Consequences of the *Exxon Valdez* Oil Spill on Black Oystercatchers Inhabiting Prince William Sound, Alaska

Dissertation

Presented in Partial Fulfillment of the Requirements
for the Degree Doctor of Philosophy in the
Graduate School of the Ohio State University

by

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ABSTRACT

Approximately 40% (16.7 million L) of the Prudhoe Bay crude oil spilled by the *T/V Exxon Valdez*, in 1989, in northern Prince William Sound, Alaska (Sound), was deposited along 563 km of shoreline of the Sound. Black oystercatchers (*Haematopus bachmani*) are completely dependent upon marine shorelines for their life's requirements and were at risk from direct and indirect effects of the *Exxon Valdez* oil spill (Spill). Therefore, I undertook a study to determine the immediate effects of the Spill on reproduction of oystercatchers and to determine if oil persisting in fine sediments and mussel beds, into 1992 and 1993, was negatively affecting adult reproduction and growth rates of chicks. I also collected information on the habitat requirements of black oystercatchers.

Although direct mortality of adults was, most likely, minimal (only 2 oystercatcher carcasses were retrieved in the Sound and no oiled adults were observed on Green Island in 1989), black oystercatchers nesting along shorelines that were impacted by oil had their breeding season disrupted in 1989. On Green Island, 39% of the breeding population was not able to maintain nests into June. Oystercatcher pairs that tolerated heavily or moderately oiled nest sites and were able to hatch eggs experienced greater losses of chicks than pairs that nested at unoiled shorelines. Disturbances to shorelines,

by oil impact and associated clean up operations, also reduced productivity into 1990; by 1991, however, productivity had recovered and reoccupation of oiled shoreline habitats had occurred.

Gradual, gravelly shorelines indicated high quality habitat for breeding black oystercatchers; these shoreline types supported the highest densities in the Sound. In areas of steep rocky shorelines, the presence of numerous offshore islets (≤ 100 m diameter) and mixed sand and gravel beaches were good indicators of oystercatcher habitat.

The importance of mussels as a food source for adults and young and the reliance on fine sediment shorelines for foraging sites indicated that black oystercatchers could be exposed to oil persisting on shorelines. Where shoreline oil remained in 1992 and 1993, elevated hydrocarbon concentrations in the feces of chicks indicated that black oystercatchers were being exposed to persistent oil. Petroleum hydrocarbon concentrations in sediments collected from oystercatcher foraging sites were highest in Herring Bay, in Bay of Isles, and on northwest Knight Island.

Exposure to persistent oil changed the ability of adults to provision their young, possibly lowered the body condition of renesting females, and lowered the ability of chicks to gain weight. Effects of exposure to persistent oil on breeding success were overshadowed by the influence of egg and chick predators.

Because areas of persistent shoreline oil exposure were restricted, widely distributed, and highly variable, effects on the black oystercatcher population, on the short term, were minimal. The longevity of oystercatchers minimizes the importance of a single, or dual, season breeding failure. Remediation or cleaning of remaining oiled mussel beds in Prince William Sound should have a minimal impact on black oystercatcher population as whole. Additional restoration actions for oystercatchers should include the acquisition of low elevation, gravelly shorelines.

To the fond memories of

Vernon Ora Dennis (1895-1972) and

Everitt Ludden Jefferson (1905-1992)

ACKNOWLEDGMENTS

The research described in this paper was supported by the Exxon Valdez Oil Spill Trustee Council. However, the findings and conclusions presented by the author are his own and do not necessarily reflect the views or position of the Trustee Council or previous investigators. B. E. Sharp supervised fieldwork in 1989 and prepared an interim report on the findings. M. M. Cody conducted fieldwork on Green Island in 1990. P. A. Martin, J. R. Bart, A. Gunther, D. B. Irons, D. R. Nysewander, K. K. Laing, and S. P. Klosiewski and provided valuable input into the study design and analysis. Logistical support for this study was provided by the U.S. Fish and Wildlife Service, Region 7, Nongame Migratory Bird Project; K. D. Wohl, M. McWhorter, S. Kalxdorff and J. D. Ramey assisted with various logistical details. M. M. Babcock, P. M. Rounds, and B. Wright (Natl. Oceanic Atmos. Adm., Auke Bay Laboratory) assisted in collecting and analyzing mussel and sediment samples. M. East and D. Gray, Alaska Dept. Fish Game, aided in identifying oiled sites on Knight Island. Pilot W. I. Butler flew the winter survey. Without the assistance of R. Turner, M. L. DeZeeuw, D. I. Weeks, D. R. Weeks and S. E. McClellan, these projects would not have been completed. S. P. Kendall furnished boat survey data from 1990 and 1993. Dana Bruden furnished moral support to finish this dissertation and also provided statistical consultation. This dissertation

benefited from the comments of J. R. Bart, T. A. Bookhout, A. R. DeGange, D. M. Fry, R. E. Gill, J. D. Harder, D. Heineman, G. L. Hunt, S. P. Klosiewski, K. L. Oakley, D. D. Roby, R. T. Sayre, R. B. Spies and J. A. Wiens.

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INTRODUCTION

The *Exxon Valdez* Oil Spill

On March 24, 1989, the *T/V Exxon Valdez* ran aground in northern Prince William Sound, Alaska (Sound), and released 42 million L of Prudhoe Bay crude oil into the marine environment. Within 7 days, an oil slick extended 120 km across the western Sound. Approximately 40% (16.7 million L) of the spilled oil was eventually deposited along 563 km of shoreline in the Sound (Galt et al. 1991). Crude oil that washed ashore was more persistent than oil that remained suspended in the water column. Shoreline oiling was heaviest along western Montague Strait and eastern Knight Island Passage; however, some shoreline segments in this area escaped oiling. Within 2 weeks after the spill, only 5% of the spilled oil remained floating in the Sound. About 25% of the oil suspended in the water column exited the Sound and fouled a lesser amount of shoreline along the Kenai and Alaska peninsulas. Only 2% of the spilled oil reached the Alaska Peninsula. Remaining oil (35%) either evaporated or dispersed into the water column (Galt et al. 1991). Thus, shorelines within the Sound were one of the marine environments most negatively affected by the *Exxon Valdez* oil spill (Spill).

More than 10,000 workers were deployed in 1989 to clean up remaining floating and beached oil (U. S. Environmental Protection Agency 1989). Between 1989 and 1992 numerous shoreline treatment methods were applied: mechanical, manual scrubbing, spot swabbing, high-pressure washing, and bioremediation (Piper 1993). For mechanical treatments, sediments were agitated by tractors, backhoes, or other machines to release buried oil. In the bioremediation process, petrophilic microorganisms were applied to oiled shorelines to increase the natural rate of oil degradation. The choice of cleanup technique depended on the shoreline type (e.g., gravel, rocky), weather, condition of the oil present on the shoreline, and presence of potentially affected natural resources (U. S. Environmental Protection Agency 1989). Individual segments of shoreline often received a variety of treatments. By the time cleanup operations were initiated in April 1989, however, many wildlife species already had been exposed to spilled oil.

Effects of Oil on Birds

Crude oil spilled into the sea can have acute and dramatic effects on marine birds (reviewed in Bourne 1968, Holmes and Cronshaw 1977, Clark 1984). Duration and extent of exposure, origin of oil, bird species, and life stage determine the magnitude of the effect of oil exposure on marine birds. Pelagic species are more vulnerable to direct, lethal effects of spilled oil than are species that inhabit shorelines (Vermeer and Vermeer 1975). When its plumage becomes heavily oiled, a bird loses the ability to retain heat and may die of hypothermia (Hartung 1967; reviewed in Jenssen 1995) or may lose buoyancy and drown (Holmes and Cronshaw 1977). Cold climates accentuate the susceptibility of marine birds to hypothermia. Further mortality can result if oiled birds consume contaminated prey or preen contaminated feathers (Hartung 1963). Transfer of oil from the plumage or feet of an incubating bird to its eggs can kill the embryos (McGill and Richmond 1979, Lewis and Malecki 1984).

Estimates of the direct mortality of marine birds, mainly pelagic foraging species, caused by the *Exxon Valdez* oil slick ranged from 100,000-300,000 individuals (Piatt et al. 1990) to 375,000-435,000 individuals (Ecological Consulting, Inc. 1991). Although mortality caused by an oil spill can be massive, like that of the *Exxon Valdez* oil spill, negative effects on marine bird populations are often short-term (Clark 1984, Butler et al. 1988). Ford et al. (1982), however, suggested that pronounced, long-term effects in seabird populations could occur and were dependent on the extent and timing of a spill. Long-term effects of the *Exxon Valdez* on populations of marine birds are still being addressed.

Besides direct, lethal effects of oil contamination, ingestion of petroleum can negatively affect the reproduction of marine birds. Most studies of the effects of external or internal exposure to oil have involved experimentally treating captive or free-living birds. Principal among indirect effects on adults is the abandonment of eggs or chicks and a reduction in parental care. Adult Leach's storm-petrels (*Oceanodroma leucorhea*) and wedge-tailed shearwaters (*Puffinus pacificus*), internally and externally exposed to oil (Trivelpiece et al. 1984, Fry et al. 1986, Butler et al. 1988), and south polar skuas (*Catharacta maccormicki*) exposed to spilled crude oil (Eppley and Rubega 1989, 1990, but see Trivelpiece et al. 1990) deserted their chicks or reduced their parental care. Adult Leach's storm-petrels exposed to oil, internally and externally, had elevated metabolic rates and may have neglected parental duties to meet their own energy demands (Butler et al. 1986). Fry et al. (1986) suggested that internal and external oil exposure disrupted the pair bond of wedge-tailed shearwaters and that the disruption continued into the following breeding season. Reduction in the amount of parental care provided to chicks by oil-exposed adults, relative to non-exposed adults, resulted in lower growth rates and lower survival of chicks (Trivelpiece et al. 1984, Fry et al. 1986, Butler et al. 1988).

Hatching success and clutch initiation can also be negatively influenced by oil exposure. Hatching success of wedge-tailed shearwaters and Leach's storm-petrels exposed, internally and externally, to crude oil was reduced (Fry et al. 1986, Butler et al. 1988). In storm-petrels, external exposure had a greater effect on hatching success than did internal exposure (Butler et al. 1988). Cassin's auklets (*Ptychoramphus aleuticus*) internally exposed to crude oil reduced their nesting attempts but did not have reduced hatching success; Ainley et al. (1981) suggested that effects of oil ingestion of a female's ability to lay eggs and the ability of eggs to hatch operated independently.

Negative effects of internal oil exposure on the growth rates of marine bird chicks is well documented. Herring gull (*Larus argentatus*), black guillemot (*Cepphus grylle*), Leach's storm-petrel, and fork-tailed storm-petrel (*Oceanodroma furcata*) chicks internally exposed to oil gained weight slower than did control chicks (Miller et al. 1978, Butler and Lukasiewicz 1979, Peakall et al. 1980, 1982, 1983, Trivelpiece et al. 1984, Boersma et al. 1988). In general, growth of the wing, tarsus, and culmen did not differ between exposed and control chicks. Negative effects of oil exposure on weight gain were most pronounced during the early life of the chicks.

Development of a number of pathological conditions caused by consumption or inhalation of oil can reduce the overall physical condition of a marine bird (summarized in Fry and Lowenstine 1985). Internal oil exposure caused hypertrophy of adrenal or nasal glands in herring gulls, black guillemots, and Leach's storm-petrels (Miller et al. 1978, Peakall et al. 1980, 1982, 1983, Trivelpiece et al. 1984) and increased the P-450 cytochrome content in livers of herring gulls (Miller et al. 1978, Peakall et al. 1983, Lee et al. 1985). Ingested oil also impaired the osmoregulatory ability and nutrient absorption ability of herring gull chicks (Miller et al. 1978, Peakall et al. 1983). Necrosis of the intestine, kidneys, and liver, and damage to the blood (hemosiderosis) were found in spill-oiled common murres (*Uria aalge*) and thick-billed murres (*Uria lomvia*) that died during rehabilitation (Khan and Ryan 1991, Khan and Nag 1993). Internally-exposed Cassin's auklets and spill-oiled common murres exhibited hemolytic anemia, hemosiderosis, and renal necrosis (Fry and Lowenstine 1985). Hemolytic anemia also developed in young herring gulls and Atlantic puffins (*Fratercula arctica*) that were experimentally dosed with crude oil (Leighton et al. 1983). Blood serum of bald eagles (*Haliaeetus leucocephalus*) inhabiting oiled areas of the Sound showed only mild anemia that was not thought to seriously compromise the health of the population (Redig et al. 1990).

Although bird species that inhabit shorelines, particularly rocky shorelines, are less susceptible than pelagic species to the acute, lethal effects of a marine oil spill, their prolonged exposure to oiled substrates nonetheless makes them vulnerable to the direct and indirect consequences of shoreline oiling. Adults and chicks may contact beached oil, consume contaminated prey, or be disturbed by cleanup operations (Maccarone and Bizorad 1994); these effects are most often sublethal. Wintering purple sandpipers (*Calidris maritima*), a rocky shoreline specialist, were lightly coated with oil during a spill in Nova Scotia but did not die (Smith and Bleakney 1969) as were gulls (*Larus*) oiled along the coast of England (Dixon and Dixon 1976). Birds probably removed the light coating of oil from their plumage by preening.

The Black Oystercatcher

The black oystercatcher (*Haematopus bachmani*) is a large marine shorebird (length = 44 cm) that is found along rocky shores of the eastern North Pacific and is a fairly common breeding species in the Sound (Isleib and Kessel 1973). Black oystercatchers are completely dependent upon marine shorelines for their life's requirements. They nest above the high tide line and forage in the intertidal zone to obtain food for themselves and to provision their young (Webster 1941a). Black oystercatchers feed predominantly on mussels and limpets but also consume clams, cockles, and other benthic bivalves (Webster 1941b, Hartwick 1976, Andres and Falxa 1995). Because bivalves readily accumulate petroleum hydrocarbons (National Resource Council 1985) and because of its tidal zone dependency, the black oystercatcher was at risk from direct and indirect effects of the *Exxon Valdez* oil spill. Information from boat surveys in 1989 indicated that the Spill reduced the population of black oystercatchers inhabiting the Sound (Klosiewski and Laing 1994). Therefore, I analyzed data collected in 1989 and 1990 to determine the immediate effects of the Spill on reproduction of black oystercatchers and initiated work in 1991 to determine if any negative effects on reproduction were continuing. Because habitat protection was identified as an important restoration option for species injured by the Spill, I collected information on the habitat requirements of black oystercatchers to guide habitat restoration and acquisition. In 1992 and 1993, I focused on pairs inhabiting Knight Island to determine if oil persisting in fine sediments and mussel beds was negatively affecting growth rates of chicks and reproduction of black oystercatchers. Few studies have documented the immediate, or persistent, nonlethal effects of spilled crude oil on a population of free-living birds. Herein, I report the results of my field studies and analyses.

CHAPTER 1

IMMEDIATE EFFECTS OF THE *EXXON VALDEZ* OIL SPILL ON BREEDING BLACK OYSTERCATCHERS

Introduction

The spillage of 42 million L of Prudhoe Bay Crude oil by the *T/V Exxon Valdez* eventually fouled 563 km of shoreline in Prince William Sound, Alaska (Galt et al. 1991). The heaviest deposits of oil were made along shorelines of islands in the central Sound. Oil that contacts the shoreline persists longer than oil that remains suspended in the water column (National Research Council 1985, Galt et al. 1991), and species inhabiting shorelines, such as the black oystercatcher were particularly vulnerable to the direct and indirect consequences of oil that washed ashore.

Because they are completely dependent upon marine shorelines for their life's requirements, black oystercatchers could be negatively affected by shoreline oiling; oystercatchers nest just above the high tide line and forage in the intertidal zone. In Alaska, diets of adults and chicks consist of benthic and epibenthic molluscs (Webster 1941b), mainly mussels (*Mytilus*) and limpets (*Techtura* and *Lottia*). Of all marine molluscs, bivalves most readily accumulate petroleum hydrocarbons (National

Resource Council 1985). Therefore, black oystercatchers had the potential of being directly, through physical contact, and indirectly, through ingestion of contaminated prey, exposed to oil spilled from the *T/V Exxon Valdez*.

To determine if black oystercatchers were immediately affected by the Spill, I analyzed components of reproductive success and changes in the breeding population from 1989 to 1991. Mortality and hydrocarbon contamination of mussels residing within oystercatcher feeding sites were estimated to determine the effects of the Spill on oystercatcher prey.

Study Area and Methods

Ringed by mountains exceeding 3960 m, Prince William Sound, Alaska, (approximately 60°30'N, 147°00'W) encloses the northerly waters of the Gulf of Alaska. Formed by glacial processes, convoluted shorelines, deep fjords (maximum water depth is >870 m), and numerous islands are characteristic of the Sound. Shoreline habitats range from low wave-energy tidal flats to steep, high wave-energy rock walls. Shorelines are influenced by a large amplitude, diurnal tide (mean tidal range is 3.8 m). Summers are cool and winters are moderate; mean annual temperatures range from 2.1° to 4.5°C among areas of the Sound. All seasons are wet (mean annual precipitation ranges from 158 to 445 cm) and windy. Uplands are dominated by rock cliffs, bogs and hemlock (*Tsuga*)-Sitka spruce (*Picea sitchensis*) forests and most surrounding uplands lie within the boundaries of the Chugach National Forest.

Surveys of breeding oystercatcher pairs were made around Green Island and in the vicinity of Port Chalmers, Montague Island (Fig 1). Shorelines on Green and Montague islands consist of tidal flats, mixed sand and gravel beaches, and gravel beaches and are interspersed with rocky, wave-cut platforms. Virtually all shoreline around Green Island was oiled within 6 days after the *T/V Exxon Valdez* ran aground (*Exxon Valdez* Oil Spill Damage Assessment Geoprocessing Group 1989, 1990). The extent of oiling on Green Island varied from light (<10%) to heavy (>50%) among segments of shoreline; shoreline around Port Chalmers remained unoiled. Cleanup methods used on Green Island beaches in 1989 and 1990 included manual pickup, hot water/high pressure washing, and bioremediation. In 1990, nest sites were categorized as disturbed (if any treatment occurred between 15 May and 3 August) or undisturbed by cleanup operations. Virtually all beaches on Green Island were cleaned sometime during 1989.

Field work was conducted from 4 June to 12 July 1989 and from 30 May to 14 August 1991. In 1990, Green Island was visited on 22-23 June and 2-3 August. Two-person crews initially searched the shoreline by boat, or on foot, to determine the presence of breeding oystercatchers. Except for a small area on Montague Island, the same area was surveyed in 1989 and 1991 (analyses account for this discrepancy). For each oystercatcher, the location, breeding status (single, pair, pair with nest), and number of eggs or chicks were recorded. Nests were approached cautiously to avoid attracting aerial predators and most nests were approached from the water to avoid attracting mammalian predators. Nest sites were revisited every 3-10 days to assess their fate.

Several components of breeding success of the black oystercatcher population that inhabited oiled and unoiled shoreline were estimated. Besides enumerating the number of pairs present, I calculated the percentage of all pairs that had active nests. The mean number of eggs produced per pair (all pairs present), the mean clutch size of a pair, and the mean nest success of all pairs (no. hatched ≥1 chick/no. of pairs) were also estimated. Because some pairs initiated nesting before the start of the study (in both years) and nesting phenology was unknown, the maximum number of eggs produced by a pair (within a clutch) was used to calculate eggs/pair and clutch size. I calculated chick loss (no. chicks hatched - no. chicks alive at the end of the study) for 1989 and productivity (no. chicks fledged/no. pairs present) for 1991. For 1990, productivity was estimated for a small sample of pairs that nested on disturbed and undisturbed shorelines on Green Island. The ultimate outcome of each pair, that resulted from either the first or second nesting attempt, was used for estimates of productivity.

To document the effect of the spill on prey populations of oystercatchers, the proportions of living and dead bay mussels (*Mytilus trossulus*) were measured on 1 m^2 quadrats on Green and Montague islands during 1989. Unequal numbers of quadrats were sequentially surveyed on 2 to 4 transects selectively placed in known oystercatcher feeding sites.

Mussels were collected from oystercatcher feeding sites in 1989 for petroleum hydrocarbon analysis. Mussel samples were removed from the shell, placed in sterile jars, and immediately refrigerated. Tissue samples were sent to the Geochemical and Environmental Research Group of the Texas A&M University for gas chromatography-mass spectroscopy determination of aliphatic hydrocarbon (AH) and polycyclic aromatic hydrocarbons (PAH) concentrations. Extraction of tissue samples followed the NOAA Status and Trends Method (Macleod et al. 1985) with minor revisions (Wade et al. 1988). Briefly, tissue samples were homogenized with a Teckmar Tissumizer and a 1- to 10-g sample (wet weight) was extracted with the tissumizer by adding surrogate standards, Na_2SO_4 and methylene chloride, in a centrifuge tube. The tissue extracts were purified by silica/alumina column chromatography to isolate the AH and PAH fractions. The PAH fraction was further purified by HPLC to remove interfering lipids. The quantitative analyses were performed by capillary gas chromatography (CGC) with a flame ionization detector for aliphatic hydrocarbons and a mass spectrometer detector in the SIM mode for aromatic hydrocarbons (Wade et al. 1988).

Dry weight was determined by weighing approximately 1 g of wet sample and placing it into a clean, labeled, preweighed 10-ml beaker. The beaker was placed in a forced air oven at approximately 75° C for 24 hr. After drying, the sample was weighed and % dry weight calculated. Details of analytical methods are in GERG standard operating procedures SOP-8901 to SOP-8905 (*Exxon Valdez* Oil Spill Public Information Office, Anchorage, Alaska).

Indices were calculated to determine if petrogenic hydrocarbons were present in mussel tissues (Manen 1990) and included the pristane:phytane ratio and the Carbon Preference Index; concentrations of the unresolved complex mixture (branched aliphatics), total aliphatic compounds, and total aromatic compounds were also examined (Table 1).

I used a randomization procedure (Manly 1991, including Manly's RT program) to test for differences (D_o) between oiled and unoiled islands in 1989 (two-sample test) and between years on Green Island (one-sample paired test, excluding zero values). The one-sample test is equivalent to contingency table analysis (Snedecor and Cochran 1980:121-124). The number of randomizations (i) for a test was determined as (2^n) for one-sample tests, as ($n!/n_1!n_2!$) for two-sample tests, or was set at a maximum of 5,000 iterations. I also calculated means, or proportions, and standard errors (normal approximations) for all estimated parameters. For comparisons between oiled and unoiled sites in 1989, I constructed alternative hypotheses to indicate that oiling caused a negative effect but used alternative hypotheses of no difference for between-year tests for the oiled site. Because I was primarily interested in testing two differences, that between oiled and unoiled pairs in 1989 and that between years for the oiled site, nominal statistical significance was set at the $P \leq 0.025$ level.

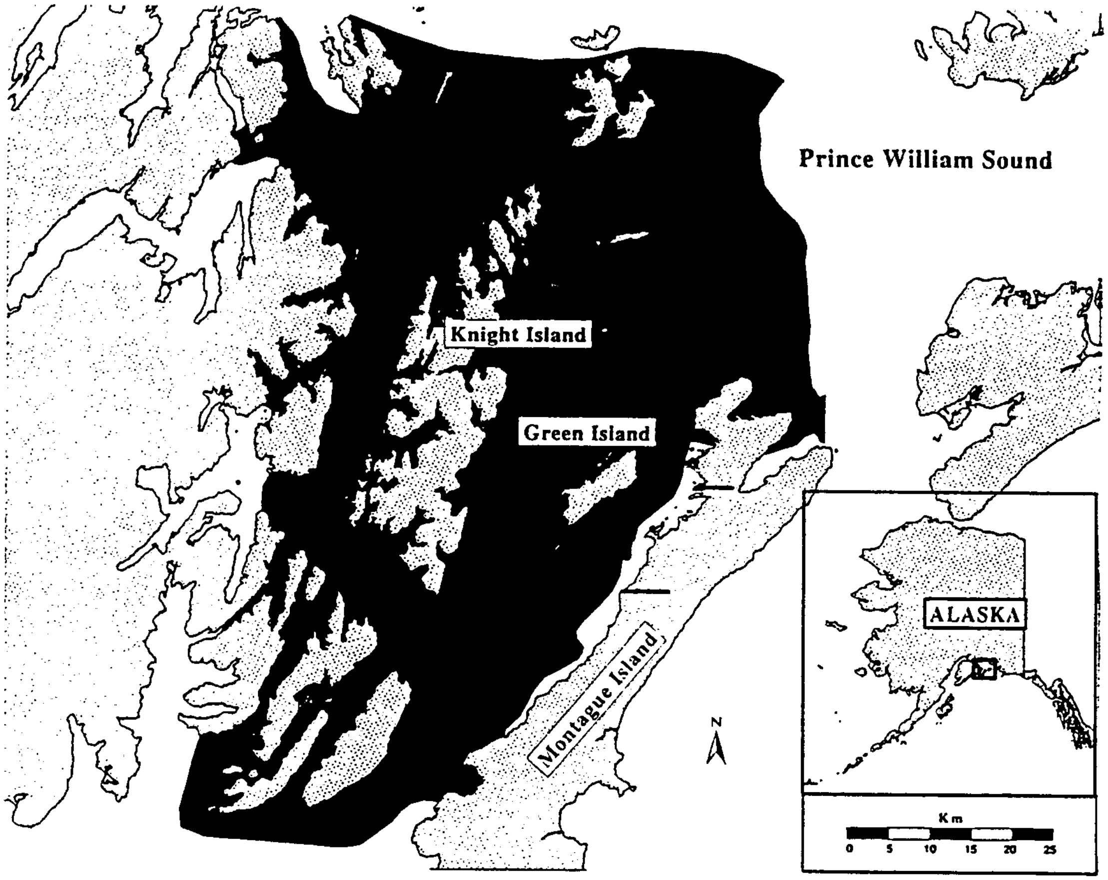

Figure 1. Location of black oystercatcher study area on Green and Montague islands and the area affected by the *Exxon Valdez* oil spill (dark), on March 24, 1989, in Prince William Sound, Alaska.

Table 1. Description of indices used in analyzing petroleum hydrocarbons (ng•g^{-1} dry weight, g•g^{-1} for UCM) in mussel tissue samples and the expected differences between oiled and unoiled matrices (from Manen 1990).

Hydrocarbon index	Index calculation	Expected direction
pristane:phytane ratio	pristane/ phytane	oil<non
carbon preference index	2(C27 + C29)/ (C26 + 2(C28) + C30)	oil<non
total aliphatics	cumulative sum of all aliphatics	oil>non
total aromatics	cumulative sum of all PAHs	oil>non
unresolved complex mixture		oil>non

Results

The number of breeding black oystercatcher pairs occupying Green Island (oiled) increased from 28 pairs in 1989 to 38 pairs in 1991 (36%), whereas the number of breeding pairs on Montague Island (unoiled) changed little (-5%) between 1989 (20 pairs) and 1991 (19 pairs). A significantly lesser percentage of pairs maintained active nests on Green Island in 1989 (61%) than pairs did in 1991 (95%; $D_o = 34\%$ (pair-wise), $i = 2{,}048$, $P = 0.001$) or than pairs did that nested on Montague Island in 1989 (91%; $D_o = 30\%$, $i = 5{,}000$, $P = 0.002$). Consequently, pairs breeding on Green Island in 1989 had significantly fewer eggs present in their nests during June than pairs had on Montague Island in 1989 ($D_o = -0.9$ eggs, $i = 5{,}000$, $P = 0.006$) or than pairs had on Green Island in 1991 ($D_o = -1.1$ eggs (pair-wise), $i = 5{,}000$, $P = 0.001$). The percentage of active pairs in the population and the number of eggs found per pair were similar between years on Montague Island (Table 2). For pairs that were able to maintain nests on Green Island in 1989, clutch size did not differ significantly ($P > 0.2$) from 1991 or from pairs on M-ontague Island in 1989 (Table 2).

The loss of eggs from nests on Green Island led to low nest success. Success of pairs nesting on Green Island in 1989 (44%) was significantly lower than it was in 1991 (71%; $D_o = 40\%$ (pair-wise), $i = 4{,}096$, $P = 0.005$) but was not lower than nest success of pairs on Montague Island in 1989 ($D_o = 9\%$, $i = 5{,}000$, $P = 0.4$).

Table 2. Measures of breeding success (mean ± SE, (*n*)) of black oystercatcher pairs nesting at oiled and unoiled sites in Prince William Sound, Alaska - 1989, 1991.

	Pairs with eggs (%)	Eggs per pair[1]	Clutch size	Nest success (%)
Green Island (oiled)				
1989	60.7 ± 9.2 (28)	1.54 ± 0.26 (27)	2.53 ± 0.15 (17)	44.0 ± 9.9 (25)
1991	94.7 ± 3.6 (38)	2.41 ± 0.14 (37)	2.54 ± 0.10 (35)	71.1 ± 7.3 (38)
Montague Island (unoiled)				
1989	91.0 ± 6.1 (22)	2.38 ± 0.21 (21)	2.63 ± 0.14 (19)	52.9 ± 12.1 (17)
1991	94.7 ± 5.1 (19)	2.10 ± 0.19 (18)	2.18 ± 0.15 (17)	68.2 ± 10.7 (19)

[1] all pairs present

Some oystercatcher pairs were able to maintain nests and hatch young on Green Island in 1989. However, chicks were lost at a significantly greater rate from nests on Green Island (42%, $n = 10$) than were lost from nests on Montague Island (0%, $n = 4$; $D_o = 42\%$, $i = 1{,}001$, $P = 0.024$). Although studies in 1989 were concluded before chicks fledged, the mean number of days chicks were exposed to oil or predation was similar for oiled (12.8 days) and unoiled sites (14.7 days). By 1991, productivity of pairs on Green Island (1.3 chicks fledged/pair, $n = 32$) tended to exceed productivity of pairs on Montague Island (0.6 chick/pair, $n = 17$; $D_o = 0.7$ chick/pair, $i = 5{,}000$, $P = 0.044$). Disturbance associated with cleanup operations affected breeding pairs on Green Island in 1990; productivity of pairs nesting at disturbed sites was significantly lower than productivity of pairs nesting at undisturbed sites (Fig. 2; D_o = 1.4 chicks/pair, i = 5,000, P = 0.010). By 1991, productivity of pairs nesting at those same sites did not differ (Fig. 2; D_o = 0.7 chick/pair, i = 5,000, P = 0.185).

The percentage of dead mussels found in quadrats on Green Island (3.5%) was greater than in quadrats on Montague Island (1.6%) in 1989 (D_o= 1.82, P = 0.0028, i = 5,000). All oil contamination measures of mussel tissue samples from Green Island had significantly higher petroleum hydrocarbon values or concentrations than did mussel samples collected on Montague Island in 1989 (Table 3). Crude oil from the *T/V Exxon Valdez* appeared to be the source of hydrocarbon contamination in mussel tissues in 1989 (Fig. 3).

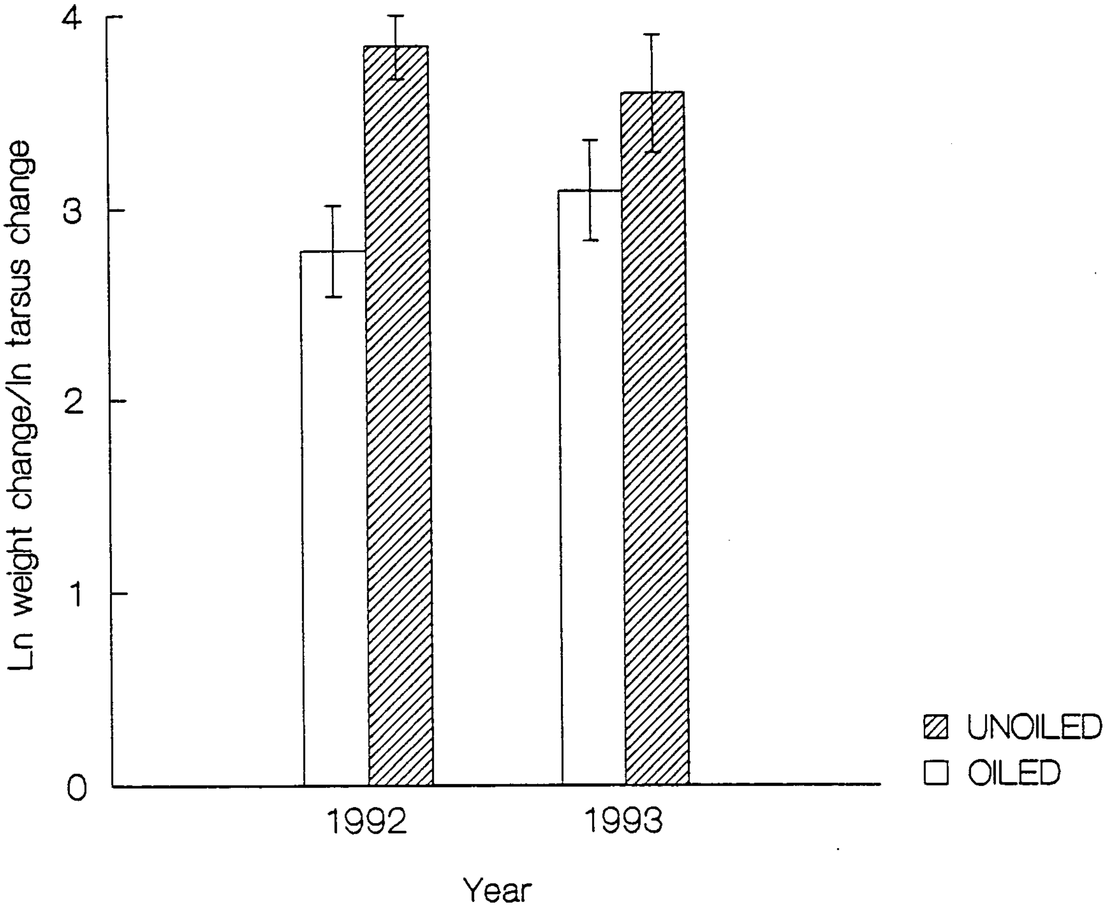

Figure 2. Mean (±1 SE) productivity (chicks/pair) of black oystercatchers nesting at sites disturbed by cleanup operations (*n* = 10) and of those nesting at undisturbed sites (*n* = 6) on Green Island (oiled), Prince William Sound, Alaska, 1990-1991.

Table 3. Mean (±1 SE) petroleum hydrocarbon concentrations (ng•g^{-1} dry weight, g•g^{-1} for UCM) and index values and significance of difference tests (D_o) for mussel samples collected on Green and Montague islands, Prince William Sound, Alaska, in 1989.

	Oiled sites (*n* = 30)		Unoiled sites (*n* = 25)			
Hydrocarbon index	mean	SE	mean	SE	D_o	*P*-value
pristane:phytane	1.49	0.15	19.98	4.33	18.49	0.0002
carbon preference	1.05	0.05	2.39	0.23	1.34	0.0002
total aliphatics	9130	3892	1321	168	-7809	0.0002
total aromatics	321.0	106.6	10.3	2.5	-310.6	0.0002
unresolved complex mixture	171.4	58.6	7.4	1.7	-164.0	0.0002

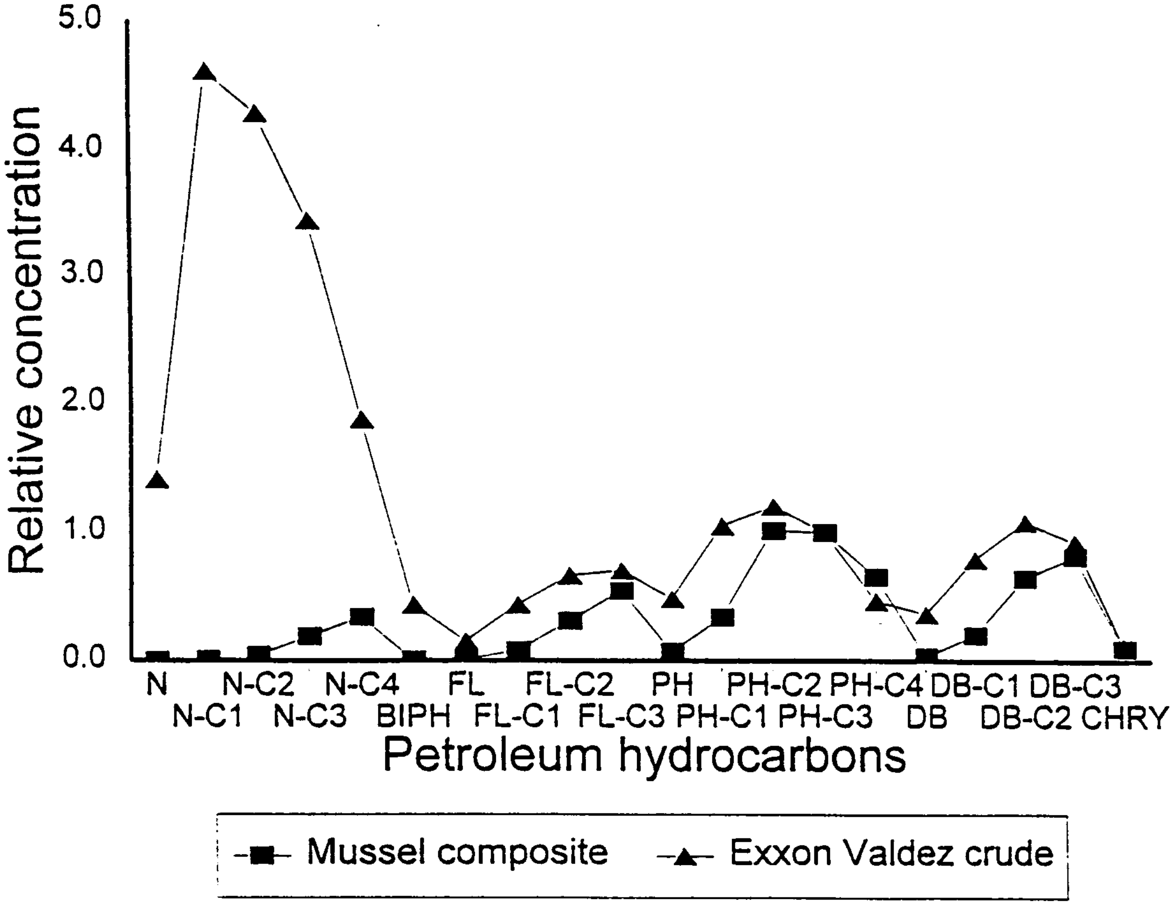

Figure 3. Relative concentrations (ng•g^{-1} dry weight, standardized by C3-phenanthrene) of polycyclic aromatic hydrocarbons in *Exxon Valdez* crude oil and mussel tissue samples (mean) collected from Green Island, Alaska, in 1989 (N = naphthalene, BIPH = biphenyl, FL = fluorene, PH = phenanthrene, BD = dibenzothiophene, CHRY = chrysene).

Discussion

The number of pairs occupying Green Island continued to increase into 1992 (43% from 1989) and into 1993 (61% from 1989; Fig. 4). This rapid change is indicative of a population where large-scale immigration is occurring. Changes in the number of breeding pairs, from 1991 (36%) to 1993, greatly exceeded estimates of population growth of oystercatchers that could be attributable to birth and death processes; annual increases in perturbed populations of Eurasian oystercatchers (*Haematopus ostralegus*), the only documented population growth rates for any *Haematopus* species, averaged only 6% (Nur and Ainley 1992). The percentage of non-breeders in the Green Island population remained low (<10%) after, and during, 1991. Although black oystercatchers were reoccupying other oiled areas in the Sound by 1991 where they were absent during 1989 and 1990 (Day et al. 1995), the number of breeding pairs inhabiting Knight Island (oiled) decreased 8% between 1991 and 1993. Thus, reoccupation by black oystercatchers of habitats disturbed by the Spill varied spatially across the Sound.

Marine birds that inhabit shorelines, particularly rocky shorelines, are less vulnerable than pelagic species to the direct, lethal effects of plumage oiling (Vermeer and Vermeer 1975); shoreline species do not usually become completely coated with oil. Wintering purple sandpipers, a rocky shoreline specialist, were lightly coated with oil during a spill in Nova Scotia but did not die (Smith and Bleakney 1969). Because few oystercatchers occupied territories at the time of the spill, direct mortality of adults was, most likely, minimal. In fact, only 2 oystercatcher carcasses were retrieved in the Sound and no oiled adults were observed on Green Island during 1989 (M. M. Cody, pers. obs).

Most individuals do not maintain year-round territories in the Sound, and failed breeders may have migrated prematurely to wintering areas located in unoiled areas of the Sound (see Chapter 2).

Clearly, however, breeding was disrupted in 1989. A high proportion of pairs (39%) did not maintain nests into June; clutches are initiated as early as the first week of May in the Sound (K. Kuletz, U.S. Fish Wildl. Serv., pers. commun). The proximate cause for the failure of many pairs to maintain nests, or fledge chicks, in 1989 is not known. Although oil was observed on the legs of adult oystercatchers, no eggs appeared oiled in 1989 (M. M. Cody, B. E. Sharp, pers. obs.). A greater loss of chicks, coupled with the observation of oiled chicks in 1989, indicates that some chicks died as a result of contacting oil on nesting or foraging beaches. This result was strengthened by the fact that the unoiled site, Montague Island, supported a greater number of predators and had greater egg and chick losses than Green Island in all years after 1989 (B. E. Sharp pers. obs.; pers. obs.). Otherwise, Green and Montague Island provided similar habitat for oystercatchers; both were dominated by low-sloping shorelines, favored by oystercatchers for foraging (see Chapters 2, 3) and had similar breeding pair densities in 1991 (0.46 and 0.49 bird/km).

The ability of females to produce eggs on Green Island in 1989 indicates that impairments to egg production (Grau et al. 1977, Ainley et al. 1981), caused by consuming contaminated prey or preening oiled feathers, were mild. Concentrations of petroleum hydrocarbons in bay mussels, an important food source for black oystercatchers, were elevated on Green Island in 1989 and provided a pathway of exposure via ingestion. Aromatic hydrocarbon concentrations were much higher than those reported from unpolluted sites (Rainio et al. 1986). Many individual samples had aliphatic hydrocarbon concentrations that greatly exceeded ambient concentrations in mussels collected from Kachemak Bay, Alaska, (Shaw and Wiggs 1980) and those collected from Port Valdez, Alaska, that were subjected to chronic petroleum discharge (Shaw et al. 1986). High levels of productivity of pairs after 1990 suggests that ingestion of oil had little effect on the subsequent reproductive performance of oystercatchers on Green Island.

Evidence from Green Island in 1990 indicates that disturbance from cleanup operations was directly responsible for nest failure and chick mortality of oystercatcher pairs. More than 10,000 workers were deployed in 1989 to clean up floating and beached oil (U. S. Environmental Protection Agency 1989) and they certainly represented a major disturbance to species inhabiting shorelines. In the absence of disturbance, productivity increased. At most, 27 chicks were produced on Green Island in 1989, whereas 50 chicks were produced in 1991. Oystercatchers live long (≥15 years; Andres and Falxa 1995), and the effects on the population of a single, or dual, breeding season failure are minimal. No reduction in the survival of bald eagles nesting in oiled areas of the Sound was documented, and recovery to pre-Spill levels was estimated to be 3 years (Bowman et al. 1995). Although breeding is temporarily disrupted, cleanup and restoration of oiled beaches probably benefit long-lived species that inhabit shorelines.

Although the Spill, and associated cleanup operations, disrupted breeding and caused egg and chick mortality in 1989 and 1990, the magnitude of the effect of shoreline oiling is difficult to assess across the entire oystercatcher population. Differences in habitat selection and corresponding breeding density and the important influence of predation on productivity of black oystercatchers confound a determination of precise estimates of Spill effects on black oystercatcher populations. These differences could lead to differences in the recovery of breeding success, and ultimately recruitment, among segments of the population.

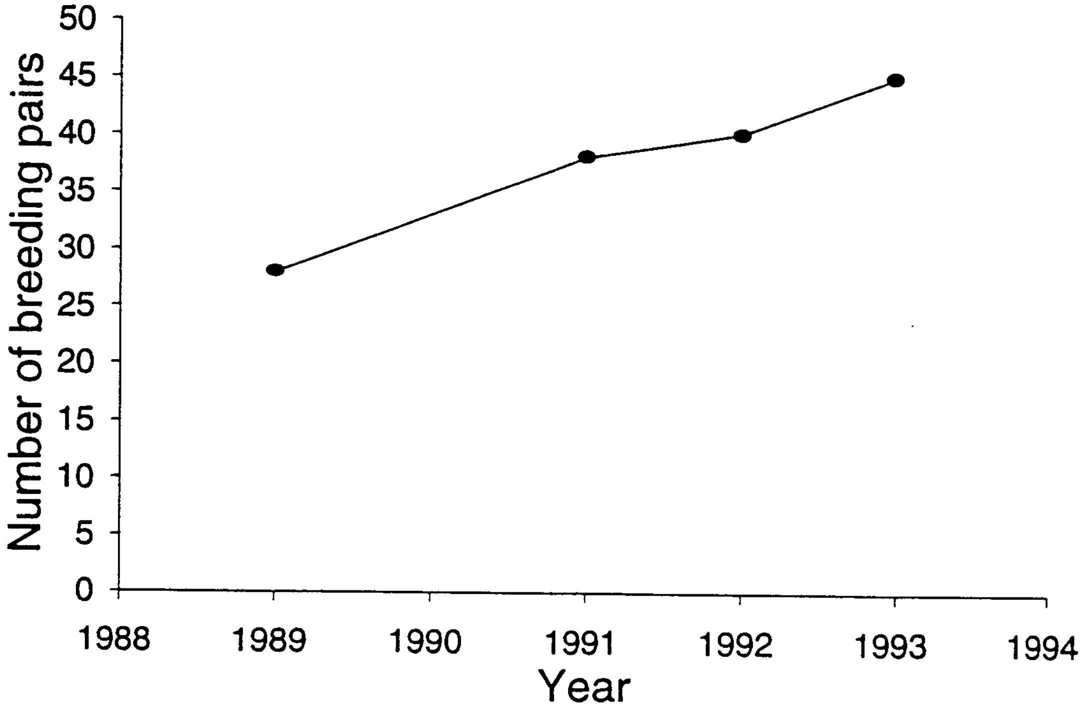

Figure 4. Reoccupation of Green Island by breeding black oystercatcher pairs after the *Exxon Valdez* oil spill, in March 1989, in Prince William Sound, Alaska.

CHAPTER 2

HABITAT REQUIREMENTS OF BREEDING AND WINTERING BLACK OYSTERCATCHERS

Introduction

The black oystercatcher is generally considered an inhabitant of rocky shorelines in southcoastal Alaska (Isleib and Kessel 1973) and throughout its range (Hayman et al. 1986, Paulson 1993). Indeed, its southern range limit coincides with a transition from rocky shores to sandy beaches along the Pacific coast (Jehl 1985), and individuals are absent from sandy or muddy shorelines in the Pacific Northwest (Paulson 1993). Besides nesting on solid, rocky substrates of boulders and small islands (Paulson 1993), black oystercatchers also use gravel and cobble beaches to nest and feed. Pairs nested on glacial moraines (gravel/cobble) in Glacier Bay, Alaska, (Lentfer and Maier 1995) and large winter flocks fed on sand and gravel tidal flats, which supported an abundant mussel population, in British Columbia (Hartwick and Blaylock 1979). Despite these general impressions, no study has determined the habitat requirements of black oystercatchers on a large scale anywhere within their range. For example, the most well-studied population of breeding black oystercatchers inhabits a mere 7.7-ha island (Cleland Island) in British Columbia (Hartwick 1974, Groves 1984, Vermeer et al. 1992a).

After preliminary assessment conducted in 1989 (Klosiewski and Laing 1994), the black oystercatcher population inhabiting the Sound was determined to have been injured by oil spilled from the *T/V Exxon Valdez*. Oil that washed ashore directly killed adults and chicks and negatively affected their reproduction (see Chapter 1); 563 km of shoreline were fouled by oil in the Sound. Shoreline oil also indirectly affected oystercatchers by killing or contaminating their intertidal food sources. Because acquisition and restoration of shoreline habitat were identified as possible actions to aid the recovery of injured species (*Exxon Valdez* Oil Spill Trustee Council 1991), a knowledge of the habitat requirements of injured species was needed before such restoration actions could be initiated. Therefore, I undertook a study to determine the general habitat requirements of breeding black oystercatcher pairs and, to a limited extent, the habitat requirements of wintering oystercatcher flocks.

Study Area and Methods

I mapped the locations of all breeding black oystercatcher pairs in the vicinity of Green, Montague, and Knight islands from 1991 to 1993 (Figures 5-7). The study area was divided into natural, geographic units to produce estimates of the oystercatcher density. The maximum number of distinct nesting sites/breeding pairs occurring in these units, over all 3 years, was used to calculate density (this measure of density should give a good assessment of suitable breeding habitat within a unit). Designation of shoreline types within the unit was taken from maps generated by the *Exxon Valdez* Oil Spill Damage Assessment Geoprocessing Group (1989, 1990) and included: exposed wave-cut platforms, exposed rocky, sheltered rocky, gravel, mixed sand and gravel, and tidal flats. Tidal flats were restricted to the Port Chalmers area of Montague Island and to Gibbon Anchorage on Green Island and were very similar in composition to mixed sand and gravel beaches. Tidal flats, exposed wave-cut platforms, and mixed sand and gravel beaches were considered gradual shoreline types. I used 15-min topographic maps to measure the maximum shoreline elevation at points 0.3 km perpendicular to the shoreline where each 1.609 km (1 mile) section-line intersected the waters of the Sound. Only section-lines >0.3 km apart were included in the sample. I also used 15-min topographic maps to count the number of offshore rocks and islets (width ≤100 m) present within 0.3 km of the shore in each unit. I recorded the locations of all unique nest sites used by pairs over the 3-year period as along straight sections of shoreline of large islands, on points of large islands, on small islands (100-500 m in diameter), or on islets (≤100 m diameter). Periodic observations of adult foraging behavior and habitat use were made in conjunction with other aspects of this study (see Chapter 3). I used Manly's (1991) two-sample randomization procedure to test for differences in oystercatcher density between island groups and I used regression analysis to evaluate habitat predictors of oystercatcher density. I incorporated quantitative information into a general, qualitative habitat model for black oystercatchers.

I used transect counts of oystercatchers collected on Sound-wide boat surveys to test the habitat model I generated from intensive counts of breeding pairs made on

Knight, Green, and Montague islands. Boat survey procedures involved counting all oystercatchers observed on a set of randomly-selected shoreline transects that were distributed throughout oiled (in 1989) and unoiled areas of Sound. Although transect lengths varied between 2.5 and 15.8 km, no relationship existed between transect length and numbers of birds encountered (Klosiewski and Laing 1994). Procedural details of boat surveys are provided in Klosiewski and Laing (1994) and in Agler et al. (1994). Counts from 1990, 1991, and 1993 were used but counts from 1989 were ignored (survey effort was reduced and shoreline disturbance was high in 1989). Based on information gathered on Knight Island (see results), designation of shoreline type and counts of offshore islets were made for transects in the western Sound as described above; long stretches of gradual shoreline were absent in this area of the Sound. Because information on shoreline type was not available for the northeastern area of the Sound, I classified the elevation of each shoreline transect as gradual or steep. Gradual shorelines were those where the elevation within 0.3 km of the shoreline did not reach the 62-m contour along >33% of the transect. Steep shorelines were those where the elevation exceeded 62 m within the same bounds. Because numerous transects contained counts of 0 birds, I used nonlinear regression (Probit model, where $y = 0,1$; Agresti 1990:102-104) to test the relationship between habitat variables and the occurrence of black oystercatchers on transects.

On 11 February 1994, a U. S. Fish and Wildlife Service pilot-biologist and I conducted a fixed-wing aerial survey to locate and enumerate wintering flocks of oystercatchers. We surveyed areas in the Sound where wintering flocks were observed in previous years or where concentrations were found in late August 1993. Shorelines in the following areas were surveyed: 1) Port Etches/Constantine Harbor — Hinchinbrook Island; 2) Rocky Bay, Zaikof Bay, Stockdale Harbor, Port Chalmers — Montague Island; 3) Gibbon Anchorage — Green Island; 4) Little Green Island; 5) Channel Island; 6) Bay of Isles — Knight Island; and 7) north Chenega Island. The survey was conducted from 1200 to 1500 hr under clear skies, with a high tide, and with moderate winds (15-40 km/hr).

Results

The Knight Island vicinity was characterized by steep, rocky shorelines and differed markedly from the gradual, gravelly shorelines of the Green and Montague Island vicinity (Table 4). Gradual shorelines (wave-cut platforms, mixed sand and gravel beaches, tidal flats) constituted 63% of the shoreline on Green and Montague islands, whereas steep shorelines (exposed rocky, sheltered rocky) constituted 62% of the shoreline on Knight Island. Consequently, the density of breeding pairs on Green and Montague islands was a magnitude higher than the density on Knight Island (Table 4) and the difference in density was highly significant ($D_o = 1.36$ pairs/km, $i = 5{,}000$, $P = 0.0002$). The randomization test procedure did not weight densities by shoreline length in geographic units; a weighted estimator was not required because the number of pairs was not linearly related to shoreline length. Between island groups, breeding oystercatcher pairs differed in their use of shoreline habitat for nesting and foraging (Table 5). Most nest sites used by pairs on Knight Island (80%, $n = 56$) were located on islets (≤ 100 m in diameter), whereas most nest sites on Green and Montague islands (97%, $n = 79$) were located on points or along straight sections of large and small islands. In both areas, however, gradual shorelines were important for foraging. Therefore, I used the density of islets and the proportion of gradual shoreline as predictors of breeding pair density on Knight Island.

On Knight Island, breeding pair density of black oystercatchers was highly dependent (Table 6; $F = 26.8$, $P = 0.001$, $n = 14$, $R^2_{adj} = 0.80$) on the proportion of gradual shoreline ($t = 3.69$, $P = 0.004$) and on the density of islets ($t = 7.24$, $P = 0.001$) in geographic units. The density of islets alone was a good predictor of oystercatcher density ($F = 19.7$, $P = 0.001$, $n = 14$) but explained much less of the variation in oystercatcher density than did the 3-parameter model ($R^2_{adj} = 0.59$). Explanatory variables were not correlated and data did not violate assumptions of fitting a linear model.

Density of oystercatchers calculated from boat survey transects located along the steep shorelines in the northeastern Sound ($\bar{x} = 0.04$ bird/km, $n = 20$) differed significantly ($D_o = 0.28$ bird/km, $i = 5,000$, $P = 0.0002$) from oystercatcher density calculated from transects located along gradual shorelines ($\bar{x} = 0.32$ bird/km, $n = 20$). In the western Sound, the proportion of gradual shorelines (Wald's $X^2 = 4.33$, $P = 0.037$) and the density of islets (Wald's $X^2 = 6.06$, $P = 0.014$) were significant predictors of oystercatcher occurrence (Table 7; $n = 91$). Predictors were not correlated, and inspection of residuals indicated no violations of model assumptions.

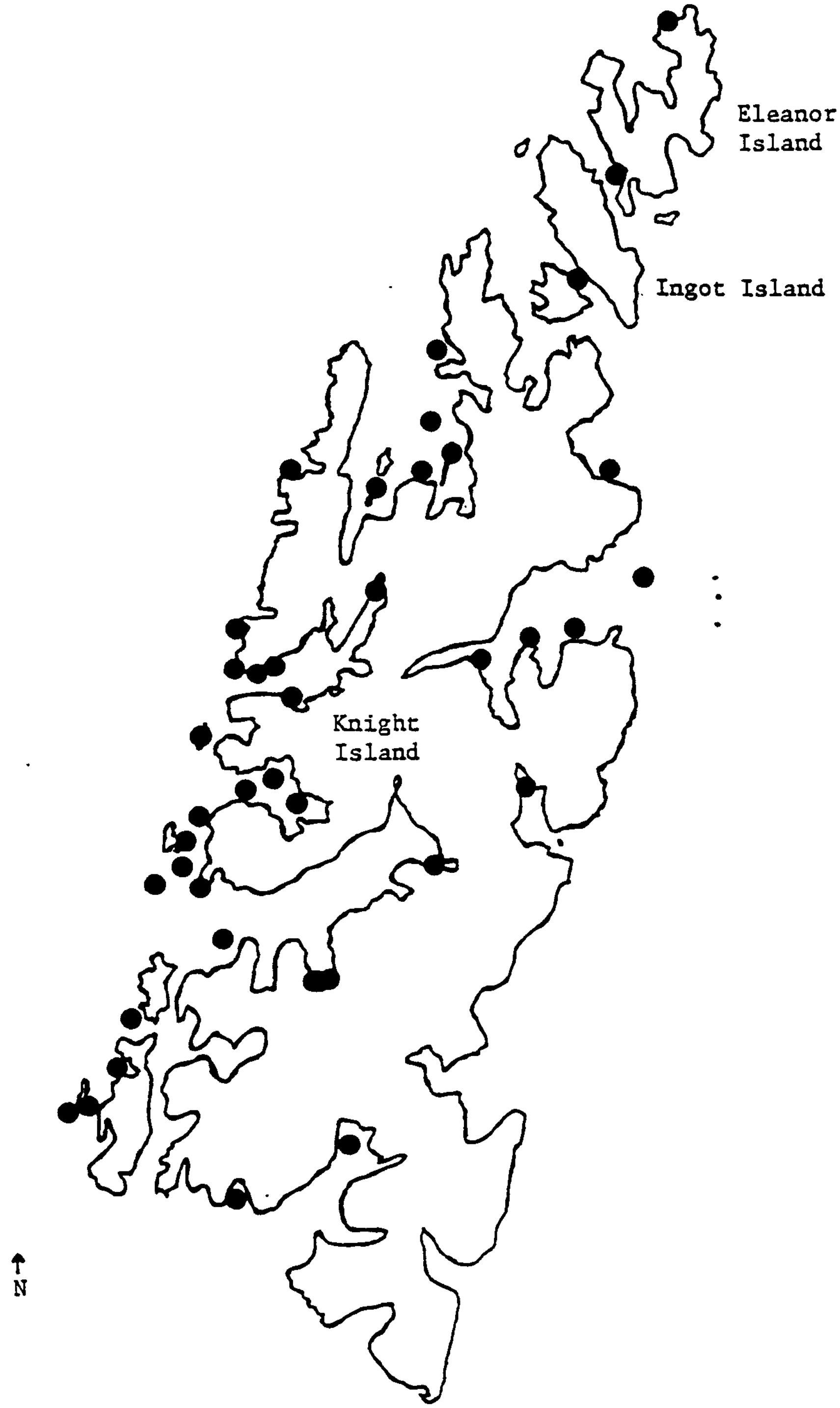

Figure 5. Locations of black oystercatcher nest sites in the vicinity of Knight Island, Prince William Sound, Alaska, 1991-1993.

Figure 6. Locations of black oystercatcher nest sites in the vicinity of Green Island, Prince William Sound, Alaska, 1991-1993.

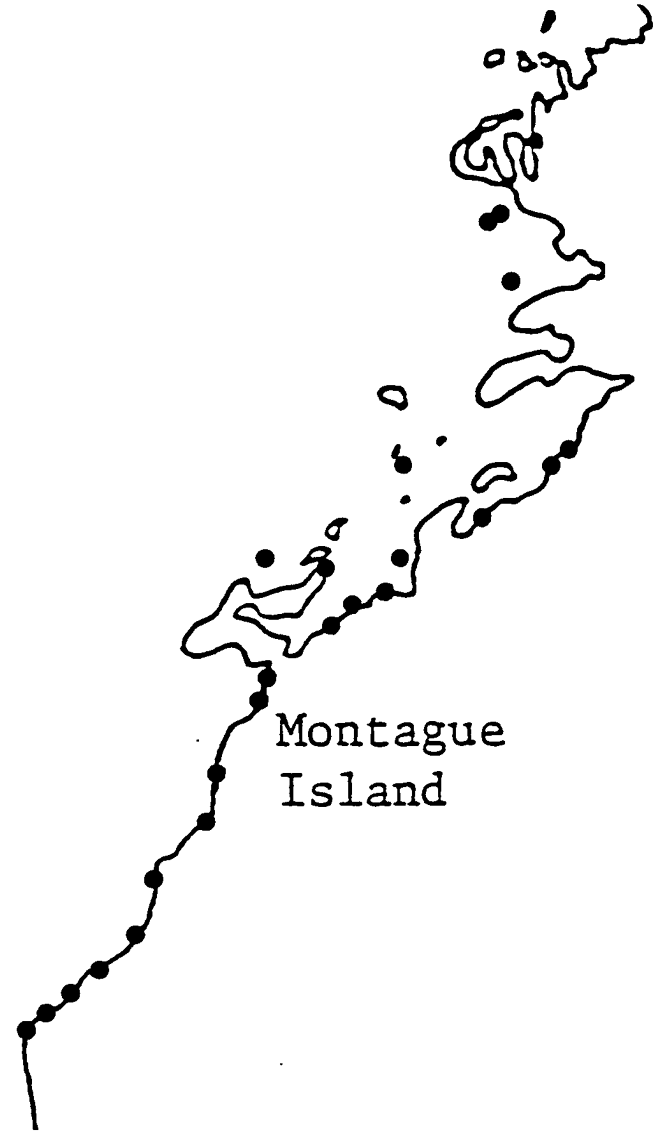

Figure 7. Locations of black oystercatcher nest sites in the vicinity of Montague Island, Prince William Sound, Alaska, 1991-1993.

Table 4. Shoreline characteristics of Knight and Green/Montague islands and density of breeding pairs of black oystercatchers, Prince William Sound, Alaska, 1991-1993.

				Percentage of shoreline				
Island area	Total km of shoreline	Mean no. of islets/km	Mean elevation (km)	exposed/ sheltered rocky	wave-cut platforms	gravel	mixed sand and gravel/ tidal flats	Mean breeding pairs/km (*n*)
Green/Montague	109	15.0	0.034	0.2	27.4	34.8	35.7	0.63 (69)
Knight	499	22.9	0.144	61.6	6.6	5.2	26.2	0.08 (40)

Table 5. Characteristic use of shoreline types by black oystercatchers for foraging and nesting among islands of Prince William Sound, Alaska, 1991-1993.

Island	Nesting	Foraging
Montague/Channel	gravel beach berm	wave-cut platforms
Gibbon Anchorage/ Port Chalmers	wave-cut platforms	mixed sand and gravel beaches
Green/Little Green	wave-cut platforms	wave-cut platforms
Knight (bays)	sheltered rocky	mixed sand and gravel beaches
Knight (outer shorelines)	exposed rocky	exposed rocky

Table 6. Parameter estimates and standard errors for proportion of gradual shoreline and islet density (islet) and standard errors of predicted values (SE[$\hat{y}$]) for Knight Island and Sound-wide models of breeding black oystercatcher density, 1990-1993.

Model	Variable	Coefficient	SE (coef.)	SE ($\hat{y}$)
Knight Island pairs - linear	constant	-0.053	0.024	0.029
	gradual	0.206	0.056	
	islet	0.049	0.007	
Knight Island pairs - linear	constant	0.018	0.020	0.042
	islet	0.040	0.009	
Sound-wide counts - Probit	constant	-0.930	0.296	-
	gradual	1.099	0.528	
	islet	0.210	0.085	

Five flocks containing 157 oystercatchers, and averaging 31 individuals, were found in the eastern Sound during the 1994 winter aerial survey (Table 7). No birds were located at areas on Knight Island or Chenega Island where flocks were observed in August 1993. Generally, oystercatcher flocks observed in February contained fewer members than flocks observed in the same areas in mid- to late August. All oystercatcher flocks in the eastern Sound occurred in areas where gradual shorelines were prevalent.

Table 7. Location and number of individuals in black oystercatcher flocks located in eastern Prince William Sound, Alaska, in February, 1994.

Location of flock	Number of individuals
Constantine Harbor, Hinchinbrook Island	30
Rocky Bay, Montague Island	12
Stockdale Harbor, Montague Island	40
Port Chalmers, Montague Island	30
Gibbon Anchorage, Green Island	45
Total of all areas	157

Discussion

Sound-wide information from boat survey transects corroborated the prediction made from local, Knight and Green/Montague islands information that 1) densities of black oystercatchers would be higher along gradual shorelines than along steep shorelines, and 2) densities in areas of steep shorelines would be dependent on the density of offshore islets and the proportion of gradual shorelines. This agreement is strengthened by the fact that boat-survey methods were less accurate, and potentially more variable, in enumerating breeding birds. Boat survey estimates on sections of Knight Island were lower than my pair counts, and large chicks were often included in the total counts of oystercatchers. In other portions of their range, local densities of breeding oystercatcher pairs are highest on small, flat islands: 4.6 pairs/km on Destruction Island, Washington (Nysewander 1977), 14.0-22.8 pairs/km on Cleland Island, British Columbia (Vermeer et al. 1992a), and 10.8-53.5 pairs/km in Glacier Bay, Alaska (Lentfer and Maier 1995).

Gradual, gravelly shorelines most likely expose a greater surface area for foraging during a falling tide than do steep, rocky shorelines and hence render more prey available to oystercatchers at any given time. Some prey species, mussels for example, might occur in high densities on gradual shorelines (see Chapter 3) and may be an important food source for wintering black oystercatchers (Hartwick and Blaylock 1979). Use of gradual shorelines is limited in winter, however, and flocks exhibit strong fidelity to specific wintering sites. Constantine Harbor, Hinchinbrook Island, has hosted wintering flocks since at least 1971 (Isleib and Kessel 1973). Flocks wintering on Kodiak Island, Alaska, use bays that are characterized by gradual shorelines and are present year after year (Christmas Bird Count data). Counts of oystercatchers made during boat surveys of Montague Island and Green Island in February 1996 (T. Bowman, U.S. Fish Wildl. Serv., unpubl. data) differed by only 3 individuals from my aerial count made in 1994, and flocks were found in the same areas in both years. Counts made in February suggest that most of the breeding population (75%) migrates out of the Sound to spend the winter. Clearly, delineation and protection of winter concentration areas in, and beyond, the Sound would ensure the recovery of the black oystercatcher populations injured by the Spill.

In areas of steep shorelines, oystercatchers rely on small offshore islets to provide adequate nesting and foraging sites. Local and Sound-wide models were consistent in their prediction that oystercatcher density is dependent on the density of islets along a shoreline. The high use of offshore islets as nest sites by breeding pairs suggests that oystercatcher density along steep shorelines may be limited by the presence of islets that are appropriate for nesting. Predation on eggs and young is significant and is probably a strong selective force for nesting on offshore islets (Hartwick 1974, Nysewander 1977, Campbell et al. 1990) and may contribute to local patterns of distribution among shoreline habitats.

Suitable shoreline on Knight Island may have been unoccupied by breeding pairs because of high predator densities; common raven (*Corvus corax*) density was much higher on Knight Island than on Green Island (see Chapter 3). Colonies of large *Larus* gulls can also affect oystercatcher population dynamics and habitat use. On Cleland

Island, British Columbia, a recent increase in the number of glaucous-winged gulls (*L. glaucescens*) has been coupled with a decrease in the number of breeding black oystercatchers (Vermeer et al 1992b). Black oystercatcher pairs in the Sound either avoided nesting around glaucous-winged gull colonies or never raised a successful brood when they did nest nearby.

Besides the shoreline characteristics I have identified, other factors may influence habitat use by black oystercatchers. The negative effects of human or feral animal disturbance on breeding pairs are well documented. Although disturbance by humans and domestic animals precluded black oystercatchers from nesting on South Farallon Island, California, for 100 years, 20 pairs reestablished territories within 5-7 years after major disturbances were eliminated (Ainley and Lewis 1974). A similar response was noted on Destruction Island, Washington, where oystercatchers increased from 4 to 12 pairs within 7 years of lighthouse automation (Nysewander 1977). Increased human activity around Sitka, Alaska, was thought to have contributed to a decline from 102 individuals in 1940 to 4 individuals in 1985 (J. D. Webster, pers. commun.).

Black oystercatchers may be absent from otherwise suitable shorelines because of the presence of introduced Arctic foxes (*Alopex lagopus*) and red foxes (*Vulpes vulpes*); occurrence of black oystercatchers on islands in the Aleutian chain, Alaska, is a good indicator of the absence of foxes (E. Bailey, U.S. Fish Wildl. Serv., pers. commun.). Although previously absent as breeders, black oystercatchers nested along shorelines in the Shumagin Islands the first year after foxes were removed from the islands (Byrd et al. 1996).

Because numerous factors can influence habitat use by black oystercatchers, I integrated information on shoreline characteristics and shoreline disturbance into a qualitative, decision tree-type model to evaluate potential breeding habitat (Figure 8). Because the effects of gull colonies are somewhat ambiguous and because predator populations are difficult to measure, I did not include these factors in the model. Information on shoreline characteristics and gross disturbance should be easy to obtain for most coastal areas of Alaska. The model will provide a tool for managers to assess the value of shoreline segments to black oystercatchers and to help guide habitat acquisition and restoration for this injured species. The model should be applicable throughout the Spill area and may apply to all coastal areas of southeastern and southcentral Alaska. Information recently collected on shoreline boat surveys by the U. S. Fish and Wildlife Service could be used to test the applicability of the model to Southeast Alaska and to refine the structure of a quantitative, predictive model of black oystercatcher habitat use.

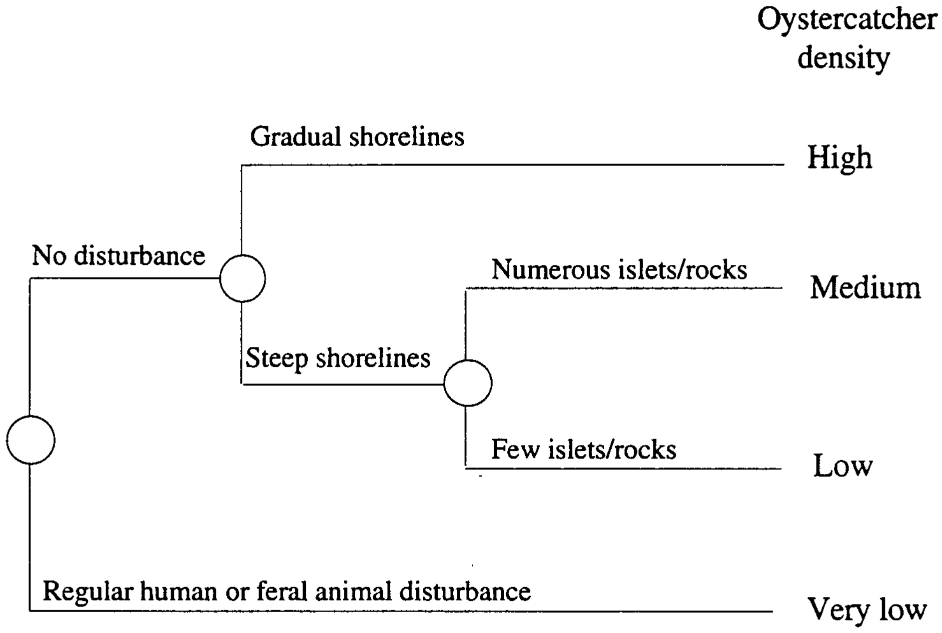

Figure 8. Qualitative decision tree model to assess potential black oystercatcher breeding habitat in areas of southcoastal Alaska that were affected by the *Exxon Valdez* oil spill.

CHAPTER 3

EFFECTS OF PERSISTENT SHORELINE OIL ON THE REPRODUCTIVE SUCCESS, CHICK GROWTH RATES, AND FORAGING ECOLOGY OF BLACK OYSTERCATCHERS

Introduction

The persistence of oil spilled by the *T/V Exxon Valdez* in some mussel beds of Prince William Sound, Alaska, raised questions regarding the chronic exposure of intertidal mussel consumers to petroleum hydrocarbons. Because mussels are important food for intertidal consumers and are an important structuring element of intertidal communities, the *Exxon Valdez* Oil Spill Interagency Shoreline Cleanup Committee intentionally avoided cleaning oiled mussel beds in 1989 (Babcock et al. 1994). In 1991, potential and known oiled mussel beds were identified by the National Oceanic and Atmospheric Administration (NOAA) and the Alaska Department of Natural Resources in 1991 (Weiner and Slocomb 1991). In 1992, personnel from the Alaska Department of Fish and Game (*Exxon Valdez* Oil Spill Restoration Project Number 71) reviewed U.S. Coast Guard cleanup files to identify additional mussel beds that might contain persistent oil. Samples collected in 1991 from sediments underlying mussel beds in the Sound had total hydrocarbon concentrations that were >10,000 ppm wet-weight oil equivalents (measured by ultraviolet fluorescence; P. M. Rounds, NOAA, pers. commun.). Within the Knight Island vicinity, sediments with the highest hydrocarbon concentrations were located on Disk and Eleanor islands and in Herring Bay and Bay of Isles. Mussel beds that retained oil into 1991 (persistently oiled) most often occurred on low wave-energy, porous substrates; after a spill, these shoreline types are prone to long-term oil retention (National Research Council 1985). If mussels were contaminated from oil remaining in sediments, avian consumers might be exposed to persistent oil through mussel ingestion.

Although ingestion of contaminated prey is seldom lethal, the deleterious, sub-acute effects of oil ingestion on adults and chicks are well documented (see Introduction). Most studies addressing the effects of oil ingestion on birds have involved dosing either free-living or captive individuals with known quantities of oil. Virtually no studies have documented the effects of long-term exposure of birds to oil that persists on shorelines after an oil spill. Would birds exposed to persistent shoreline oil, via trophic pathways, be as negatively affected as dosed individuals?

Breeding black oystercatcher pairs establish well-defined, composite feeding and nesting territories and occupy the same sites year after year. Oystercatchers exclusively use marine shorelines for feeding. Throughout the oystercatcher's range, mussels constitute an important component of its diet and are consumed by adults and fed to young (Andres and Falxa 1995). The obligate use of intertidal foraging areas coupled with intensive use of mussels as a food source make black oystercatchers susceptible to oil that persisted in mussel beds and other fine sediments into 1992 and 1993. Therefore, I undertook a study to assess the potential of exposure of black oystercatchers to oil, via diet and foraging behavior, and to determine the effects of persistent shoreline oil on foraging behavior, reproductive success, and growth of chicks.

Study Area and Methods

Study Area

In 1991, I located 35 composite nesting/foraging sites of breeding black oystercatcher pairs along 430 km of shoreline of the Knight Island archipelago. I compared these sites to known or suspected locations of oiled mussel beds to identify the potential for exposure. Oystercatcher foraging and nesting sites that overlapped known and potentially oiled mussel beds were chosen for study; remaining oystercatcher sites in the Knight Island area were used for comparison. Northern and eastern shorelines of the Knight Island archipelago were heavily impacted by oil spilled from the *T/V Exxon Valdez* in 1989 (see Fig. 9) and had a high potential for providing continued exposure of oystercatchers to oil. To determine broad scale effects of persistent shoreline oil on the oystercatcher population, I divided the study area into 2 areas; one with persistently oiled shorelines present in the local vicinity and the other with no persistently oil shorelines present in the vicinity (Fig. 9). Exact locations of all nest sites were archived with the U. S. Fish and Wildlife Service, Nongame Migratory Bird Project, Anchorage, Alaska. All areas designated as persistently-oiled were heavily impacted by oil in 1989.

Field work in the Knight Island archipelago was conducted from May to August during 1992 and 1993; some information collected in 1991 was also included (see

Chapter 1). Limited information also was collected on oystercatchers nesting on Green Island during this period (see Fig. 6). Although Green Island was heavily impacted by oil in 1989, little residual oil remained in shoreline sediments in 1992 (M. M. Babcock, NOAA, pers. commun.; pers. obs.).

Potential for Exposure

To further determine the potential for exposure of black oystercatchers to persistent shoreline oil, I quantified their foraging behavior and habitat use. Prey choice of adults foraging during tidal minima was measured to determine their use of mussels as a food source. Consumption of shelled and de-shelled prey items was enumerated; observers practiced identifying de-shelled prey items by flesh color and shape prior to data collection. Lengths of prey items (scaled to tenths of the bill length) were also estimated; size determination of prey items has been accomplished in other oystercatcher studies (Cayford and Goss-Custard 1990). Specific foraging sites were delineated and distances (m) from the nest to the centers of foraging sites were determined. Shoreline substrates of foraging sites were recorded as gravel (mixed sand and gravel beaches, gravel beaches) or rocky (sheltered rocky shores, exposed rocky shores, exposed wave-cut platforms).

Mussel densities at oystercatcher foraging sites, within the rockweed (*Fucus*) zone, were determined by counting individuals ≥ 10 mm, in situ, in 9-10 randomly selected, circular plots (177 cm^2). Preliminary observations made in 1991 indicated that oystercatchers seldom eat mussels <10 mm. Mussels also were counted in 3 plots placed at 10-m intervals, within the rockweed zone, to each side of the foraging site (for an additional 6 plots). Shoreline slope, determined by compass clinometer, and substrate were recorded at each plot.

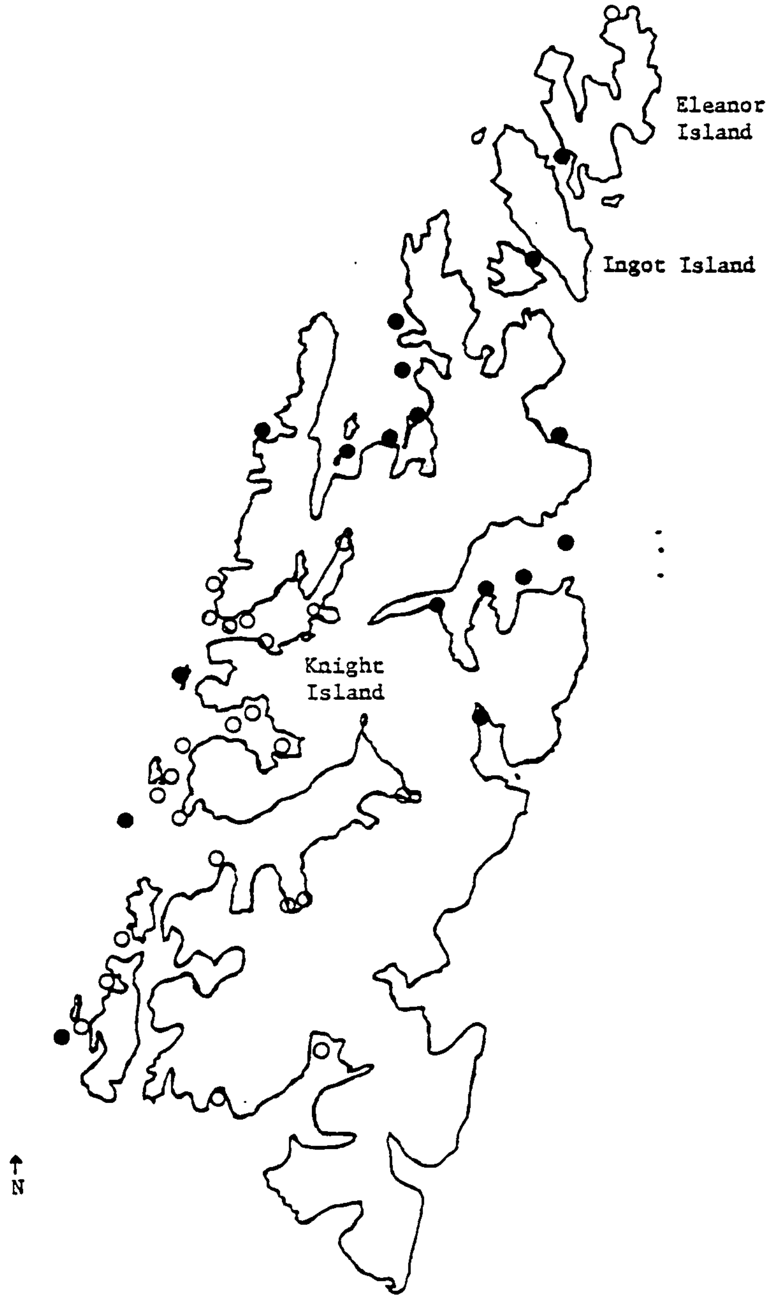

Figure 9. Locations of nest sites of black oystercatchers in areas of persistently oiled shoreline (●) and those in areas of unoiled shoreline (○) on Knight Island, Prince William Sound, Alaska, 1992-1993.

Foraging sites suspected of containing oil were searched for visible and olfactory indications of oil. If oil was detected, triplicate sediment samples, each consisting of 6-10 subsamples, were collected along a 15- to 50-m transect laid through the densest part of the mussel bed 0 to 2 cm below the mussels. Triplicate mussel samples (each consisting of 6-10 mussels) also were collected from the same areas as the sediments. Samples were kept frozen in the field and transferred to NOAA's Auke Bay Laboratory at the end of August. Sediment samples were screened by UV fluorescence and mussel samples were analyzed by gas chromatography-mass spectroscopy (see Babcock et al. 1994 for details).

To determine if oystercatchers would avoid consuming oiled prey, I presented adults with a platter of 2 similar-sized, living limpets (*Techtura scutum*) ranging from 20 mm to 25 mm. After gluing the limpets to a flat rock, I coated one of the individuals with weathered Prudhoe Bay crude oil and presented the other naturally. The coated individual was always the slightly larger of the 2 limpets. The platter was periodically checked to determine which limpet had been taken. Only 1 trial was conducted with each oystercatcher pair.

Effects on Foraging Behavior

To determine the effects of oil exposure on black oystercatcher foraging behavior, I made time budget observations of adults. Pairs were observed during an interval 2 hr prior to low tide and 1 hr after low tide, the period when most foraging activity occurs. Continuous watches were kept on each member of the pair and behaviors were recorded as: rest/preen, forage, locomotion (did not include movement associated while directly engaged in other behaviors), incubate/brood, and aggression/interaction (towards other oystercatchers, competitors, or predators). The proportion of time spent foraging, relative to all activities, was calculated for each pair at each nest site. Observations of prey types and prey sizes captured and consumed by adult oystercatchers and the duration of foraging bouts were also recorded during low tide periods. The proportion of time adults spent foraging, foraging rate (number of prey ingested/min), and biomass intake (prey weight/min) were calculated.

Prey size was converted to biomass by regressing wet weights against flesh lengths. Varying numbers of common prey were segregated into 5-mm (limpets, mussels) or 10-mm (clams, and chitons) size classes and the mean weight (g) and length (mm) for each size class were calculated. I used natural log models to regress mean wet weights against mean tissue lengths (Table 8). For limpets, separate regressions for *Lottia pelta*, *Techtura persona*, and *T. scutum* were determined. A combined estimate for all limpet species of the "average" limpet wet weight for each size class was produced by weighting the estimates for each size class by the representation of the species in that size class as determined from shell collections and data from Coastal Habitat Study 1 (A. J. Hooten, Univ. Alas., unpubl. data). Mean chiton weights were calculated in the same way. Although caloric values differ among prey types, caloric content between grazing and filter-feeding prey varied <5% (Cummins and Wuycheck 1971), whereas biomass between prey types varied by several orders of magnitude. Thus, biomass consumed was used as an index of energy available to feeding oystercatchers. Lengths (mm) of mussels and limpets eaten by oystercatchers were measured from shells discarded at their nest sites and were compared to sizes of individuals available in the environment (Coastal Habitat Study data).

Effects on Breeding Success

Components of breeding success were estimated to determine if persistent shoreline oil affected the breeding of black oystercatchers. Two- or 3-person crews initially searched shorelines by boat or on foot to determine the presence of breeding oystercatchers. When nesting pairs were located, the number of eggs or chicks, maximum lengths and widths of eggs, and behavioral response of the adults were recorded. Behaviors were recorded as: aggressive toward intruders, distraction (injury feigning) display, displacement incubation or brooding, standing silently away from nest, or standing and calling away from nest. The 2 first-named behaviors (aggressive, distraction) were considered active, whereas the remainder were considered passive. All nests were approached cautiously to avoid attracting avian predators, and most nests were approached from the water to avoid attracting mammalian predators. Nest sites were revisited every 3-10 days to monitor their outcome.

I calculated the following measures of breeding success: clutch size (no. of eggs/nest attempts), hatching success (no. eggs hatched/no. eggs laid), fledging success (no. chicks fledged/no. eggs laid), nest success (no. of nests that hatched ≥1 chick/total no. of nests), productivity (no. of young/no. of nests), and proportion of females that renested after losing eggs or chicks ≤7 days old. Except for clutch size, where I

Table 8. Length-weight regressions for dominant prey items consumed by black oystercatchers in Prince William Sound, Alaska, 1992.

	Regression equation	R^2
limpets		
Lottia pelta	$\ln(wt) = -8.439 + 3.004 \ln(lg)$	0.985
Techtura persona	$\ln(wt) = -7.902 + 2.840 \ln(lg)$	0.991
T. scutum	$\ln(wt) = -9.276 + 3.268 \ln(lg)$	0.987
mussels		
Mytilus	$\ln(wt) = -10.003 + 2.953 \ln(lg)$	0.995
Modiolus	$\ln(wt) = -9.354 + 2.636 \ln(lg)$	0.997
clams (*Saxidomus*, *Protothaca*)	$\ln(wt) = -10.357 + 3.368 \ln(lg)$	0.998
chitons		
Katharina	$\ln(wt) = -6.775 + 2.246 \ln(lg)$	0.985
Mopalia	$\ln(wt) = -3.789 + 0.118 \ln(lg)$	0.961

determined size for first and second clutches, the ultimate outcome of each pair's nesting effort, which resulted from either the first or second nesting attempt, was used for all analyses of reproductive success.

To determine the abundance of predators living in the vicinity of oystercatcher pairs, I conducted a series of point counts around each nest site. Points were located at 1- km intervals centered on the nest site and each nest site had a maximum of 3 points. At each point, I recorded the presence or absence of common ravens during a 3-min period. Previous observations revealed that adult oystercatchers reacted vigorously to the presence of common ravens in their territories, and I suspected that ravens were the most important predator on eggs and chicks.

I measured egg volumes to indirectly determine if breeding females were exposed, through ingestion, to persistent oil. Maximum length (L) and breadth (B) of eggs were measured in the field, and Nol et al.'s (1984) formula (volume = $0.51LB^2$) was used to covert egg measurements to egg volumes. To reduce the confounding effect of body size on egg size, I calculated the relative difference (%) between the largest ($_1$) and smallest ($_n$) egg within a clutch ($[egg_1 - egg_n]/egg_1$; where n = 2- or 3-egg clutches) laid by females during their first or second nesting attempt. Because the numbers of 2- and 3-egg clutches were similar between oiling groups, one measure of mean relative difference (%) was calculated for all clutch sizes.

Effects on Chicks

Oystercatcher chicks were appropriate subjects for studying exposure of intertidal consumers to persistent shoreline oil, because early life stages of birds are most sensitive to the negative effects of exposure to oil (Holmes and Cronshaw 1977) and chicks are easy to handle. Body weights (g), tarsus lengths (mm), and bill lengths (mm) of chicks at each nest site were measured at 7- to 10-day intervals. Chick ages were determined from plumage characteristics reported by Webster (1942). Position in the tidal cycle when measurements were made and brood size also were recorded. Instantaneous change (r) in body weight and tarsus length (Butler and Lukasiewicz 1979) was calculated according to the following equation (Ricklefs 1983):

$$r = \frac{\ln(m_2) - \ln(m_1)}{t_2 - t_1} \qquad (1)$$

where,

m_x = *body weight or tarsus length at time x, and*
t_x = *date of measurement*

Instantaneous weight change (r) was regressed against chick age for chicks ≤20 days (about ½ the age at fledging) and for those >20 days; persistent oiling was treated as an indicator covariable. If measurements were obtained from ≥2 chicks at a nest site, indices were averaged to produce a mean growth change for each nest site; measurements were also averaged across years.

To alleviate the strong age dependence of weight gain in chicks, I calculated the ratio of the natural log of daily changes in weight to the natural log of daily changes in tarsus length for within-year comparisons. This metric (r_w/r_t) should quantify the surplus energy available for weight gain in oystercatcher chicks. Use of energy for the growth of structures that are critical for the survival of the individual (e.g., legs) reduces energy available for weight gain (O'Connor 1984). This type of growth index is commonly used in contaminant studies to determine toxin-related growth impairments (Hoffman et al. 1993). For sites where sediment samples were collected, weights of chicks at 14 days of age were compared to sediment hydrocarbon concentrations. The dates that adults and their young departed their nest sites were recorded.

Observations of prey type and size fed to chicks were made during low tide periods. Each observation period began approximately 2 hr before low tide and ended 1 hr after low tide. Timed observations commenced with the first delivery of a prey item by an adult. For each subsequent delivery, the time, type of prey item, and length of the prey item were recorded (as described above for adults). To compare between nest sites, I standardized food consumption by dividing the rate of prey biomass delivered by the total biomass of each brood ([prey biomass/time]/chick biomass).

Fecal samples of chicks were collected in 1993 to directly determine if oystercatchers were being exposed to oil persisting in the mussel beds and fine sediments in the Sound. Samples were collected at 7- to 10-day intervals once chicks were $\geq$7 days old. Chicks were placed in a cleansed teflon pan for a maximum of 10 min to collect feces. Excreta were transferred by a clean, disposable pipette to an HC-grade sterilized jar containing $\approx$20 ml of dichloromethane. All reusable equipment underwent standard detergent-acetone-hexane washing between sample collections. These procedures generally follow Fry (ND). Samples were frozen in the field and remained frozen during transportation and shipping. Multiple samples were collected from as many chick-nest sites as possible to maximize the chance of detecting hydrocarbons in the feces of oystercatcher chicks. All within-nest site samples were pooled prior to petroleum hydrocarbon analysis.

Fecal samples were sent to the Geochemical and Environmental Research Group (GERG) of the Texas &M University for gas chromatography-mass spectroscopy determination of aliphatic hydrocarbon (AH) and polycyclic aromatic hydrocarbon (PAH) concentrations. Extraction of fecal tissue samples followed the NOAA Status and Trends Method (Macleod et al. 1985) with minor revisions (Wade et al. 1988). Briefly, fecal tissue samples were homogenized with a Teckmar Tissumizer and 1- to 10-g sample (wet weight) was extracted with the tissumizer by adding surrogate standards, Na_2SO_4 and methylene chloride, in a centrifuge tube. The fecal tissue extracts were purified by silica/alumina column chromatography to isolate the AH and PAH fractions. The PAH fraction was further purified by HPLC to remove interfering lipids. Quantitative analyses were performed by capillary gas chromatography (CGC) with a flame ionization detector for aliphatic hydrocarbons and a mass spectrometer detector in the SIM mode for aromatic hydrocarbons (Wade et al. 1988). Details of analytical methods are in GERG standard operating procedures, SOP-8901 to SOP-8905 (*Exxon Valdez* Oil Spill Public Information Office, Anchorage, Alaska).

Precise dry weights of samples were not determined in the laboratory; therefore, I used ratios of hydrocarbon (HC) concentrations to determine oil exposure. The following measures are well-documented indicators of petroleum hydrocarbon contamination (Farrington et al. 1973, Hall and Coon 1988, Manen 1990, Paasivirta 1991:165-179): C17 (n-heptadecane)/phytane, odd/even HCs (C24-C29), unresolved complex mixture (branched aliphatics)/resolved aliphatic HCs, and PAHs (polycyclic aromatic hydrocarbons)/all resolved HCs. Because the purpose of these analyses was to compare petroleum hydrocarbon indices between persistently oiled and unoiled groups, minimum detection limits were subtracted from reported hydrocarbon concentrations prior to calculations.

Statistical Analysis

The primary sampling (or experimental) unit for all tests of the effect of persistent shoreline oil on black oystercatchers was a nest site or breeding pair, except for feeding rates for which an individual oystercatcher was the primary sampling unit. Comparisons were made between areas of persistently oiled shoreline and areas of unoiled shoreline. Relationships between sediment hydrocarbon concentration and HC indices of feces or chick weight were examined at individual nest sites. Several estimators and testing procedures were used to compare between persistently oiled and unoiled groups (Table 9); the ratio estimator (Snedecor and Cochran 1980:455-457), binomial estimator (Fisher's exact test; Agresti 1990:59-62), and a two-sample randomization test (Manly 1991:43-63) were used most often. Alternative hypotheses generally were constructed to indicate that oiling caused a negative effect on estimated parameters. Satterthwaite's approximation for testing 2 population means was used to determine degrees of freedom for all *t*-tests (Snedecor and Cochran 1980:97). The number of iterations (*I*) that were run in randomization tests was set at 5,000 unless sample sizes were small; then, the number of complete iterations was calculated as $n!/(n_1!n_2!)$. *P*-values of test statistics were examined to determine the magnitude of the difference between persistently oiled and unoiled groups. Multiple tests of the common null hypothesis of no persistent oiling effect within this study, and among several Spill studies testing this general hypothesis, made setting a realistic apriori α-level for test statistics problematical. Therefore, I used *P*-values of tests to examine the magnitude of persistent oiling effects but set nominal significance level at the standard $P \leq 0.05$.

Extraneous independent factors (age, brood size, tidal height), along with oiling category, were incorporated into a covariate, interaction linear regression model of chick growth (with weight change as the dependent variable). Linear model-checking procedures involved visual inspection of residuals plotted against predicted values, correlation of residuals and predicted values, and calculation of Cook's D and tolerances. I used Kendall's test of concordance to examine the relationship between hydrocarbon concentrations in sediment samples and those in fecal samples. Contributions of pairs or nest sites to hydrocarbon contamination and foraging behavior datasets are shown in Table 10 (persistently oiled) and Table 11 (unoiled). Regression analyses were conducted with the statistical package SYSTAT®, *t*-tests were calculated in a Quattro Pro® macro, and randomization tests were performed in Manly's RT program.

Table 9. Estimated parameters, years analyzed, expected outcome if oiled, and estimator or statistical test used for comparisons of foraging ecology and reproductive success between black oystercatchers nesting on persistently oiled and unoiled shorelines in Prince William Sound, Alaska, 1991-1993.

Estimated Parameter	Years in analysis	Outcome unoiled:oiled	Estimator/test
adults' time spent foraging	1992-93	±	randomization
mussel consumption	1992-93	±	ratio
foraging bout duration	1992-93	>	randomization
adults' behavior toward intruders	1992	>	binomial
clutch size	1992-93	>	randomization
hatching, fledging, nesting success	1991-93	>	ratio
productivity	1991-93	>	ratio
females renesting	1992-93	>	randomization
egg volume	1992-93	<	randomization
chick growth rate	1991-93	>	regression
food delivery rates	1991-93	±	ratio
fecal hydrocarbon concentration	1992-93	> or <	randomization

Table 10. Location of nest sites and information used for analyses of petroleum hydrocarbon contamination and foraging behavior of black oystercatchers inhabiting areas of persistently oiled shoreline in Prince William Sound, Alaska, 1991-1993.

Location	Nest site	Sediment sample	Fecal sample	Chick weight	Chick delivery	Adult foraging
Aguliak Island	86					
Bay of Isles	50					
	53	X	X	X		X
	54			X	X	
	96	X				X
Block Island	30	X	X	X	X	X
Disk Island	39				X	X
Herring Bay	45	X	X	X	X	
	46	X	X	X	X	X
	47	X	X	X	X	X
	48			X	X	X
	98	X	X	X	X	X
Marsha Bay	97					
NE Knight Island	49					
New Year's Island	80	X	X	X	X	X
NW Knight Island	312	X	X		X	
Squire Island	76	X	X	X		

Table 11. Location of nest sites and information used for analyses of petroleum hydrocarbon contamination and foraging behavior of black oystercatchers inhabiting areas of unoiled shoreline in Prince William Sound, Alaska, 1991-1993.

Location	Nest site	Fecal sample	Chick weight	Chick delivery	Adult foraging
Eleanor Island	38	X	X	X	
Drier Bay	73	X	X	X	X
	74				
	81		X	X	X
	82		X	X	X
	83	X	X	X	
Johnson Bay	41				
	94		X		
	95				
Lower Herring Bay	87	X	X	X	X
	88	X	X	X	
	89	X	X	X	X
	90				X
	91	X	X	X	X
	92				
	93				X
Lucky Bay	206				
Mummy Bay	207	X	X	X	
Mummy Island	75				
	78				
	79				
Squirrel Island	43				X
	84	X	X	X	
	208	X	X		

Results

Potential for Exposure

Bay mussels were an important component of the diet of adult black oystercatchers inhabiting areas of persistently oiled shoreline on northern and western Knight Island. Of all prey items (*n* = 1,082) selected by foraging adult oystercatchers (*n* = 13 pairs), the majority were mussels (54.6%). Limpets, clams, chitons, and horse mussels (*Modiolus*) constituted the remainder of the diet (Table 12). Clams, however, constituted most (45.4%) of the biomass of invertebrate prey captured by adults. Adults most often captured mussels and clams in shallow water (<10 cm) by stabbing and severing the adductor muscles of individuals that had their valves gaped. Adults captured larger individual mussels (Fig. 10) and limpets (Fig 11) in a greater proportion than those available in the intertidal environment.

Table 12. Numbers and biomass of prey items comprising the diet of adult black oystercatchers in areas of persistent shoreline oil in Prince William Sound, Alaska, 1991-1993.

Prey species	Number (%) *n* = 1,082	Biomass (%) *n* = 811.1 g
bay mussels	54.6	26.6
limpets	28.0	15.4
clams	10.9	45.4
chitons	3.5	11.3
horse mussels	3.0	1.3

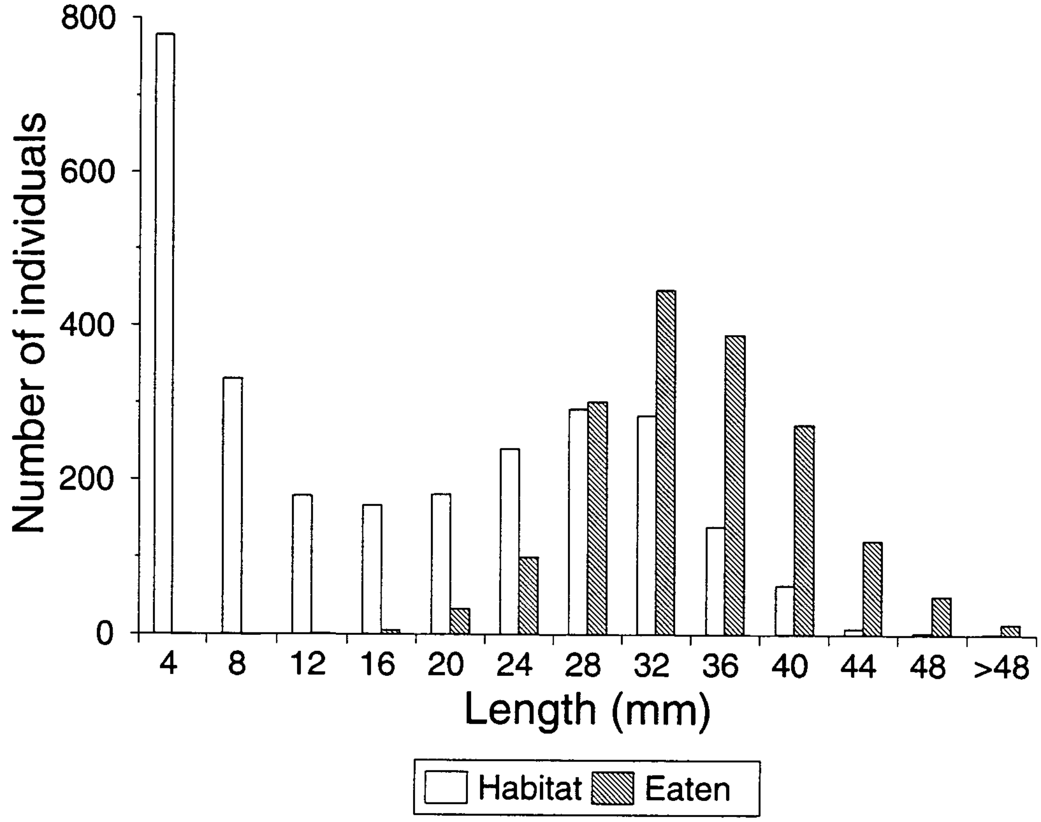

Figure 10. Lengths (mm) of bay mussels consumed (*n* = 1,727) by black oystercatchers and those available in the intertidal environment (*n* = 2,660) of Prince William Sound, Alaska, 1991-1993.

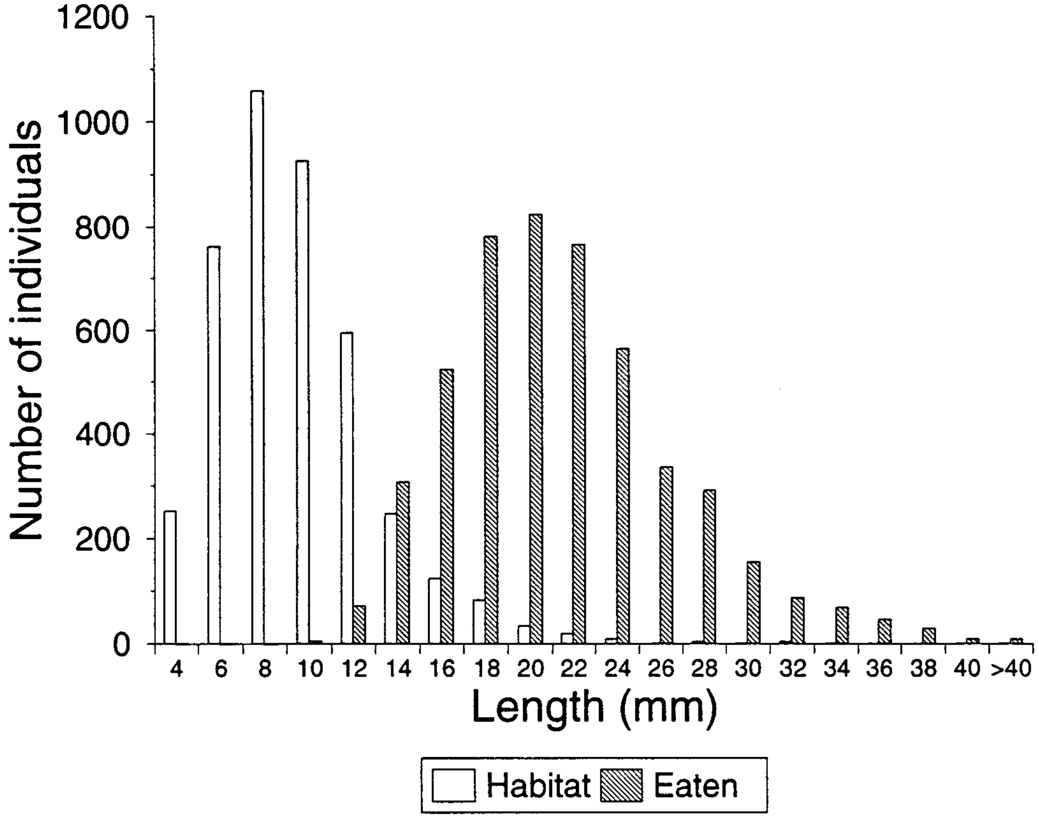

Figure 11. Lengths (mm) of limpets consumed (*n* = 4,860) by black oystercatchers and those available in the intertidal environment (*n* = 4,113) of Prince William Sound, Alaska, 1991-1993.

Black oystercatchers sought fine sediment, mixed sand and gravel substrates for foraging. The majority (61.5%) of foraging commutes (*n* = 52) by oystercatchers (*n* = 10 pairs) that were >100 m were made to mixed sand and gravel shorelines. Mixed sand and gravel beaches, however, constituted only 24.6% of the shoreline in areas where oystercatchers foraged. The density of mussels on foraging sites ($815/m^2$ ± 145, mean ± SE, *n* = 8 sites) was greater than the density of mussels on surrounding shorelines (262 ± 92) where oystercatchers did not forage. The gradual slope (13°) of mixed sand and gravel foraging sites compared to often rocky, non-foraging sites (slope = 49°) made more mussel prey available, by providing more substrate surface area, to oystercatchers foraging in these areas.

Sediment and mussel samples were collected from 10 oystercatcher foraging sites that contained detectable signs of oil in 1992 and 1993. Hydrocarbon concentrations at all 10 oiled sites exceeded the concentration of an unoiled control site in Barnes Cove, Knight Island (Table 13). Most of the oiled oystercatcher foraging sites contained only moderately dense aggregations of mussels (<1,000 mussels/m^2) and were often located in tombolos composed of fine sediments (spaces between offshore rocks). Sediments with the highest concentrations of hydrocarbons occurred at foraging sites on northern and eastern Knight Island. Information on hydrocarbon concentrations of mussels collected at these sites is not yet available.

Although only a few, late season trials were conducted, an interesting result regarding selection/avoidance of oiled limpets emerged. When platters were retrieved within 4 hrs of placement (*n* = 2), only unoiled limpets were removed. If platters were left for 36 hours ($n = 3$), both limpets were removed. The experiment needed further replication to determine whether oystercatchers can or can not discern oiled prey, and thus if they can avoid the deleterious effects of consuming oiled prey.

Effects on Foraging Behavior

Overall, oystercatcher adults breeding in areas of persistently oiled shoreline spent the same amount of time foraging as did pairs breeding in areas of unoiled shoreline (Table 14). Adults differed somewhat in the time they spent foraging during incubation ($D_o = -0.57$, $I = 5{,}000$, $P = 0.159$) but not during brood-rearing ($D_o = -0.05$, $I = 5{,}000$, $P = 0.50$). During individual feeding bouts, adults spent the same amount of time foraging on mixed sand and gravel substrates in areas of persistently oiled shoreline as did adults in areas of unoiled shoreline ($D_o = -1.10$, $I = 5{,}000$, $P = 0.77$). However, at two locations where territories included known oiled and unoiled foraging substrates, adults spent less time feeding on oiled sites than they did on unoiled substrates; a pair nesting in Bay of Isles spent much less time foraging on oiled substrates ($D_o = 9.14$, $\mathrm{I} = 495$, $P = 0.01$) as did a pair nesting on Block Island ($D_o = 6.25$, $\mathrm{I} = 5{,}000$, $P = 0.05$).

Adults breeding in areas of unoiled shoreline consumed significantly more mussel individuals ($t = 4.16$, df = 14, $P = 0.001$) but not more mussel biomass ($t = 1.40$, df = 10, $P = 0.2$) than did adults in areas of persistently oiled shoreline (Table 14). Variation in biomass intake was high among individuals within the unoiled group.

Table 13. Location, shoreline segment and hydrocarbon (HC) concentrations (oil equivalents, μg/g wet weight) of sediments collected from black oystercatcher foraging and unoiled sites on Knight Island, Prince William Sound, Alaska, 1992-1993.

Location	Segment number	Nest site	Year	HC concentration mean	HC concentration SE
Bay of Isles	KN203A	96	1992	6435.82	670.68
Bay of Isles	KNO16A	53	1992	4698.85	922.68
Bay of Isles	KN005A	96	1992-93	1042.67	301.04
Block Island	EL011A	30	1992-93	1162.17	445.86
Herring Bay	KN144B	47	1992	481.31	114.22
Herring Bay	KN114A	45	1992-93	13875.56	1979.48
Ingot Island	IN031B	39	1992	12515.02	828.32
New Year's Island	NY001A	80	1992	940.46	352.91
Squire Island	SQ004A	76	1992	18.91	1.10
NW Knight Island	KN505A	312	1993	7800.33	5698.58
Barnes Cove (unoiled control)	KN575A		1992	0.33	0.27

Table 14. Foraging behavior of black oystercatcher pairs (*n*) breeding at persistently oiled sites and unoiled sites in Prince William Sound, Alaska, 1992-1993.

	Oiled			Unoiled		
Foraging behavior	mean	SE	*n*	mean	SE	*n*
time spent (% hr) foraging (incubation stage)	21.5	13.7	9	18.8	13.0	9
time spent (% hr) foraging (chick-rearing stage)	35.1	18.1	7	32.9	13.0	13
foraging bout duration (min, incubation stage)	11.9	0.41	9	14.6	1.8	7
no. of mussels consumed per hr (all stages)	30.2	5.7	8	58.1	3.5	10
biomass of mussels consumed per hr (all stages)	13.1	2.5	8	21.0	5.0	10

Effects on Breeding Success

Few measures of breeding success differed significantly between pairs nesting in areas of persistently oiled shoreline and those nesting in areas of unoiled shoreline (Table 15). Unexpectedly, point estimates of hatching, fledging, and nesting success and productivity were all greater, but not significantly, for pairs nesting in areas of persistently oiled shoreline than for those nesting in areas of unoiled shoreline. The percentage of renesting females differed little between persistently oiled and unoiled areas. Only the relative difference in egg volumes of second clutches differed significantly (D_o = 15.6%, I = 2,002, $P = 0.015$) between females nesting in areas of persistently oiled shoreline and those nesting in areas of unoiled shoreline. Productivity in persistently oiled areas of Knight Island decreased from 1991 to 1993 but remained virtually the same in unoiled areas (Fig. 12). A significantly lesser proportion of adults (50%) nesting in areas persistently oiled areas responded actively to intruders than did adults nesting in unoiled areas (80%; Fisher's exact test, $n = 18$, $P = 0.006$). Adults and young in persistently oiled areas remained at their nest site for about the same amount of time after hatching as did families in unoiled areas (Table 15).

Pairs nesting on Green Island (oiled in 1989) were more productive (0.81 young/pair ± 0.11, $n = 49$) than pairs nesting in areas of persistently oiled shoreline on Knight Island. In 1992, productivity was 1.15 young/pair on Green Island and 0.39 young/pair for all areas (persistently oiled and unoiled) of Knight Island. During this year, common ravens were detected at 22% of the point counts conducted on Knight Island ($n = 78$) but at only 5% of the points on Green Island ($n = 20$). The high mobility of common ravens makes the nests and young of many black oystercatcher pairs nesting in the Knight Island area susceptible to their predation.

Effects on Chicks

Chicks raised at persistently oiled nest sites gained weight slower than chicks raised at unoiled nest sites. With age as a significant covariate in a regression model, persistent shoreline oiling had a significant effect on instantaneous weight change of chicks ≤20 days old ($F_{1,17} = 6.07$, $P = 0.025$) and of chicks >20 days old ($F_{1,11} = 5.47$, $P = 0.039$). Additionally, a significant ($P = 0.02$) interaction between oil and age was evident for weight change in chicks ≤20 days old. Neither brood size nor tidal height helped explain additional variation in weight change. By 30 days old, growth rates of all chicks were projected to be similar (Fig. 13). Weight gain, relative to tarsus growth, of chicks raised on persistently oiled shorelines was significantly lower than relative weight gain of chicks raised on unoiled shorelines in 1992 ($D_o = -1.14$, $I = 5{,}000$, $P = 0.001$) but only tended to be lower in 1993 ($D_o = -0.50$, $I = 5{,}000$, $P = 0.105$; Fig. 14). Chicks raised in areas of high sediment hydrocarbon concentrations tended to weigh less at 14 days of age than did chicks raised in areas of low sediment hydrocarbon concentrations (Fig. 15).

Table 15. Measures of breeding success of black oystercatcher pairs nesting at persistently oiled and unoiled sites in Prince William Sound, Alaska, 1992-1993.

	Oiled			Unoiled		
Reproductive variable	mean	SE	*n*	mean	SE	*n*
active behavioral response by adults to intruders (%)	50.0	17.6	8	80.0	12.7	10
clutch size, 1st attempt (eggs)	2.34	0.17	16	2.61	0.09	22
clutch size, 2nd attempt (eggs)	2.71	0.18	7	2.33	0.24	13
hatching success (%)	49.6	9.3	16	31.1	6.0	24
fledging success (%)	17.1	4.7	16	11.7	3.7	24
nesting success (%)	30.9	7.4	16	24.2	5.3	24
productivity (young/pair)	0.48	0.13	16	0.37	0.11	24
females renested (%) (either year)	63.6	14.5	11	65.0	10.7	20
egg volume, 1st clutch (% difference)	5.1	0.6	13	5.1	0.9	22
egg volume, 2nd clutch (% difference)	19.8	11.6	5	4.2	1.1	9
length of stay after hatch (days)	54.9	7.4	7	49.6	7.0	5

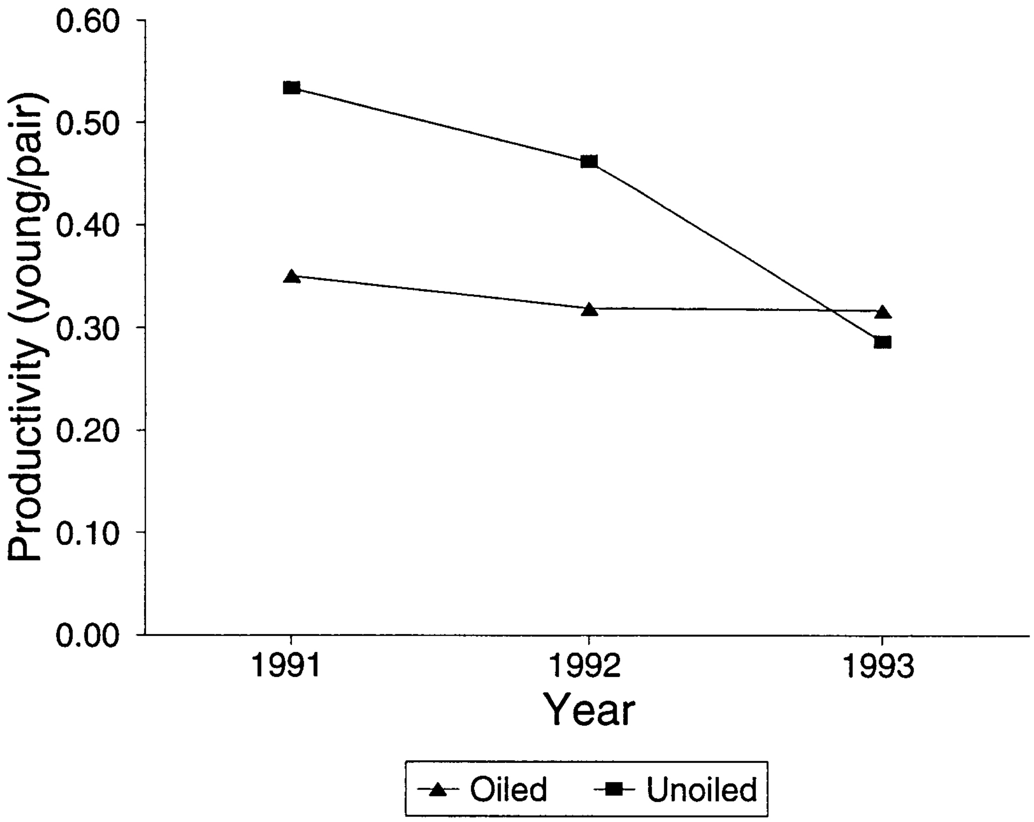

Figure 12. Changes in productivity (young/pair) of black oystercatchers inhabiting areas of persistently oiled shoreline and those inhabiting unoiled areas on Knight Island, Prince William Sound, Alaska, 1991-1993.

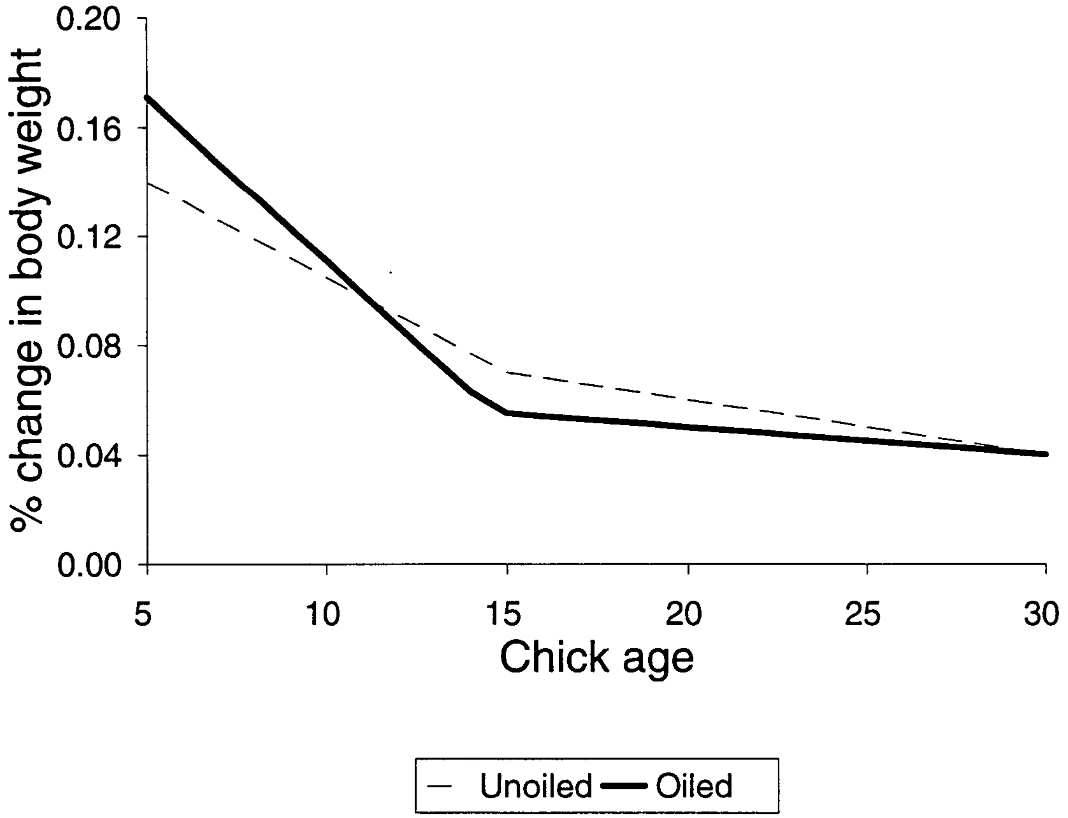

Figure 13. Fitted growth curves for chicks ≤20 days and >20 days old raised in areas of persistently oiled shoreline and in areas of unoiled shoreline in Prince William Sound, Alaska, 1991-1993.

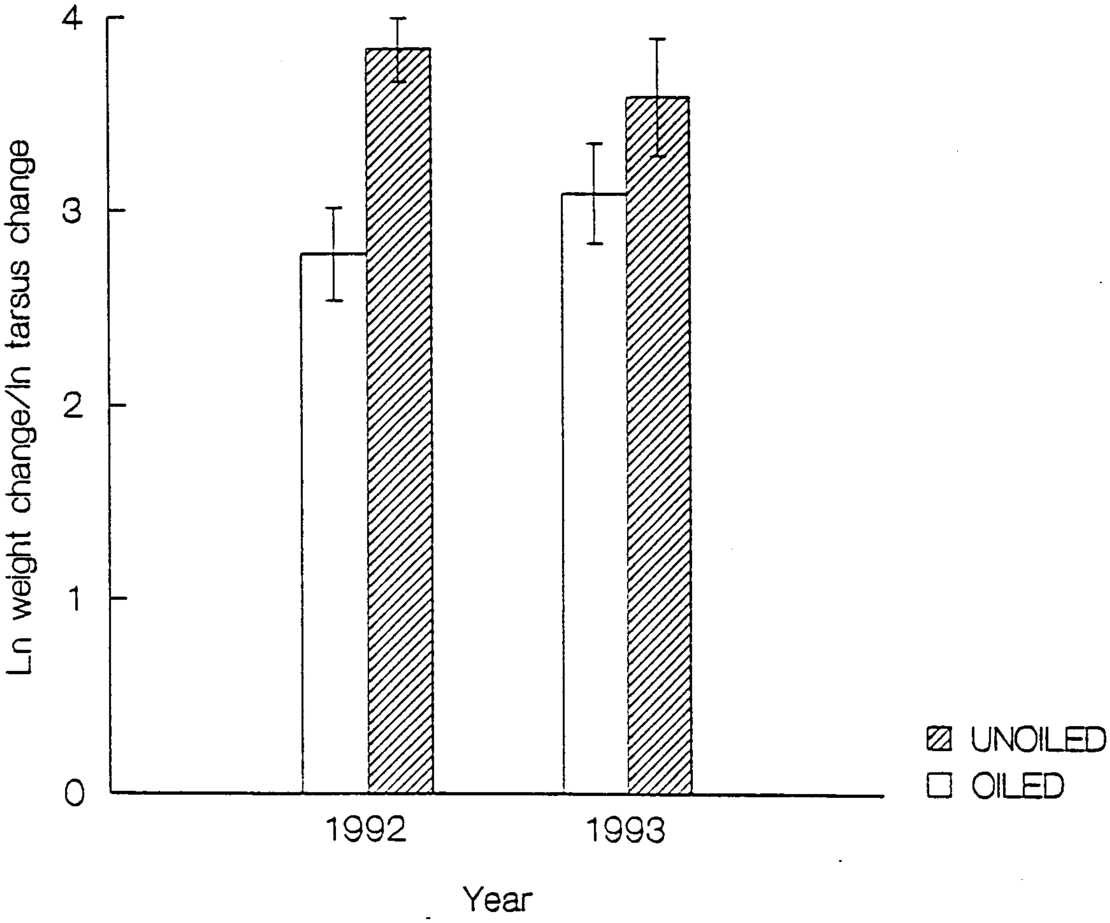

Figure 14. Daily change in weight (g) relative to daily change in tarsus length (mm) of chicks raised on persistently oiled and unoiled shorelines in the area of Knight Island, Prince William Sound, Alaska, 1992-1993.

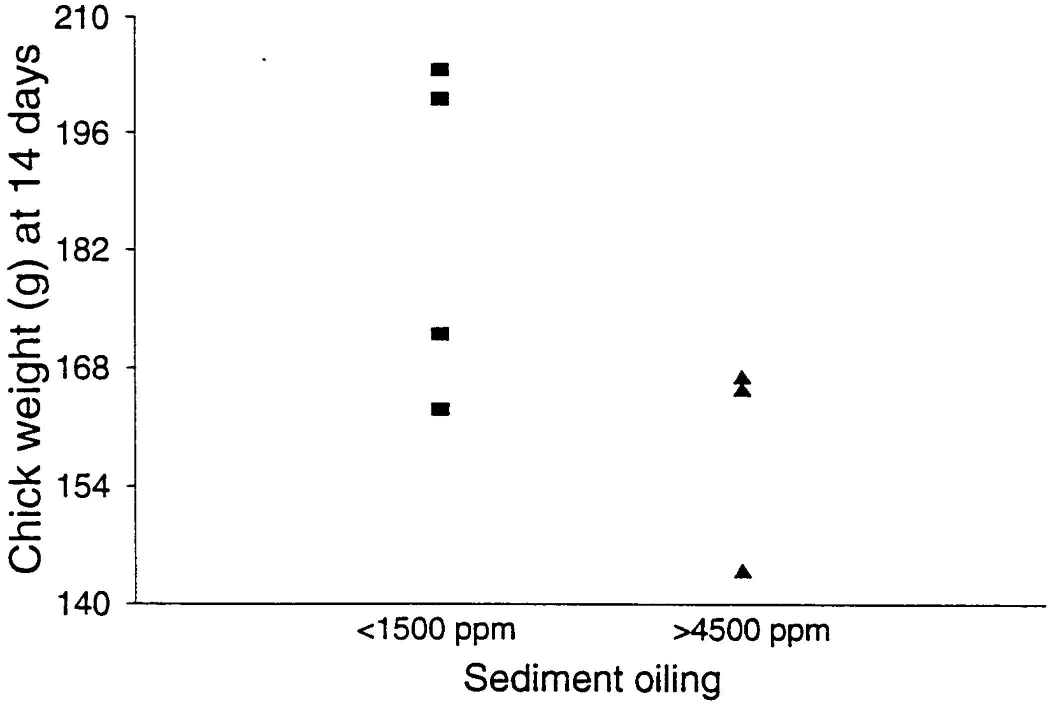

Figure 15. Weights (g) of chicks 14 days old raised in areas of high sediment hydrocarbon concentrations and in areas of low sediment hydrocarbon concentrations on Knight Island, Prince William Sound, Alaska, 1992-1993.

Adults delivered food at a somewhat faster rate in areas of unoiled shoreline than did adults in oiled areas ($t = 1.52$, df = 18, $P = 0.15$) and brought somewhat more biomass in each delivery (Table 16; $D_o = 0.69$, I = 5,000, $P = 0.112$). Prey biomass, relative to brood biomass, delivered to chicks by adults was highly variable, and therefore did not differ between oiling groups (Table 16). Because oystercatchers foraged on a variety of shoreline substrates, prey selection was quite variable among pairs (Table 17); individual preference for certain prey items might also contribute to variability among pairs within an oiling group.

Mussels constituted 42.4% of all items and 23.3% of all biomass of food brought to chicks raised in the Knight Island area. Adults foraging in unoiled areas tended to deliver a greater number of mussels ($t = 1.82$, df = 12, $P = 0.09$) but not more biomass of mussels ($t = 0.44$, df = 13, $P > 0.5$) than did adults foraging in oiled areas (Table 16).

Table 16. Numbers of individuals and biomass of prey delivered to black oystercatcher chicks raised in persistently oiled (*n* = 83 hr observation) and unoiled areas (*n* = 79 hr observation) of Knight Island, Prince William Sound, Alaska, 1991-1993.

	Oiled (*n* = 10)			Unoiled (*n* = 11)		
Foraging behavior	mean	SE	CV	mean	SE	CV
prey mass/chick mass (% per hr)	2.19	0.91	0.41	4.21	1.44	0.34
no. of prey deliveries per hr	5.11	1.11	0.22	7.94	1.49	0.19
biomass per delivery	1.28	0.29	0.23	1.97	0.45	0.23
no. of mussels delivered per hr	1.57	0.39	0.25	3.83	1.18	0.31
mussel mass (g) delivered per hr	0.46	0.17	0.40	1.25	0.47	0.38

Table 17. Percentages of prey items delivered to chicks by adult oystercatchers at nest sites in unoiled areas of Prince William Sound, Alaska, 1991-1993.

	Percentage of prey deliveries			
Nest site	mussels	limpets	clams	chitons
Eleanor Island (*n* = 32)	12.5	59.4	-	28.1
Northeast Cove (*n* = 23)	30.4	34.8	34.8	-
Rocky Point (*n* = 69)	-	100.0	-	-
Lower Herring Bay (*n* = 159)	73.0	25.8	0.6	0.6
Mallard Bay (*n* = 69)	47.8	52.2	-	-

Several indices of hydrocarbon concentrations calculated from feces of chicks living in areas of persistently oiled shoreline differed from those living in unoiled areas (Table 18). Differences in indices between oiled and unoiled groups were evident for C17 (n-heptadecane)/phytane ($D_o = 5.41$, $I = 5{,}000$, $P = 0.028$), odd/even aliphatics ($D_o = 3.21$, $I = 5{,}000$, $P = 0.006$), and unresolved complex mixture (UCM)/resolved aliphatics ($D_o = -11.55$, $I = 5{,}000$, $P = 0.025$). Proportions of PAHs in feces did not differ between oiling groups ($D_o = 0.67\%$, $I = 5{,}000$, $P = 0.608$).

Hydrocarbon indices of chicks' feces were somewhat concordant with sediment concentrations where they were raised (Table 19). The highest proportions of PAHs and highest vales of the UCM/resolved aliphatics index in chicks' feces were recorded from sites where sediment hydrocarbon concentrations were >6,000 ppm.

Table 18. Indices of concentrations of petroleum hydrocarbons in feces of black oystercatcher chicks raised in areas of persistently oiled and unoiled shorelines in Prince William Sound, Alaska, 1993.

	Oiled			Unoiled		
HC index	mean	SE	*n*	mean	SE	*n*
C17/phytane	7.00	1.34	8	12.42	2.65	9
odd/even (C24-C29)	4.12	0.59	8	7.33	0.95	12
UCM/resolved aliphatics	22.65	4.79	10	11.1	3.08	10
PAH (%)	6.37	2.11	9	7.05	0.93	12

Table 19. Kendall's K statistic, sample size, and *P*-value for measures of concordance among sediment hydrocarbon concentrations and hydrocarbon contamination indices of feces collected from black oystercatcher chicks in Prince William Sound, Alaska, 1993.

Hydrocarbon index	Kendall's K statistic	*n*	*P*
C17/phytane	7	6	0.136
odd/even (C24-C29)	5	7	0.281
UCM/resolved aliphatics	18	8	0.016
PAH (%)	10	8	0.138

Discussion

Because nesting and foraging territories of black oystercatchers breeding in the Knight Island archipelago overlapped potentially oiled mussel beds and fine sediments, some pairs were susceptible to the effects of persistent shoreline oil. Their reliance on mussels as a food source and their high use of mixed sand and gravel shorelines for foraging indicated that black oystercatchers had the potential to be exposed to oil persisting in the sediments of Prince William Sound via a trophic pathway. Hydrocarbon concentrations in sediments collected in 1992 corresponded with hydrocarbon concentrations in mussels collected at the same sites; however, sediment concentrations showed little relationship to overlying mussel concentrations in 1991 (Babcock et al. 1994). Hydrocarbon concentrations in sediment samples collected from oystercatcher foraging sites indicated that pairs breeding in Bay of Isles, in Herring Bay, and on northwestern Knight Island were most susceptible to persistent hydrocarbon exposure. All of these contaminated sites consisted of fine sediments or were in tombolos. Hydrocarbon concentrations in sediments underlying mussel beds in these types of areas remained relatively unchanged between 1991 and 1992; however, oil released into the water by oiled sediments did not disperse widely and had little affect on surrounding shoreline areas (Babcock et al. 1994). The byssal mats of dense mussel beds form a relatively impermeable barrier that limits the dispersion of sediment oil underlying these beds (Babcock et al. 1994). Distribution of hydrocarbon components in sediment samples confirmed that *Exxon Valdez* crude oil was the source of persistent oil in Prince William Sound (Babcock et al. 1994).

Although oystercatcher adults inhabiting areas with persistently oiled shorelines spent the same amount of time foraging as adults inhabiting areas with unoiled shorelines, they tended to spend less time feeding on oiled substrates. Pairs with territories that contained persistently oiled and unoiled foraging sites spent less time feeding at the former; mussel densities on persistently oiled and unoiled substrates were similar and did not appear to influence the use of these areas. The lower use of persistently oiled substrates suggests that adult oystercatchers might be able to detect contaminated sediments or prey. Although adults were observed to reject prey items only on persistently oiled substrates, adult oystercatchers did not always discriminate against oiled limpets offered in choice experiments. The presence of persistent shoreline oil might have dissuaded adult oystercatchers from consuming mussels; adults consumed fewer mussels, but not biomass, than adults foraging in areas of unoiled shorelines. The use of multiple foraging sites and the high variability of hydrocarbon concentrations in sediments, both within specific shoreline segment and among sites (Babcock et al. 1994), might mitigate negative effects of persistent shoreline oil on adult oystercatcher foraging behavior.

Any differences in breeding success between oystercatchers nesting in areas of persistently oiled shorelines and those nesting in areas of unoiled shorelines were, most likely, overridden by the effects of predation on eggs and young, mainly by common ravens. The ability of ravens to remember food caches is well known (Heinrich 1989). I suspect that common ravens were the cause of complete and rapid depredation of small (>20 nests) colonies of arctic terns (*Sterna paradisea*) and mew gulls (*Larus canus*) at several sites on Knight Island. Throughout the Sound, black oystercatchers often nest in the vicinity of arctic terns. The decrease in productivity of pairs nesting in the oiled area of Knight Island might be attributable to a decrease in human activity in these areas. Far more researchers were afield in Herring Bay and Bay of Isles in 1991 and might have disrupted the predatory behavior of ravens in these areas. In fact, all 5 oystercatcher pairs breeding in Herring Bay in 1991 successfully fledged young but were less successful in subsequent years. I was not able to detect an increased presence of ravens in areas of unoiled shorelines that might account for lower productivity in these areas. Additionally, other environmental measures were similar between areas of persistently oiled and unoiled shorelines and were not consistent with the expectation of good or poor quality habitat (Table 20).

Table 20. Characteristics of the environment in areas of persistently oiled and unoiled shoreline on Knight Island, Prince William Sound, Alaska, 1992-1993.

	Shoreline	
Environmental feature	Unoiled	Oiled
mean shoreline elevation (m)	155.3	133.3
mixed sand and gravel (%)	40.3	27.2
islets/km	1.65	2.07
raven occurrence (% points)	20.5	29.2
oystercatcher pairs/km	0.098	0.086

Although no apparent differences exist between the reproductive success of pairs breeding in persistently oiled areas of Knight Island and those breeding in unoiled areas, some evidence suggests a possible physiological effect of persistent shoreline oil on nesting adults. Lower relative egg volumes of second clutches suggest a possible effect of persistent oil on the condition of nesting females. Ingestion of oil has impaired the ability of females to produce eggs in other marine bird species (Ainley et al. 1981). Because females often arrive on their territories with energy reserves to produce first clutches, effects of oil ingestion should be more pronounced during the formation of second clutches; females have been exposed to persistent oil in the Sound for at least month before the initiation of a second clutch.

The high proportion of pairs breeding in persistently oiled areas that exhibited a passive response to intruders was similar to changes in the behavior of breeding waterfowl that were dosed with petroleum (Cavanaugh et al. 1983). In 1992, human disturbance was minimal, and equivalent, in oiled and unoiled areas and should not have caused differences in the behavior of adult oystercatchers. Direct measures of petroleum hydrocarbon contamination in adults are needed to corroborate these suggestions.

Oystercatcher chicks were directly, and possibly indirectly, affected by the presence of persistent shoreline oil in the territories where they were raised. Slower growth rates of chicks were apparent across areas of persistently oiled shoreline on Knight Island and corresponded to sediment hydrocarbon concentrations at specific nest sites. The pronounced effect of oil ingestion on growth rates of young chicks has been documented in other seabirds (Butler and Lukasiewicz 1979, Boersma et al. 1988). In these studies, differences in growth rates between oil-dosed and control chicks converged as chicks aged, and dosing had little effect on the growth of appendages.

Reduced parental care, manifested in somewhat low delivery rates of food to chicks, could be attributable to adult behavioral changes resulting from the ingestion of petroleum hydrocarbons. Complete abandonment of nest sites by birds exposed to environmental oil has been documented (Eppley and Rubega 1990), and growth rates of chicks have been negatively affected, as a result of relaxed parental care, by dosing parents (Trivelpiece et al. 1984). Sibling rivalry within broods also may influence growth of chicks (Groves 1984); I observed the death of 1 chick due to starvation. Any reductions in provisioning chicks, however, did not translate to a reduction in productivity. High abundance of intertidal prey and a long brood-rearing season could offset short-term inadequacies of parental care. Nothing, however, is known about the effects of reduced growth rate, and perhaps a slower acquisition of self sufficiency, on the overwinter survival of young oystercatchers and their recruitment into the population. Young oystercatchers were less efficient foragers than adults for most of their first winter (Falxa 1992). Additionally, little information exists on the cumulative effects of oil ingestion on free-living birds that consistently occupy contaminated sites. Specifically, what are the temporal patterns of hydrocarbon uptake and assimilation and the effects on population dynamics?

The presence of elevated hydrocarbon indices in the feces of chicks provided direct evidence that black oystercatchers were exposed to oil persisting on shorelines of the Sound. Although percentages of PAHs in the feces of chicks did not differ between persistently oiled and unoiled groups, the highest proportions of PAHs were found in chicks that lived in areas with high concentrations of hydrocarbons in the sediment. Consequently, exposure of chicks to persistent shoreline oil was greatest in Herring Bay, in Bay of Isles, and on northwest Knight Island. Within these areas, exposure to persistent shoreline oil was highly variable; outside these areas exposure was minimal.

The restricted, local effects of oil exposure are corroborated by the lack of any large scale population changes in the oiled area of the Sound (Agler et al. 1994). Because exposure was localized, treatment of oiled mussel beds persisting in Prince William could occur without causing large scale disturbances to the breeding oystercatcher population. Reoccupation of oiled habitats on Green Island indicates that black oystercatchers can recover from major shoreline perturbations and, in the event of future shoreline oiling, all affected shorelines should receive initial cleaning to eliminate concern for exposure to oil that might persist in fine sediments or underneath mussel beds.

CONCLUSIONS

Because few oystercatchers occupied territories at the time of the Spill, direct mortality of adults was, most likely, minimal; only 2 oystercatcher carcasses were retrieved in the Sound and no oiled adults were observed on Green Island in 1989. However, black oystercatchers nesting along shorelines that were impacted by oil spilled from the *T/V Exxon Valdez* had their breeding season disrupted in 1989. On Green Island, 39% of the breeding population were not able to maintain nests into June. Oystercatcher pairs that tolerated heavily or moderately oiled nest sites and were able to hatch eggs experienced greater losses of chicks than pairs that nested at unoiled shorelines. Disturbances to shorelines, by oil impact and associated clean up operations, reduced productivity into 1990. By 1991, productivity had recovered and reoccupation of shoreline habitats was occurring. The longevity of oystercatchers minimizes the importance of a single, or dual, season breeding failure.

Gradual, gravelly shorelines appear to be favored by breeding black oystercatchers and should indicate high quality oystercatcher habitat; these shoreline types support the highest densities in the Sound and throughout their range. In areas of steep rocky shorelines, the presence of numerous offshore islets and mixed sand and gravel beaches are good indicators of oystercatcher habitat. Because oystercatchers congregate into flocks in winter and use traditional sites, delineation of these areas and enumeration of their use by oystercatcher use is needed to ensure their protection. Environmental perturbations that affect wintering flocks could have much different consequences for black oystercatchers than those of the *Exxon Valdez* oil spill.

The use of mussels as an important food source for adults and young and the reliance on fine sediment shorelines for foraging sites indicated that black oystercatchers breeding in Prince William Sound could be exposed to persistent shoreline oil. Petroleum hydrocarbon concentrations in sediments collected from oystercatcher foraging sites were highest in Herring Bay, in Bay of Isles, and on northwest Knight Island. Persistent shoreline oil, however, was patchily distributed and affected relatively few oystercatcher pairs. Therefore, remediation or cleaning of remaining oiled mussel beds in Prince William Sound should have a minimal impact on black oystercatcher population as whole.

Where shoreline oil remained in 1991-1993, elevated hydrocarbon concentrations in the feces of chicks indicated that black oystercatchers were being exposed to persistent oil. Exposure to persistent oil changed the ability of adults to provision their young, possibly lowered the body condition of renesting females, and lowered the ability of chicks to gain weight. Effects of exposure to persistent oil on breeding success were overshadowed by the influence of egg and chick predators. Because areas of persistent shoreline oil exposure were restricted, widely distributed, and highly variable, effects on the black oystercatcher population, on the short term, were minimal. Nothing is known, however, about the long-term effects of continued exposure to oil persisting in the environment or how reduced chick growth rates translate into effects on recruitment.

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