produced by Research Planning Institute (RPI, 1979, 1983a, 1983b, 1985, 1986) for the National Oceanic and Atmospheric Administration and digitized by Environmental Services Research Institute.

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A Reconstruction of Pink Salmon Wild Stock Runs in Prince William Sound

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Assessing the effects of the *Exxon Valdez* oil spill on the Prince William Sound wild pink salmon (*Oncorhynchus gorbuscha*) fishery requires knowledge of the spatial and temporal distributions of the individual salmon stocks. This knowledge is important because geographic and economic factors dictate that the harvest occur in mixed-stock areas. The incidence of large hatchery runs co-occurring with the wild stocks in the fishery and the lack of information on individual stock contributions to the catch in these areas make the management task more difficult. Estimates of stock-specific catches are required to determine the abundance of individual stocks. Run reconstruction, known as the "poor man's stock ID," builds a history of a stock's movement through a fishery, providing stock-specific information with few data requirements.

We develop a multi-stock/multi-district run reconstruction of wild pink salmon stocks using catch, effort, tagging and escapement data to estimate stock-specific run characteristics. The reconstruction works backward beginning with the fish in their spawning streams and projecting them backward through the fishery to the time when they enter the sound. This method is preferable to a forward projection because it requires fewer assumptions about the distribution of entry to the fishery. The completed reconstruction provides a seasonal history of each stock by estimating daily stock abundances in each district, stock-specific contributions to the catch in each district, district-specific catchabilities, movement rates between districts and initial abundances of each stock.

A continuous database of daily catch and effort and weekly escapement records in Prince William Sound exists for the years between 1968 and the present. It is necessary to identify and remove hatchery contributions from the catch records. This can be accomplished for recent years with coded wire tag information. Effort is defined as one seine boat fishing for one day. Purse seine fishing gear is used to harvest pink salmon in the sound except in District 223 where a gillnet fishery co-occurs with the seine fishery. In this case the gillnet effort is standardized to seine effort units. Information from coded wire and radio tagging experiments is used to establish salmon movement and rates. To avoid undue complexity, a stock is defined as all the pink salmon spawning in the streams of a management district. For the purposes of damage assessment and because the data are more accurate, we reconstruct the 1990 and 1991 pink salmon runs.

We begin the reconstruction with the daily entry of fish into the streams assuming that a fish enters only one stream and remains there until it dies. In Prince William Sound, salmon hold at the stream mouth for a number of days before ascending. During this time they are not subject to harvest. Accordingly, the stream entry for stock is delayed before being backed into the fishery as escapement.

We model the migration of stock using a transition matrix, assuming that salmon movement in Prince William Sound is directed, stock-specific and constant over the season.

The sum of the daily stock-specific catches is the daily catch in that district. Because of the backward nature of the reconstruction, catches are modeled as an inverse survival, becoming additions to the stock abundances. This conveniently avoids the possibility of removing more fish than are available. We accumulate the salmon in pools in each district as we back them out to the Gulf of Alaska. The magnitude of these pools is a function of input from escapement, catch and movement between districts and output to the gulf. The daily movement of salmon to the fishery from the gulf is modeled as a fraction of the pool of fish in their district of entry (gateway).

A stock's initial abundance is the sum of its total escapement and its stock-specific catch contributions. The initial abundance of all wild pink salmon stocks is the sum of the stock abundances.

The parameters of the model are: district-specific catchabilities, gateway-residence times and stock-specific migration matrices. The lack of stock identification information prevents us from distinguishing between removals due to catch and removals due to migration. Thus the catchabilities and the movement rates cannot be estimated simultaneously. For this reason, the migration matrices are assumed to be known and the model estimates the catchabilities and the gateway-residence times which are constrained by total run size. The model is programmed in Fortran and parameter values are estimated by minimizing the sum of squared residuals using a nonlinear least squares routine in IMSL.

Intertidal/Supratidal Site Selection Utilizing A Geographic Information System

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A goal of the Coastal Habitat Injury Assessment (CHIA) is to quantify injuries to biological resources inhabiting the intertidal zone throughout the *Exxon Valdez* oil spill area. The CHIA was designed to provide information for the Natural Resources Damage Assessment program. A Geographic Information System (GIS) was employed by the CHIA to identify candidate study sites. A GIS was used for the following reasons: (1) the large amount of shoreline (potentially in excess of 1,244 miles; 2,002 kilometers) contaminated by the spill, (2) the extreme heterogeneity of shoreline types and degrees of oiling in the spill-affected area, (3) the need to embody the various shoreline habitats and degrees of oiling in the study, and (4) to allow for extrapolation of injury determinations to the universe of all sites in the spill-affected area.

Candidate sites were stratified into 15 habitat/oiling categories and randomly drawn from the GEO (GEO 1989) Arc/Info (ESRI) data base with probability proportional to size. Shoreline lengths of sites ranged from 100-600 meters; longer sites had a higher probability of candidate selection. Sites less than 100 m were not selected because they were judged to be too small to allow repeated annual visits with new sample plots over the period of the study. The GIS data layers utilized in GEO for site selection included: (1) the mean high water shoreline digitized from U.S. Geological Survey 1:63,360 quadrangles in the spill-affected area, (2) the Environmental Sensitivity Index (ESI) maps (Hayes & Ruby 1979; RPI 1983a, 1983b, 1985, 1986) which classified the shoreline into 19 geomorphologic types, and (3) the Oil Spill Impact (OSI) maps (ADEC 1989) which classified the shoreline into five degrees of oiling.

From a universe of 21,362 potential sites encompassing 4,950 miles (9,173 kilometers) of shoreline, 424 candidate sites were drawn and 240 sites were ground-truthed in 1989 to validate GIS-depicted geomorphologic and shoreline oiling information and to determine if they were accessible by the supratidal/intertidal survey crews. Candidate sites represented both oiled (treated) and unoiled (control) conditions.

Ground-truthing revealed 119 sites that did not fit the GIS assigned habitat/oiling categories. Discrepancies between the GIS classifications and ground-truthed classifications can be attributed to: (1) the strong dependence on aerial overflight information in the ESI and OSI data bases which was not always consistent with characteristics we observed in the intertidal zone, (2) the lag time of 2 to 4 months between the shoreline oiling data in the OSI data base and our site selection surveys, and (3) a digitizing error which misclassified 20 candidate sites in the fine-textured category. Assessable sites that did not fit the GIS assigned categories were reclassified into the observed category. The end number of sites varied for each of the categories. When the number of sites was over the target sample size for a category, the desired number was drawn at random without replacement. In the case of the sheltered estuarine category, the number of sites was under the target value.

This methodology maintained the probability based sample; however, extreme variation existed in the strata. For example, the control strata in Prince William Sound included many sites on the mainland with low salinity while oiled strata contained mostly sites on the islands with higher salinity. This problem was rectified during the winter of 1989-1990 by retaining all sites reclassified into the moderately-heavily oiled strata and pairing each with a control site based on physical characteristics and proximity to the oiled site.

Ultimately, a total of 97 study sites (50 treated and 47 controls) were selected for intensive sampling of intertidal biota by the CHIA. Sites were distributed throughout the spill-affected area, consisting of 37 sites in the Prince William Sound area, 27 sites in the Cook Inlet/ Kenai Peninsula area, and 33 sites in the Kodiak/Alaska Peninsula area. This study design maintained the probability based inferences to the effects of oil on the intertidal biota in the "adjusted universe" of accessible, moderately-heavily oiled sites greater than 100 m subject to the protocol by which paired control sites were selected.

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Identification of Marbled Murrelet Nesting Habitat in Southcentral Alaska to Guide Restoration Efforts Following the *Exxon Valdez* Oil Spill

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The *Exxon Valdez* oil spill zone is an important population center for the marbled murrelet (*Brachyramphus marmoratus*), a small diving seabird. The marbled murrelet suffered substantial injury from the spill (Kuletz, in prep), and its population in Prince William Sound has declined by 67% since surveys in the early 1970s (Laing and Klosiewski in prep).

In Washington, Oregon and California, the marbled murrelet is listed as threatened under the Endangered Species Act. In these states, marbled murrelet nests have only been found in old-growth conifers, and the only remaining populations of marbled murrelets are in waters adjacent to remnant coastal old-growth forests (Marshall 1988).

Because there is evidence that the loss of critical nesting habitat contributed to the decline of marbled murrelets at lower latitudes, protection of nesting habitat has been proposed to assist natural recovery of the murrelet population in the spill zone. However, little is known about the murrelet in Alaska, and until 1990, there had been no effort to study breeding biology of the marbled murrelet in southcentral Alaska. The only known nests at that time were six ground nests that had been found opportunistically (Day et al. 1983) and one tree nest in southeast Alaska (Quinlan and Hughes 1990).

The goal of this project was to describe upland habitat used by murrelets in the spill zone to guide acquisition of timber rights and for the development of management guidelines to assist murrelet recovery.

The objectives were to: 1) Determine marbled murrelet habitat requirements and develop criteria for documenting occupied nesting habitat within forested portions of the spill zone, and 2) Survey portions of the spill zone to investigate upland murrelet use in the full spectrum of available habitat. During 1991 and 1992, in cooperation with the U.S. Forest Service, we studied murrelet activity during the breeding season throughout western Prince William Sound and described nesting habitat at Naked Island, located in the center of Prince William Sound.

The basic method used to survey murrelets has been the 'dawn watch', whereby observers record murrelet vocalizations and flight patterns around dawn, when murrelets fly to their inland nests (Paton et al. 1990). Because this protocol had not been used in Alaska, we conducted a pilot study in 1990 (Kuletz 1991) and determined that the 'intensive dawn watch' was best suited to our field situation.

We also tested for weather effects and diel (daily) and seasonal variability by conducting intensive dawn watches at three sites above Cabin Bay, Naked Island at bi-monthly intervals from late May to mid August in 1991 and 1992. To locate nests, we used a modified dawn watch (Naslund et al. 1990), whereby multiple observers are used to pinpoint the suspected nesting site over several mornings. Nest or landing trees were climbed at the end of the season for measurements and collection of nest samples. During all dawn watches we recorded flight behavior and vocalizations to analyze *a posteriori* murrelet activity at sites with known nests, no known nests and flight corridors.

We tested murrelet activity among habitat types in two stages. In 1991, timber type data for Naked, Storey and Peak islands, available in a geographic information system (ARC INFO) from the U.S. Forest Service, were used to describe four forest types based on volume class and stand class, ranging from a low volume, small/young age class to a high volume, large/older age class. We randomly selected 80 sites among the three islands, and a single dawn watch was conducted at 73 of these sites between June and early August, 1991. Forest characteristics were also recorded for each site, and included the diameter at breast height (DBH), height and condition of the 10 nearest overstory trees. The number and type of murrelet detections were used to test for differences in murrelet activity among forest types.

In 1992, dawn watches were conducted at 68 randomly selected sites in western Prince William Sound to determine if there was a non-random distribution of murrelet activity at inland sites. Because timber type coverage was not available, these analyses will be done *a posteriori* among habitat types. The potential for murrelet occupation of non-forested sites will be evaluated with the 1992 data from western Prince William Sound.

At the regularly monitored Cabin Bay sites, peak murrelet activity occurred 1 hour before official dawn; at lower latitudes, peak activity is at official dawn. On clear days, murrelet activity began earlier, and on mornings with low clouds and fog murrelet activity continued for a longer period after dawn. A peak in murrelet detections occurred between mid and late July, and detections declined abruptly after early August. Thus, in southcentral Alaska dawn watches should begin at least 90 minutes before official dawn and surveys should be conducted from early May to early August. Because of the increase in murrelet activity in late July, survey effort should be distributed evenly throughout the summer among sampling strata, or a weighting factor applied.

On Naked Island in 1991 and 1992, we located seven active marbled murrelet nests, three trees with nest cups, and eleven trees where murrelets landed during the dawn activity period. Based on the characteristics of these 21 trees, marbled murrelets in Prince William Sound appear to nest in old-growth, moss-covered conifers. Analysis of behavioral data among occupied and unoccupied sites will refine the interpretation of behaviors observed at other locations.

Among the 1991 randomly chosen sites on Naked, Peak and Storey islands, we found a positive relationship between high volume, larger age-class forest and murrelet nesting activity. On-site measurements of overstory tree DBH for these sites was also positively correlated with the number of murrelet detections. However, not all high volume forests had murrelet activity. Factors such as slope, aspect, elevation, canopy closure, number and size of moss platforms and other tree and stand characteristics may prove to be important criteria. Thus, although we can now define potential murrelet nesting habitat, the value of a specific land parcel as murrelet nesting habitat can only be determined by a site-specific survey.

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Homing and Straying Patterns of Coded Wire Tagged Pink Salmon in Prince William Sound.

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Homing and straying patterns of coded-wire tagged pink salmon *Oncorhynchus gorbuscha* originating from four hatcheries and six streams in Prince William Sound were assessed for the 1991 return year. Prince William Sound hatcheries released 615 million pink salmon fry in 1990. Of these, 1,032,000 fry were tagged with coded-wire tags using 32 unique tag codes. Outmigrations from six study streams yielded 2,120,000 wild stock pink salmon fry of which 258,000 were tagged. Coded tags identified a fish release location, it's release or outmigration date and, for hatchery bred salmon, the rearing strategy employed prior to release.

Initially, tagged juvenile salmon were recovered during their early marine life and were used to compare growth and survival for salmon from oiled and unoiled areas of Prince William Sound. Tagged adults returning in 1991 were recovered in commercial fisheries allowing managers to assess the contribution of an individual hatchery's or stream's production to the overall commercial catch and to compare the ocean survival of salmon stocks of known oil exposure history.

While enumerating the wild pink salmon escapements to 46 streams in Prince William Sound, over 788,000 spawned out pink salmon carcasses were examined for the presence of a coded-wire tag. In addition, over 90% of the 1991 broodstock collections at all four hatcheries were similarly inspected for the presence of a coded-wire tag. One hundred and sixteen tagged pink salmon of hatchery origin were recovered in 27 of the 46 streams examined.

Straying pink salmon from Wally H. Noerenberg Hatchery (WHN) on Esther Island comprised 55% (64 tagged fish) of the total number of hatchery strays recovered and were found in 20 of 46 streams examined. Eighteen of these tagged fish from WHN were recovered from a single stream. Pink salmon from Armin F. Koernig Hatchery (AFK) in southwest Prince William Sound comprised 28% (33 tagged fish) of the total numbers of hatchery strays and were recovered in 16 streams. One AFK tagged pink salmon was recovered at Irish Creek located in eastern Prince William Sound approximately 90 miles from AFK hatchery.

Pink salmon from Solomon Gulch Hatchery (SGH) in Valdez comprised 5% (six tagged fish) of the stray hatchery tagged salmon and were recovered in three streams. Pink salmon from Cannery Creek Hatchery (CCH) comprised 11% (13 tagged fish) of the total number of hatchery strays and were recovered in 11 streams. It should be noted that a majority of the tag recovery effort was directed towards streams in the western, oil impacted areas of Prince William Sound close to where wild stock tags were applied in 1990. Therefore streams nearer to both Cannery Creek and Solomon Gulch Hatcheries did not receive comparable tag recovery effort as did streams in oil impacted areas.

In examining broodstock collections

for evidence of straying, 12 tagged pink salmon out of a total of 1241 broodstock recoveries were found to be from a hatchery other than their natal location and 3 were found to be from tagged wild stocks. No AFK tagged pink salmon were recovered at other hatcheries although one WHN tagged fish and one tagged wild fish were recovered at AFK. Three CCH, six SGH and one tagged wild fish were recovered in the broodstock collection at WHN. Conversely, two WHN tagged fish were recovered in the CCH broodstock.

At SGH, one tagged wild fish whose natal stream was over 80 miles away was recovered in the SGH broodstock. A total of 616 tagged pink salmon originating from the six wild stock tagging sites were recovered from streams. Three of the wild stock tagging sites (Loomis Creek, Herring Creek and Hayden Creek) were located in oiled locations and three (Totemoff Creek, Cathead Creek and O'Brien Creek) were in unoiled locations, all in western Prince William Sound. For fish tagged at Loomis Creek, 150 of 163 fish (92%) were recovered in their natal stream.

The remaining 13 fish were recovered at eight different streams located from 1 to 15 miles away from Loomis Creek. Of those fish tagged at Hayden Creek on LaTouche Island, 84 of 94 (89%) were recovered at their natal stream. The remaining ten fish were recovered in seven different streams located between 2 and 20 miles from Hayden Creek. At Herring Creek on Knight Island (the most heavily oiled wild stock tagging site), 54 of 117 tagged fish (46%) were recovered on site. The remaining 63 tags were recovered at 18 different streams from 6 to 25 miles away. Fourteen tagged fish from Herring Bay Creek were recovered at Loomis Creek and an additional 20 tagged fish from Herring Creek were recovered in a single stream in Eshamy Bay approximately 9 miles away.

At Totemoff Creek on Chenega Island, 108 of 140 tagged fish (77%) were recovered on site. The remaining 32 fish were recovered at 14 different creeks. Two tagged fish from Totemoff Creek were recovered at streams in eastern Prince William Sound some 75 miles from Totemoff Creek. Eleven tagged fish from Totemoff Creek were recovered at a single creek less than 1 mile from Totemoff Creek. At O'Brien Creek, 26 of 29 tagged fish (89%) were recovered on site while the remaining 3 fish were recovered at three different streams located between 4 and 25 miles away. At Cathead Creek, 36 of 73 tagged fish (49%) were recovered on site and the remaining 37 fish were recovered at 16 different creeks located from 3 to 20 miles away.

In Prince William Sound, all hatchery bred pink salmon are tagged at a ratio of approximately 1:560 while the six wild stocks were tagged at ratios ranging from 1:3 to 1:15. By extrapolating the number of hatchery tagged fish recovered in a stream to be representative of their untagged cohorts, then it appears hatchery fish contributed approximately 59% of the total escapement to Loomis Creek; 16% to the Herring Creek escapement; 27% to both the Hayden and O'Brien Creek escapements; 12% of the Totemoff Creek escapement; and 13% of the Cathead Creek escapement. Of all streams examined daily in 1991, Loomis Creek had the greatest number of fish straying into the creek (20 hatchery tagged fish and 16 wild tagged fish). At the same time those fish tagged at Loomis Creek strayed the least of the tagged wild stocks. Conversely, fish tagged at

Cathead Creek strayed more than any other tagged wild stocks while receiving the fewest number of stray hatchery (2) and wild (0) tagged fish. Although our understanding of the magnitude of the straying phenomena is limited geographically to those streams extensively examined in 1991, some patterns in straying are evident. Pink salmon from WHN and AFK hatcheries showed a tendency to stray into streams near the hatcheries and along migration corridors in southwestern Prince William Sound. Of the six stream recoveries of stray tagged-fish from SGH, four were from a remote release near Bligh Island.

The numerous wild stocks in Alaska contain the genetic resources necessary for continued production of salmon under shifting environmental conditions. The Genetic policy of Alaska (Davis, 1985) acknowledges that genetic diversity buffers biological systems against disaster, either natural or human-induced. Maintaining genetic diversity both within and between local populations is essential for the long-term sustained production of Alaska salmon (Mathisen, 1991).

There is evidence that the offspring of hatchery-produced pink salmon may be less viable in the wild than those from local wild fish (Windsor, 1990). Further evidence indicates that rapid expansion of hatchery production coupled with increased exploitation rates usually results in the eventual collapse of the wild stocks (Mathisen, 1991). Increased exploitation of wild stocks by the commercial fishing fleet coupled with the combined effects of the oil spill and straying by hatchery bred salmon into wild populations has put significant numbers of wild pink salmon populations at risk. For example, wild pink salmon in Prince William Sound, particularly in the oil impacted southwest districts, are unique in that they are predominantly (75%) intertidal spawners, a characteristic which enhances these populations' chances for continued reproductive success especially during harsh winter conditions. However, this characteristic also placed them at great risk for exposure to oil from the Exxon Valdez oil spill and subsequent cleanup activities. Because of the significant amount of straying by hatchery stocks, the potential exists for the eventual displacement of this and other unique adaptations of wild pink salmon populations. The long term productivity of Prince William Sound's wild pink salmon stocks will depend upon the conservation of the genetic diversity among and within those stocks.

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Assessment of Intertidal Algal Populations in Prince William Sound with an Airborne Multispectral Scanner

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One of the goals of the Coastal Habitat Injury Assessment (CHIA) studies is to quantify the damage caused by the *Exxon Valdez* oil spill to intertidal algal populations. The inherent spatial variability of intertidal algal populations suggested that digital remote sensing techniques might be the best way to sample the large geographic area affected by the *Exxon Valdez* spill.

Satellite imagery from the French SPOT satellite has been used to map intertidal algal populations on the Brittany coastline (Moussa et al., 1989). The 20 meter resolution of the SPOT sensor is adequate for this coastline because the topography is fairly flat and the tidal range is over 12 meters which creates a very wide intertidal zone.

The steep, narrow beaches in Prince William Sound would not be adequately mapped by the currently available commercial satellite sensor systems such as Thematic Mapper (TM) and SPOT. A previous study in southeast Alaska (Polcyn et al., 1978) had shown that an airborne scanner could be used to map intertidal algae.

For this study we selected the Compact Airborne Spectrographic Imager (CASI) because the instrument is easily transportable and can be installed on aircraft of opportunity. In addition, processing of the data can be done on microcomputers in the field which greatly assists in ground truthing the imagery. This instrument can achieve a spatial resolution of 1 square meter per pixel and a spectral resolution of up to 15 spectral bands in its spatial image mode.

The purpose of the study was to determine whether oil induced injuries to *Fucus* and other marine plants could be detected using CASI imagery, and whether these injuries could be quantified and mapped on shorelines throughout the spill area.

The specific objectives were to (1) determine whether the CASI could detect significant differences between the reflectance spectra of *Fucus* and other marine plants in oiled and unoiled sites, (2) determine whether the CASI data could be correlated with quantitative data being collected by the CHIA including estimates of percent cover and biomass of *Fucus*, and (3) determine the feasibility of using the CASI to detect and quantify oiled induced injuries to intertidal algal populations throughout the region affected by the *Exxon Valdez* oil spill.

The field work for this study took place in and around Herring Bay on Knight Island, Prince William Sound. Two field visits were made in late July and early August 1990. On the first visit, low clouds and heavy rain prevented flying, but spectral reflectance data was obtained in the laboratory for plant samples taken from six experimental beaches. On the second visit, the weather was again poor, but airborne image data was obtained for 3 pairs of oiled and cleaned (treated) and unoiled (control) beaches, under heavy overcast skies and light rain.

Spectral differences between *Fucus* and *Neorhodomela*, the dominant algal species in the rocky intertidal of Prince William Sound, were found with the CASI in the laboratory study. The unique spectral signature of the marine phanerogam, *Zostera marina*, also allowed this plant to be easily distinguished. Spectral differences were also noted between *Fucus* plants collected on oiled and unoiled beaches. Juvenile plants from oiled beaches were nearly twice as bright (higher reflectance) than those from unoiled sites. Mature plants from the lower intertidal of the oiled site were also brighter than those from the same tidal level at the control site. Mature plants from the upper intertidal levels of the oiled and unoiled sites had similar reflectances.

Based on the spectral differences observed in the laboratory studies, seven spectral bands were chosen for data collection with the airborne CASI. A number of different spectral band combinations were used to quantify the amount of algal biomass on the beaches. These included the "Fluorescence Line Height" algorithm developed by Gower (1978) and Borstad et al (1985) and a number of simple band ratios. A ratio of band 7 (750.5-759.9 nm) to band 6 (665.9-680.6 nm) correlated well with visual estimates of *Fucus* percent cover made in the field ($r^2 = 0.84$).

The total algal cover on the three treated beaches and their paired controls was mapped using the CASI imagery. Because of the low signal levels, algal cover was estimated with a simple supervised classification method that used the relationship between band 7 (750.5 - 759.9 nm) and band 3 (598.4 - 614.7 nm) and checked visually against a true color composite using bands 3, 2 (538.4 - 561.7 nm), and 1 (478.9 - 500.3 nm). This method did not account for differences in percent vegetative cover in any one pixel, but gave the total area covered by at least some vegetation. The lower threshold of what constituted vegetative was a subjective decision of the operator in this calculation.

Comparison of algal cover showed that the control beaches had a average of 12% greater algal cover than the treated beaches. This algal cover included both *Fucus* and *Neorhodomela*, because the airborne CASI imagery could not differentiate between them. Subtidal *Zostera* was recognized, but no ground data was available to confirm the percent coverage.

The discrimination of different algal species and the different age classes of *Fucus* that had been shown to be possible in the laboratory studies was not possible due to the poor weather conditions at the time of the aerial surveys. The low light levels caused by the heavy overcast and rain caused low signal to noise ratios which hampered spectral analysis. In addition, at this time of year in Herring Bay the intertidal algal species have bleached to a uniform brown color which makes species discrimination by spectral methods difficult.

This study has shown that CASI imagery can be used to quantify intertidal algal populations in Prince William Sound. Low altitude aerial photographs provide higher spatial resolution than the CASI imagery, but have very low spectral resolution, a low dynamic range, and must be redigitized in order to be used in Geographical Information System databases. Future studies of this nature should be carried out in the spring of the year when the different algal species could be more easily separated on their spectral characteristics. Digital classification methods should be investigated further, with the goal of obtaining *Fucus* specific indices. This will require more *in situ* data identifying areas of different types of plants and close coordination between biologists and remote sensing specialists. The ability to process image data in the field is very useful in accomplishing this task.

The use of this type of data for mapping large areas of intertidal habitat will require linking Global Positioning System (GPS) navigation data with the CASI imagery. This will allow the data to be referenced to a standard geographic coordinate system and combined with other datasets in the various GIS databases that have been established for this region during the *Exxon Valdez* spill response. Further consideration should also be given to the geometric problems caused by contorted, non-horizontal beach surfaces. It may be possible to acquire aerial photography at the same time as the CASI imagery and develop digital terrain maps of the beaches onto which the CASI imagery could be superimposed. This would provide better comparison of algal cover between beaches, but would significantly increase the costs of the surveys.

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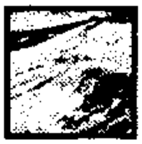
Stable Carbon Isotope Ratios of Prince William Sound Subtidal Sediments, Prior and Subsequent to the *Exxon Valdez* Oil Spill



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Numerous investigations have demonstrated the usefulness of stable carbon isotope ratios ($d^{13}C$) of organic matter in sediments and waters in identifying marine regions contaminated with petroleum (e.g., Calder and Parker, 1968; Spies and DesMarais, 1983; Anderson *et al.*, 1983; Eganhouse and Kaplan, 1988). The premise in these investigations was that carbon derived from various organic pools has a characteristic $d^{13}C$ value, e.g., the $d^{13}C$ of terrigenous C3 plants = 25 ‰ (Hong, 1986; Naidu *et al.*, 1992), marine phytodetritus = -21 ‰ (Fry and Sherr, 1984), seagrasses = -10 ‰ (McConnaughey and McRoy, 1979), and Prudhoe Bay crude oil = -30 ‰ (Magoon and Claypool, 1981).

In principle, therefore, the $d^{13}C$ of marine sediments could, based on an isotope mixing equation (Calder and Parker, 1968; Eganhouse and Kaplan, 1988), help to estimate the proportion in the sediment of organic matter derived from various natural or anthropogenic pools. Based on the above premise, we have attempted to examine the possibility of subtidal sediment contamination by *Exxon Valdez* crude oil in Prince William Sound.

In 1979, 1980 and 1981, 24 prespill sediment samples were collected with a van Veen grab from 37 to 106 m depths throughout Prince William Sound. In 1990 and 1991 two suites of sediment samples, one 'unoiled' and the other 'oiled', were collected from the relatively shallow (<20m) and deep (40-100m) regions of the southwest Sound, using SCUBA divers or a van Veen grab. All samples were stored frozen until ready for analysis. The oiled sites were selected from areas that had adjacent eelgrass and silled fjord habitat shorelines moderately to heavily oiled during the summer and fall of 1989.

The $d^{13}C$ analysis was made on carbonate-free sediments, using a VG 602E mass spectrometer (Naidu *et al.*, in press). The results are expressed relative to the PDB Standard, with a precison of 0.2 ‰. The mean $d^{13}C$ values of the time-series prespill, oiled and unoiled samples were statistically compared using nonparametric tests. Differences between means at $p>0.05$ were considered insignificant.

The mean $d^{13}C$ of the prespill sediments of 1979 (-22.2 ‰), 1980 (-22.1 ‰) and 1981 (-22.8 ‰) were all similar. The integrated mean $d^{13}C$ values of all the prespill sediments (-22.3 ‰) was similar to the mean $d^{13}C$ values for 1990 shallow and deep oiled sediments (-21.9 ‰ and 22.1 ‰, respectively), and the 1991 shallow and deep oiled sediments (-22.2 ‰ and -22.3 ‰, respectively). No significant differences were detected between the mean $d^{13}C$ of 1990 sediment samples from oiled and control sites of either the shallow or deep-waters. However, in 1991, the mean $d^{13}C$ of shallow (-22.2 ‰) and deep (-22.3 ‰) oiled sediments was lower than that of the shallow (-20.4 ‰) and deep (-21.6 ‰) unoiled sediments.

The finding of similar $d^{13}C$ values in prespill and oiled sediments, from shallow and deepwater sites, was contrary to our expectations.

Initially, we postulated that the d^{13}C of prespill sediments in the Sound would be relatively higher (less negative values) than the values for the post-spill oiled sediments. We assumed that any marked contamination of sediments from the Sound with Prudhoe Bay crude oil would shift the d^{13}C of the oiled sediments to more negative values. The discrepancy between our postulation and the analyzed d^{13}C values for prespill and oiled sediments suggests that oiled sediments were not markedly contaminated with oil.

Alternatively, it is possible that petroleum intercalated into the sediments was overwhelmingly diluted by natural organic material (e.g., eelgrass debris). As noted previously, lower d^{13}C values were determined for the 1991 shallow and deep oiled sediments, in comparison with shallow and deep unoiled sediments. It is possible that the source of the lower d^{13}C values in the 1991 sediments is petroleum from the adjacent heavily-oiled beaches. Perhaps sufficient oil had accumulated in the subtidal region by 1991 so that an isotopic signature of oil could finally be detected there. Thus, it appears that at least some oil reworked from the beaches, either by storm waves or tides, is carried offshore and may accumulate in the subtidal region.

In conclusion, we believe that in Prince William Sound sediments, unless heavily contaminated with petroleum, d^{13}C values are of limited use to assess the extent of sediment contamination by crude oil. It is suggested that additional d^{13}C analysis, using GC-IRMS, on the methanol and benzene soluble material (e.g., saturated and aromatic hydrocarbons) of prespill, unoiled and oiled sediments (Anderson *et al.*, 1983), could provide a more useful index of detecting petroleum contamination of the Prince William Sound sediments than d^{13}C analysis on gross organics of sediments.

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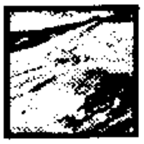

Response to the *Exxon Valdez* Oil Spill: A New Method to Test the Effects of Residual Oil on Intertidal Recolonization

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Following the *Exxon Valdez* oil spill, the Alaska Department of Environmental Conservation (ADEC) evaluated the effects of stranded oil on rocky intertidal communities. Field experiments, begun in the winter of 1989, tested the ability of intertidal organisms to colonize oiled substrates. This paper reports on a new field method which paired natural substrates using *in situ* levels of oil. Studies of intertidal communities face the basic problem of variability (Thomas, 1978). In this study, we expected colonization patterns to vary due to the range of intertidal microhabitats encountered as well as differences in surface texture among rocks used as test substrates. A paired design was chosen because of its potential to account for much of this variability (Sokal and Rohlf, 1981) and reduce the need to match control or reference beaches with oiled beaches, a process that can be problematic and can introduce further variance (Mann and Clark, 1978; Thomas, 1978). In this study, pairing was achieved by removing oil from one half of each rock, creating an oiled side and a reference side. Placement of the rocks on an unimpacted beach allowed a controlled examination of the colonization process without the threat of reoiling.

This approach builds on methods used previously to investigate oil spill-related effects on intertidal communities. Previous methods have included: cleaning substrates (Nelson, 1981; Crapp, 1971), manipulating substrates (Straughan, 1971), and pairing oiled and clean substrates (Straughan, 1971). However, to our knowledge, this is the first method to manipulate natural substrates containing *in situ* levels of oil.

Oiled rocks were gathered from two beaches in Prince William Sound impacted by the stranded oil, one beach on Smith Island (designated as beach segment SM-005 by ADEC) and the second on Eleanor Island (segment EL-057). These beaches were subject to a variety of treatment efforts to remove the oil, including flooding, pressure washing, handwiping, and bioremediation (enhancement of microbial degradation). Despite these removal efforts, which cleaned exposed surfaces, it was easy to obtain oiled rocks for this study by turning rocks over. In mid September, 1989, the Smith Island beach was 70% covered by a 30m wide band of mobile, sticky oil, while at Eleanor Island, although most of the free oil had been removed, rocks were still stained and subsurface oil remained mobile in gravely areas. The Smith Island beach is a relatively high-energy beach due to its exposure to a long fetch from the north; boulders and large cobbles on the beach are well-rounded, reflecting the impact of storms and wave action. In contrast, the Eleanor Island segment is more sheltered and the rocks are more angular. The unimpacted site, Gull Island, southeast of the *Exxon Valdez* grounding site, had a flourishing community of *Fucus*, filamentous algae, barnacles, limpets, snails, mussels, and other organisms typically found in this area (Feder and Bryson-Schwafel, 1988). No stranded oil was observed during any of the visits to this site.

Between September 15 and 18, 1989, rocks were collected from the three beaches, cleaned, and placed back on the beaches in accordance with the following design. Rocks of visually similar levels of oiling were collected. At EL-057, two levels of staining (high and low) were chosen visually. Staining is used here as defined by ADEC (1991) "Oil < 0.1 mm thick, cannot be easily scratched off with fingernail." Stained rocks were cleaned by dipping one half of each rock into methylene chloride. They were then placed randomly into the mid to upper tide area (as judged by the presence of barnacles and other biota) of Gull Island and affixed to the rocky substrate using marine epoxy putty (Z-Spar©). Rocks were retrieved March 17, 1990.

Measurements of algal colonization were made by dividing each rock into two halves corresponding to the oiled and cleaned halves. Five microscope fields were examined on each half; each field was viewed with a grid containing 25 squares of 1 mm^2. Algal coverage was recorded as the number of squares covered and converted to percent cover.

Using the paired data, following a square-root transformation, algal cover on the cleaned side was regressed against algal cover on the oiled side. The slope, was significantly different from one ($p=0.003$; slope=0.28 after backtransformation), indicating a reduction in percent cover on the oiled sides of about 72%.

The ecological significance of these and other results are reported in detail elsewhere. Here, we describe the statistical utility of the paired design which was evaluated by comparing the results of both nonparametric and parametric tests. All tests indicated a significant ($p<0.05$) difference between percent cover of algae on the two sides of the rocks. The paired t-Test had the lowest p value (0.001). For the unpaired t-Test, p was 0.029, and for the nonparametric tests (Wilcoxon Matched Pairs Test and Mann-Whitney U Test), probabilities were 0.005 and 0.046 respectively. It was not appropriate to conduct a 2-way ANOVA (with site origin and presence of oil as main effects) because one of the sites had zero variance in percent cover on the oiled sides (i.e., there was no algal settlement on the oiled sides of the Smith Island rocks).

The inhibition of algal colonization on oiled sides was very pronounced which probably explains why all of our analyses produced significant results. Our paired design, however, did achieve a greater ability to demonstrate the effect of the residual oil and has been used in subsequent studies.

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The Alaska Heritage Stewardship Program

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The *Exxon Valdez* oil spill brought hundreds of people to the relatively remote beaches of southcentral Alaska for cleanup activities. One effect of this influx was increased awareness of, and access to, archaeological sites which had been protected by their isolation. This awareness prompted an increase in vandalism and damage to sites which has continued since cleanup.

Vandalism is often caused by individuals interested in archaeology but unaware of the damage caused by disturbing sites. Information lost from archaeological sites is irretrievable, and damaged sites cannot be restored to their original condition. Mitigation of such damage generally involves excavation to recover information before further loss occurs. This approach is expensive and time consuming and does not address the ultimate cause of the damage.

Successful archaeological stewardship programs in Arizona, Texas, and Arkansas prevent vandalism through public education and regular patrols of threatened sites (Arizona Site Steward Program, 1992; Texas Archeological Stewardship Network, 1992; Arkansas Archeological Survey, 1992). The Alaska program, focusing on the spill area, is being developed by the U. S. Fish and Wildlife Service and the State Office of History and Archaeology, with help from the Forest Service, National Park Service and the Aluutiq Cultural Center on Kodiak Island.

The program is intended to be self sustaining and locally based. Interested individuals and organizations will volunteer with participating land managers or owners and receive training. The volunteers will patrol sites, reporting any disturbance and engage in other preservation activities. Governmental involvement will be limited to necessary administrative and record keeping functions, advisory and technical assistance and, if necessary, law enforcement activities.

Public reception has been positive and enthusiastic with interest throughout the spill area as well as the Aleutians, Ketchikan and the Seward Peninsula. Pilot programs are being organized in Kodiak, Homer and Prince William Sound villages and stewards will be active in summer 1993.

The poster shows examples of site vandalism with a brief explanation of the importance of archaeological context. Copies of the Steward Handbook and Fieldbook will be available to illustrate the duties and goals of the program. Photographs of the types of activities stewards will engage in follow. These include site monitoring, collecting oral histories, documenting private artifact collections, and contributing to public education during events like Alaska Archaeology Week.

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Instrumental Neutron Activation Analysis of Nestling Peregrine Falcon Feathers Collected From Prince William Sound and Norton Sound, Alaska, Breeding Populations in 1990

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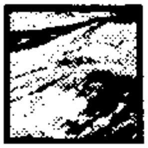
Secondary remex feathers were collected from nestling Peregrine Falcons (*Falco peregrinus pealei*) in Prince William Sound (n = 32) and Norton Sound (n = 13), Alaska, in 1990. Feather samples were analyzed for trace element content using Instrumental Neutron Activation Analysis.

Concentrations of Sodium, Magnesium, Aluminum, Sulfur, Chlorine, Calcium, Titanium, Vanadium, Manganese, Copper, Iodine, and Nickel were quantified and compared between the two locales and with known concentrations for peregrine breeding populations on Langara Island, British Columbia.

Of primary concern were Vanadium and Nickel concentrations. Both Vanadium and Nickel are considered "signature elements" for Prudhoe Bay crude oil, and concentrations in peregrine feathers may indicate exposure as a result of the *Exxon Valdez* oil spill.

Concentrations of Nickel were not present in sufficient quantities to be quantified in the feather samples analyzed from both locales. Concentrations of Vanadium were approximately 1.5 times greater in Prince William Sound samples than in those representing Norton Sound. Overall, trace element concentrations quantified for Prince William Sound were higher than those for Norton Sound and other regions in Alaska.

Concentrations of Chlorine in Prince William Sound feather samples, for instance, were almost 4 times greater than Norton Sound samples, and substantially higher as well than concentrations known for other regions in Alaska. Variation in trace element concentrations was also noted for areas sampled within Prince William Sound as well as Norton Sound.



Hydrocarbons in Intertidal Sediments and Mussels from Prince William Sound, Alaska, 1977-1980: Characterization and Probable Sources

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We collected and analyzed samples of sediments and mussels (*Mytilus trossulus*) for alkane and aromatic hydrocarbons from eight sampling stations adjacent to the oil tanker vessel transportation corridor through Prince William Sound, Alaska, during the period 1977 to 1980, to determine baseline concentrations of these analytes prior to any pollution that might result from oil tanker traffic through the Sound. We evaluated inter-annual (between years) variability of these analytes in sediments and in mussels using two-factor analysis of variance (ANOVA) of logarithm-transformed hydrocarbon concentrations determined in duplicate samples collected in June, 1977 and in June, 1978 at six of the sampling stations.

Intra-annual (within the year) variability was similarly evaluated using ANOVA on results of chemical analyses of duplicate samples collected in May, June, and August, 1978 at seven of the sampling stations. To facilitate comparison with future work, total organic carbon and grain size distribution was determined in the sediment samples, the lipid content was determined in the mussel samples, and the surface seawater temperature and salinity was determined at each sampling station and time.

The results of the hydrocarbon analyses indicate chronic, low-level hydrocarbon contamination that probably originates from small fuel spills, ballast water discharges, and fuel-combustion exhaust emissions of occasional vessel activity adjacent to three of the sampling stations: Constantine Harbor, Rocky Bay, and Mineral Flats, in decreasing order of contamination, respectively. Contamination at these three stations is indicated by the diversity of aromatic hydrocarbons found in sediments at concentrations that are generally less than 10 ng/g dry sediment weight, but above detection limits (< 1.0 ng/g) of these analytes in sediments.

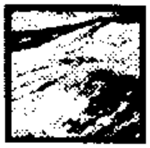
In contrast, the remaining five sampling stations showed no indication of petroleum hydrocarbon contamination, primarily because few aromatic hydrocarbons were detected at these stations, and detected aromatic hydrocarbons were present only sporadically and at concentrations that were generally near detection limits. Exceptions are perylene, which was found at concentrations well above detection limits at all sampling stations outside Port Valdez, and which probably has natural sources; and phenanthrene, which was found sporadically at all sampling stations and which may also have natural sources, in addition to the hydrocarbon contamination sources indicated at the three polluted stations. Concentrations of aromatic hydrocarbons are too frequently below detection limits at most of the sampling stations to evaluate intra- and inter-annual variability using ANOVA.

Concentrations of individual n-alkanes vary substantially in sediments and in mussels. The most abundant n-alkanes in sediments include odd carbon-numbered alkanes of molecular weight greater than tetradecane (C-14). Concentrations of these n-alkanes are generally in the range of 10 to 100 ng/g dry sediment weight, and exceed 1000 ng/g at Constantine Harbor. The most abundant n-alkanes in mussels include decane (C-10) through heptadecane (C-17), and pristane, at concentrations generally ranging from 10 to several hundred ng/g dry tissue weight.

Sources of alkanes in sediments include terrigenous plant waxes, marine plankton, and possibly marine macrophytic algae at all the sampling stations, and petroleum-derived alkanes in addition at Constantine Harbor. Terrigenous plant waxes in sediments are indicated by high abundances of odd carbon-numbered n-alkanes of molecular weight greater than nonadecane (C-19) compared with even carbon-numbered n-alkanes in these sediments, and by slight but significant intra-annual variability of these odd carbon-numbered alkanes in sediments, which probably arises from seasonal deposition of senescent leaves. Marine planktonic and algal sources of pristane and normal alkanes is indicated by the presence of these alkanes in sediments and in mussels, and by the relatively high abundances of pristane, pentadecane (C-15), and heptadecane (C-17) in sediments and in mussels.

Pristane, pentadecane (C-15), and heptadecane (C-17) vary significantly ($P < 0.001$) in sediments, or in mussels, or in both, intra-annually or inter-annually. Pristane variability in sediments and in mussels is significantly correlated, and is probably due to variability of populations of calanoid copepods within and among years in Prince William Sound. Neither pentadecane variability nor heptadecane variability are correlated in sediments and mussels, suggesting multiple biological sources of these alkanes.

These results indicate that, except in areas affected by localized vessel traffic, intertidal sediments and mussels in Prince William Sound are remarkably free of petroleum-contaminant hydrocarbons during the period of this study. The hydrocarbons found in sediments and mussels unaffected by vessel traffic can be adequately explained by known, natural sources. As a result, sediments and mussels contaminated by crude oil from the *Exxon Valdez* oil spill should be particularly apparent, due to the general absence of other confounding sources of petroleum hydrocarbons.



Meiofaunal Recolonization Experiment with Oiled Sediments: Major Meiofauna Taxa

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An in situ experiment was initiated in 1990 in Herring Bay, Prince William Sound, to study the effects of the *Exxon Valdez* oil spill on recolonization by meiobenthos (small, bottom-dwelling organisms). The study site (60°28'0"N latitude, 147°41'12"W longitude) was on a section of shoreline designated as heavily oiled by the Alaska Department of Environmental Conservation; oil remaining from the *Exxon Valdez* spill was obviously present. Temperature and salinity were measured with a self-contained sensing device mounted on a tripod at an elevation of -0.1 m; both remained relatively constant over the first 28 days of the experiment at 8.0±0.1°C and 29.1±0.1 ppt.

Exxon Valdez crude oil was added and mixed into azoic sediments resulting in two concentrations, 0.5% and 1.7% crude oil, and the resulting mixture was added to triplicate colonization trays (all 13 x 28 x 33 cm). In addition, non-oiled azoic sediments treated similarly were added to triplicate trays, and samples were also collected from untreated surrounding sediments to examine treatment effects and the ambient meiofauna community which would probably be the origin of colonizers.

Trays were placed flush with the sediment surface on beaches along a transect paralleling the -0.6 m tidal level, along the upper margins of an eel grass bed. Triplicate samples were collected with hand-held corers (modified 60 ml plastic syringes) at random locations along X and Y axes within each tray during aerial exposure at low tide on days 0, 1, 2, 29, 90 and 443 after initiation on April 25, 1990.

Cores for meiofauna analysis were preserved in 10% buffered formalin and returned to the laboratory. Hydrocarbon samples were collected with a 3-cm-diameter chrome plated brass tube and placed into hydrocarbon-free glass jars with Teflon lids and frozen until analysis. In the laboratory, meiofauna passing through a 0.500 mesh sieve but retained on a 0.063 mesh sieve were separated from detritus with a sucrose flotation/centrifugation technique (Fleeger, 1979).

All organisms were identified to major taxon with a stereo dissection microscope and enumerated with ruled trays. Predominant taxa (mainly nematodes) were subsampled when they occurred in high densities using a technique which employs a triply-balanced square design (Sherman et al., 1984). Here, we report on the predominant meiofauna taxa from these collections, particularly the nematodes, harpacticoid copepods, copepod nauplii, ostracods and bivalve larvae. Other meiofauna taxa that occurred in substantial numbers on some dates include turbellarians, halocarid mites, gastropod larvae and polychaetes.

Hydrocarbon concentrations in the sediments correlated well with the percent oil added to the treatments. Hydrocarbon, aromatic and alkane concentrations declined rapidly during the first 30 days of the experiment and became asymptotic. The unoiled sediment treatments were contaminated with very small quantities of hydrocarbons, which also declined over time. Hydrocarbon concentrations in ambient sediments were similar to those in the unoiled treatments.

The experiment was initiated during the late spring, which generally is a period of active meiofauna recruitment in Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990; McGregor, 1990). Colonization was rapid for many true meiofaunal taxa (but not macrobenthic larvae) which occur in the surface sediments, as densities in trays were not significantly different from surrounding sediment collections by day two, except in the high oil sediments. Generally, high oil treatments had a reduced density compared to low and control sediments until day 29.

After initial colonization, experimental effects independent of treatment were apparent for most taxa. The effects resulted in densities (for most meiofauna taxa) higher in the experimental treatments than densities measured synoptically in the surrounding sediments. Modifications of biotic interactions generated by colonization of an azoic habitat, or emigration/immigration phenomena, may be responsible for the experimental effects.

The type of competitive, agonistic or predator-prey interaction which were altered may explain the variation in magnitude and timing of experimental effects among taxa. The azoic colonization trays may have decreased competition for some taxa, provided others an escape from predation, or influenced both interactions for some taxa. Some predatory meiofauna may have had altered prey availability, while bactivorous meiofauna may have experienced increased prey in the oiled sediments.

Harpacticoid copepods are important food items for the early life history stages of many marine fish and crustaceans; because of their importance in marine food webs, they are treated in detail in a separate presentation (Fleeger et al., 1993). In our study, harpacticoids were diverse with > 40 species encountered. Species analysis of the harpacticoid community indicated that sediments and colonization trays were inhabited primarily by phytal copepods associated with the adjacent eel grass and algal mats habitats. We assume the same relationship may have occurred with other surface meiofauna taxa, although they were not identified to species.

The average density of combined live copepods and copepodites were similar in all treatments by day 29, but averaged two to three times higher in experimental treatments than in surrounding natural sediments by day 90. The elevated densities in the experimental treatments persisted on day 443 the following year. The biotic mechanism(s) responsible for the increased densities cannot be determined by our single-factorial experiment. Copepods may have actively selected the experimental trays, or may have had enhanced survival and production after immigration.

Copepods which were assumed to be dead (as determined by deterioration or missing appendages) at the time of collection were counted separately. Dead copepods were present in the sediments used in the experiments as a result of the repeated freezing and thawing technique used to render the sediments azoic, and created some methodological problems.

Dead copepods were present in all samples through day 90, but were higher in experimental trays and rare in ambient sediments. The highest density of dead copepods were found in the high oil treatment on day 29, but few significant differences existed among the treatments throughout the experiment due to high variance.

Decreased availability or active selection against immigration into the experimental trays by copepod nauplii was evident early in the experiment. Nauplii occurred in highest density in the ambient sediments on day 2 (124±79.core-1), while at the same time nauplii densities in the treatments were extremely low (4.8±5.8, 7.4±5.2 and 22.3±18.9 in high oil, low oil and unoiled sediments, respectively). Higher densities in the experimental treatments were not found until day 29, when no significant differences occurred among the treatments (142±136, 103±80, 84±73 and 74±75.core-1 for high oil, low oil, unoiled and ambient sediments, respectively).

Nematodes were the numerically predominant taxon on most sampling dates in all treatments, with average densities varying from a minimum of not significantly more than zero at the beginning of the experiment, to a maximum average of 698.core-1 on day 443 in the unoiled trays. As with many other taxa, nematodes had higher average densities in experimental trays in comparison to ambient sediments after day 90, with lowest values in the high oil and highest values in the unoiled treatments.

Ostracods occurred in low average densities in all treatments and colonized rapidly. By day 2, no significant differences occurred among the treatments or ambient sediments. Densities remained low in the high oil treatment through day 443, The highest densities encountered were an order of magnitude higher, in the low oil treatment on day 443, which was also the date of highest density of ostracods in the unoiled treatment and in ambient sediments. Lower densities of ostracods have also been found in some heavily oiled bays in Prince William Sound the year following the oil spill (Shirley et al., 1993).

Halocarid mites, which are often predatory and sometimes predominate in meiofaunal communities in the high intertidal among algae, responded almost identically as the ostracods. They colonized rapidly, had higher densities in the experimental pans than in the ambient sediments, but never attained high densities in any treatment.

Pronounced seasonal changes in density occurred for all taxa in the ambient natural sediments and in the experimental treatments. The seasonal changes varied among taxa in timing and magnitude and reflect seasonal recruitment and mortality events. Natural seasonal and interannual (between years) variation in meiofaunal community composition and density, as is common for most marine metazoans, confound analysis of treatment effects.

Changes in density of temporary meiofauna (the larvae of macrobenthic invertebrates, e.g., bivalves, gastropods and polychaets) occur in pulses related to planktonic settlement in the intertidal zone in Alaska (McGregor, 1990). Bivalve larvae were rare in our cores until day 90, when they were abundant in all treatments and in ambient sediments. Average density of bivalve larvae was 3-4 times higher in the experimental trays than in ambient sediments, suggesting active selection by the larvae or higher post-settlement survival rates.

Our data demonstrate that over small spatial scales, meiofauna recolonize azoic sediments in the intertidal rapidly following an oil spill, but highly oiled sediments reduce recolonization rates of major meiofauna taxa and have effects that are persistent for more than a year for some taxa.

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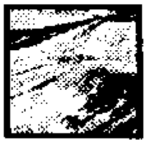

Responses of Intertidal and Subtidal Meiofauna to the Prince William Sound Oil Spill

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The effects of oil spills on meiofauna (small animals on the ocean floor) are poorly known, even though they are primary food items for many newly settling or postmetamorphic fish and macroinvertebrates. We examined the responses of intertidal (0 m tidal level) and subtidal (-6 m tidal level) meiofauna in Prince William Sound to the *Exxon Valdez* oil spill by comparing community composition and density in oiled and unoiled bays.

Meiofauna were quantitatively sampled in 10 bays within Prince William Sound on five dates in 1989-1990, beginning approximately six weeks after the initial spill. Five bays were unoiled and five received varying amounts of oiling. Samples were collected from five bays on all sampling dates and four bays were sampled on all but one date. Eight samples were collected with hand-held piston corers (modified syringes) to approximately the same sediment depth (2.5 cm) at each tidal level and study site on each date. All subtidal samples and most (72%) of the intertidal samples were collected by SCUBA divers; other samples were collected during low tide exposure. Samples were collected at randomly determined intervals along transects paralleling the shoreline at the selected tidal heights. Divers collected samples in front of their swim path to avoid disturbing surface sediments. A total of 380 cores in 1989 and 304 cores in 1990 were collected for analysis of meiofauna; an additional 288 cores collected in 1991 are partially analyzed. Additional cores from the transects were collected synoptically for sediment and hydrocarbon analysis. Hydrocarbon samples were collected with a 3 cm-diameter chrome-plated brass tube, placed into hydrocarbon-free glass jars with Teflon lids and frozen until analysis. Cores collected for bacterial activity and abundance along our transects at some sites on selected dates during 1989 for a separate study provide additional correlative information.

Cores for meiofauna analysis were preserved in 10% buffered formalin and returned to the laboratory. Meiofauna passing through a 0.500 mesh sieve but retained on a 0.063 mesh sieve were separated from detritus with a sucrose flotation-centrifugation technique (Fleeger, 1979) and stained with rose bengal. All organisms were identified to major taxon with a stereo dissection microscope and enumerated within ruled trays. Predominant taxa (mainly nematodes) were subsampled when they occurred in high densities using a technique which employs a triply-balanced square design (Sherman et al., 1984).

Here, we report on seasonal changes in density of the predominant meiofauna taxa from oiled and unoiled bays, particularly the nematodes, harpacticoid copepods, copepod nauplii, ostracods and bivalve larvae. Other meiofauna taxa that occurred in substantial numbers on some dates included turbellarians, halocarid mites, gastropod larvae and polychaets.

Pronounced seasonal changes in density occurred for all taxa, particularly among the temporary meiofauna (larvae of macrobenthic species) reflecting seasonal recruitment patterns which varied among the taxa. Changes in density of temporary meiofauna (the larvae of macrobenthic invertebrates, e.g., bivalves, gastropods and polychaets) occur in pulses related to planktonic settlement in the intertidal zone in Alaska. Their densities often decline markedly over short time intervals due to predation and rapid growth which excludes them from being members of the meiofauna (McGregor, 1990). Our infrequent sampling intervals hinder comparative use of their densities.

Nematodes were consistently the numerically predominant taxon at all sites; their density varied significantly between bays, sampling dates and tidal heights. No consistent trends were obvious in the density of nematodes between oiled and unoiled bays in the intertidal or subtidal. One of the heavily oiled bays, Herring Bay, had the lowest nematode densities in the intertidal zone throughout 1989 in comparison to other bays. Intertidal nematode density in Herring Bay increased to levels not significantly different from other bays by September 1989 and remained at levels comparable to other bays in 1990 and 1991.

Harpacticoid copepods (the sum of adults and copepodites) generally had higher densities in the intertidal than in the subtidal zone for all bays. Our measurements of densities of the entire harpacticoid community are simplistic, as the harpacticoid community of Prince William Sound is diverse. In related meiofauna recolonization studies conducted in Herring Bay, more than 40 species were identified (Fleeger et al., 1993). Most oiled bays had declines in average copepod density in June, 1989 in comparison to samples collected at the same locations in May; however, the same seasonal decline was evident in several unoiled bays. The lowest densities of harpacticoids measured during the study were in oiled bays, however, some of the highest densities were found in other oiled bays. Most bays, oiled and unoiled, had a trend of declining densities of harpacticoids from April through September in the intertidal, with generally the opposite trend for the subtidal harpacticoids. Similar seasonal cycles have been reported for harpacticoid copepods in both the intertidal (McGregor, 1990) and subtidal in southeastern Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990).

We know from field experiments in Prince William Sound that some harpacticoid species can recolonize azoic oiled and unoiled sediments quickly (<30 days) over small spatial scales, and that their densities may become higher in oiled sediments than in adjacent, unoiled sediments (Fleeger et al., 1993; Shirley et al., 1993). Similar, but also varying, responses of harpacticoids to oil spills have been reported from other habitats (Fleeger and Chandler, 1983; Decker and Fleeger, 1984).

Ostracods responses were similar to harpacticoid copepods. Several oiled bays (Herring, Iktua, Sleepy) had lower ostracod densities in the intertidal zone than unoiled bays (Eshamy, Ewan, Paddy) during the initial sampling series, while subtidal ostracod densities were not significantly different among oiled and unoiled bays. A midsummer depression in density was present in all bays in both the intertidal and subtidal; however, increases in ostracod densities occurred in the unoiled bays in the September samples, but generally did not occur in the oiled bays. The same phenomenon was observed for ostracods in recolonization in situ experiments, where their densities remained depressed the subsequent year.

In summary, it is probable that the initial depression of the meiofauna community in response to oiling were not measured because of the time lapse (six weeks) between the spill and sampling. The oil spill in Prince William Sound occurred at a time of annual recruitment for many meiofauna species in Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990; McGregor, 1990) and concentration of many volatile hydrocarbon components decrease rapidly (Shirley et al. 1993). Putative oil effects may have been evident in the initial samples following the spill in some bays, primarily in intertidal samples. Nematode densities were relatively unaffected, although the lowest densities recorded were in oiled bays. Harpacticoid copepods generally had lower densities (with a notable exception) in the intertidal zone in oiled bays on the initial sampling date after the spill. The same relationship was not true in the subtidal, which appeared to have decreased densities of harpacticoids in some oiled bays on the subsequent sampling date.

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