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NINE YEARS AFTER THE *EXXON VALDEZ* OIL SPILL: EFFECTS ON MARINE BIRD POPULATIONS IN PRINCE WILLIAM SOUND, ALASKA[1]

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Abstract. We compared post *Exxon Valdez* oil-spill densities of marine birds in Prince William Sound from 1989–1991, 1993, 1996, and 1998 to pre-spill densities from 1984–1985. Post-spill densities of several species of marine birds were lower than expected in the oiled area of Prince William Sound when compared to densities in the unoiled area. These negative effects continued through 1998 for five taxa: cormorants, goldeneyes, mergansers, Pigeon Guillemot (*Cepphus columba*), and murres. Black Oystercatchers (*Haematopus bachmani*) and Harlequin Ducks (*Histrionicus histrionicus*) exhibited negative effects in 1990 and 1991. Loons showed a weak negative effect in 1993. Black-legged Kittiwakes (*Rissa tridactyla*) showed relative decreases in 1989, 1996, and 1998 which may have been caused by shifts in foraging distribution rather than declines in populations. Glaucous-winged Gulls (*Larus glaucescens*) showed positive effects in most post-spill years. Murrelets and terns showed relative increases in 1993, 1996, and 1998. Generally, taxa that dive for their food were negatively affected, whereas taxa that feed at the surface were not. Effects for some taxa were dependent upon the spatial scale at which they were analyzed. Movements of birds and the mosaic pattern of oiling reduced our ability to detect oil-spill effects, therefore our results may be conservative. Several marine bird species were negatively affected at the population level and have not recovered to pre-spill levels nine years after the oil spill. The reason for lack of recovery may be related to persistent oil remaining in the environment and reduced forage fish abundance.

Key words: loons, oil spill, oiling impacts, oystercatchers, seabirds, waterfowl.

INTRODUCTION

Due to concern about potential environmental effects of oil development in Prince William Sound (PWS), the U.S. Fish and Wildlife Service assessed marine bird populations in PWS in 1972 (Dwyer et al. 1976) and again in 1984–1985 (Irons et al. 1988). On Good Friday, 24 March 1989, *T/V Exxon Valdez* ran onto Bligh Reef in PWS, approximately 60 km from Valdez. About 4×10^6 liters of North Slope crude oil entered the waters of PWS before the remainder of the cargo could be off-loaded to another oil tanker. The spill was the largest recorded in U.S. waters and there was much concern about its effects. About 30,000 oiled bird carcasses were found in the spill area by 25 September 1989, the most birds ever picked up after an oil spill (Piatt and Lensink 1989, Piatt et al. 1990). Large numbers of carcasses of diving birds, such as loons, grebes, cormorants, sea ducks, murres, murrelets, and Pigeon Guillemots (*Cepphus columba*), and surface feeding birds such as Procellarids and gulls were found (Piatt et al. 1990). There were several estimates of the total marine bird mortality (Piatt et al. 1990, Ecological Consulting, Inc. 1991), but Piatt and Ford's (1996) best estimate was that about 250,000 birds died, 74% of which were murres.

The magnitude of lethal oil-spill effects on marine birds can be determined using three general approaches: (1) measure the differences in pre- and post-spill populations, (2) estimate from carcass loss and recovery rates at the time of the spill, and (3) extrapolate from carcass loss/recovery experiments from other spills (Piatt and Ford 1996). The first approach can be used to look at immediate and long-term effects, but requires pre-spill data. The latter two approaches

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are best to determine immediate effects and do not require pre-spill data. Statistical methods for determining the effects of an environmental perturbation using pre-perturbation data have been developed and refined in the past two decades. Green (1979) and Skalski and McKenzie (1982) developed the BACI (Before, After, Control, Impact) design to evaluate the effects of planned development, which has since been modified (Stewart-Oaten et al. 1986, 1992, Wiens and Parker 1995). Because there were data on bird populations in PWS before the spill (1984–1985), the effects on the populations could be investigated using a BACI type design. The BACI study design was fulfilled by comparing marine bird densities before the spill to marine bird densities after the spill. Unoiled areas of PWS served as a control and oiled areas served as the impacted zone.

To evaluate potential effects of the *Exxon Valdez* oil spill on summer residents in PWS, the U.S. Fish and Wildlife Service conducted bird surveys in 1989, 1990, 1991, 1993, 1996, and 1998. The objective of this study was to determine whether the oil spill affected the summertime densities of marine birds in the path of the oil spill in PWS and to assess the duration of the impacts.

METHODS

STUDY AREA

Prince William Sound is a protected body of water (ca. 10,000 km^2) located in the northern Gulf of Alaska. It is characterized by highly convoluted shorelines composed of deep fiords and large islands with tides as great as 6 m. The marine bird fauna of PWS is rich and diverse (Isleib and Kessel 1973). The 1972 summertime marine-bird population estimates of PWS were 629,000 (Klosiewski and Laing 1994). The study area used in the present analyses included waters of PWS within 200 m of shore (Fig. 1). We used shoreline data because those transects were surveyed by the same method before and after the oil spill.

SURVEY METHODS

During the summers of 1984 and 1985, Irons et al. (1988) surveyed the entire shoreline of PWS except for the southern sides of Montague and Hinchinbrook Islands and a few transects that were missed. The shoreline was divided into 772 transects. Transects were located by geographic features, such as points of land, to facilitate orientation in the field and to separate the shoreline by habitat type. All transects were 200 m wide, but varied in length, the mean transect length was 6 km, and they ranged from 1 to 30 km.

Survey methodology developed for surveys in 1984–1985 (Irons et al. 1988) was used throughout this study. Surveys were conducted from 7.7-m boats traveling at speeds of 10–20 km hr^{-1}. Two observers on each boat counted all birds and mammals detected in a sampling window 100 m on either side, 100 m ahead, and 100 m overhead of the vessel. Observers also recorded birds and mammals sighted on land within 100 m of the shore. Observers scanned continuously and used binoculars to aid in species identification. Most transects were surveyed when wave height was < 0.3 m; no surveys were conducted when wave height was > 0.6 m.

Post-spill surveys were conducted in July of 1989, 1990, 1991 (Klosiewski and Laing 1994), 1993 (Agler et al. 1994), 1996 (Agler and Kendall 1997), and 1998 (Lance et al. 1999). These surveys all used the same methodology as used by Irons et al. (1988), however only a portion of the PWS shoreline was surveyed post-spill. Klosiewski and Laing (1994) randomly selected 25% (187) of the total 742 shoreline transects for the post-spill surveys in 1989. An additional 25 shoreline transects from western PWS were randomly selected and added in July 1990 to increase the precision of estimates from the oiled zone. Observers in all years were experienced at identifying marine bird species and were trained using the same protocol.

DATA ANALYSIS

Pre- and post-spill bird densities were estimated from surveys that were conducted on the same transects before and after the spill. We chose transects ($n = 146$) that were surveyed during a comparable period (in July and early August, when bird numbers are relatively constant, K. J. Kuletz, unpubl. data) pre- and post-spill. To determine which transects were oiled, we used data from the Shoreline Cleanup Assessment Team in 1989 (these data were agreed upon by government and Exxon-sponsored scientists to be the best assessment of oiled shorelines). The distribution of the unoiled transects were such that 21% were within the general oiled area and 73% were within a 20-km buffer around the oiled area. The rest of the transects were scattered in

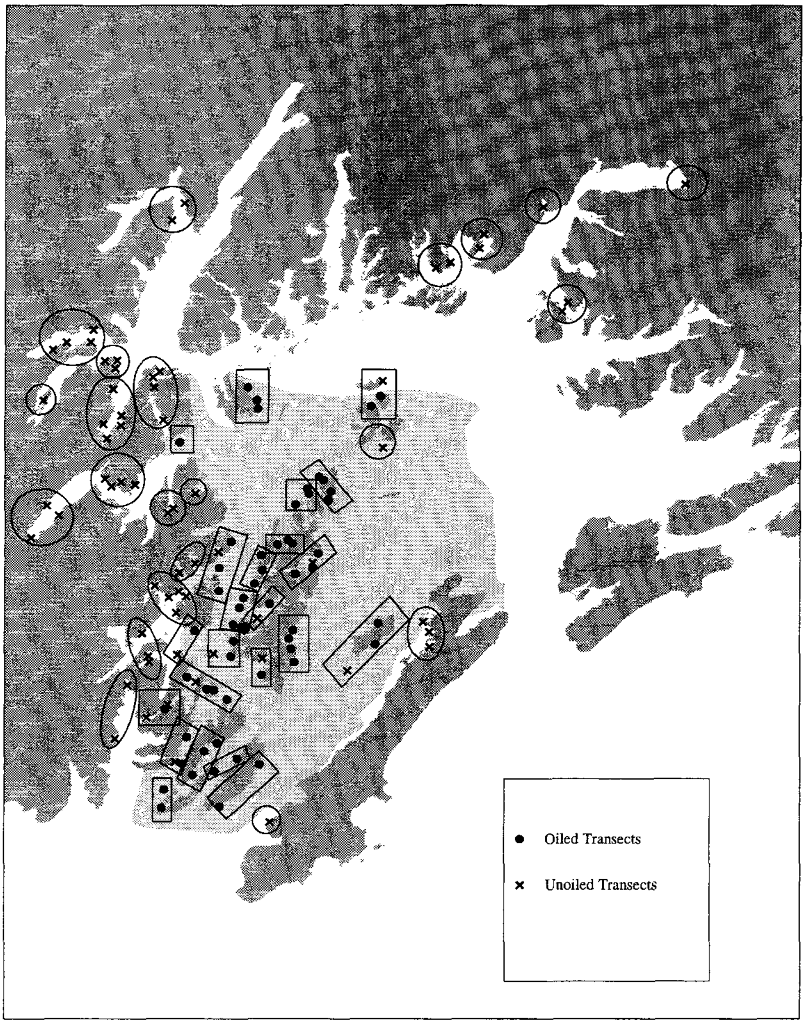

FIGURE 1. Map of Prince William Sound, Alaska, showing locations of 123 transects that were used for analyses at the fine scale, how the transects (x and ●) were combined into 45 groups for the medium scale, and the overall area that was oiled by the *Exxon Valdez* oil spill, which was used for the coarse scale. Transects marked with a ● were oiled and transects marked with an "x" were not oiled. Groups enclosed with a rectangle were oiled and groups enclosed by a circle were unoiled. The stippling indicates the greater oiled area.

the western and northern portion of PWS (Fig. 1).

The BACI design is dependent upon having a comparable reference area to compare to the oiled area. Beaches are not oiled in a random fashion, so the investigator is faced with the problem of selecting a reference area that is similar to the oiled area. It was fortunate that not all of PWS was oiled so that the unoiled portions could be used as a reference area; however, even within PWS all areas are not the same. Densities of some birds are different on islands and in fiords (Irons et al. 1985). However, the BACI analysis does not require that the oiled and unoiled areas are the same, just that changes, in the absence of an oil spill, would be similar.

To help ensure that our reference area was similar to the oiled area, we used cluster analysis to select a group of transects with similar prespill bird densities that was then split into oiled and reference groups. Euclidean distance was used as the similarity metric, and average linkage was used to join clusters (SAS Institute Inc. 1988). We chose transects that clustered together at or below the Euclidean distance of 1.0, resulting in a subset of 123 transects in a single cluster. In 1989, only 108 of the 123 transects were used because fewer transects were surveyed in 1989 than in later years.

We also examined shoreline types of transects in both zones to help determine whether the reference and oiled areas were similar. We used the designations from the Prince William Sound Environmental Sensitivity Maps (produced by Research Planning Inc., Columbia, South Carolina) to categorize the shoreline type for each transect. One hundred and eighteen of the 123 transects fell into one of four categories. When more than one shoreline type occurred in a transect, the most prevalent type was used. Analysis indicated that the frequencies of shoreline types in the oiled and unoiled areas were not different (χ^2_3 = 4.1, P = 0.25). The shoreline categories and the number of transects in the oiled and reference areas, respectively, were as follows: exposed rocky shores (18, 9), exposed wave-cut platforms in bedrock (14, 6), gravel beaches (25, 19), and sheltered rocky shores (12, 15).

Fourteen taxa were analyzed for oil spill effects in this study. We chose to analyze species or species groups that had ca. 25 or more individuals spread over several transects in pre-spill surveys, similar to the criteria used by Murphy et al. (1997). Some bird species that were similar in appearance and vulnerability to oil (King et al. 1979) were grouped by genus for analyses (Appendix 1).

When comparing oiled areas to unoiled reference areas, the ability to detect oil spill effects on birds is affected by the magnitude of the birds' movements and the mosaic pattern of oiling that occurred in PWS. Individual birds whose home ranges bisected the oiled-unoiled border reduced our ability to detect oil spill effects. The influence of birds' movements varied according to the scale that the birds moved, therefore it was important to analyze the data at the proper spatial scale.

To investigate the consequence of spatial scale on detecting oil spill effects, we analyzed the data at three different spatial scales: coarse, medium, and fine. Our coarse scale considered all shorelines within the outer boundary of the general oiled area ("oiled"; Klosiewski and Laing 1994). The medium scale was created by combining one to five transects into groups of transects to create areas similar in size to the bays used by Murphy et al. (1997). The fine scale simply used a single transect as the sample unit. To compare results from our study (where data were analyzed at three scales) to other studies (where data were analyzed at one scale), we determined that a taxon exhibited an oil spill effect only if there were at least three significant results for that taxon rather than one. The chi square analysis on shoreline types in oiled and reference areas was conducted using the medium scale.

We decided *a priori* to use an unconventional alpha level of 0.20 to help balance the Type I and Type II errors and to allow us to compare our results to studies of the short-term effects of the *Exxon Valdez* oil spill on marine bird populations, where an alpha level of 0.20 was used (Wiens et al. 1996, Day et al. 1997, Murphy et al. 1997). A consequence of conducting many statistical tests is that by chance alone some of the results may be statistically significant. Accordingly, in this paper we looked at patterns and the strength of significant results and interpreted those patterns in light of our knowledge of life histories of the affected taxon.

We compared the pre-spill bird densities to bird densities for each post-spill year. Because of the nature of the data it was necessary to use two different statistical methods to analyze the data at three scales. For the fine and medium scales, we used a two-tailed t-test, and for the coarse scale, we used a ratio estimator with a two-tailed z-test.

We followed a similar approach used by Murphy et al. (1997) for testing for oil spill effects at fine and medium scales. We used a BACI-type design (Green 1979) and did a paired comparison on the bird densities measured in the same transects (fine scale) or on the same group of transects (medium scale) before and after the oil spill, then compared the mean differences for the oiled area and reference area. If the bird densities were lower in the oiled area post-spill than expected based on the pre-spill/post-spill change in the reference area, it was considered a negative oil spill effect. If the bird densities were higher, it was considered a positive effect. Recovery of an injured taxa was defined as lack of an effect (Murphy et al. 1997). This approach to detecting effects and recovery puts the burden of proof on the data to demonstrate an effect, but not to demonstrate recovery, which is a fairly liberal definition of recovery and not consistent with the requirements to show an effect.

The constant 0.167 was added to all density estimates to avoid calculating a log of zero, and adjusted densities (N km^{-2} of transect), d, were then transformed by ln(d) (Murphy et al. 1997).

To determine the amount of change pre- to post-spill at the fine and medium scales, δ_i, we subtracted the log(bird density) for each transect or group, pre-spill, from the log(bird density) for the corresponding transect or group, post-spill:

$$\delta_I = \ln[d(\text{post-spill})] - \ln(d(\text{pre-spill})]$$

Standard two-sample two-tailed *t*-tests were used to compare the mean of the differences, $\bar{\delta}_o$ and $\bar{\delta}_u$, between oiled and reference areas, respectively. To detect oil spill effects at the coarse scale, we again used a BACI analysis for all transects in an "oiled" area relative to all transects in a reference area for pre- and post-spill. We used the estimator for the ratio of random variables (ratios of totals of bird counts to area surveyed in an "oiled" area relative to a reference area, pre- and post-spill) (Cochran 1977). Data were not transformed to logarithms. The statistical methods are not easily referenced to standard textbooks and are described in more detail in Appendix 2.

Power of the statistical tests was calculated for a 50% reduction (or equivalently a two-fold increase, after Murphy et al. 1997) in densities relative to the mean differences in the reference area, pre-spill versus post-spill for each taxa for each year. Methods based on normal theory for approximating power of two-sample *t*-tests and *z*-tests were used (Zar 1984). Estimated variances for the oiled and reference areas were used in the approximations.

Two taxa (Black Oystercatcher and Pigeon Guillemot) had ≥ 50% power to detect these effects for all three scales and all years (Appendix 3). Six taxa (loons, cormorants, scoters, goldeneyes, Bald Eagles, and murres) had ≥ 50% power to detect effects for all years at the fine and coarse scales. All taxa had at least ≥ 50% power to detect effects at the fine scale (Appendix 3). Scientific names of birds are given in Appendix 1.

RESULTS

OIL SPILL EFFECTS

General patterns and persistence of effects. Fourteen marine bird taxa were analyzed for oil spill effects. The effect was considered negative if bird densities were lower in the oiled area after the oil spill than expected based on observed changes in the reference area. The effect was considered positive if bird densities were higher in the oiled area after the oil spill than expected based on observed changes in the reference area. We considered there to be no effect if bird densities were not different in the oiled area after the oil spill than expected based on observed changes in the reference area. If bird populations changed by random chance, we would expect to see 33% of the taxa to fall into each category. Of the birds analyzed, nine taxa (64%) showed a negative effect, two (14%) showed no effect, and three (21%) showed a positive effect (Fig. 2, Appendix 4). Loons, cormorants, Harlequin Ducks, goldeneyes, mergansers, Black Oystercatchers, Black-legged Kittiwakes, murres, and Pigeon Guillemots were negatively affected. Scoters and Mew Gulls showed no effect. Glaucous-winged Gulls, murrelets and terns showed a positive effect.

Of the nine taxa that showed negative effects, several continued to show effects through 1998. Pigeon Guillemots, murres, cormorants, goldeneyes, and mergansers showed negative effects in most years from 1989 to 1998 (Fig. 2). Harlequin Ducks showed negative effects in 1990 and 1991. Black Oystercatchers showed negative effects in 1990, 1991, and 1998. Loons showed weak evidence of a negative effect in 1989 and 1993. Black-legged Kittiwakes showed negative effects in 1989, 1996, and 1998, with a positive effect in 1993.

Effects relative to foraging style. The oiling effects relative to foraging style were dramatic. Seven of the nine taxa that feed by diving underwater showed negative oiling effects (Fig. 2, Appendix 1). Of the four taxa that feed at the surface of the water, two showed a positive oiling effect, one showed no effect, and one showed a negative effect. Black Oystercatchers, which forage on molluscs and other invertebrates in the intertidal, showed a negative oiling effect.

Comparison of spatial scales. The total number of significant negative effects detected were slightly greater at the medium scale than at the fine and coarse scales. Significant negative effects numbered 29, 24, and 25, respectively (Fig. 2). At the taxon level, there were some obvious differences in the effects that were detected

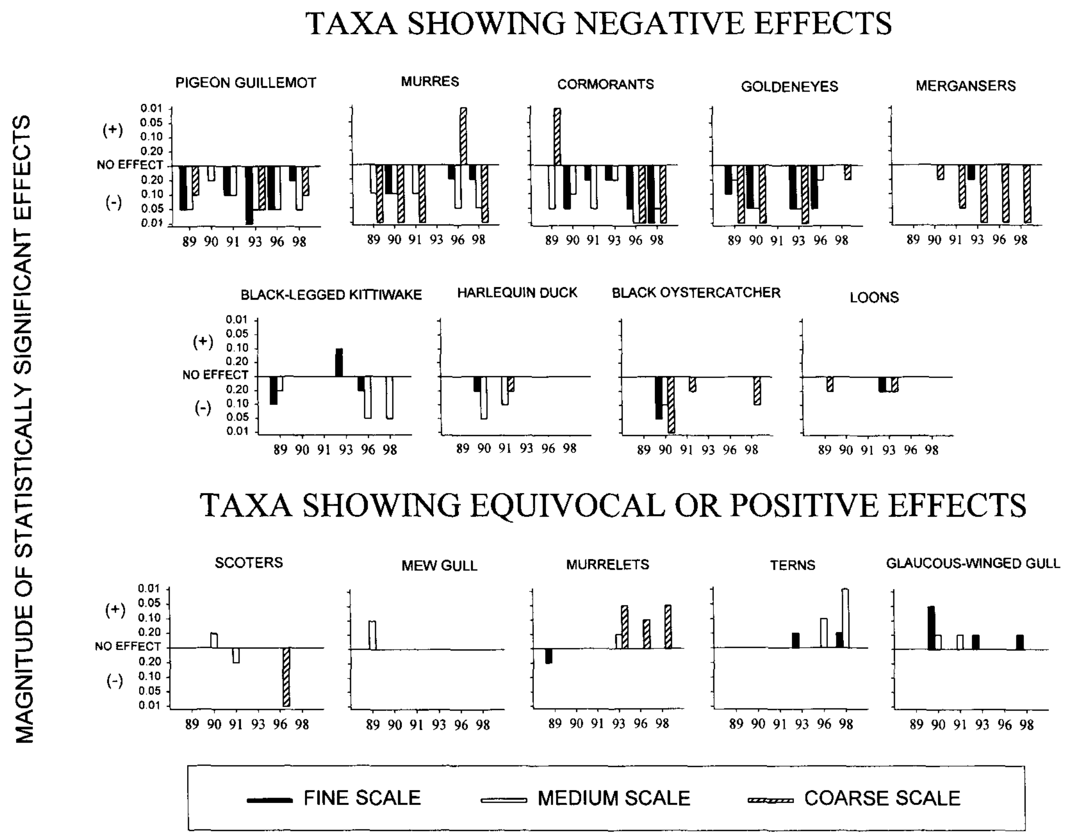

FIGURE 2. Magnitude and duration of statistically significant oil spill effects for 14 taxa analyzed at three spatial scales (fine, medium, coarse) during six post-spill surveys conducted from 1989 to 1998. Results were determined by BACI analyses, which were done by comparing marine bird densities pre- to post-spill between oiled and reference transects in Prince William Sound, Alaska. The direction of the vertical bar indicates whether the effect was positive (+) or negative (−). The length of the bar indicates the strength (P-value) of the result: from $P \leq 0.01$ (longest bars) to $P \leq 0.2$ (shortest bars); P-value given on x-axis. No bars for a given year indicates that no effects were detected at any of the three spatial scales during that year.

among scales. Cormorants and Pigeon Guillemots, which forage over short distances during the summer (Kuletz 1983, Birt et al. 1987), exhibited stronger effects at finer scales, whereas murres, which forage over wide ranges (Schneider and Hunt 1984), showed stronger effects over broader scales. Mergansers, which may travel large distances during summer to molt (Palmer 1976), showed stronger effects at the coarse scale.

DISCUSSION

Inherent in the BACI analyses are three assumptions: (1) that birds in the reference area were not affected by the oil spill, (2) that the birds in the spill area and in the reference area are closed populations, and (3) that changes in bird density in the reference area reflect changes that would have occurred in the oiled area had the spill not taken place. We expect that assumption three was generally met, but for some taxa that eat forage fish it may have been violated (see section below on detecting oil spills in a changing environment). The effect of a violation of assumption three could exaggerate or obscure oil spill effects. We recognize that assumptions one and two were likely violated. The effect of these violations would be to reduce our ability to detect oil spill effects using a BACI analysis, which would cause our estimates of oiling impacts to be conservative.

STRENGTH, DURATION, AND POTENTIAL CAUSE OF NEGATIVE EFFECTS

Although 9 of the 14 taxa showed a negative oil spill effect, the strength and duration of these effects varied among taxa. We conclude that cormorants, goldeneyes, mergansers, murres, and Pigeon Guillemots exhibited strong evidence of negative oil spill effects nine years after the oil spill. Harlequin Ducks and Black Oystercatchers displayed strong evidence of negative oil spill effects a few years after the spill and may be recovering. Black-legged Kittiwakes demonstrated sporadic negative effects. These results combined with data on the changes in the sizes of kittiwake colonies (D. B. Irons, unpubl. data) indicate that observed effects were probably the result of changes in foraging distribution of birds rather than a change in breeding numbers. Kittiwakes are capable of foraging broadly and may have avoided oiled areas in 1989, 1996 and 1998 (see Irons 1996). It is not known whether these changes in foraging distribution were influenced by the oil spill. Loons exhibited weak evidence of a negative effect.

Six of the taxa showed no effect or a positive effect. Scoters and Mew Gulls demonstrated no effect. Glaucous-winged Gulls displayed strong evidence of a positive effect. The reason for this is not clear. Murphy et al. (1997) suggested that boats cleaning up the oil spill may have attracted gulls and caused an increase in the oiled area. The increase in murrelets and terns four years after the spill may be related to an increase in sand lance (*Ammodytes hexapterus*) in the oiled area. Murrelets and terns eat many sand lance in the Gulf of Alaska (Sanger 1987, Kuletz et al. 1997) and may have responded to the increase in prey in recent years. Independent data on the abundance of sand lance schools in PWS from 1995 to 1998 show a relative increase in the oiled area (Brown et al. 1999, E. D. Brown, unpubl. data).

The results of this study demonstrated that their was no indication of recovery in the number of birds for several taxa nine years after the oil spill. Lack of an increase in numbers can occur because fecundity, survival, or immigration is not sufficient to allow recovery. Although the present study did not investigate reasons and mechanisms for persistent effects, other studies provide insight of potential mechanisms.

Exxon Valdez oil has persisted on some shorelines in PWS and Shelikof Strait for several years after the spill. *Exxon Valdez* oil has been found on the shores of PWS and entering the water as late as 1997 (Hayes and Michel 1999). Four years after the spill, residual oil in protected PWS mussel beds had been a source of chronic contamination of mussels, and contamination was expected to continue for several years (Babcock et al. 1996). Furthermore, *Exxon Valdez* oil deposited outside PWS in Shelikof Strait was only slightly weathered because after the oil left PWS much more of it turned to mousse, which resists weathering (Irvine et al. 1999).

Birds living in the oiled area ingested more oil than birds living in the reference area through 1999. The Nearshore Vertebrate Predator Project (Holland-Bartels et al. 1998) assessed continued exposure of birds and otters to oil using expression of cytochrome P4501A, an enzyme induced by polynuclear aromatic hydrocarbons or halogenated aromatic hydrocarbons. Holland-Bartels et al. (1998) compared P4501A levels in animals from the oiled and reference areas and found significantly higher levels of P4501A in Pigeon Guillemot, Harlequin Duck, and Barrow's Goldeneye that resided in the oiled area than in birds that resided in the reference area. Significant differences also were found in sea otters (*Enhydra lutris*) and river otters (*Lutra canadensis*). However, it is not possible to identify whether or not these hydrocarbons are from *Exxon Valdez* oil, they may be from some other source, such as discharge from other vessels or natural sources.

Other studies have compared the fecundity and survival of birds in oiled and reference areas. Harlequin Duck survival was lower in the oiled area than in the reference area (Holland-Bartels et al.1998). Pigeon Guillemot fecundity was lower in the oiled area post-spill than prespill (G. H. Golet, unpubl. data).

There is evidence that high quality prey (i.e., sand lance, Pacific herring [*Clupea pallasii*], and capelin [*Mallotus villosus*]) for birds were less abundant in PWS for a number of years after the spill than pre-spill. High-lipid fish were less available for Pigeon Guillemots and Marbled Murrelets after the spill than before the spill (Kuletz et al. 1997, Golet et al. 2000). Juvenile Pacific herring abundance declined in PWS after the spill (Brown et al. 1996). Reasons for these declines are not clear, but there is evidence that oil (Brown et al. 1996) and natural causes (Kuletz et al. 1997, Agler et al. 1999, Pearson et al.

1999) played a role. Overall, results of these studies suggest that persistent oil in the environment and reduced prey abundance may be affecting the recovery of marine birds in PWS.

COMPARISON TO OTHER STUDIES

The *Exxon Valdez* oil spill was a major perturbation and attracted much attention. There have been three other papers published on the short-term effects of the oil spill on marine birds in PWS using at-sea survey data: Wiens et al. (1996), Murphy et al. (1997), and Day et al. (1997). We compared our results to Murphy et al. (1997) because they also used pre-spill and post-spill data to determine oil spill effects. Wiens et al. (1996) and Day et al. (1997) used only post-spill data. Murphy et al. (1997) used data from the same pre-spill study (Irons et al. 1988) that we did, and compared data using a BACI-type analysis. However, they used data from 10 bays collected over three years and we used data from 123 transects collected in six years over a nine-year span. Murphy et al. (1997) also chose a different oil/unoiled criterion than the present study. Murphy et al. (1997) used an oiling index (range 0–400) and considered bays with an index value of <100 to be unoiled. The present study considered a transect that had any oil on it to be oiled. This difference in categorization of oiling affected 26% of the transects in the present study.

Generally the results of the two studies were similar and suggest that differences that do emerge may be due to the sample size and power involved in the studies. Of the nine taxa that were analyzed by both studies, Murphy et al. (1997) found that three (33%) of the taxa examined were negatively affected. Our study found that six (66%) of the taxa were negatively affected. Murphy et al. (1997) had a sample size of 10, and we had a sample size of 45, at the medium scale. Murphy et al. (1997) and our study determined the power to detect a 50% decline or a 100% increase for each species for each year. Generally, the power for our study was higher than that of Murphy et al. (1997), but there was much variation among species in both studies.

Comparisons among the studies at the taxon level is difficult because several taxa that we analyzed were not analyzed by Murphy et al. (1997) and vice versa. Murphy found three taxa to be negatively affected. We found negative effects on those three taxa and we found negative effects for six other taxa, of these Murphy et al. (1997) analyzed data for only two of the taxa: Black-legged Kittiwake and Harlequin Duck. It also is difficult to compare the duration of effects between the two studies because Murphy et al. (1997) collected data for only three post-spill years and we report on data that were collected over nine post-spill years. Murphy et al. (1997) found that the number of negative effects decreased from two to none by 1991, suggesting that recovery was occurring. Our study found results similar to Murphy et al. at the medium scale for the first three years. However, in 1993, 1996, and 1998, effects persisted and the indications of recovery had disappeared for many taxa (Fig. 2).

Prior to the *Exxon Valdez* oil spill, oil spill effects on marine birds were generally detected by either finding oiled carcasses on beaches (Bourne 1968, Stowe and Underwood 1984) or by a change in the number of breeding seabirds at one or more colonies (Stowe 1982) rather than a change in bird populations found in and around an oiled area (Harrison and Buck 1967), and most studies lasted only a year or two. The situation of the *Exxon Valdez* oil spill was different. There were pre-spill data on several taxa of marine birds in and around the area that was oiled and we were able to collect data over nine post-spill years. As a result, we were able to conduct a comprehensive study of potential oil spill effects on several bird taxa and determine whether effects lingered. The persistent effects found in several taxa were somewhat unexpected given that few earlier studies detected long-term effects. However, it should be noted that long-term effects (i.e., through 1998) of the *Exxon Valdez* oil spill were also detected on survival rates of sea otters in Prince William Sound (Monson et al. 2000).

The effects of the *Exxon Valdez* oil spill on marine birds have been detectable over nine years for several potential reasons. First, we continued to look for effects for nine years. Second, the spill occurred in PWS, a partially enclosed body of water, and much oil was deposited on hundreds of kilometers of shoreline rather than drifting unimpeded out to sea (O'Clair et al. 1996). Third, oil remained on the shorelines for years after the spill (Hayes and Michel 1999, Irvine 1999). Fourth, recovery of piscivorous taxa in PWS may be slow because of poor feeding conditions (Brown et al. 1996, Agler et al. 1999, Golet et al. 2000).

OIL SPILL EFFECTS RELATIVE TO FORAGING STYLE

King et al. (1979) ranked several species of marine birds according to their vulnerability to oil; their rankings were based on 20 factors that affect survival. Species that dive underwater for food were ranked as more susceptible to oiling than surface-feeding species. The disparity in rankings between divers and non-divers was largely due to behavioral differences involving foraging, resting, and escape responses. Divers were thought to be more susceptible to oiling than non-divers because they spend more time resting on the water, and when foraging divers dive under the water they may re-surface in oil. Also, their escape response is to dive, which increases the chances of surfacing in oil, whereas the non-diving species fly to escape. Additionally, non-divers may avoid foraging in heavily oiled areas because prey are difficult to see from the air when the surface is covered with oil (Irons 1996).

The results from the present study are consistent with rankings of King et al. (1979). Most of the species that dive for their food showed a negative oil spill effect, whereas only one of the surface-feeding species showed a negative effect (Fig. 2, Appendix 1). Piatt et al. (1990) and Murphy et al. (1997) also found that diving species were more affected by the *Exxon Valdez* oil spill than non-diving species. However, it should be recognized that the King et al. (1979) vulnerability rankings generally refer to immediate oiling effects and not long-term effects. Immediate effects are often from birds becoming oiled and long-term effects may be related to other factors such as oiled prey.

EFFECTS OF SCALE AND OILING PATTERN IN DETECTING OIL SPILL EFFECTS ON BIRDS

It has long been recognized that there are scale-dependent problems associated with assessing avian populations (Wiens 1981). Assessing the effects of an oil spill on avian populations also has scale-dependent issues. Problems arise when birds move in and out of oiled areas. In this study we grouped data at three different spatial scales to investigate the effect of scale on detecting effects of oil spills. The results showed that effects were different at different scales for some taxa and these differences appeared to be related to the scale at which birds travel to forage or molt. To help understand factors that influence the detection of oil spill effects, we have outlined three general properties involving the influence of scale, bird movement, and pattern of oiling. These properties mainly apply to BACI study designs and relate to whether or not birds in the reference area are affected by the oil spill. There are also two assumptions: (1) that birds which enter oiled areas are negatively affected and birds that do not enter oiled areas are not affected, and (2) that birds do not actively try to avoid oil. These general properties are:

(1) As the size of a bird's home range increases, the ability to detect oil spill effects decreases.

(2) As the number of borders between oiled and unoiled areas (i.e., the number of unoiled areas within a greater oil spill region) increases, the ability to detect effects on mobile species decreases.

(3) The scale at which the data are analyzed affects the ability to detect oil spill effects on birds when there are pockets of unoiled areas within a greater oiled region. There are two situations when sampling at the incorrect scale would reduce the ability to detect oil spill effects because some birds in unoiled pockets would be unaffected but considered oiled and vice versa. The first case would occur when birds' home ranges are much smaller than the unoiled pocket and the sampling unit is larger than the unoiled pocket. In this case birds in the unoiled pocket would be unaffected, but would be considered oiled because the scale of the sampling unit was too large. The second case would occur when birds' home ranges are larger than the unoiled pocket and the sampling unit is smaller than the unoiled pocket. In this case all birds in the unoiled pocket likely would be oiled, but the pocket would be considered unoiled because the scale of the sampling unit was too small. The results of these confounding situations is that for birds like Pigeon Guillemots with small home ranges we would be less likely to detect oil spill effects at our coarse scale than our fine scale, and for birds like murres with large home ranges we would be less likely to detect oil spill effects at our fine scale than our coarse scale.

Given these general properties, we recognize that our ability to detect oil spill effects of the *Exxon Valdez* oil spill was confounded because the mosaic pattern of oiling created many borders between oiled and unoiled areas. Also, we were less likely to detect effects for birds like Black-legged Kittiwakes and murres, which have large home ranges, than for Pigeon Guillemots and cormorants, which have small home ranges. Birds with small home ranges showed more oil spill effects when using a small spatial scale for analyses, and birds with large home ranges showed more oil spill effects when using a large spatial scale for analyses. We can conclude that when there are unoiled pockets within an oiled area, the chances of detecting oil spill effects will be greatest if the data are collected and analyzed at a spatial scale that matches the birds' home range.

DETECTING OIL SPILL EFFECTS IN A CHANGING ENVIRONMENT

The ability to detect oil spill effects on birds may be complicated by natural variation in populations (Wiens and Parker 1995). The *Exxon Valdez* oil spill provides an example of this. Many of the pre-spill data that were available on birds within the spill area were collected in the 1970s. Many of the murre colonies were counted only in the 1970s (Piatt and Anderson 1996) and some data on marine bird numbers were collected in PWS in 1972 (Dwyer et al. 1976, Klosiewski and Laing 1994).

It was not recognized at the time of the spill, but we now know that the Gulf of Alaska (GOA) experienced a climatic shift about 1978. There was an abrupt change in sea-surface temperature and in several indicators of long-term climatic variability in the GOA (Francis et al. 1998). Coincident with that change, some important prey species of marine birds changed. For example, capelin decreased and pollock (*Theragra chalcogramma*) increased in abundance (Piatt and Anderson 1996, Francis et al. 1998). Apparently as a result of declining high-quality prey (e.g., capelin), many species of marine birds that depend upon schooling forage fish declined in PWS and the GOA. Agler et al. (1999) found that 14 of 17 piscivorous marine bird taxa declined in PWS from 1972 to 1989, and that 17 of 21 marine bird taxa declined from 1976 to 1986 in the GOA along the Kenai Peninsula. However, birds that depend on benthic invertebrates for food, such as Harlequin Duck and goldeneyes, did not decline over this period. Piatt and Anderson (1996) found that several murre colonies outside the spill area declined from the late 1970s to 1989. It appears that the climatic shift did not affect PWS equally. Suryan and Irons (unpubl. data) found that the number of nesting kittiwakes in southern PWS declined from 1972 to 1985, while numbers increased in northern PWS. They attributed this change to a change in food availability that may have been associated with the 1978 climatic shift.

In the midst of a large-scale climatic shift, how can we detect oil spill effects? Three important factors helped us separate oil spill effects from the climatic shift. First, in PWS we had data that were collected in 1984 and 1985, only a few years before the spill, whereas the climatic shift occurred about 1978 and most of the declines associated with that shift had abated by 1984 (D. B. Irons, unpubl. data). Second, the suite of species that declined after the climatic shift and the suite of species that declined after the oil spill were largely different. Most of the species that declined from the climatic shift consume schooling forage fish and many species that are nearshore benthos feeders did not decline (Agler et al. 1999). Many species that declined from the oil spill are nearshore benthos feeders and several species that consume schooling forage fish showed no effect or a positive effect from the oil spill. Third, the oil spill and the climatic shift occurred at different spatial scales. Within the spill area, the oil spill contaminated some beaches, but left adjacent beaches untouched by oil, creating a patchwork pattern of oiling. The climatic shift occurred at the scale of the entire GOA and perhaps larger (Francis et al. 1998). Our findings that some species with small home ranges showed greater effects at small scales than at large scales is consistent with a perturbation of the scale and pattern of the oil spill and not the scale of the climatic shift.

In conclusion, we found that 64% of the 14 taxa analyzed exhibited negative oil spill effects and 36% of the taxa showed persistent effects nine years after the spill. Most taxa that were affected dive for their food. The spatial scale at which analyses were done affected the results for some taxa. The effects lasted longer than those reported by many other oil spill studies. The reason for this may be related to the persistence of oil and reduced levels of forage fish in Prince William Sound.

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APPENDIX 1. Common and scientific names and foraging mode of bird species/species groups mentioned in text.

Species/Species group	Common name	Scientific name	Foraging mode
Loons	Red-throated Loon	*Gaia stellata*	diver
	Pacific Loon	*Gavia pacifica*	diver
	Common Loon	*Gavia immer*	diver
	Yellow-billed Loon	*Gavia adamsii*	diver
Cormorants	Double-crested Cormorant	*Phalacrocorax auritus*	diver
	Pelagic Cormorant	*Phalacrocorax pelagicus*	diver
	Red-faced Cormorant	*Phalacrocorax urile*	diver
Harlequin Duck	Harlequin Duck	*Histrionicus histrionicus*	diver
Scoters	Black Scoter	*Melanitta nigra*	diver
	Surf Scoter	*Melanitta perspicillata*	diver
	White-wing Scoter	*Melanitta fusca*	diver
Goldeneyes	Common Goldeneye	*Bucephala clangula*	diver
	Barrow's Goldeneye	*Bucephala islandica*	diver
Mergansers	Common Merganser	*Mergus merganser*	diver
	Red-breasted Merganser	*Mergus serrator*	diver
Black Oystercatcher	Black Oystercatcher	*Haematopus bachmani*	intertidal feeder
Mew Gull	Mew Gull	*Larus canus*	surface feeder
Glaucous-winged Gull	Glaucous-winged Gull	*Larus glaucescens*	surface feeder
Black-legged Kittiwake	Black-legged Kittiwake	*Rissa tridactyla*	surface feeder
Terns	Caspian Tern	*Sterna caspia*	surface feeder
	Arctic Tern	*Sterna paradisaea*	surface feeder
	Aleutian Tern	*Sterna aleutica*	surface feeder
Murres	Common Murre	*Uria aalge*	diver
Pigeon Guillemot	Pigeon Guillemot	*Cepphus columba*	diver
Murrelets	Marbled Murrelet	*Brachyramphus marmoratus*	diver
	Kittlitz's Murrelet	*Brachyramphus brevirostris*	diver

APPENDIX 2

Statistical methodology used to detect oil spill effects at the coarse scale. A BACI design for all transects in an "oiled" area relative to all transects in a reference area for pre- and post-spill was used with the estimator for the ratio of random variables (ratios of totals of bird counts to area surveyed in an "oiled" area relative to a reference area, pre- and post-spill) (Cochran 1977).

The statistical methods are not easily referenced to standard textbooks and are described in more detail in the following paragraphs. The general estimator of a ratio is the ratio of means (or, equivalently, ratio of totals)

$$\hat{R} = \frac{\bar{y}}{\bar{x}}$$

with corresponding estimated variance

$$v(\hat{R}) = \left(\frac{1}{n}\right)(\hat{R}^2)\left[\frac{s_y^2}{\bar{y}^2} + \frac{s_x^2}{\bar{x}^2} - \frac{2r_{xy}s_y s_x}{\bar{y}\bar{x}}\right]$$

where s^2_y, s^2_x and r_{xy} are respectively the sample variance of the y's (bird counts), x's (area surveyed), and the sample correlation of the x's and y's. Define the following:

$\hat{R}_{oa} = \dfrac{\bar{y}_{oa}}{\bar{x}_{oa}}$ ratio of the mean number of birds to mean area of transects for the oiled area after the spill

$\hat{R}_{ob} = \dfrac{\bar{y}_{ob}}{\bar{x}_{ob}}$ ratio of the mean number of birds to mean area of transects for oiled area before the spill

$\hat{R}_{ra} = \dfrac{\bar{y}_{ra}}{\bar{x}_{ra}}$ ratio of the mean number of birds to mean area of transects for reference area after the spill and

$\hat{R}_{rb} = \dfrac{\bar{y}_{rb}}{\bar{x}_{rb}}$ ratio of the mean number of birds to mean area of transects for reference area before the spill

The variances of the ratios are calculated by applying the above formula, $v(\hat{R})$. Define:

$$\hat{R}_o = \frac{\hat{R}_{oa}}{\hat{R}_{ob}} \quad \text{and} \quad \hat{R}_r = \frac{\hat{R}_{ra}}{\hat{R}_{rb}}$$

The variance of R_o (variance of R_r is calculated same way) was estimated by:

$$v(\hat{R}_o) = (\hat{R}_{oa}{}^2)\left[\frac{v(\hat{R}_{oa})}{\hat{R}_{oa}{}^2} + \frac{v(\hat{R}_{ob})}{\hat{R}_{ob}{}^2}\right].$$

Finally, the estimated oil spill effect is given by

$$\hat{R} = \frac{\hat{R}_o}{\hat{R}_r}$$

Values greater than 1.0 indicate a positive oil spill effect and values less than 1.0 indicate a negative oil spill effect. The variance of $\hat{R}$ was estimated by a second application of the formula above for $v(\hat{R}_o)$. A two-tailed z-test was then conducted using the same significance levels as for the fine and medium scales to determine whether the estimated effect was significantly different from 1.0.

APPENDIX 3. Results of power analyses for the pre-spill, post-spill comparisons of bird densities in Prince William Sound during the summer. Power was calculated assuming a 50% reduction for a 100% increase for each taxon for each year. Power was calculated for each spatial scale for analyses that were conducted. Pre-spill data were collected in 1984–1985 (Irons et al. 1988).

		Pre-spill and post-spill comparisons					
Species/Taxon	Scale	1989	1990	1991	1993	1996	1998
Loons	Fine	1.00	1.00	1.00	1.00	1.00	0.88
	Medium	0.97	0.99	0.91	0.91	0.96	0.88
	Coarse	0.43	0.25	0.41	0.41	0.45	0.42
Cormorants	Fine	0.99	1.00	0.99	0.99	0.99	0.98
	Medium	0.83	0.85	0.83	0.90	0.90	0.98
	Coarse	0.63	0.53	0.59	0.80	0.73	0.52
Harlequin duck	Fine	0.76	0.86	0.81	0.80	0.76	0.64
	Medium	0.47	0.62	0.48	0.52	0.52	0.64
	Coarse	0.26	0.62	0.61	0.23	0.25	0.33
Scoters	Fine	1.00	1.00	0.98	1.00	1.00	0.99
	Medium	0.71	0.83	0.66	0.91	0.79	0.99
	Coarse	0.20	0.23	0.21	0.21	0.34	0.27
Goldeneyes	Fine	0.99	1.00	1.00	1.00	1.00	0.99
	Medium	0.79	1.00	1.00	0.97	0.99	0.99
	Coarse	0.30	0.32	0.30	0.33	0.34	0.30
Mergansers	Fine	0.69	0.85	0.80	0.69	0.71	0.64
	Medium	0.43	0.53	0.52	0.41	0.44	0.64
	Coarse	0.62	0.70	0.55	0.72	0.67	0.65
Black Oystercatcher	Fine	0.90	0.96	0.95	0.94	0.94	0.74
	Medium	0.72	0.73	0.79	0.71	0.69	0.74
	Coarse	0.53	0.69	0.67	0.58	0.26	0.45
Mew Gull	Fine	0.67	0.76	0.68	0.76	0.72	0.57
	Medium	0.47	0.52	0.44	0.50	0.48	0.56
	Coarse	0.26	0.31	0.77	0.28	0.71	0.32
Glaucous-winged Gull	Fine	0.58	0.61	0.63	0.53	0.63	0.50
	Medium	0.41	0.54	0.43	0.35	0.43	0.49
	Coarse	0.25	0.26	0.28	0.27	0.64	0.30
Black-legged Kittiwake	Fine	0.65	0.74	0.76	0.77	0.62	0.58
	Medium	0.58	0.51	0.52	0.62	0.41	0.58
	Coarse	0.29	0.28	0.32	0.30	0.74	0.52
Terns	Fine	0.76	0.79	0.84	0.71	0.80	0.97
	Medium	0.50	0.51	0.61	0.40	0.50	0.75
	Coarse	0.23	0.28	0.29	0.20	0.26	0.29
Murres	Fine	0.97	0.97	0.95	0.95	0.95	0.97
	Medium	0.78	0.74	0.64	0.68	0.83	0.98
	Coarse	0.31	0.28	0.29	0.20	0.30	0.31
Pigeon Guillemot	Fine	0.79	0.83	0.80	0.83	0.77	0.64
	Medium	0.61	0.65	0.63	0.80	0.70	0.63
	Coarse	0.70	0.82	0.87	0.78	0.80	0.81
Murrelets	Fine	0.60	0.65	0.67	0.68	0.72	0.56
	Medium	0.48	0.55	0.55	0.59	0.54	0.55
	Coarse	0.40	0.38	0.37	0.52	0.48	0.50

APPENDIX 4. Comparison of changes in marine bird densities pre- to post-spill between oiled and reference transects in Prince William Sound, Alaska. Pre-spill counts were made in 1984–1985 by Irons et al. (1988). Post-spill counts were made in six years from 1989 to 1998. Results of analyses are indicated as follows: * $P \leq 0.20$, ** $P \leq 0.10$, *** $P \leq 0.05$, and **** $P \leq 0.01$. Response refers to our conclusion as to how a taxon was affected by the oil spill.

		Percent difference					
Taxon (Response)	Scale	1989	1990	1991	1993	1996	1998
Pigeon Guillemot	Fine	−55***	−24	−42**	−68***	−54***	−47*
(Negative)	Medium	−66***	−43*	−55**	−51***	−56***	−65***
	Coarse	−50**	−29	−15	−51***	−37	−51**
Murres	Fine	−23	−32**	−25	−7	−26*	−30*
(Negative)	Medium	−47**	−47**	−54**	−27	−51***	−56***
	Coarse	−100****	−100***	−98****	1	100****	−100****
Cormorants	Fine	−2	−37***	−24*	−25*	−33***	−38****
(Negative)	Medium	−46***	−41**	−47***	−33*	−53****	−49***
	Coarse	100****	19	132	−59	−84****	−89****
Goldeneyes	Fine	−29**	−23***	−8	−32***	−19***	−13
(Negative)	Medium	−44*	−29***	−5	−45***	−24*	−25
	Coarse	−92****	−94***	−50	−90***	−50	−64*
Mergansers	Fine	−38	−17	−16	−43*	−27	−28
(Negative)	Medium	−34	−24	−45	−54	−43	−49
	Coarse	−19	−46*	−61***	−64***	−64****	−67****
Black-legged Kittiwake	Fine	−50**	11	−27	79**	−50*	−49
(Negative)	Medium	−53*	−23	−44	−6	−79***	−64***
	Coarse	121	101	79	67	−30	−6
Harlequin Duck	Fine	10	−36*	−28	−11	−24	−9
(Negative)	Medium	37	−61***	−65**	−18	−30	−24
	Coarse	182	−28	−50*	216	134	74
Black Oystercatcher	Fine	6	−40***	−6	26	−1	−11
(Negative)	Medium	−12	−47**	−28	4	−20	−15
	Coarse	−15	−83***	−44*	−24	0	−51**
Loons	Fine	6	14	0	−19*	−3	−11
(Negative)	Medium	−6	13	−22	−55*	−18	−23
	Coarse	−60*	0	−5	−58*	−25	−18
Scoters	Fine	4	11	−22	5	−17	−2
(None)	Medium	9	50*	−42*	34	−19	2
	Coarse	71	71	209	266	−100****	85
Mew Gull	Fine	15	40	−33	−13	4	−27
(None)	Medium	172**	44	−37	−4	16	−20
	Coarse	149	107	−2	35	44	24
Murrelets	Fine	−46*	12	−19	44	35	36
(Positive)	Medium	−9	43	−13	87*	42	4
	Coarse	43	51	10	147***	100**	144***
Terns	Fine	−15	2	−1	78*	30	142*
(Positive)	Medium	33	35	28	104	148**	401****
	Coarse	603	690	192	322	471	4,102
Glaucous-winged Gull	Fine	22	175***	44	98*	−10	101*
(Positive)	Medium	37	112*	124*	163	−34	111
	Coarse	136	44	33	33	−43	16