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EFFECTS OF THE *EXXON VALDEZ* OIL SPILL ON BIRDS: COMPARISONS OF PRE- AND POST-SPILL SURVEYS IN PRINCE WILLIAM SOUND, ALASKA[1]

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Abstract. We used data from pre- and post-spill surveys to assess the effects of the *Exxon Valdez* oil spill on the abundance and distribution of birds in Prince William Sound, Alaska. We conducted post-spill surveys during mid-summer (1989–1991) in 10 bays that had been surveyed prior to the spill (1984–1985) and that had experienced different levels of initial oiling from the spill (unoiled to heavily oiled). We evaluated whether there were changes in overall abundance across all bays between the pre-spill and post-spill sampling periods, and changes in abundance in unoiled/lightly oiled bays versus moderately/heavily oiled bays that would suggest oiling impacts. Of 12 taxa examined for changes in overall abundance, 7 showed no significant change, 2 (Bald Eagle and Glaucous-winged Gull) increased in abundance, and 3 (Red-necked Grebe, Pelagic Cormorant, and Pigeon Guillemot) decreased in abundance during all three post-spill years. Of the 11 taxa examined for differences in use of oiled versus unoiled habitats, 7 showed no significant response, 1 (Black-legged Kittiwake) exhibited a positive response to oiling, and 3 (Pelagic Cormorant, Black Oystercatcher, and Pigeon Guillemot) exhibited negative responses to initial oiling. We conclude that the impacts of this oil spill on abundance and distribution of birds were most evident in 1989, the year of the spill, and were most pronounced for Pigeon Guillemots. By 1991, signs of recovery were evident for all taxa that showed initial oiling impacts.

Key words: Prince William Sound, Alaska, Exxon Valdez, *oiling impacts, oil spill, marine birds, before–after comparisons.*

INTRODUCTION

When the *Exxon Valdez* went aground in Prince William Sound (PWS), Alaska, on 24 March 1989, the ensuing spill of approximately 41,000,000 L of Alaska North Slope crude oil became the largest marine oil spill in U. S. history. Marine birds, which are among the most conspicuous victims of oil spills at sea, experienced substantial mortality: nearly 30,000 carcasses were collected throughout the spill-affected area (PWS, Kenai Peninsula, Kodiak Island archipelago) during the spring and summer of 1989 (Piatt et al. 1990). Simulation models that used functions relating carcass recovery rates to total mortality produced a "best estimate" of total mortality of 375,000–435,000 birds (Ecological Consulting, Inc., 1991). Although not all scientists agree that mortality was this high (e.g., Erikson 1995), there is little doubt that many thousands of birds were killed as a direct consequence of the spill (Fry 1993, Parrish and Boersma 1995, Wiens 1995), mostly near large seabird colonies in the Gulf of Alaska. Perhaps as many as 42,000 birds died in PWS (Piatt et al. 1990, Ecological Consulting, Inc. 1991), with the greatest mortality documented for sea ducks, cormorants, murres, grebes, and murrelets.

We initiated a research program in June 1989 to examine the effects of the oil spill on marine birds. One aspect of this research evaluated both impacts to and recovery of use of oil-affected habitats by birds in PWS and along the Kenai Peninsula during a 3-year period (1989–1991) after the spill (Day et al. 1995, in press; Wiens et al. 1996). During that 3-year period, 15 sampling cruises were conducted during different seasons, and analyses of impacts and recovery reported by these studies were based exclusively on post-spill data. Another way to gauge the ef-

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fects of an oil spill on birds is to compare abundances of birds before and after the spill. If mortality or displacement from contaminated habitats was great, we would expect reduced densities of birds, particularly in the most heavily oiled sites. Here, we compare data collected in mid-summer during 3 years following the *Exxon Valdez* oil spill (1989–1991) with historical data collected before the oil spill.

Studies in PWS conducted in 1972 and 1973 (Dwyer et al. 1976) and during the summers of 1984 and 1985 (Irons et al. 1988) provide the only systematic pre-spill information on avian abundance and distribution. The data collected by Dwyer et al. had poor geographic correspondence with our study area and were not useful for making historical comparisons. The data collected by Irons et al (1988) had complete geographic overlap with our study area and good temporal overlap with surveys we conducted during mid-summer (July–August). We acquired the original survey data from the U. S. Fish and Wildlife Service and determined that 320 km of shoreline in the 10 bays that we were sampling also had been surveyed by Irons et al. (1988).

In this paper, we use mid-summer data from Irons et al. (1988) and from surveys that we conducted in 1989–1991 to evaluate two hypotheses:

$H_0$1: Densities of birds in the 10 study bays did not differ between 1984/1985 (pre-spill) and 1989–1991 (post-spill).

$H_0$2: Changes in densities of birds from pre-spill levels did not differ between oiled and unoiled bays during 1989–1991.

The first hypothesis was a simple "before–after" baseline comparison (Wiens and Parker 1995) that evaluated whether there were significant changes in overall densities of birds in the study area. We made no distinction between sampling sites (bays) that were oiled and those that were unoiled and did not separate oiling effects from natural factors that might have contributed to overall population changes between the two time periods.

The second hypothesis addressed a fundamentally different question, one about impacts caused by oiling. Among-bay differences in habitat, natural temporal variation, and interobserver differences were accounted for by comparing Irons et al.'s (1988) data from each sampling site with the post-spill data for that site, thereby allowing a clear assessment of effects of oiling on bird densities. This analysis has been described as a "pre/post pairs" design (Wiens and Parker 1995) and approximates a BACI (before–after/control–impact) design (Skalski and McKenzie 1982, Stewart-Oaten et al. 1986, 1992, Schroeter et al. 1993), which was developed for evaluating effects of planned perturbations (e.g., large-scale industrial developments).

METHODS

STUDY AREA

Prince William Sound (PWS) is a semi-enclosed body of salt water located in the northern Gulf of Alaska (Fig. 1). It is surrounded by the Chugach Mountains and is characterized by tidewater glaciers, fjords with steep, forested slopes and rocky shorelines, and numerous islands, islets, and intertidal rocks. The climate of the region is cool-maritime with frequent and intense storms; precipitation is highest in fall and winter (Isleib and Kessel 1973, Royer et al. 1990).

Approximately 16% of the 4,800 km of shoreline in PWS was oiled by the *Exxon Valdez* oil spill (Neff et al. 1995). Oiling of shorelines was discontinuous and patchy, and only some of the bays in the general path of the spill were inundated with oil. We selected 10 bays for study that were typical of western PWS: they were fjords with rocky shorelines, although they differed in size, shape, exposure to larger water masses, and other habitat features. Nine of these 10 bays were on or near the Knight Island group (Fig. 1) and were in the general path of the spill (Galt et al. 1991). The study bays ranged in area from 1.5 to 24.3 km^2; four were unoiled or lightly oiled, and six were moderately to heavily oiled. Of the 10 bays that we surveyed, 9 were surveyed by Irons et al. (1988) in 1984 and the 10th (Galena Bay) was surveyed in 1985; hence, the single survey of the 10 study bays is referred to here as 1984/1985.

We used bays as our basic sampling unit because they are discrete areas that can be described quantitatively in terms of initial exposure to oil. Bays were not chosen randomly, but instead were selected to be representative of the types of bays found in western PWS and to establish a gradient of oiling conditions (Day et al. 1995).

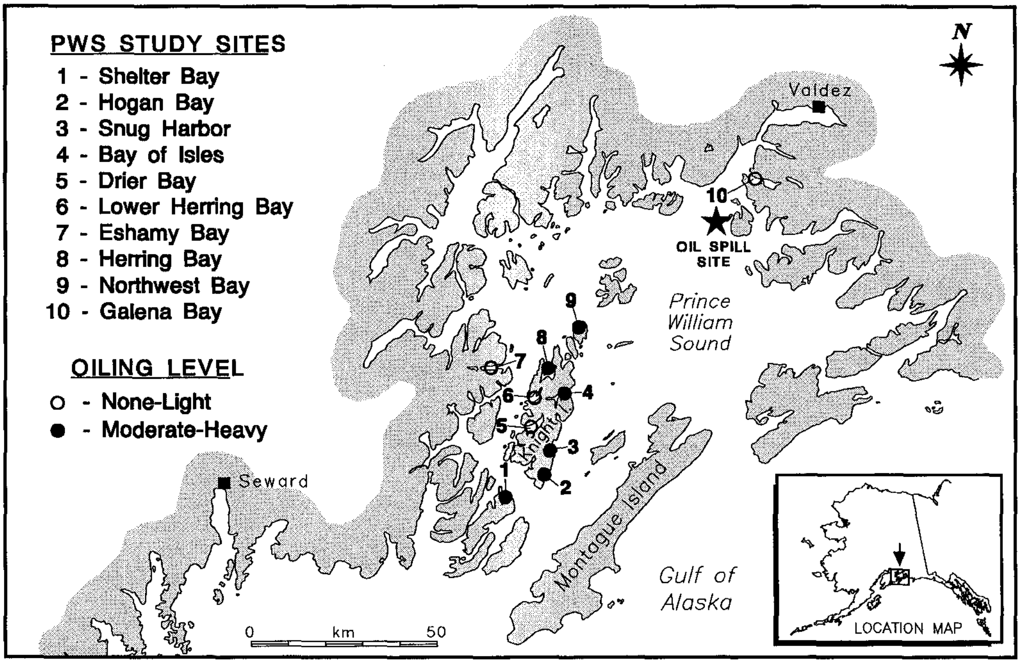

FIGURE 1. Locations of study bays in Prince William Sound, Alaska.

DATA COLLECTION

Our survey procedures generally were the same as those used by Irons et al. (1988). In both studies, nearshore areas were surveyed during midsummer (~20 July to 10 August) by slowly driving a small boat along the shoreline of a bay and identifying and counting all birds seen on the water within 200 m of shore and in the air over this zone. Although including flying birds in transect survey counts may bias density estimates (Tasker et al. 1981), we did not have a choice as to whether we would include flying birds because they were included in the pre-spill data set. We do not, however, consider inclusion of flying birds to have been a major source of bias in this instance because (1) we were not conducting our surveys at sea, where this problem is most acute, (2) accurate population estimates were not a goal of this research program, and (3) we accounted for natural variability by using replicate surveys. We also carefully searched for and counted birds in the intertidal and supratidal (≤ 100 m from high-tide line) zones; these zones appear to have been surveyed thoroughly by Irons et al. (1988) only for Bald Eagles and Black Oystercatchers (see Appendix 1 for scientific names of bird species). We typically drove our boats ~50 m from shore, although the actual distance from the shoreline often was greater because of the need to avoid intertidal and subtidal rocks. Irons et al. (1988) reported driving their boats ~100 m from shore; overall, we think that similar survey tracks were followed by both survey teams. Because both teams used experienced observers and traveled slowly during surveys, we do not think differences in distances would affect survey results.

We developed an oiling index to provide a continuous measure of the initial exposure of each bay to oil (Day et al. 1995). Data used to calculate the oiling index had been mapped in the field by Shoreline Cleanup Assessment Teams in 1989, and thus represented a mutually agreed upon (by government and Exxon scientists) and consistent system for measuring the amount of oil on the shorelines. Given the fjord-like structure of the bays and substantial tidal ranges that occur in PWS, we believe that these quantitative measures of oil on the shorelines provided a reasonable indication of the amount of oil to which avian habitats in a bay initially were exposed. Oiling on shorelines was categorized at five levels, ranging from "no oil" to "heavy oil" (Neff et al. 1995). For each bay, we

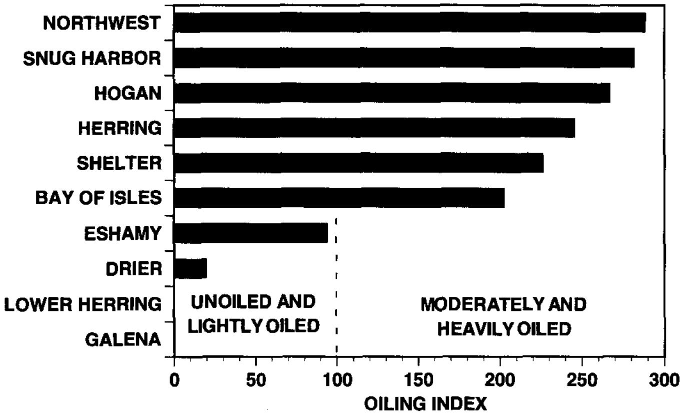

FIGURE 2. Oiling index values for the study bays in Prince William Sound, Alaska (modified from Day et al. 1995). Oiling indices reflect the initial oiling exposure for each bay. Dashed line separates unoiled and lightly oiled bays from moderately and heavily oiled bays.

calculated an oiling index value (see Day et al. 1995) that could range between 0 (100% no oil) and 400 (100% heavy oil). Actual index values calculated for the bays surveyed in PWS ranged between 0 and 288.2 (Fig. 2). For analyses, we combined the four unoiled and lightly oiled bays (index values < 100) to form a group of "unoiled" bays and the six moderately and heavily oiled bays (index values ≥ 200) to form a group of "oiled" bays.

DATA ANALYSIS AND INTERPRETATION

Although the length of shoreline surveyed in some bays differed slightly between 1984/1985 and 1989–1991, we standardized all counts of birds from nearshore surveys to linear densities (birds/km of shoreline surveyed) to facilitate comparisons. If Irons et al. (1988) recorded < *ca.* 20 individuals of a taxon in the 10 study bays, that taxon was omitted from our analyses. Consequently, data that we collected on shorebirds other than oystercatchers, wading birds, and corvids could not be compared with historical data.

Irons et al. (1988) visited each bay only once. We conducted multiple (3–5) visits to a bay on each cruise, so we averaged density estimates to calculate mean densities for each year. We expected the effects of oiling to be multiplicative (i.e., effects would be proportional to population levels in the various bays) and not additive (i.e., effects would be constant regardless of population levels in the various bays). Because statistical tests assume additivity, it thus was necessary to make the relationships additive by log-transforming the density estimates before conducting the statistical tests (Stewart-Oaten et al. 1986). We added a constant of 0.167 to all density estimates to avoid calculating a log of zero (Mosteller and Tukey 1977).

For each taxon, we calculated the amount of change (Δ) in densities that occurred in each bay (*i*) during each post-spill year (*after*) relative to the pre-spill baseline (*before*):

$$\Delta_i = \ln \text{density } (\textit{after}) - \ln \text{density } (\textit{before}). \quad (1)$$

These log-transformed changes in densities then were used to calculate the percent change between pre-spill and post-spill years across all bays and the percent difference in densities between oiled and unoiled bays in each post-spill year relative to the pre-spill baseline.

The overall change in densities between pre-spill and post-spill years across all bays for a given taxon was calculated by taking a mean of the changes for all 10 bays:

$$\text{overall change } (\bar{\Delta}) = \frac{\sum \Delta_i}{n}, \quad \text{where } n = 10. \quad (2)$$

The percent change across all bays and the percent difference between oiled and unoiled bays then were calculated as:

$$\text{percent change} = (e^{\bar{\Delta}} - 1) \times 100. \quad (3)$$

We used a one-sample *t*-test (Zar 1984) for

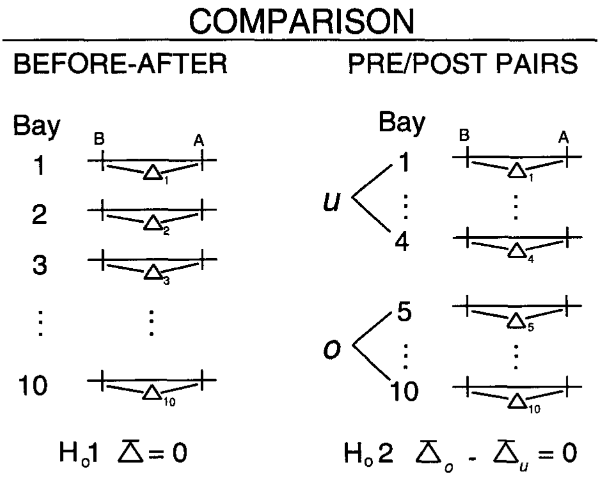

FIGURE 3. Approaches used to examine the effects of the *Exxon Valdez* oil spill on birds using historical data (Irons et al. 1988) and data collected during 3 years after the spill (1989–1991) in 10 bays in Prince William Sound, Alaska. The before–after comparisons (H_01) evaluated whether the mean differences (D) between pre-spill (B) bird densities and post-spill (A) densities differed from zero. The pre/post pairs comparisons (H_02) evaluated whether the mean difference between pre-spill (B) bird densities and post-spill (A) densities in oiled (*o*) bays differed from mean differences in unoiled bays (*u*).

each taxon to evaluate whether there was a significant change in density over all bays (H_01: $\bar{\Delta} = 0$; Fig. 3). We considered the overall abundance to have decreased (increased) across all bays if the mean change in densities was significantly lower (higher) than zero.

The difference in densities between oiled and unoiled bays for a given taxon was calculated as the difference between the mean difference for oiled (*o*) bays and the mean difference for unoiled (*u*) bays:

$$\text{difference in density} = \bar{\Delta}_o - \bar{\Delta}_u, \text{ where } n = 6 \text{ and } 4, \text{ respectively.} \quad (4)$$

The percent difference between oiled and unoiled bays then was calculated as:

$$\text{percent difference} = (e^{(\bar{\Delta}_o - \bar{\Delta}_u)} \times 100. \quad (5)$$

We used a two-sample *t*-test (Zar 1984) for each taxon to evaluate whether changes in mean densities differed between oiled and unoiled bays (H_02: $\bar{\Delta}_o - \bar{\Delta}_u = 0$; Fig. 3). We conducted tests only on those taxa that occurred in at least two bays in each of the two oiling categories during the post-spill years. We considered a negative (positive) oiling impact to have occurred if the mean change in densities between one or more of the post-spill years and the pre-spill year was significantly lower (higher) in the oiled bays than in the unoiled bays.

The combination of high variance and small sample size could reduce power to detect an oiling effect and increase the likelihood of making Type II errors (i.e., failing to detect a spill impact that actually did occur). To counter this problem, we established *a priori* that statistical significance for all comparisons would be tested with a two-tailed $\alpha = 0.20$. Thus, we sought to increase the likelihood of detecting oiling effects in these statistical analyses, realizing that by doing so we also increased the probability of detecting some oiling effects that did not actually occur (Type I errors) (Shrader-Frechette and McCoy 1993, Mapstone 1995, Wiens and Parker 1995). To aid in the interpretation of test results, we also tallied results at $\alpha = 0.05$ and 0.10.

Recovery was evaluated for all taxa that showed initial (1989) negative impacts by examining both trends in densities and statistics in 1990 and 1991. That is, we evaluated the percent change in densities between pre- and post-spill years, the results of statistical comparisons, and the power analyses to determine if a detected impact had diminished in subsequent years.

Finally, we conducted power analyses to assesses the probability of Type II errors and, hence, our ability to detect impacts. Power increases with increasing α level and sample size and with decreasing variance (Osenberg et al.

TABLE 1. Pre-spill counts, percent changes, results of *t*-tests, and overall responses of birds based on comparisons of densities during mid-summer before (1984/1985; Irons et al. 1988) and after (1989–1991) the *Exxon Valdez* oil spill in 10 bays in Prince William Sound, Alaska. A negative (positive) percent change indicates that densities decreased (increased) in the post-spill year. Percent change was calculated from log-transformed data. Significance codes for *t*-tests: * = $P \leq 0.20$, ** = $P \leq 0.10$, *** = $P \leq 0.05$.

Species/taxon	Pre-spill count	Percent change			Overall response
		1989	1990	1991	
Red-necked Grebe	19	−17.7**	−17.7**	−17.4**	Decrease
Pelagic Cormorant	28	−32.0***	−22.0***	−18.4***	Decrease
Harlequin Duck	103	−13.5	−6.4	−11.9	None
Common Merganser	101	9.5	−7.6	13.6	None
Bald Eagle	81	−3.9	55.8***	50.9***	Increase
Black Oystercatcher	24	−7.3	−9.4	5.3	None
Bonaparte's Gull	131	−2.4	11.6	−4.3	None
Mew Gull	825	−8.1	20.8	−33.2	None
Glaucous-winged Gull	579	75.9*	248.2***	197.4***	Increase
Black-legged Kittiwake	920	52.4	−4.4	38.8	None
Pigeon Guillemot	179	−55.7***	−56.5***	−37.5***	Decrease
"Total murrelets"	798	1.7	15.3	69.1**	None

1994). It also is a function of the amount of change (Δ) to be detected, because it is easier to detect a large change than a small one. Our interest in conducting power analyses was in evaluating the strength of, and therefore our confidence in, the statistical results. To facilitate comparing our tests among different taxa, we conducted the power analyses using a standardized amount of change. Because of the large amount of oil spilled, there were expectations of dramatic impacts on many taxa (e.g., Piatt et al. 1990, Fry 1993); therefore, the expected effect size would be large. Moreover, the degree of natural variability in marine bird abundance in these environments may be great (Zwiefelhofer and Forsell 1989). Therefore, we computed power for a hypothetical change of either a 100% increase or a 50% decrease in abundance (i.e., a two-fold increase or decrease).

Power was calculated by the procedures described in Winer et al. (1991). Sample standard deviations were used in all before–after analyses, and pooled variance estimates were used in the pre/post pairs analyses. Power was computed with $\alpha = 0.20$. Results were classified as high power (≥ 90% chance of detecting a two-fold change), moderate power (60–89% chance), and low power (< 60% chance).

RESULTS

OVERALL CHANGES IN ABUNDANCE

General patterns. Twelve taxa occurred in sufficient abundance in 1984/1985 for us to evaluate changes in overall abundance between pre- and post-spill years. During a single survey of the 10 study bays in 1984/1985, Irons et al. (1988) recorded 3,788 individuals of these 12 taxa, with numbers for individual taxa ranging from 920 for Black-legged Kittiwakes to 19 for Red-necked Grebes (Table 1). During our surveys of these same bays, we recorded averages of 3,756, 5,175, and 3,592 individuals of these 12 taxa in 1989, 1990, and 1991, respectively. Because we surveyed approximately 11% more shoreline than did Irons et al. (1988), we adjusted the abundance totals to reflect that difference. The adjusted totals indicate that numbers of birds were lower in 1989 and 1991 (−12 and −16%, respectively) and were higher in 1990 (+22%) than in 1984/1985.

Percent changes in overall abundance in the 10 study bays between 1984/1985 and the 3 post-spill years for individual taxa ranged from a 57% decrease by Pigeon Guillemots in 1990 to a 248% increase by Glaucous-winged Gulls in that same year (Table 1). In 1989, 8 of 12 (67%) taxa had decreased in overall abundance from the pre-spill baseline, whereas 4 (33%) had increased. In 1990, 7 of these 12 taxa (58%) had decreased in overall abundance from the pre-spill baseline and 5 (42%) had increased. By 1991, six (50%) taxa had decreased and six (50%) had increased in abundance from the pre-spill baseline.

Before–after comparisons. In the 1984/1985 vs. 1989–1991 comparisons, 6 (50%) of the 12

TABLE 2. Percent differences, results of t-tests, and overall responses of birds to oiling based on relative changes in densities during mid-summer in oiled ($n = 6$) and unoiled ($n = 4$) bays after (1989–1991) the *Exxon Valdez* oil spill in Prince William Sound, Alaska. A negative (positive) percent difference indicates that densities in oiled bays were lower (higher) than in unoiled bays during a particular post-spill year. Percent difference was calculated from log-transformed data. Significance codes for t-tests: * = $P \leq 0.20$, ** = $P \leq 0.10$, *** = $P \leq 0.05$.

Species/taxon	Percent difference			Overall response
	1989	1990	1991	
Red-necked Grebe	−7.6	−7.6	−8.4	None
Pelagic Cormorant	na[a]	−26.7*	na	Negative
Harlequin Duck	13.9	4.6	1.7	None
Common Merganser	−49.2	36.8	−7.4	None
Bald Eagle	−3.9	−8.8	27.2	None
Black Oystercatcher	−27.5*	−13.8	−11.5	Negative
Mew Gull	21.2	122.8	63.9	None
Glaucous-winged Gull	52.8	13.1	193.4*	None
Black-legged Kittiwake	215.0*	305.7*	132.2	Positive
Pigeon Guillemot	−51.6**	−51.2*	−21.8	Negative
"Total murrelets"	−47.9	−30.1	11.7	None

[a] na = no analysis.

taxa examined showed no significant changes in abundance during any of the 3 post-spill years (Table 1). During all 3 post-spill years, Red-necked Grebes, Pelagic Cormorants, and Pigeon Guillemots decreased significantly, whereas Glaucous-winged Gulls increased significantly. Two additional taxa (Bald Eagle and "total murrelets") did not differ significantly in 1989 but did in 1990 and/or 1991 (Table 1). We used "total murrelets" for these comparisons because Irons et al. (1988) recorded 44% of the murrelets in our 10 study bays as "unidentified murrelets." Our observations during 1989–1991 were of > 99% Marbled Murrelets and < 1% Kittlitz's Murrelets. Bald Eagles were classified as exhibiting an overall increase because they were significantly more abundant in both 1990 and 1991.

OILING EFFECTS

General patterns. Differences in densities between oiled and unoiled bays relative to the pre-spill baseline ranged from a 52% decrease by Pigeon Guillemots in 1989 to a 306% increase by Black-legged Kittiwakes in 1990 (Table 2). In 1989, 6 (60%) of 10 taxa examined had decreased in abundance in oiled bays relative to unoiled bays, whereas 4 (40%) of 10 taxa had increased (Table 2). In 1990, 6 (55%) of 11 taxa decreased in oiled bays relative to unoiled bays, whereas 5 (45%) taxa increased. In 1991, 4 (40%) of 10 taxa analyzed had decreased in oiled bays relative to unoiled bays, whereas 6 (60%) had increased.

Pre/post pairs comparisons. In 1989, 3 (30%) of 10 taxa examined decreased significantly in oiled versus unoiled bays relative to pre-spill baseline, indicating oiling effects (Table 2). Black Oystercatchers and Pigeon Guillemots exhibited negative responses to oiling, whereas Black-legged Kittiwakes exhibited a positive response. In 1990, 3 (27%) of 11 taxa examined exhibited significant oiling effects, with Pelagic Cormorants and Pigeon Guillemots responding negatively and Black-legged Kittiwakes responding positively. In 1991, only 1 (10%) of 10 taxa (Glaucous-winged Gull) exhibited a significant oiling effect, and it responded positively (Table 2). Thus, significant positive responses to oiling were detected in all 3 post-spill years, whereas negative responses were found only in 1989 and 1990. Three species responded negatively to oiling in at least 1 year, but only Pigeon Guillemots showed significant negative responses in more than 1 year.

NATURAL VARIATION AND STATISTICAL POWER

Natural variation. One of the most challenging aspects of using historical data for impact analyses is sorting out the effects of a perturbation from natural temporal variability that occurs independently within the system (Osenberg et al. 1994, Wiens, in press). Natural variability can

TABLE 3. Power analyses for before–after comparisons (H_01) of densities of birds during mid-summer 1984/1985 and 1989–1991 and for pre/post pairs comparisons (H_02) of densities during mid-summer in oiled (n = 6) and unoiled (n = 4) bays after (1989–1991) the *Exxon Valdez* oil spill in Prince William Sound, Alaska. Power was calculated for a hypothetical two-fold decrease or increase in density between years for the before–after comparison and for a hypothetical two-fold difference in density between oiled and unoiled bays for the pre/post pairs comparisons. Pre-spill (1984/1985) data are from Irons et al. (1988).

	Before–after comparisons			Pre/post pairs comparisons		
Species/taxon	1989	1990	1991	1989	1990	1991
Red-necked Grebe	1.00	1.00	1.00	1.00	1.00	1.00
Pelagic Cormorant	1.00	1.00	1.00	na[a]	1.00	na
Harlequin Duck	1.00	0.95	0.85	0.70	0.55	0.50
Common Merganser	0.85	0.85	0.95	0.30	0.45	0.55
Bald Eagle	1.00	1.00	1.00	0.70	0.85	0.80
Black Oystercatcher	1.00	1.00	1.00	0.95	0.90	1.00
Bonaparte's Gull	1.00	1.00	1.00	na	na	na
Mew Gull	0.55	0.70	0.60	0.30	0.40	0.30
Glaucous-winged Gull	0.80	0.95	0.90	0.40	0.55	0.45
Black-legged Kittiwake	0.70	0.60	0.70	0.40	0.35	0.35
Pigeon Guillemot	1.00	0.95	1.00	0.75	0.65	0.70
"Total murrelets"	0.40	0.70	0.70	0.30	0.40	0.50

[a] na = no analysis.

be measured, however, if there is an adequate time series of data prior to the perturbation.

Unfortunately, such a multi-year data set of bird abundance collected prior to the spill does not exist for PWS. Because of the importance of this issue, however, we examined the natural annual variability in abundance (densities) of seabirds around Kodiak Island (located southwest of PWS in the Gulf of Alaska), using data collected during fall and winter in 1979–1983 (Zwiefelhofer and Forsell 1989). For both fall and winter data sets, we calculated the percentage of annual change in abundance over successive years for 19 taxa for which a complete 5-year data series was available (primarily from Uyak and Uganik bays; the procedure for calculating percent change followed that described in the Methods section for before–after comparisons in PWS). The degree of annual variability differed among taxa and between seasons, but annual changes in both seasons combined exceeded a two-fold increase or decrease in overall abundance in ~21% of the comparisons (n = 152 for all taxa, seasons, and years). This threshold of a two-fold change in abundance was exceeded in at least one annual comparison by 9 (47%) of 19 taxa in fall and by 8 (42%) of 19 taxa in winter.

From these results, we conclude that natural annual variability often is high for marine birds during fall and winter in southcoastal Alaska. Consequently, one's ability to detect statistically significant effects on overall population levels from environmental perturbations is confounded by this high variability. These results also indicate that the abundance of birds in bays can fluctuate substantially between years and that an evaluation of overall changes in abundance between years is likely to produce statistically significant results as frequently as 20% of the time, even in the absence of a perturbation. Undoubtedly, there are differences between the variability measured during fall and winter on Kodiak Island and that which occurs in PWS during summer. The Kodiak data are useful nonetheless because they involve the same general suite of taxa in habitats that are fairly similar to those in PWS.

Statistical power. Power analyses for the before–after analyses indicated that our ability to detect significant changes in densities between 1984/1985 and the 3 post-spill years was moderate (60–89%) to high (≥ 90%) for most taxa: mean power for the 36 comparisons was 88% (Table 3). Overall, power was high for 22 of the 36 comparisons, moderate for 12, and low (< 60%) only for 2 comparisons (Mew Gulls and total murrelets in 1984/1985 vs. 1989). Mew Gulls showed nonsignificant declines in 2 of 3 post-spill years, but there was no consistent pattern from year to year that would suggest an impact. Total murrelets increased significantly in 1991 but had nonsignificant increases (with low power) in 1989 and 1990.

TABLE 4. Responses of marine birds to oiling based on pre/post pairs comparisons, power analyses, and general patterns of abundance in 10 bays in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill, mid-summer, 1989–1991.

Response to oiling	Taxon
No significant response and moderate–high power	Red-necked Grebe Harlequin Duck Bald Eagle
No significant response, but low power and negative trends[a]	Common Merganser "total murrelets"
No significant response, but low power and positive trends[b]	Mew Gull Glaucous-winged Gull
Significant negative response	Pelagic Cormorant Black Oystercatcher Pigeon Guillemot
Significant positive response	Black-legged Kittiwake

[a] Negative trend defined as > −10% relative difference between oiled and unoiled bays (see Table 2).
[b] Positive trend defined as > 10% relative difference between oiled and unoiled bays (see Table 2).

Power analyses for the pre/post pairs comparisons indicated that our ability to detect significant changes in densities between oiled and unoiled bays was low to moderate for most taxa: mean power for 31 comparisons was 61% (Table 3). Power was high for 7 (23%) of 31 comparisons, moderate for 7 (23%) comparisons, and low for 17 (55%) comparisons. Power was low in at least 1 year for 6 (55%) of the 11 taxa analyzed. Four of these six taxa showed positive responses to oiling, whereas two (Common Merganser and total murrelets) appeared to have had at least initial negative responses to oiling.

CLASSIFICATION OF OILING RESPONSES

We classified the responses of each taxon to oiling based on (in decreasing order of importance) the results of the pre/post pairs analyses (including the temporal pattern of significant results), the power analyses, and the general patterns of abundance in oiled and unoiled bays. Based on these criteria, we concluded that the taxa we examined fell into five fairly distinct response categories that ranged from no significant response and moderate–high power to significant negative response (Table 4).

For 7 (64%) of 11 taxa, no significant overall effects could be attributed to oiling. For 4 of these 7 taxa, however, statistical power was low and there was > 10% relative difference between oiled and unoiled bays (Tables 2 and 4). Thus, only Red-necked Grebes, Harlequin Ducks, and Bald Eagles were categorized as "no significant response and moderate–high power." Common Mergansers, Mew Gulls, and total murrelets also showed no significant response to oiling, but power was low and relative densities in oiled bays were > 10% different from those in unoiled bays. Densities in oiled bays were higher than in the unoiled bays relative to the pre-spill baseline for Mew Gulls, so this species was classified as "no significant response but low power and positive trends." Glaucous-winged Gulls also were placed in this response category because they had low power in all 3 years and nonsignificant positive trends in 1989 and 1990. Although a significant positive response was detected in 1991, the available evidence did not justify classifying this species as positively impacted. Black-legged Kittiwakes, on the other hand, did show significant positive responses to oiling in 1989 and 1990 and were the only species classified as "significant positive response" (Table 4).

Densities of Common Mergansers and total murrelets were lower in oiled than in unoiled bays relative to the pre-spill baseline; thus, these two taxa were classified as "no significant response but low power and negative trends" (Table 4). Pelagic Cormorants, Black Oystercatchers, and Pigeon Guillemots responded negatively to oiling during at least 1 of 3 post-spill years and were classified as "significant negative response." Pelagic Cormorants showed a negative response only in 1990 but were not present in enough bays in 1989 and 1991 to conduct analyses. Consequently, there were insufficient data to evaluate whether recovery was occurring. Black Oystercatchers and Pigeon Guillemots were negatively impacted in 1989 (and 1990 for Pigeon Guillemots) but showed no significant negative response in 1991. We suspect, therefore, that recovery was underway by that year, although the relative difference between oiled and unoiled bays for both of these species still was $> 10\%$ in 1991 (Table 2).

It is noteworthy that all seven pre/post pairs comparisons that were significant had P values > 0.075 and ≤ 0.20 (Table 2). Thus, with the possible exception of the 1989 pre/post pairs comparison for Pigeon Guillemots ($P = 0.075$), if we had partitioned α in any way or used more restrictive decision-making criteria (e.g., $\alpha = 0.05$), none of the pre/post pairs comparisons would have been significant. Our use of a large α, however, represents an emerging trend in impact assessments that considers Type II errors (i.e., failing to detect an impact that did occur) to be equally or more problematic than Type I errors (i.e., falsely identifying an impact that did not occur) (Eberhardt and Thomas 1991, Shrader-Frechette and McCoy 1993, Mapstone 1995).

DISCUSSION

Oil spills at sea can affect marine birds along three primary pathways, namely, by altering population size and structure, reproduction, and habitat occupancy and use (Wiens 1995). The analyses presented here addressed effects of the oil spill on overall bird abundance and distribution, and therefore provide insight into effects on the first and third of these pathways. The before–after analyses (H_01) indicated that half of the 12 taxa that we examined exhibited no significant changes in overall abundance during mid-summer between 1984/1985 and 1989–1991. Fewer taxa showed negative trends in abundance compared to the pre-spill baseline during each successive year after the spill (8 taxa declined in 1989, 7 in 1990, and 6 in 1991), suggesting that declines in abundance were most pronounced in the year of the spill and that recovery was occurring in 1990 and 1991. Overall, we documented few dramatic declines or increases in the abundance of most bird species during the three summers after the *Exxon Valdez* oil spill. In addition, power analyses indicated that we should have good confidence in these results.

Species that were significantly less abundant during all 3 post-spill years than they were prior to the spill included Red-necked Grebes, Pelagic Cormorants, and Pigeon Guillemots. The decreases in abundances for Red-necked Grebes and Pelagic Cormorants must be placed in a seasonal context, however. Red-necked Grebes are present in PWS primarily during winter and are uncommon visitors during summer (Isleib and Kessel 1973). Indeed, Irons et al. (1988) saw only 19 Red-necked Grebes in their pre-spill surveys of the 10 study bays (Table 1), and 12 of those grebes occurred in 1 bay. Although we recorded only one Red-necked Grebe during our 1989–1991 mid-summer surveys, we question the biological significance of this apparent decline because this species generally is uncommon in PWS during summer and our statistical evaluations of annual changes are based on scant data. Pelagic Cormorants also are much more common in PWS during winter (Isleib and Kessel 1973), and Irons et al. (1988) counted only 28 in the 10 study bays during summer 1984/1985. Pigeon Guillemots, on the other hand, are more common in PWS during summer (Isleib and Kessel 1973), so both the pre-spill and post-spill data sets were adequate to support the conclusion of significant and persistent overall declines in abundance during each of the 3 post-spill years. Pigeon Guillemots, however, were thought to be declining in PWS prior to the spill (Oakley and Kuletz 1993). The fact that the magnitude of the difference in Pigeon Guillemot abundance decreased in each successive post-spill year, however, suggests that there was an initial decline in overall abundance immediately after the spill but that numbers of birds were increasing 1–2 years later.

In the pre/post pairs analysis (H_02), the confounding effects of natural variation were reduced because the historical data were used to estimate the differences between oiled and unoiled bays that would exist in the absence of the perturbation (Stewart-Oaten et al. 1992). Our final classification of overall impacts for the 11 taxa examined indicated that 7 taxa showed no significant response to oiling, 3 (Pelagic Cormorant, Black Oystercatcher and Pigeon Guillemot) showed significant negative responses to oiling, and 1 (Black-legged Kittiwake) had a significant positive response to oiling. Statistical power was a greater consideration in these analyses than in the before–after comparisons, because there were fewer degrees of freedom after partitioning the data into oiled and unoiled categories. The consequence of low power would be to increase the likelihood of failing to detect significant differences (Type II errors). Four of the 7 taxa that showed no significant response to oiling had low power or trends that led us to reconsider those statistical results (Table 3). Of these questionable taxa, Common Mergansers and total murrelets exhibited negative trends in 1989, suggesting that these taxa initially responded negatively to oiling. Thus, an inclusive accounting of the taxa that were affected negatively by the oil spill includes Pelagic Cormorants, Common Mergansers, Black Oystercatchers, Pigeon Guillemots, and total murrelets (Table 4). Most of these impacts were evident only in 1989, however, and signs of recovery were evident for all taxa for which we had data in subsequent years. Pigeon Guillemots showed the most persistent negative effects, with significant decreases in oiled bays in 1989 and 1990 and a nonsignificant but substantial decrease (22%) in 1991. Considering that the decreases in 1989 and 1990 were significant and > 50%, the 1991 results suggest that recovery was beginning.

Black-legged Kittiwakes, Mew Gulls, and Glaucous-winged Gulls appeared to be attracted to spill cleanup activities. Their apparent positive response to oiling probably was a secondary consequence of the spill, as some of the most heavily oiled bays provided anchorage for the hundreds of vessels and their crews that participated in the shoreline cleanup.

The results of the pre/post pairs analyses (H_02) can be compared directly with the results of two other analyses that addressed the same general question of oiling impacts to birds in PWS. Klosiewski and Laing (1994) conducted nearshore surveys in PWS during summer 1989-1991 in a subset of the areas that had been surveyed by Irons et al. (1988) in 1984/1985. Although their study had only partial geographic overlap with our study area, their survey and analytical techniques were similar to ours and also can be described as pre/post pairs analyses (Wiens and Parker 1995). Based on comparisons with Irons et al. (1988), Klosiewski and Laing (1994) found that six taxa ("loons," Harlequin Duck, "scoters," Black Oystercatcher, Mew Gull, and Arctic Tern) exhibited negative oiling impacts (Table 5). The number of impacted taxa ranged from zero in 1989 to five in 1991; however, if the same two-tailed α that was used in this study ($\alpha = 0.20$) is applied to their results, those numbers change to one taxon (Black Oystercatcher) in 1989 and six taxa (including murres) in 1991.

Day et al. (1995, in press) also used the same post-spill data set that we used but did not use historical data. Instead, Day et al. assessed oiling impacts with two analytical techniques described by Wiens and Parker (1995) as "single-time gradient analysis" and "multiple-time trend-by-time analysis." Although they assessed impacts and recovery based on six sampling periods spaced throughout the entire year, only analyses of nearshore survey data from mid-summer sampling periods in 1989–1991 were directly comparable to our analyses. During mid-summer, Day et al. recorded negative oiling impacts for five species: Pelagic Cormorants, Harlequin Ducks, Common Mergansers, Black Oystercatchers, and Mew Gulls (Table 5).

Thus, in addition to the three significant and two probable taxa that we identified as negatively impacted during mid-summer, four other taxa ("loons," Harlequin Ducks, and "scoters" [Klosiewski and Laing 1994]; Harlequin Ducks and Mew Gulls [Day et al. 1995, in press]) were identified as negatively impacted by the two other studies evaluating spill effects on birds (Table 5). When these studies are viewed collectively, they suggest that up to 10 (38%) of the 26 taxa studied were negatively impacted by the spill in summer. By mid-summer 1991, however, there was evidence of recovery from at least one study for all of these taxa except "loons" and "scoters." However, neither of these taxa was abundant in our study bays during summer either before or after the spill and neither was evaluated in this study or in mid-summer by Day et al. (Table 5).

Based on year-round sampling, Day et al. (1995) found evidence of initial impacts to 19 (54%) of the 35 species evaluated in 1989, whereas the percentage of impacted species declined to 40% in 1990 and 10% in 1991. In comparison, this study showed that 20–40% of the species were impacted in mid-summer 1989 and that no species showed significant negative impacts by 1991, although several species still showed some (nonsignificant) evidence of persistent effects. Thus, the overall results of these two studies were quite similar (Table 5).

Although there was broad concurrence among all three studies in the classification of taxa that were not impacted and in the percentage of taxa that were impacted, there was a lack of concurrence in the classification of negatively impacted taxa. This lack of concurrence may be related to how the three studies classified oiled areas in

TABLE 5. Comparison of the results of three studies that assessed impacts of the *Exxon Valdez* oil spill on marine-oriented birds during mid-summer in Prince William Sound, Alaska, 1989–1991. Negatively impacted taxa that showed evidence of recovery are depicted in bold-faced type.

	Historical comparisons		
Species/taxon	This study	USFWS study[a]	Habitat-use study[b]
No Evidence of Impact			
Red-necked Grebe	No impact	na[c]	na
"Goldeneyes"	na	No impact	na
Bald Eagle	No impact	No impact	No impact
Wandering Tattler	na	na	No impact
Spotted Sandpiper	na	na	No impact
Red-necked Phalarope	na	No impact	No impact
"Shorebirds"	na	No impact	na
Glaucous-winged Gull	No impact?[d]	No impact	No impact
"Murres"	na	No impact	na
"Total murrelets"[e]	No impact?	No impact	No impact
Horned Puffin	na	No impact	na
Belted Kingfisher	na	na	No impact
Steller's Jay	na	na	No impact
Black-billed Magpie	na	na	No impact
Northwestern Crow	na	na	No impact
Common Raven	na	na	No impact
Evidence of Impact			
"Loons"	na	Negative impact	na
Pelagic Cormorant[f]	Negative impact	No impact	**Negative impact**
Harlequin Duck	No impact	Negative impact	**Negative impact**
"Scoters"	na	Negative impact	na
Common Merganser[g]	No impact?	No impact	**Negative impact**
Black Oystercatcher	**Negative impact**	Negative impact	**Negative impact**
Mew Gull	No impact?	Negative impact	Negative impact
Black-legged Kittiwake	Positive impact	No impact	Positive impact
Arctic Tern	na	Negative impact	No impact
Pigeon Guillemot	**Negative impact**	No impact	Positive impact

[a] U.S. Fish and Wildlife Service comparisons with 1984/1985 baseline (Klosiewski and Laing 1994).
[b] Day et al. (1995, in press); includes nearshore data from mid-summer (July/August) cruises in 1989–1991.
[c] na = no analysis.
[d] ? = taxon that had nonsignificant differences but low power (see Table 3).
[e] Analyzed as Marbled Murrelets by Day et al. (1995, in press).
[f] Analyzed as "cormorants" by Klosiewski and Laing (1994).
[g] Analyzed as "mergansers" by Klosiewski and Laing (1994).

Prince William Sound and how those classifications affected data analyses. Day et al. (1995) used an oiling index based on the amount of oil that initially contaminated the shorelines of the study bays to describe the relative amount of oil in each bay as a quantitative, continuous variable. This strategy was highly accurate and facilitated the use of regression statistics in hypothesis testing, which were more powerful for detecting impacts than were the procedures used in both of the other studies. In this study, we used the same oiling index as Day et al., but then categorized each bay as oiled or unoiled (Fig. 2) to accommodate the BACI-like analytical design. Klosiewski and Laing (1994) also categorized areas as oiled and unoiled for use in a BACI-like analysis, but they considered any shoreline that was in the general path of the spill to be oiled, whether or not field surveys showed that it actually was oiled. Although the analytical procedures used in this study and by Klosiewski and Laing were similar, differences in impact classifications for individual taxa probably were related to differences between the studies in the classification of shoreline oiling. The distribution of oil on shorelines in the general path of the spill was highly variable (to the extent that some entire bays were unoiled), and a quantitatively based measure such as the oiling index we used would seem to provide a more accurate assay of oiling conditions. Indeed, Baker et al. (1990) considered this level of resolution to be critical, stating that "consideration of the overall, long-term impact of a particular spill must take into account the relative proportions of oiled and unoiled habitats in the area of interest."

CONCLUSIONS

Our evaluation of pre- and post-spill abundances of birds in 10 bays in Prince William Sound indicated that 3 of 11 taxa evaluated had declined significantly from numbers recorded 4–5 years before the spill. Although this analysis could not factor out natural sources of variation, there was no evidence of large-scale, post-spill changes in overall abundance of most taxa. Our inability to reject the null hypothesis of no large-scale, post-spill changes in overall abundance of most taxa in our 10 study bays (even using a broad statistical definition of impact) suggests that most bird populations in PWS during mid-summer were not impacted catastrophically. The Pigeon Guillemot was the one species that was both common during summer in PWS and showed persistent declines in overall abundance relative to the pre-spill baseline.

Our analysis of oiling impacts indicated that the effects of the spill on bird abundance in oiled bays during mid-summer were most pronounced in 1989 and that, by 1991, most marine bird species in PWS were using oiled bays at levels that were indistinguishable from levels of use in unoiled bays. Once again, Pigeon Guillemots showed the greatest negative impacts and the fewest signs of recovery.

The conclusions from both analyses are strengthened by the fact that we sampled many of the most heavily oiled bays in the entire spill area, thereby increasing our chances of detecting impacts if they were present. These conclusions also are supported by Day et al. (1995, in press), who evaluated effects of the spill on habitat use, and by Wiens et al. (1996), who evaluated effects on bird communities. The analyses supporting those papers were based upon the same post-spill data used in this paper, although their analyses did not incorporate comparisons with pre-spill data. Regardless, both Day et al. and Wiens et al. found that impacts were substantial and most apparent in 1989, but that recovery was well underway by 1991. Our conclusions also are in agreement with assessments of impacts and recovery by researchers who have studied other large marine oil spills (Clark 1982, Baker et al. 1990, Mielke 1990, Ritchie and O'Sullivan 1994). For example, Clark (1982) noted that most communities of organisms in temperate waters will recover from massive oil spills within 2 years, with some effects detectable up to 10 years after the spill.

Our research indicates that, despite substantial mortality in the aftermath of the spill and significant impacts to the distribution of up to 40% of the taxa in PWS during the first summer following the spill, the abundance of birds during mid-summer generally was not significantly affected and redistribution of birds back into heavily oiled bays was occurring for nearly all taxa by the second summer after the spill. The redistribution of birds into the heavily oiled bays indicates that these habitats are at least in the process of recovering and that the birds themselves find the habitat suitable for occupancy. Because habitat recovery is a prerequisite for recovery of population or reproductive measures (Morrison 1986, Wiens 1995), the prognosis for recovery in these other dimensions appears to be good.

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APPENDIX 1. Scientific names of bird species mentioned in text.

Common name	Scientific name
Red-necked Grebe	*Podiceps grisegena*
Pelagic Cormorant	*Phalacrocorax pelagicus*
Harlequin Duck	*Histrionicus histrionicus*
Common Merganser	*Mergus merganser*
Bald Eagle	*Haliaeetus leucocephalus*
Black Oystercatcher	*Haematopus bachmani*
Wandering Tattler	*Heteroscelus incanus*
Spotted Sandpiper	*Actitis macularia*
Red-necked Phalarope	*Phalaropus lobatus*
Mew Gull	*Larus canus*
Glaucous-winged Gull	*Larus glaucescens*
Black-legged Kittiwake	*Rissa tridactyla*
Arctic Tern	*Sterna paradisaea*
Pigeon Guillemot	*Cepphus columba*
Marbled Murrelet	*Brachyramphus marmoratus*
Kittlitz's Murrelet	*Brachyramphus brevirostris*
Horned Puffin	*Fratercula corniculata*
Belted Kingfisher	*Ceryle alcyon*
Steller's Jay	*Cyanocitta stelleri*
Black-billed Magpie	*Pica pica*
Northwestern Crow	*Corvus caurinus*
Common Raven	*Corvus corax*