When the supertanker *Exxon Valdez* ran aground on Bligh Reef in Prince William Sound, Alaska, on the morning of 24 March 1989, it aroused widespread concern about possible environmental devastation. Within hours, some 41 million liters of crude oil were released into the marine ecosystem, making this spill the largest in US history. Eventually, oil was found more than 900 km from the spill site (Figure 1), and roughly 2100 km of shoreline were contaminated with oil (Neff et al. 1995).

Reports of mortality of marine birds were immediate, and images of oiled seabirds figured prominently in media coverage of the spill. Within a few months, more than 30,000 oiled carcasses had been retrieved from the water and beaches in the spill area, and estimates of overall mortality were substantially greater (Piatt et al. 1990). The magnitude of these mortality estimates led some scientists to declare that some breeding colonies had suffered major losses, that the breeding activity of some species was disrupted or failed entirely, that the intertidal habitats on which many species depend were severely impacted, and that the recovery of the local seabird populations might take decades to as much as a century, if a recovery was even possible (*Exxon Valdez* Oil Spill Trustee Council 1993, Fry 1993, Heinemann 1993, Nysewander et al. 1993a, Piatt et al. 1990).

Studies of the effects of the *Exxon Valdez* oil spill on seabirds were initiated shortly following the spill by researchers working for the State of Alaska and several federal agencies (the "Trustees") or supported by Exxon. By mid-summer 1989, however, litigation became a priority, and as a result the two groups conducted their studies separately, each with little knowledge of what the other group was doing. Reports of many of these studies have now been made public, and enough information is available to develop a general understanding of the effects of the oil spill on seabirds.

Figure 1. Map of the northern Gulf of Alaska and Prince William Sound, Alaska, showing locations mentioned in the text and the extent of the *Exxon Valdez* oil spill (in gray). Modified from *Exxon Valdez* Oil Spill Trustees (1992).

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assessment of how the spill affected seabirds and whether the initial concerns were justified. The studies also provide some insights into how the scientific process itself may be affected by such well-publicized environmental accidents and into the relationships among preconceptions, advocacy, and science. In this article, I discuss these findings and consequences, emphasizing the work in which I was directly involved. This work was funded by Exxon but was conducted and interpreted independently.

Seabirds and oil

Seabirds spend much of their lives at the air–water or land–water interface, where floating oil accumulates. This habit, combined with other features of the behavior, ecology, and life history of seabirds, suggests that they should be particularly vulnerable to oil spills. However, spills differ in the type of oil spilled, currents and weather conditions at the time of the spill, and the activities, abundances, and distributions of seabirds in the spill area. Consequently, there is little relationship between measures such as spill volume and seabird mortality (Burger 1993).

Oil spills may affect seabird populations in several different ways. Losses of individuals, through direct or indirect mortality or emigration from the spill area, may alter population size and structure. The reproductive performance of those birds that do remain may be altered. Finally, changes in the condition of the habitat may lead individuals to move elsewhere, influencing habitat occupancy and use.

Effects on population size and structure

Murre. Some 74% of the oiled carcasses retrieved following the Exxon Valdez spill were murres (Uria spp.; Piatt et al. 1990). Carcasses retrieved on beaches represent only a fraction of the mortality caused by a spill, however, and the magnitude of this apparent mortality indicated that impacts on breeding populations might be severe. Counts of birds in attendance at colonies (i.e., occupying nesting ledges) or flying or on the water near colonies had been conducted at most murre colonies in the Gulf of Alaska during the 1970s. These counts were intended only to document colony locations and provide coarse estimates of their sizes, but they were used by several investigators to make quantitative assessments of population changes after the Exxon Valdez spill. For example, comparisons of counts at murre colonies in the spill path by the US Fish and Wildlife Service (USFWS; Dragoo et al. 1993, Nysewander et al. 1993b) indicated that breeding numbers at several colonies were significantly reduced after the spill relative to counts in the 1970s (Figure 2). Noting the absence of concurrent declines in breeding populations of colonies at Middleton Island and the Semidi Islands, which are outside of the spill area (Figure 1), Nysewander et al. (1993a) concluded that the reductions within the spill area were caused by the Exxon Valdez spill. However, Middleton Island and the Semidis are exposed to different oceanographic conditions than are colonies in the spill path, and therefore their suitability as reference colonies is questionable (Piatt and Anderson in press, Wiens 1995).

Not all postspill surveys recorded population decreases, however. Two years after the spill, Erikson (1995) surveyed 32 of the 36 murre colonies in the spill area (four small colonies were not surveyed). At an unnamed colony in the Chiswell Island group, for example, the two estimates from surveys conducted 3 years and 13 years before the spill differed considerably (Figure 3). Postspill counts conducted by USFWS in 1989–1991 generally exceeded the prespill counts, although there was much day-to-day variation in estimated colony attendance. Erikson’s single postspill survey recorded fewer birds than were seen in the USFWS surveys, but the estimate was well within the range of the prespill counts.

These observations illustrate some of the limitations of such prespill versus postspill comparisons. Historical (prespill) surveys are usually few and old. If more than one historical data set is available, it is not always clear which set should form the baseline for comparison. The substantial daily variation among counts also complicates comparisons, because often only single counts are compared. These complications led Erikson (1995) to group colonies for analysis based on their level of oiling risk (a function of proximity to the spill path and the state of the oil when it passed through an area). If murres were assembling near their breeding colonies at the time of the

Figure 2. Estimated changes in abundances of common murre (Uria aalge) at breeding colonies in the Gulf of Alaska from surveys conducted during the 1970s and 1980s, before the Exxon Valdez oil spill, to postspill surveys. Colonies in the spill path are indicated by solid bars, those outside of the spill area by hatched bars. The colony sequence is from Prince William Sound (left) to the tip of the Alaska Peninsula (right). After Piatt and Anderson (in press).

Figure 3. Murre colony attendance estimates at an unnamed island in the Chiswell Island group made before the Exxon Valdez oil spill compared with postspill estimates made by the US Fish and Wildlife Service (USFWS) and by D. Erikson. The 1990 and 1991 USFWS data are from surveys conducted on separate days over eight-day and six-day periods, respectively. The Chiswell Islands were close to the main spill trajectory (Figure 1) and were classified by Erikson in a high oiling risk category. After Erikson (1995).
spill, as has been suggested (Piatt et al. 1990), those colonies with greater oiling risk should have suffered greater losses in comparison with prespill levels. Such was not the case: postspill colony attendance levels were similar to those recorded before the spill, and prespill versus postspill changes in colony attendance did not differ significantly among oiling risk categories.

The largest murre colonies in the spill area are on the Barren Islands (Figure 1). Oil remained in this area for several weeks, and seabird mortality was considered great (Piatt et al. 1990). Fry (1993) suggested that 60%-80% of the breeding adults “were engulfed, carried away and killed by the oil.” However, the results of Boersma and her colleagues, who had studied seabirds on East Amatuli Island (one of the Barren Islands) during the late 1970s and then again after the spill, raise doubts about Fry’s suggestions. Counts made more than a decade before the oil spill indicated that somewhere between 19,000 and 61,000 birds (with perhaps a “best guess” of 25,000 birds) were breeding on East Amatuli. Counts in 1990-1992 recorded roughly 31,000-35,000 birds in attendance (Boersma et al. 1995). The wide variation in prespill counts, which was due to differences in methods and observers as well as to real variations in murre abundance, makes it difficult to determine whether the population actually changed at all after the spill.

People working with Boersma on East Amatuli between 1976 and 1981 had also taken photographs of breeding murres in various locations. Boersma and colleagues assembled seven of these historical photographs that could be matched with photographs taken in 1990 or 1991. Comparisons of paired photographs (e.g., Figure 4) indicated that the number of birds was the same or greater in the 1990s than it was in the 1970s and 1980s.

These results from East Amatuli failed to reveal the sort of drastic population declines that were reported by USFWS (Dragoo et al. 1993, Nysewander et al. 1993b) or were expected from carcass-based mortality estimates. The differences among studies may stem from several factors. It has already been noted that many of the historical estimates were based on single surveys of colonies, which are subject to variations associated with weather, methods, observers, time of day, and season (Boersma et al. 1995). Some comparisons used historical data in which prespill abundances were erroneously overestimated (e.g., one count was inadvertently adjusted twice to account for breeding birds that were at sea when the count was made; Erikson 1995). Such errors would increase the likelihood of documenting large postspill reductions. Nevertheless, mortality of the magnitude projected for the Exxon Valdez spill should overwhelm many of these complications. How can the mortality estimates derived from carcass counts (e.g., Ecological Consulting, Inc. 1991, Piatt et al. 1990) be reconciled with the observations of Boersma et al. (1995) and Erikson (1995), particularly on the Barren Islands?

One possibility is that the model projections of mortality based on carcass counts (e.g., Ecological Consulting, Inc. 1991) were simply too high. In addition, many murre colonies have a pool of nonbreeding individuals that remain at sea in the vicinity of the colony (Birkhead and Hudson 1977). Individuals from this pool may have filled the vacancies created by the deaths of breeding adults, or the mortality might have affected the nonbreeding pool rather than breeding adults. Because the sizes of such at-sea pools cannot be estimated satisfactorily, changes in such pools would be difficult to detect. Another possibility is that the birds killed in a particular location were not necessarily destined to breed in nearby colonies. The ocean around the Barren Islands, for example, is especially productive and is a regional hot-spot for seabirds. Perhaps birds from colonies outside of the spill area were present when the oil passed through. If so, mortality would be spread among colonies over a larger region, and it would be difficult to document spill-associated changes at particular colonies.

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Figure 4. Photographs of the same murre nesting ledges on East Amatuli Island of the Barren Islands taken in 1976–1977, before the Exxon Valdez oil spill (left; photo: M. Amaral), and in 1991, after the spill (right; photo: A. Kettle). Photographs courtesy of P. D. Boersma.
Long-term, region-wide population trends may also make it difficult to detect specific effects of the Exxon Valdez spill. For example, from comparisons of prespill and postspill populations, Piatt and Anderson (in press) concluded that substantial declines (greater than 20% reduction) occurred from the 1970s to 1990s at 13 of 16 major murre colonies in the northern Gulf of Alaska. Eight of these 13 colonies were outside of the spill path (Figure 2). During this time period there were major changes in the flow patterns of the Alaska Coastal Current and sea temperatures in the Gulf of Alaska, perhaps triggered by the 1976 El Niño event (Royer 1993). These oceanographic changes were accompanied by shifts in fish populations and in seabird diets (Piatt and Anderson in press). Thus, seabird populations may have been under food stress and already declining before the oil spill occurred. Unfortunately, the quantity and quality of prespill colony data do not enable one to distinguish between effects of the Exxon Valdez spill and natural responses to changes in the marine environment (Boersma et al. 1995, Piatt and Anderson in press). The difficulty of detecting the effects of an environmental accident such as an oil spill against a background of substantial natural variation due to environmental changes continues to complicate the analysis of environmental impacts of all sorts.

Other birds. Studies of other marine birds in the spill area were less intensive than those of murres. Counter to popular perception, however, these studies also suggest that long-term effects on population levels were either minimal or could not be linked with certainty to the Exxon Valdez spill. For example, the number of breeding black oystercatchers (Haematopus bachmani; Figure 5) increased from 1989 through 1991, suggesting a recovery from spill effects, although Andres (1993) concluded that direct, lethal effects of the spill had probably been minimal. Both pigeon guillemots (Cepphus columba; Figure 6) and marbled murrelets (Brachyramphus marmoratus) were less abundant after the spill than in the late 1970s, but populations of both species had declined in Prince William Sound during the 1980s, so these changes could not be attributed solely to the oil spill (Kuletz 1993, Oakley and Kuletz 1993). Oakley and Kuletz (1993) indicated that postspill reductions of pigeon guillemots were greater in oiled than in unoiled areas, however, which suggested a spill effect.

Two additional studies evaluated changes for large numbers of species by comparing surveys along shorelines in Prince William Sound conducted in the mid-1980s with matched surveys in 1989–1991. Although one might expect numbers to change over this time period due to natural variability, a decline in oiled areas relative to unoiled areas could indicate spill effects. Klosiewski and Laing (1994) found that between 1984–1985 and 1989–1991, populations of loons (Gavia spp.), harlequin ducks (Histrionicus histrionicus), scoters (Melanitta spp.), black oystercatchers, mew gulls (Larus canus), and arctic terns (Sterna paradisaea) decreased at sites within the overall spill zone (oiled areas) more than at sites outside the spill zone, whereas 14 other species showed no evidence of spill-related changes.

The distribution of oil in the spill area was patchy, however, and the birds in the overall spill zone were therefore exposed to different degrees of oiling. Using a subset of the same 1984–1985 data that Klosiewski and Laing (1994) used in their comparisons, Murphy...
et al. compared individual bays that were either heavily or moderately oiled with bays that were lightly oiled or unoiled. These workers found that declines were significantly greater in heavily or moderately oiled bays than in lightly oiled or unoiled bays for pelagic cormorants (Phalacrocorax pelagicus), black oystercatchers, and pigeon guillemots. Another 11 species showed no statistically significant evidence of negative spill impacts. Both oystercatchers and guillemots showed clear evidence of recovery by 1991, whereas data were insufficient to assess recovery for cormorants. In both the Klosiewski and Laing (1994) and the Murphy et al. studies, most of the species analyzed showed no systematic reduction in abundance in oiled (however defined) versus unoiled areas, and when recovery was assessed, most affected species did not show persistent effects.

In these comparisons, one must assume that changes over time are equivalent among areas and that environmental factors other than oiling do not differ systematically between the oiled and unoiled samples. This assumption is less likely to be violated if sites within a single location are compared (as Oakley and Kuletz [1993] did) or if oiling levels of sites are specified quantitatively (as in Murphy et al.) than if the comparisons are based on broad regions that are likely to differ in a host of environmental features (as in Klosiewski and Laing 1994). Thus, the greater number of apparent spill impacts demonstrated in Klosiewski and Laing’s study in comparison with that of Murphy et al. may be due at least in part to the confounding effects of broad-scale variation in other environmental factors.

Effects on reproductive performance

Murres. Because of the magnitude of their estimated losses from breeding colonies, investigations of reproduction also concentrated on murres. Initial studies conducted by USFWS suggested that there was total reproductive failure at some colonies in the spill path in 1989, that the onset of egg laying (breeding phenology) in other colonies was delayed by as much as a month, and that reproductive success remained below normal and phenology continued to be late at least through 1991 (Dragoo et al. 1993, Nysewander et al. 1993a, b). On the basis of these observations and the preliminary indications that breeding populations had declined abruptly following the oil spill, Nysewander et al. (1993a) and Fry (1993) proposed the following scenario. If densities of birds on breeding ledges were reduced, some of the missing adults might have been replaced by young, inexperienced birds breeding for the first time. These changes in population structure and social organization could lead to delayed courtship and egg laying, a reduction in breeding synchrony within colonies, and increased risk of predation on eggs or young. As a result, reproductive success could be reduced and, because of the late breeding, many of the remaining chicks could be susceptible to fall and winter storms. Fry (1993) asserted that, because of delayed breeding phenology in 1990–1992, winter storms “swept more than 100,000 young chicks off the cliffs to their deaths.” Other statements set the losses due to delayed reproduction at more than 300,000 chicks (Exxon Valdez Oil Spill Trustee Council 1993). Fry (1993) feared that, should these patterns become entrained in the populations, the breeding failures could lead to the eventual extinction of some colonies.

Most of this argument is based on speculation rather than firm evidence. Projections of chicks lost due to delayed breeding are based on assumptions, not observations, and the age of breeding individuals in these colonies is unknown. The only testable components of this scenario are whether breeding densities were in fact reduced, breeding phenology was delayed, and reproductive success was markedly lower following the spill.

The studies of Boersma et al. (1995) in the Barren Islands provide some perspective on these issues. These workers found no long-term reduction in breeding density: murres attendance levels at the East Amatuli colonies a year or more after the spill were within the range of prespill estimates, and counts of birds were similar on identifiable nesting ledges in matched prespill and postspill photographs (e.g., Figure 4).

What about breeding phenology? At one 25-m² plot on Light Rock (East Amatuli Island) where breeding had been monitored during the 1970s, phenology varied among years and was not markedly later in 1990–1992 than in some (but not all) of the prespill years. During their studies in 1991 and following years, Boersma et al. (1995) also used time-lapse photography from automated cameras on Light Rock to record breeding phenology and success. The photographic records indicated that phenology may differ considerably in different areas of the same colony. In 1991, for example, phenology in two nesting areas located in different habitats less than 20 m apart differed by more than a week.

Because phenology is so variable in time and space, it is difficult to establish what is normal for a breeding colony, much less for many colonies in a region. The initial claims of spill-related delays in breeding phenology were based especially on 1989 data, before Boersma et al. (1995) had initiated their studies. Although murre breeding phenology at the Barren Islands may have been late in 1989 relative to breeding times during the 1970s or to that at Middleton Island and the Semidis in 1989, phenology was also later in 1989 at other colonies outside of the spill area than it had been in other years (Piatt and Anderson in press). Moreover, some of the first-egg dates used by USFWS investigators were actually dates of the first visit to a colony on which eggs were observed; in those cases, eggs must have been laid some time earlier. There are also indications that nesting phenology of murres in the northern Gulf of Alaska may have been perhaps two weeks

References

S. M. Murphy, R. H. Day, J. A. Wiens, and K. R. Parker, submitted manuscript.

See footnote 2.

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*P. D. Boersma et al., 1994, personal communication. University of Washington, Seattle, WA.

**P. D. Boersma, 1993, personal communication. University of Washington, Seattle, WA.


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later during the early 1990s than in the 1970s, perhaps in response to long-term oceanographic changes (Piatt and Anderson in press). Breeding on the Barren Islands did begin somewhat earlier in 1993–1994 than in the early 1990s (Boersma et al. 1995, Piatt and Anderson in press). This shift might indicate recovery from a spill effect, but it could also express the natural variability that appears to characterize this system. In any event, none of the postspill studies has shown conclusively that any changes in breeding phenology resulted directly from the oil spill.

What, then, about reproductive success? Boersma et al.’s (1995) prespill and postspill observations from the 25-m² plot on Light Rock in the Barren Islands indicated that productivity was high in 1991 (and probably in 1978), intermediate in 1979 and 1990, and low in 1992. These observations were based on direct counts, which can disturb a breeding colony and thereby increase predation losses and artificially lower reproductive output. To reduce this effect, Boersma et al. (1995) used automated cameras on Light Rock. Cameras were installed in early July 1991, when egg laying was just beginning, and were not removed until mid-October, well after chicks had fledged. These photographic records indicated that there was a substantial loss of eggs and chicks in the previously disturbed 25-m² plot in comparison to the relatively undisturbed camera area. Based on analysis of the camera records, murres produced an average of 0.64–0.70 chicks per pair (clutch size is one) in 1991, well within the range of murre chick production documented by a variety of methods elsewhere in the species’ geographic range (Boersma et al. 1995).

In contrast to these findings, Nysewander et al. (1993b) reported nearly total reproductive failure at every colony they monitored in the spill zone in 1989 and below-normal success at Puale Bay and the Barren Islands (Nord Island) in 1990 and 1991. Based on Nysewander et al.’s survey, murres at Nord Island in 1991 produced only 0.13 chicks per pair. In 1992, however, different observers recorded reproductive success at Nord Island twice that recorded in 1991 and productivity at Puale Bay that was within the normal range. The differences in results between the various studies on the Barren Islands may be due in part to variations among colonies; Light Rock may provide better nesting habitat than Nord Island. There may also be differences between the indirect assays of breeding activity used by Nysewander et al. (1993b; e.g., observations of presumed breeding postures of birds made from ships) and the direct plot counts and camera recordings used by Boersma et al. (1995).

In combination, the observations of Boersma et al. (1995) and those reported by Piatt and Anderson (in press) do not support the expectations of massive and persistent effects on murre reproduction following the Exxon Valdez spill.

Other birds. Reproduction by other seabirds has been less intensively studied than that of murres. Oakley and Kuletz (1993) found no clear effects of the spill on pigeon guillemot reproduction. Reproductive success of black-legged kittiwakes (Rissa tridactyla) was reduced compared with prespill levels in oiled colonies in 1989, but productivity in other, unoiled colonies declined in 1990 and was low throughout the region through 1992 (Irons 1993). These reductions may have been related to changes in food supplies, which may or may not have been associated with the oil spill. Breeding activity of black oystercatchers was delayed in 1989, and hatching success and chick survival were lower in oiled than in unoiled areas (Andres 1993, Sharp and Cody 1993). Studies in subsequent years indicated subtle differences between oiled and unoiled territories in egg size and feeding rates of chicks, but fledging rates did not differ (Andres 1993).

Studies of spill effects on harlequin duck reproduction are more controversial. Patten (1993) suggested that harlequin ducks in western Prince William Sound (which included the spill area) were in poorer condition in 1989 and possibly contained higher tissue hydrocarbon levels than did birds from the eastern half of the Sound (which was unoiled). Patten reported “massive reproductive failure” of birds in the spill area and postulated that reproduction was persistently depressed because the ducks were feeding on contaminated blue mussels (Mytilus trossulus) that concentrated hydrocarbon residues from unweathered, toxic oil trapped beneath the mussel beds. He concluded that “unless measures are taken to remove oil from mussel beds, it is possible that a local extinction of Harlequin Ducks may occur within the oil spill area” (Patten 1993, p. 154). These statements have been repeated in the popular press (e.g., Chadwick 1993, Edgar 1993), but because Patten has published no details or quantitative results of his studies it is not possible to assess his conclusions scientifically.

Incidental observations made during habitat-use studies by Day, Murphy, and their colleagues, however, indicated that harlequin ducks bred successfully in 1989 and subsequent years in bays that had been heavily oiled by the spill as well as in unoiled or lightly oiled bays. Moreover, blue mussels constitute only a small part of the diet of harlequin ducks, and although oil did remain in some mussel beds in Prince William Sound for several years after the spill, these sites constituted a small percentage of the total area of mussel beds in the Sound. Mussels from such beds exhibited elevated levels of polycyclic aromatic hydrocarbons (PAH), but the estimated PAH dosage that birds feeding on these beds would receive is far below the levels known to cause even sublethal effects in other species (Boehm et al. in press). It is hard to imagine that continuing low-level oil contamination in a few mussel beds could lead to the sort of widespread, persistent, and massive reproductive failure that Patten and others envisioned.

**Effects on habitat occupancy and use**

If an oil spill affects seabird population sizes or reproduction, then recovery from these effects requires...
suitable habitat. Large quantities of oil from the Exxon Valdez spill contaminated the water surface and were deposited on beaches, especially in Prince William Sound. As a result, cleanup activities and natural weathering, little evidence of oiling now remains, and these habitats have returned to normal, at least to outward appearances.

Day et al. (1995) used measurements of bird occupancy and use of areas that received different amounts of oil following the spill as an assay of habitat suitability, arguing that highly mobile organisms such as birds would avoid clearly unsuitable habitats and that occupancy therefore provides a good measure of how the birds view habitat conditions. As birds view habitat conditions, little evidence of oiling now remains, and these habitats have returned to normal, at least to outward appearances.

Day et al. (1995) used measurements of bird occupancy and use of areas that received different amounts of oil following the spill as an assay of habitat suitability, arguing that highly mobile organisms such as birds would avoid clearly unsuitable habitats and that occupancy therefore provides a good measure of how the birds view habitat conditions. As a result, it is not possible to separate their effects, but separately or together they had the potential to produce spill-related impacts on habitat use by the birds.

Day et al. (1995) found that species differed in their responses to initial oiling of bays. Some species, such as tufted puffins (Fratercula cirrhata), showed no evidence of impacts of oiling on their use of bay habitats. Other species, such as black oystercatchers, were significantly less abundant in more heavily oiled bays than in less oiled bays. Some species, such as black scoters (Melanitta nigra), and ravens (Corvus corax), either showed no spill-related impacts or recovered rapidly, so it seems unlikely that the ecological characteristics of the impacted species necessarily predispose them to suffering long-term spill effects.

These multispecies studies provide a perspective on the overall effects of the Exxon Valdez spill that is not apparent from investigations focused on single species of concern. They indicate that use of oiled habitats by many seabird species was reduced after the spill, but they also show that habitat use by other species apparently was not affected. Recovery of habitat use by most of the initially impacted species was rapid, and it appears that impacts of this spill on avian use of oil-affected habitats were not generally persistent.

Oil spills, recovery, and seabird resiliency

The Exxon Valdez oil spill had immediate effects on seabird populations. By any estimate, direct mortality was great, and reproduction and habitat use by several species were affected. Although it now appears that the spill had few persistent or devastating long-term effects on seabirds, in the years immediately after the spill there was considerable disagreement among scientists about the magnitude and extent of observed or expected impacts. There were several reasons for these disagreements.

Figure 7. Percentage of marine-oriented bird species recorded on each of several survey cruises in Prince William Sound, Alaska, that exhibited negative impacts of the Exxon Valdez oil spill on habitat occupancy and use. ES = early summer, MS = mid-summer, FA = fall, MW = mid-winter, LW = late winter, SP = spring. After Day et al. (1995).
First, some studies emphasized only the documentation of impacts, whereas other investigations sought to determine recovery as well. The primary focus of the Trustee studies, for example, was determined by US laws (e.g., Clean Water Act, Oil Pollution Act) that dictate documentation only of damages. Consequently, studies of species that did not appear to show initial impacts were not continued, and efforts to document damages in the remaining studies were intensified. This focus on damages produces an unbalanced view of spill impacts on the avian community as a whole (Wiens 1995).

Disagreements may also have been fostered by differing views about what constitutes recovery. The Exxon Valdez Trustee Council (1993) defined recovery as “a return to prespill conditions or to conditions comparable to those of nonoiled areas.” This definition implies that, in the absence of a disruption such as an oil spill, populations would be in a steady-state equilibrium, with a stable age distribution, and that spatial variation in population levels would be negligible or random (Wiens and Parker 1995). Because of the short-term and long-term environmental variations that characterize marine ecosystems, however, these requirements are biologically unrealistic. Recovery should instead be defined statistically, as the disappearance of a previously documented significant relationship between a population and a measure of initial oil exposure (Day et al. 1995, Wiens 1995). This approach recognizes the variability of the systems and makes no a priori assumptions about equilibrium, but it raises the possibility of committing Type II errors, that is, of failing to detect impacts that really occurred. Day et al. (1995) dealt with these problems in statistical tests by using broad a levels (e.g., a < 0.20 rather than the customary a < 0.05). Use of such levels would enhance the likelihood of documenting impacts and reduce the likelihood of improperly concluding that recovery had occurred.

Despite these sources of uncertainty and the considerable variation in food supplies and oceanographic conditions that characterize the spill region, it is still apparent that seabird populations were not devastated by the spill. Much the same conclusion has emerged from studies of other major oil spills. Although several spills (e.g., the Braer, Torrey Canyon, and Amoco Cadiz spills, as well as the Persian Gulf War spill) have released substantially more oil than the Exxon Valdez spill, seabird mortality was considerably less (Burger 1993, Ritchie and O'Sullivan 1994, Symens and Al Suhaibani 1995), probably because of differences in the type of oil spilled, weather conditions, or the environments in which the spills occurred.

In these other spills, major effects on population sizes or reproduction were generally absent or recovery was rapid. The wreck of the Braer on the Shetland Islands, off the coast of Scotland, in January 1993, for example, was associated with substantial decreases in the numbers of breeding shags (Phalacrocorax aristotelis) and black guillemots (Cepphus grylle), but these effects were limited and localized (Ritchie and O’Sullivan 1994). Breeding success in the following summer was not affected. Studies following the Gulf War oil spills reported similar findings (Symens and Al Suhaibani 1995). On the other hand, some spills have contributed to significant declines in populations, such as that of jackass penguins (Spheniscus demersus) off South Africa (Frost et al. 1976).

Although it is clearly difficult to generalize about the effects of oil spills on seabirds, seabird populations do appear to have considerable resiliency to the disruptions associated with environmental accidents such as oil spills. In fact, the natural variability that is the bane of ecologists attempting to assess spill effects may be the basis for this resiliency. High-latitude seabird populations have evolved in harsh and variable environments. They naturally experience episodic reproductive failures (Harris and Wanless 1991, Wooller et al. 1992) and localized mortality (so-called wrecks) associated with variations in food supplies or severe storms (Harris and Wanless 1984). Because individuals of many species have long life spans, the loss of reproductive opportunities in one or several years may have relatively little effect on long-term demographics. Pools of nonbreeding birds may provide a “buffer” from which individuals may be recruited to replace losses from breeding colonies (Klomp and Furness 1992), and movements between breeding sites may also contribute to population resiliency (Wiens 1995, Wooller et al. 1992). Seabird populations are large-scale, open systems, in which the effects of oil spills may be relatively localized and rapidly dissipated.

This apparent resiliency of seabird populations, however, should not give rise to complacency. The Exxon Valdez spill contributed to the deaths of many tens of thousands of seabirds, and there were at least short-term effects on reproduction and habitat use. Such effects cannot be disregarded, even if they may not be biologically important in the long run. Furthermore, if seabird populations experience several severe environmental changes during a generation, the effects of these disruptions are likely to be cumulative. Adding one more disruption to the effects of food shortages, winter storms, El Niño events, or long-term oceanographic changes could push a population beyond the threshold of resiliency, leading to major, long-term demographic changes. Fortunately, this scenario does not appear to have happened as a result of the Exxon Valdez spill.

### Effects of the spill on science

When an accident as dramatic as the Exxon Valdez spill occurs, many people fear the worst. Immediate effects on water quality, habitats, and wildlife are obvious. What is less obvious are the effects on the scientific process used to determine spill impacts. When an environmental accident (or, indeed, any environmentally contentious issue) creates a potential conflict between science and environmental advocacy, science may suffer.

To begin with, the operational hypothesis that an accident such as an oil spill has major impacts often replaces the statistical null hypothesis of no effect (Shrader-Frechette and McCoy 1993). Given the difficulties of designing proper studies and conducting relevant statistical
tests of either hypothesis, there may be a temptation to relax design requirements or to abandon statistical testing altogether and resort to common sense. After all, common sense tells us that an accident as big as the Exxon Valdez spill must have major, long-lasting effects. What is considered common sense, however, is often guided by preconceptions and emotions, which can lead easily to advocacy of a particular conclusion whether or not there is supporting evidence.

Advocacy can erode the objectivity and rigor of the scientific process. When one argues from “common sense” that an oil spill must have devastating ecological consequences, one is anticipating a particular result that may or may not be supported by empirical observations. Although there is nothing wrong with predicting a result (all good scientific hypotheses offer predictions), advocacy can bolster such expectations to the degree that contrary evidence is not considered or hypotheses are accepted without supporting evidence.

Litigation may act to both reinforce and counteract the weakening of science through advocacy. Litigation polarizes positions (plaintiff versus defendant) and may erect barriers against the free and open exchange of data and ideas integral to the scientific process. Lawsuits and charges were filed within weeks of the Exxon Valdez spill, for example, and as a result scientists working on opposite sides of the legal proceedings were unable to talk to one another, to develop a comprehensive design for gathering data, or to share information for more than four years, even though they were asking many of the same questions. Moreover, lawyers often want results to be presented in the simplest terms, uncomplicated by error terms or discussions of data limitations. This stipulation may generate pressures for scientists to provide data in a simplified form before they have been fully analyzed and interpreted or peer reviewed. The litigation context may also create pressures for scientists to take sides, thereby amplifying advocacy and lessening scientific credibility. Interviews with jurors after the 1994 civil trial in Alaska, for example, indicated that they were overwhelmed with conflicting testimony from scientific experts, so they largely ignored the scientific findings in reaching their verdict (Barker 1994).

It should be noted, however, that litigation does not inevitably promote advocacy and erode science. Scientific testimony in legal proceedings may be subjected to scrutiny that is more detailed and probing than typically occurs in peer review. Properly channeled, these pressures may encourage greater care and rigor in all phases of scientific research and lead to more intensive examination of results before they are publicized.

The media coverage that accompanies high-profile oil spills may also reinforce advocacy and weaken science. Unsupported and premature conclusions may be judged through the press rather than through normal peer review, and speculations can easily become converted to facts (Parrish and Boersma 1995, Wheelwright 1994). The Exxon Valdez spill provides several examples of this advocacy–science interaction. Fry’s (1993) statements that “western Prince William Sound became a dead zone overnight,” and his title, “How do you fix the loss of half a million birds?,” as well as Heinemann’s (1993) declaration that the spill was “an unprecedented catastrophe for the common murres of the northern Gulf of Alaska” seem unnecessarily dramatic, especially in view of the meager empirical support. Nysewander et al. (1993b) initially labeled as speculation their suggestion that the replacement of breeding adults at murre colonies by young inexperienced breeders might have delayed egg laying and lowered reproductive success. Later in their article, however, this speculation was stated as a conclusion (Nysewander et al. 1993b). Burger and Fry (1993, p. 260) further strengthened this conclusion: “[C]hanges in the spatial structure of the impacted colonies and in bird behaviour caused almost complete breeding failure in 1989 and 1990 and the lost production of at least 215,000 murre chicks.” In the popular press, this notion was now reported as fact: “With so few experienced breeding birds left, some colonies have failed to produce any young at all since the spill. In others, inexperienced birds are breeding a month late, leaving chicks vulnerable to predators and winter storms” (Pain 1993, p. 5).

Estimates of total seabird mortality from the Exxon Valdez spill also illustrate the influence of environmental advocacy, litigation influences, and media coverage (Parrish and Boersma 1995). Preliminary estimates placed total mortality at 100,000–300,000 birds (Figure 9; Piatt et al. 1990). Accounts in the popular press initially reported these values, then slightly higher estimates. Ford and his colleagues (Ecological
rapidly spreading oil slick is known to have killed 580,000 birds. Over 1989-1991, 375,000-435,000 carcasses on the beach. These simulations estimated mortality at 300,000–645,000, with best-guess approximations of 375,000–435,000 birds (Figure 9). Estimates subsequently reported in both the scientific and popular literature generally fell within this broad range, although a college textbook in environmental science (Miller 1992) stated that the rapidly spreading oil slick is known to have killed 580,000 birds. Over time, estimates in the popular press progressively shifted toward the high end of the range. An environmental biology textbook published in 1994 stated that total mortality could have been as high as 700,000 individuals (Miller 1994). A recent reevaluation of some of the parameter values for Ford’s model, however, led Piatt and Ford (in press) to conclude that the spill probably killed on the order of 250,000 birds (Figure 9).

Clearly, there has been a tendency among some scientists (e.g., Fry 1993) to use high-end estimates selectively, and this proclivity has been even greater among journalists and textbook writers. The estimates, however, are all founded on model projections from carcass counts. The only things that are certain are that approximately 30,000 oiled carcasses were retrieved and that this number represents some unknown fraction of the total number of birds killed by the spill.

Some of the research conducted following the Exxon Valdez oil spill was characterized by premature or incorrect claims, speculations that became facts, communication through the press rather than normal scientific channels, and a failure to examine data carefully. Unfortunately, public perception of the oil spill and its consequences is still based largely on such information, even though more rigorous and deliberative studies and reanalyses of some of the earlier work have shown that there were few obvious, long-term effects on seabirds.

What is the role of science and scientists in such large and contentious environmental accidents? Science can provide a necessary foundation for common sense, but it is the responsibility of scientists to remain objective and recognize situations in which the data indicate that what seems to be common sense is wrong. The role of the scientist, in both research and litigation, is to delineate the domain of scientifically supportable statements and to point out instances in which data are used selectively in the pursuit of advocacy. The importance of the environmental issues and the integrity of the scientific process demand no less.

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