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Exxon Valdez Oil Spill
Seabird Restoration Workshop

Edited by
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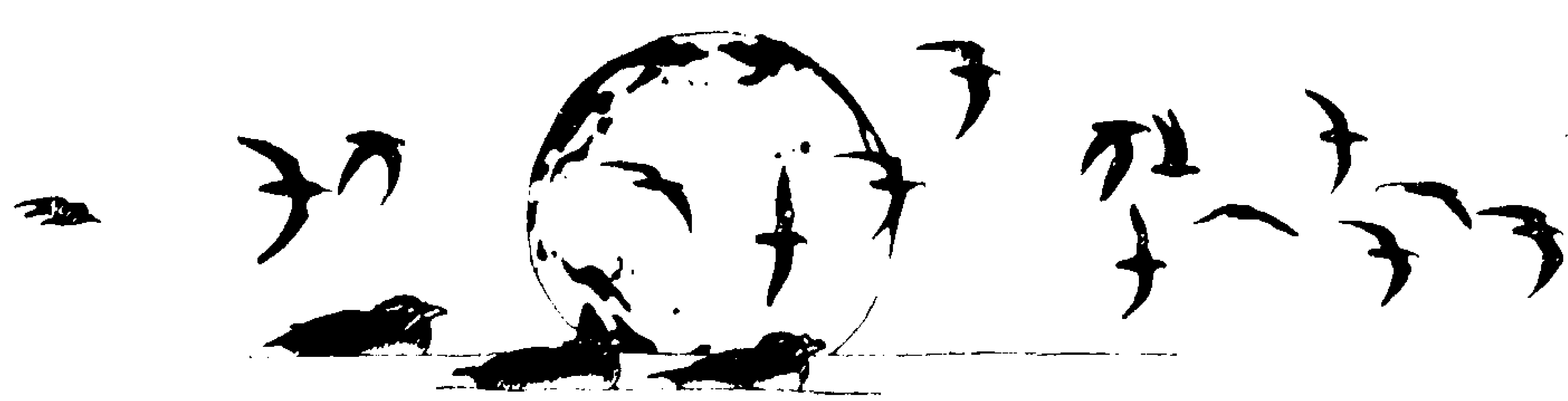

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Girdwood, Alaska

September 29-October 2, 1995

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EXECUTIVE SUMMARY

The purpose of this project was to gather knowledgeable scientists from throughout the world to attend a workshop that would identify and evaluate the techniques that can be used to restore seabird populations injured by oil spills. The workshop first addressed seabird restoration from a general perspective, and then applied the general discussions and conclusions to the specific problems of the *Exxon Valdez* oil spill (EVOS) and emphasized the seabird species considered to be "not recovering" from EVOS at the time of the workshop (common murre, harlequin duck, marbled murrelet, and pigeon guillemot). The workshop developed general policy recommendations related to EVOS, and recommendations for specific restoration techniques that should be applied to common murre, marbled murrelet, and pigeon guillemot populations in the EVOS area. The workshop also described and discussed over 20 different restoration techniques (including natural recovery) and outlined their assumptions and deficiencies. It was apparent to the workshop participants that critical baseline data are lacking for measuring injury to many seabird populations in the EVOS area, and for evaluating the efficacy of potential restoration projects. The workshop determined that such data are needed, and proposed a series of research recommendations to achieve this end. Finally, although this report contains many specifics related to EVOS, it also provides generic guidance for developing seabird restoration plans anywhere.

GENERAL POLICY RECOMMENDATIONS

- The *Exxon Valdez* Oil Spill Trustee Council (Trustee Council) should enlarge the oil spill impact area for seabirds beyond the immediate oil spill zone. Defining the oil spill area based on oiled shorelines fails to recognize the larger geographic area in which seabird populations may have been injured, and severely limits restoration options for species with high rates and distances of dispersal. Restoring seabird colonies outside the spill zone may facilitate restoration of colonies within the spill zone.
- The process by which resources are included, reclassified, or removed from the "Injured Resources" list needs to be improved. The workshop developed criteria to be used to determine if the population effects resulting from a spill are a concern, and to establish a priority list for restoration activities. We recommend that the Trustee Council adopt these criteria and use them as guidelines for identifying the seabird species or populations requiring restoration. Finally, the Trustee Council should continue to open the classification process to scientific scrutiny and review.
- New toxicants to control or eradicate introduced predators or competitors need to be registered. While the Department of Interior may be the lead agency for this recommendation, the Trustee Council should endorse this approach.
- There should be multi-year commitments by the Trustee Council on specific projects, especially field studies that measure parameters known or expected to show high annual

variability. The funding of the Alaska Predator Ecosystem Experiment (APEX) project (Restoration Project 95163) by the Trustee Council is an example of how multi-year funding should proceed.

GENERAL RESEARCH RECOMMENDATIONS

- The geographic and demographic structures of the populations need to be delineated for all nonrecovering species. Genetic and morphometric studies, as well as long-term demographic studies at representative colonies, are examples of how these data can be collected. Populations should also be modeled to assess the potential of particular restoration techniques.
- At this time the Trustee Council is making assumptions about which seabird populations were most affected by the spill. The assumption that the affected populations were in the oil spill area may be incorrect, and genetic and morphometric analysis of carcasses recovered after the spill may help identify which populations were injured and where to target restoration projects.
- The Trustee Council should fund studies that examine trophic interactions and the impacts of net fisheries on seabird population demographics to determine if human-induced alterations to trophic interactions can increase prey availability and therefore enhance the recovery of seabird populations.
- Existing resource sensitivity maps that identify critical areas requiring protection need to be updated and integrated. While individual agencies take the lead for this recommendation, the Trustee Council should endorse this approach.
- All restoration activities *and* nonrecovering species need to be monitored to determine if restoration projects are succeeding or if populations are recovering naturally.
- The workshop endorsed the idea to fund endowed chairs in marine ornithology at the University of Alaska, to assure continuing research on seabirds in all of Alaska and especially the Gulf of Alaska.

GENERAL RESTORATION RECOMMENDATIONS

The workshop determined that the following were the most promising restoration techniques: management of human impacts (e.g., reducing fisheries bycatch of seabirds, reducing breeding habitat loss resulting from habitat destruction or colony disturbance, preventing introduction of predators); habitat or nest site creation or enhancement (e.g., habitat preserves, land purchases, improvements in habitat quality); and predator control at colonies. These are broad-based techniques that benefit a suite of species. The removal of introduced exotic species from islands has the potential to restore an entire ecosystem, not just one species of seabird, while the effective management of net fisheries bycatch will benefit all species that are inadvertently taken in fishing nets.

The workshop also determined that adding birds to wild populations through captive rearing, translocation, and rehabilitation offer the lowest probability of success of all restoration techniques considered in this workshop. Among the major shortcomings of these techniques are that they are extremely labor intensive there is a relatively high risk of failure or low level of success, and they are expensive. Furthermore, these techniques are most appropriate when whole colonies have been extirpated or when populations are close to extinction.

Of particular interest were our discussions on the feasibility of enhancing food resources for seabirds through altering fisheries management practices. We determined that these techniques may be very useful in restoring seabird populations, but that not much is known about the logistical feasibility of these techniques or their population- or community-level effects. We recommended that funds be made available to research these techniques.

PRIMARY RECOMMENDATIONS TO RESTORE MARBLED MURRELETS

- Protect nesting habitat by conserving large tracts of suitable habitat, especially old-growth forests and lands around the heads of bays. Public and private forests should be managed to minimize the disturbance of nesting areas and to prevent the increase and concentration of predators.
- Reduce predation on nest contents by corvids, squirrels, and small mustelids, especially where it decreases reproductive success.
- Reduce bycatch in salmon gillnets, which annually may equal half of the mortality from the *Exxon Valdez* Oil Spill.
- Monitor population trends at sea, breeding productivity (based on at-sea surveys), activity levels at nesting locations, and annual mortality.

PRIMARY RECOMMENDATIONS TO RESTORE COMMON MURRES

- Reduce mortality and increase recruitment and breeding success by removing introduced predators from colonies, preventing the introduction of predators, reducing gillnet mortality, and reducing human disturbance at colonies.
- Examine food-web interactions to allow determination of fishery management techniques that will enhance seabird restoration. These include interactions between seabirds and hatchery-reared salmon and forage fish, the relationships between pollock harvests and murre productivity, the relationships between nearshore habitat types and sand lance spawning, and the effects of residual oil on forage fish that are an important part of common murre diets.
- Monitor the size and productivity of breeding populations and the survival and dispersal of adults.

PRIMARY RECOMMENDATIONS TO RESTORE PIGEON GUILLEMOTS

- Control egg and chick predators, including corvids, mink, and river otter.
- Create or enhance nest sites.
- Control human disturbance at colonies and investigate the degree to which guillemots are affected by gillnet fisheries.
- Monitor populations of adult birds at affected colonies as well as productivity and growth rates at target and reference colonies.

RECOMMENDATIONS TO RESTORE KITTLITZ'S MURRELETS

- Reduce disturbance at foraging sites and known nesting areas.
- Investigate and reduce gillnet mortality.

RECOMMENDATIONS FOR RESEARCH ON KITTLITZ'S MURRELETS

- Determine breeding abundance and distribution.
- Investigate breeding phenology, habitat use, and diet, and if population size and growth are limited by prey abundance.

RECOMMENDATION TO RESTORE COMMON LOONS

- Identify breeding areas of nonrecovering populations.

RECOMMENDATIONS TO RESTORE PELAGIC, DOUBLE-CRESTED, AND RED-FACED CORMORANTS

- Identify breeding colonies of nonrecovering populations.
- Conduct restoration activities similar to the primary recommendations for common murres (listed above).

This report also provides background and rationale for the workshop's recommendations, and a fuller discussion of the biological and ecosystem factors that affect these decisions. We set down specific operational goals for restoration activities, and evaluated these goals in terms of their assumptions, their constraints, and our ability to measure progress through monitoring. We also discussed the importance (and assumptions and limitations) of modeling restoration activities, and outlined population-, community-, and ecosystem-level factors that may affect restoration of seabird populations. The workshop emphasized that restoration efforts may be constrained by factors that either are uncontrolled by the restoration activities or are uncontrollable (e.g., global warming and its effect on fish distribution).

CHAPTER 1

INTRODUCTION

This report summarizes the results of a workshop held from September 29 to October 2, 1995, at the Alyeska Resort, Girdwood, Alaska, to discuss the science of seabird restoration. The Pacific Seabird Group (PSG), an international scientific society, invited experts in seabird biology and management from Great Britain, Belgium, France, New Zealand, Japan, Canada, and the United States to devote their cumulative experience totaling half a millennium to develop practical advice and recommendations on how best to restore seabird populations injured by oil spills. The workshop and this report were funded by a grant from the *Exxon Valdez* Oil Spill Trustee Council (Trustee Council) through the U.S. Department of Interior, Fish and Wildlife Service.

We present here the first comprehensive review of seabird restoration. Although the workshop emphasized the seabird species considered to be "not recovering" from the *Exxon Valdez* oil spill (EVOS) at the time of the workshop, this report also provides generic guidance for developing seabird restoration plans anywhere. The workshop first addressed seabird restoration from a general perspective, and then applied the general discussions and conclusions to the specific problems of EVOS.

The Trustee Council, and other oil spill trustee councils, each independently have struggled in their attempts to derive the most efficacious means to restore seabird populations and to allocate seabird restoration funds. Seabird restoration, as a discipline, is in its infancy and represents a new approach to seabird management. Typically, past seabird management plans have focused on cataloguing and maintaining populations or removing perturbations (e.g., alien plants and mammals) from breeding colonies, and purchasing or protecting breeding habitat (e.g., USFWS 1995). Such plans were based on research that examined the natural and anthropogenic factors that affect fluctuations in population size or breeding productivity. Only recently have seabird biologists and managers had funds at their disposal designated for the restoration of seabird populations injured by oil spills or other anthropogenic events. Because many seabird populations often show large fluctuations in numbers and have demonstrated the ability to recover naturally from a wide range of perturbations, the design of restoration plans poses a number of special problems.

This report provides comprehensive background information and a series of recommendations for the Trustee Council. Topics include a synopsis of the type of pre- and postspill data needed to design an effective restoration plan; a description of the data needed to identify injured species or populations requiring restoration; a summary of the type of monitoring activities required to evaluate the success of a restoration activity; an evaluation of over 20 specific restoration techniques; and a summary of population-, community-, and ecosystem-level factors that may affect or be affected by restoration of seabird populations. The report also recommends specific restoration techniques for species that have not recovered from EVOS and describes untested

techniques having sufficient potential that, with additional research, they may merit inclusion in future management plans.

THE *EXXON VALDEZ* OIL SPILL AND THE TRUSTEE COUNCIL

In March 1989, the tanker *Exxon Valdez* grounded in Prince William Sound, Alaska, resulting in the spread of 11 million gallons of crude oil over a wide area. The spill was the largest in the history of the United States, and during the next several months it contaminated islands, beaches, and bays in Prince William Sound, the Kenai Peninsula, the Cook Inlet, the Kodiak Archipelago, and the Alaska Peninsula. The natural resource trustees have estimated that the oil or its effects killed between 260,000 and 580,000 seabirds totaling 90 species (Piatt *et al.* 1990, NOAA *et al.* 1991). Piatt and Ford (1996) estimated that about 250,000 seabirds died; however, the actual number of seabirds killed is disputed (e.g., Parrish and Boersma 1995a, 1995b; Piatt 1995). Most seabird species in the spill area escaped with only a few mortalities, while enough individuals of other species were killed that obvious declines to their populations occurred.

The Trustee Council was established in the aftermath of the spill, and is composed of representatives from six federal and state agencies. It administers a $900 million trust fund that is to be used to restore, replace, enhance, rehabilitate, or acquire the equivalent of natural resources injured as a result of EVOS (Trustee Council 1994a, 1994b).

PACIFIC SEABIRD GROUP

PSG is an international scientific society founded in 1972 to promote the study and conservation of Pacific seabirds. PSG facilitates the exchange and distribution of information on seabirds through annual meetings, the biannual publication *Pacific Seabirds*, and periodic symposia. PSG has held symposia on the biology and management of virtually every seabird species affected by EVOS. In 1993, PSG hosted a symposium on seabird restoration following oil spills. This EVOS workshop is a microcosm of PSG's mission to advance marine ornithology by facilitating the exchange and distribution of information on seabird biology and conservation.

PSG, through its Conservation and Restoration Committees, frequently provides expert comments on seabird restoration plans throughout the Pacific coast of North America. PSG first commented on EVOS issues just weeks after the spill in 1989 when it corresponded with the Administrator of the Environmental Protection Agency. Until the trust fund was established in October 1991, the parties to the EVOS litigation released little information about the effects of the spill and there was little opportunity for public comment. Subsequently, PSG communicated frequently on the expenditure of EVOS Trust Funds. At PSG's 16th Annual Meeting in Victoria, British Columbia (February 1990), three EVOS-related papers were presented, and an EVOS-related public panel discussion was held. In 1992, PSG filed comments with the Trustee Council on the Restoration Framework, the 1992 Draft Work Plan, the Solicitation for Suggestions for the 1993 Work Plan, and the Draft 1993 Work Plan. In 1993, PSG provided written testimony to

the House Merchant Marine Committee regarding its oversight of EVOS restoration activities, and filed comments with the Trustee Council on its proposed Restoration Plan. Also in 1993, when government researchers were first able to publicly discuss their research, PSG held a symposium on EVOS and a separate session on seabird restoration. In 1994, PSG filed comments with the Trustee Council on the Draft 1994 Work Plan, the Draft Restoration Plan, the Draft Environmental Impact Statement, and the Draft 1995 Work Plan. In 1995, PSG filed comments with the Trustee Council on the Draft 1996 Work Plan. On the basis of all this effort related to EVOS, and given PSG's network with worldwide expertise in seabirds and its interest in solving practical problems related to seabird management, it follows that PSG was the ideal organization to host this workshop.

WORKSHOP BACKGROUND AND DEVELOPMENT

Soon after the Trustee Council notified PSG in November 1994 that it had received a grant, PSG's Executive Council appointed a five-person Steering Committee to direct the workshop. This committee was composed of Craig S. Harrison (PSG Vice Chair for Conservation) and Kenneth I. Warheit (Coordinator, PSG Restoration Committee), who were selected to be co-leaders; Mark Rauzon (PSG Chair); William Everett (Chair-elect); and John Piatt (past Chair). The Steering Committee hired George Divoky as the workshop's Executive Secretary.

The grant enabled PSG to provide travel funds to about 30-35 of the people attending the workshop (see pages iii-vi). The Steering Committee and PSG's Restoration Committee initially developed a list of about 100 researchers and resource managers who have worked with seabird damage assessment, monitoring, restoration, or breeding biology of seabirds. The Steering Committee issued invitations from this lengthy list with a view toward balancing the group as a whole to reflect the full spectrum of expertise, experience, and geographical dispersion. The Steering Committee believes that this process produced a workshop in which the whole was greater than the sum of its individual participants. Some invitees were unable to attend because of schedule conflicts, and many highly qualified or interested people could not be invited because the workshop was designed to facilitate discussions and debates that might have been difficult or impossible had the workshop been much larger. Each participant was invited because of his or her experience and expertise, and no one expressly represented any organization or government agency.

Initially the Steering Committee commissioned the preparation of four discussion papers that would address themes anticipated to recur in all workshop discussions. These papers were "The Role of Behavioral Ecology and Long-term Life History Studies in Seabird Restoration" (Sydeman and Nur); "The Population Ecology of Seabird Restoration: Population Dynamics and Metapopulation Models" (Nur and Sydeman); "The Role of Biotic and Abiotic Factors in Constraining or Enhancing Restoration of Seabird Populations" (Ainley and Nur); and "Seabird Restoration Techniques" (Divoky).

During late summer 1995, the Steering Committee sent these papers and other background materials to each participant. We asked participants to study pertinent literature and reports on

seabird restoration techniques, general restoration issues, and the level of impact on nonrecovering seabirds in the EVOS area (harlequin duck, common murre, pigeon guillemot, marbled murrelet). The Steering Committee also asked participants to respond to several questions regarding seabird restoration.

The Steering Committee established four discussion groups, and recruited leaders for each group. These were (1) baseline data, resource damage assessment activities, and restoration goals (Ken Briggs and John Piatt); (2) restoration activities (Dan Roby and George Divoky); (3) restoration and recovery monitoring and modeling (Ed Murphy and Craig Harrison); and (4) factors limiting recovery (Tony Gaston and Bill Everett). We encouraged discussions and contact among participants well before the workshop. Stan Senner of the EVOS office assisted with the development of workshop objectives.

At the workshop, participants met both in plenary sessions and in small groups. Group leaders endeavored to guide the discussions toward conclusions or recommendations relating to specific questions. Some responses required scientists to bring their best judgment, intuition, and knowledge of scientific principles to bear on questions for which data are currently lacking. The Steering Committee urged participants to achieve consensus or, when this was not possible, to define areas of disagreement as explicitly as possible.

The workshop participants rose admirably to the task presented to them. On several occasions, some groups worked well past midnight to resolve the thornier restoration issues. We thank each participant for making the workshop a success, and for helping to make a "great leap forward" in the science of seabird restoration.

The content of this report is a group effort and is based entirely on the discussions among the workshop participants before, during, and after the workshop. Most workshop participants drafted at least a portion of this report, and we asked each participant to review the entire report. The results and recommendations reflect the consensus of the workshop, except where divergent views are expressed. The editors acknowledge that some sections of this report are redundant. This is intentional because many readers will not read this report in its entirety.

--PSG Steering Committee

CHAPTER 2

RECOMMENDATIONS TO THE *EXXON VALDEZ* TRUSTEE COUNCIL

Part A: General Recommendations

In the following chapters we describe general restoration techniques and outline specific projects aimed at restoring populations of seabirds. Although the activities associated with some of these techniques address larger-scale issues, such as ecosystem dynamics (e.g., seabird-fish-fisheries interactions), the purpose of these techniques is to restore specific populations of seabirds. The workshop also addressed general issues or recommendations, not necessarily related to particular restoration techniques, but relevant to the overall approach to the restoration and recovery of seabird populations affected by the *Exxon Valdez* oil spill. The recommendations fall into two general categories, (1) policy and (2) research, and are discussed below.

POLICY RECOMMENDATIONS

1. Enlarge the oil spill impact area for seabirds beyond the immediate oil spill zone. – The *Exxon Valdez* oil spill area is currently defined as the "maximum extent of oiled shorelines, severely affected communities and their immediate human-use areas, and adjacent uplands to the watershed divide" (Trustee Council 1994b:map). Furthermore, the Trustee Council's Mission Statement Number 8 indicates that "[r]estoration activities will occur primarily within the spill area" (Trustee Council 1994b:14). The Trustee Council did make allowances for restoration work outside the spill area "when the most effective restoration actions for an injured migratory population are in part of its range outside the spill area." But they limited those activities by requiring that "the *vast majority* of restoration funds be focused on the spill area, where the *most serious injury* occurred and the *need for restoration* is greatest" (Trustee Council 1994b:14; emphasis added). The consensus of the workshop participants was that the current definition of the spill area excludes the larger geographic area in which seabird populations (as opposed to individual seabirds) were injured, and severely limits restoration options for those affected populations.

   First, the spill area appears to be defined by the extent of injury to shoreline habitat. Such a definition is most efficacious, since the presence or absence of oil on the beach is irrefutable evidence of contamination. However, from a biological perspective the definition is problematic. Although shoreline habitat is an integral and important part of coastal ecosystems, it represents only a portion of the habitat used by populations of mobile species. Individuals from populations breeding outside the oiled area may and probably do spend part of the year (e.g., migratory birds or marine mammals) or part of their life (e.g., plankton

larvae of relatively sedentary marine invertebrates) within the spill zone. If sufficient numbers were present in the spill area at the time of the spill, there is a real potential that these populations experienced "serious injury."

Second, there currently are no data to support the position that the "most serious injury" to all seabird species occurred within this narrowly defined spill zone. In fact, there are no data on the geographic affinities of the seabirds killed during the spill, and to assume that seabird mortality was limited to breeding populations or colonies within the spill zone is premature. Arctic and subarctic seabirds typically undertake considerable seasonal migration. The spill occurred prior to the breeding season for all species of seabirds breeding in Alaska, and the birds occurring within the spill zone at the time of the spill may have included individuals from breeding populations outside the spill zone. Although there is considerable seasonal (Harrison 1982) and year-to-year variability, Prince William Sound and adjacent areas in the Gulf of Alaska can support significant concentrations of wintering seabirds (Gould *et al.* 1982, Piatt *et al.* 1990, Agler *et al.* 1994 and 1995b, Piatt and Anderson 1996). The origin and status of these birds are not known. At the time of the spill, pelagic and nearshore seabirds in this area could be expected to include local breeding birds, breeding birds from distant localities that had not yet returned to their breeding colonies, and nonbreeding birds from any number of localities, some within and some outside the spill area. While the relative abundance of each of these groups within the spill area is not known, because most of the central and northern Bering Sea, the Chukchi Sea, and essentially all of the Arctic Ocean are usually covered with ice in March and into April, the number of birds from populations breeding outside the spill zone may be relatively high. Significant injury to any population with individuals wintering in the spill zone is conceivable.

Third, even if all the mortality resulting from the spill occurred to colonies within the spill zone, restoration of those colonies may be facilitated by activities outside the spill zone. By requiring that most restoration projects take place within the spill zone, the Trustee Council assumes a particular demographic structure to the "populations" within the spill zone. Currently there are little or no data indicating that seabirds breeding within the spill zone are genetically isolated from those outside the zone. Moreover, based on genetic research (e.g., Birt-Friesen *et al.* 1992, Friesen *et al.* 1997) and dispersal studies elsewhere (e.g., Halley and Harris 1993, Harris and Wanless 1991), there is no reason to assume that they are. Immigration among colonies for each species may occur, and the recovery of a colony in the spill area *may* result, in part, from immigrants from colonies outside the area. Similarly, colonies inside the spill area may be important sources of immigrants for colonies outside the spill area. While there are no data suggesting that this immigration-emigration process occurs in this region, studies from other regions indicate that some seabirds regularly disperse hundreds of kilometers from their natal colony (Halley and Harris 1993, Harris and Wanless 1991, Coulson and de Mévergnies 1992). Limiting restoration activities to colonies within the spill area, and thereby assuming that these colonies are demographically and genetically isolated from colonies outside the spill area, may also limit the potential for restoring affected colonies.

In summary, the composition of the winter/spring aggregation of seabirds in Prince William Sound and the Gulf of Alaska is unknown, as are the geographic structure and demography of the breeding populations within the spill. For this reason, no evidence supports the assumption that populations breeding inside the "spill area" sustained the most serious injury or that the greatest restoration needs are in the oiled area. Limiting active restoration activities to the spill area may restrain the potential for recovery by excluding populations that may have been severely injured by the spill. We recommend that the Trustee Council increase the spill area for seabirds to include Middleton Island, all of the Alaskan Peninsula, and the Aleutian Islands.

2. Improve the process by which resources are included, reclassified, or removed from the "Injured Resources" list. – The Trustee Council, along with its scientific advisors, has established a list of species, populations, and habitats that were injured by the spill and that may be appropriate for restoration actions (Trustee Council 1994b). A species, population, community, or habitat included in this list was determined by the Trustee Council to have been injured by the spill, and to have not yet recovered. Conversely, resources reclassified from "not recovering" to "recovering" or "recovered" have been found by the Trustee Council to be recovering or recovered from the spill. With the aid of public review and comment (Trustee Council 1994a, 1994b:29-30), the Trustee Council established the criteria used to determine if a species, population, or habitat was injured early in the restoration planning process. Although the Trustee Council is correct in including sublethal effects and degradation of habitat as part of the injury criteria, the workshop participants determined that these criteria must also include data on basic population demographics, community dynamics, and ecosystem health. Furthermore, to ensure that the decision to place or decline to place a species, population, community, or habitat on the injured resources list is based on biological data, we recommend a process that is more open to scientific scrutiny and review. We acknowledge that the Trustee Council adopted this approach in 1995. We also advocate a uniform policy for determining injury to any resource, noting, however, that injury determination for seabirds may require a different set of criteria than those for fish or shoreline habitats. We recommend that the Trustee Council implement the criteria that we discuss in Chapter 4 to determine which seabird species/populations sustained significant injury from any oil spill and should be the focus for restoration activities.

3. Register new toxicants. – The control or eradication of introduced exotic predators or competitors is a proven and most effective method of restoring local populations of seabirds. However, the implementation of eradication programs may be hampered by federal restrictions on the use of certain toxicants on federal land. We recommend that the relevant trustee agencies (e.g., Department of Interior) first determine what toxicants are most effective at controlling or eradicating target species without having secondary or residual negative effects to the ecosystem, and then determine if it is legal to use such toxicants on federal or state land. If it is not, we recommend that the Trustee Council take steps to register those toxicants for use on federal land, permitting their use as a means of control of predators and competitors.

RESEARCH RECOMMENDATIONS

In determining which *species* of seabirds were injured as a direct result of the *Exxon Valdez* oil spill, and which species are priority candidates for restoration, the Trustee Council has made assumptions about which *populations* were injured, as well as additional assumptions about the demographic structure of the injured populations. Likewise, in recommending specific restoration projects aimed at restoring these populations, we have made similar assumptions about the populations, as well as assumptions about the structure and functions of the community or ecosystem to which these populations belong. Ultimately, we concluded that to realistically determine which assumptions and restoration options, if any, are appropriate for a particular population, information is needed to determine how that population was affected by the spill, and how particular restoration options will affect that population. We recommend that research be conducted to help to (1) delineate the geographic and demographic structures of populations, (2) determine which populations were affected by the spill, (3) estimate the probability of natural recovery through dispersal or recruitment, and (4) understand community and ecosystem effects that may help or hinder recovery. Most participants agreed that a mechanism to fund this research could be the endowment of chairs in marine ornithology at the University of Alaska, which would serve as a long-term catalyst to conduct the research projects that we have identified.

1. Collect population demographic information. – In Chapter 3, we define populations and outline why it is important to determine the geographic boundaries and understand the demographic parameters of populations. In particular, the recovery of a seabird colony following a natural or anthropogenic disturbance will depend, in part, on the geographic and demographic structure of that colony's population. Small, isolated populations with low rates of immigration will recover more slowly than populations that are part of a larger metapopulation or that have higher rates of immigration or gene flow. There is little or no dispersal among genetically isolated colonies or subpopulations, and recovery following disturbance must be through local recruitment. Dispersal among colonies that are part of a larger population or metapopulation should be relatively high, and natural recovery following a disturbance should be relatively rapid due to the influx of immigrants.

Knowledge of the rates and distance of immigration and genetic structure of these colonies or populations would allow for a better assessment of whether active hands-on restoration is needed and, if so, what types of restoration are best prescribed. Natural recovery through immigration or high local production would indicate that no active hands-on restoration other than monitoring is needed. Furthermore, demographic analyses of populations (including genetic analyses as well as data on dispersal gathered mainly through banding efforts and annual measures of colony growth) may point to a colony or geographic region on which to concentrate restoration efforts (e.g., identification of "source" populations) or may help set restoration goals (see Chapter 3 for further discussion of seabird populations, and Chapter 6 for restoration goals).

Studies on the genetic or morphometric structure of seabird populations may identify population markers (e.g., DNA sequences or unique relative proportions of skeletal elements) that would

make it possible to identify the origin of individuals killed by oils spills. Finally, complete demographic analyses would include collecting data on the age-structure of a population (estimated through banding returns), survival of birds from all age classes, age at first breeding, and reproductive success. In order to collect most of these data, birds must be banded and permanent study sites established. These data are essential to adequately model populations so that the relative effects of different restoration options can be evaluated.

We recommend that the Trustee Council fund research on (1) the genetic and morphometric structure of seabird colonies in Prince William Sound, the Gulf of Alaska, and the Aleutian Islands, (2) population structure at representative colonies (these studies should include the banding of chicks for individual and cohort identification and the collecting of reproductive success data), and (3) modeling of the populations to help predict if natural recovery is possible, or to assess the utility of particular restoration techniques (see Chapter 11 for discussion of population models). These demographic studies should be conducted on the seabird species determined by the Trustee Council to be not recovering[1].

2. Analyze carcasses for population information. – Wiens and Parker (1995) and Wiens (1995) review methods by which oil spill impacts can be measured with some statistical rigor. Each method is based on determining statistically significant differences between affected sites and reference or control sites. In some methods, reference "sites" are the affected sites prior to the spill (i.e., baseline studies), while in others, reference sites are single or replicated sites not in the affected area, or series of sites with a gradient of effects—from no disturbance to heavily oiled. The Trustee Council also measured injury or impact by comparing prespill and postspill numbers and trends. The methods prescribed by Wiens, and Wiens and Parker, and the Trustee Council assume that the affected populations were in the oil spill area. In other words, comparing pre- and postspill data, or affected versus unaffected sites for seabird colonies within the oil spill area (see also Erikson 1995), assumes that the mortality associated with the spill is restricted to the oil spill area. We have argued above that this assumption may be incorrect and the assessment premature.

The only existing direct evidence for seabird mortality associated with *Exxon Valdez* oil spill is the oiled carcasses salvaged from beaches within the oil spill area. The majority of these carcasses have been destroyed. However, representative samples have been obtained by a few museums in North America, principally the Burke Museum, University of Washington. If populations or subpopulations of the seabirds nesting in Prince William Sound, the Gulf of Alaska, or the Aleutian Islands differ genetically and/or morphometrically (see above), it may be possible to identify what populations or subpopulations were injured by the spill (see Anker-Nilssen *et al.* 1988 and Warheit 1996 for analyses associated with the 1981 Skagerrak and the 1991 *Tenyo Maru* oil spills, respectively). By analyzing these remaining carcasses, not only will the Trustee Council have a more concrete basis by which to determine *what* populations may have been affected by the oil spill, it will also be able to estimate *how* these

[1] At the time of the workshop, the common murre, pigeon guillemot, marbled murrelet, and harlequin duck were the species of birds listed by the *Exxon Valdez* Trustee Council to be not recovering. In spring 1996, the Trustee Council moved the common murre to the "recovering" list, but added Kittlitz's murrelet, common loon, and double-crested, pelagic, and red-faced cormorants to the "not recovering" list.

populations were affected and *where* to target restoration projects. We recommend that the remaining carcasses (especially those of the common murre, pigeon guillemot, marbled murrelet, Kittlitz's murrelet, and pelagic cormorant) be analyzed for morphometric and genetic population markers (see above) to help determine their source populations. Furthermore, the age class (i.e., juvenile, subadult, adult) and sex of each carcass should be ascertained to help determine the demographic impacts of the injury.

3. <u>Examine trophic interactions, impacts of net fisheries, and community structure.</u> – Although we consider managing seabird food to be a potentially viable restoration option (see Chapter 9), there is a lack of data on trophic interactions and food availability for seabirds in the EVOS area, and on how altering such interactions and availability might enhance seabird populations. The underlying assumption in "enhancing food" as a restoration option is that the current population size for some or all of the seabird species of interest to the Trustee Council is limited by the abundance and availability of prey (see Chapters 9, 12, and 13; Piatt and Anderson 1996). One potential method of enhancing the food supply of seabirds is to reduce competition by altering commercial fishery and/or fish hatchery activities (see Chapter 9 for details). If food availability is limiting recovery, the production or maintenance of trophic competition through hatchery and fishery practices could significantly affect the ability of seabird populations to increase. Little is known about the relationships among seabird consumption of prey, fishery catch of seabird prey, and the production of salmon in local hatcheries.

Not only do humans act as competitors with seabirds for a limited food supply (i.e., direct competition through fishing and indirect competition through the production of direct competitors such as hatchery-reared salmon), they also act as predators on seabirds. Although the entanglement of seabirds in fishing nets is unintentional, its effects on seabird populations can be profound (see Chapter 9c and references therein). Much more must be learned about entanglement of seabirds in the net fisheries in Prince William Sound. We recommend that the Trustee Council provide funds to examine how modifications to the activities of fisheries (including net fisheries) and hatcheries may enhance seabird populations by increasing their food supplies, or by decreasing mortality as a result of gillnet bycatch.

In addition to those anthropogenic activities that may alter the availability of food to seabirds, natural competitors may also affect this availability and foraging success. For example, one of the goals of the EVOS restoration plan is to restore the sea otter populations in Prince William Sound. Although this goal is worthy, a growing sea otter population may affect the food availability and foraging success of local pigeon guillemots and marbled murrelets, owing to the altered subtidal and intertidal habitats that may result from otter restoration (see Chapter 12 for details on how community factors may affect restoration). Little is known on the nearshore community structure in Prince William Sound, and if interspecific competition for food resources is high, the restoration of one species (e.g., sea otters) may compromise he restoration of other species (e.g., guillemots and murrelets). We recommend that the Trustee Council continue to fund research on the nearshore community structure and food

availability (e.g., the Alaska Predator Ecosystem Experiment and Nearshore Vertebrate Predator projects).

4. Examine sources or causes of predation at marbled murrelet nests. – Later in this chapter we describe restoration options for marbled murrelets. Although reducing habitat loss either by altering forest practices or by preserving or purchasing land is singularly the most effective option for preventing further declines in marbled murrelet populations, small-scale activities, such as reducing human disturbance at campsites, may be effective in increasing marbled murrelet populations. Human disturbance at campsites (e.g., accumulation of trash) may attract corvids (crows, ravens, jays, and magpies) to the local area. Corvids are chick and egg predators of marbled murrelets, and human activity near marbled murrelet nest sites may increase corvid predation on nest contents (Singer *et al.* 1991). We recommend that the Trustee Council fund projects to determine if corvid predation associated with human activities decreases marbled murrelet reproductive success, and if such human disturbance can be controlled.

5. Develop resource sensitivity maps. – One of the concerns of those in charge after a disaster is to ensure that every effort is made to prevent any further accidents that could worsen the situation. Another oil spill in the EVOS area could set back the recovery of some or all of the resources injured by EVOS and could complicate or even negate all the restoration efforts implemented so far. Therefore, there is a need for a system to rank the seabird use (and the use by other resources) of different waters of Alaska according to their vulnerability to environmental hazards (see King and Sanger 1979, Carter *et al.* 1993, Williams *et al.* 1995). Oil vulnerability maps for the EVOS area could be used to delineate shipping routes, assign fishing areas, design oil managing facilities, deploy booms, allocate skimmers, and conduct all the other actions that would protect critical seabird areas from injury associated with another oil spill.

In the early to mid-1980s, the National Oceanic and Atmospheric Administration (NOAA) produced environmental sensitivity index atlases for many of the regions that later were affected by EVOS (e.g., RPI 1983). These indices ranked shorelines in terms of their vulnerability to detrimental effects associated with toxic substance spills, including oil, and incorporated both the physical and biological features of shoreline habitats. In 1978, the U.S. Fish and Wildlife Service (USFWS) published an Alaska seabird catalog (Sowls *et al.* 1978), which is periodically updated and maintained in a USFWS database. However, these indices, catalogs, and maps (as well as other databases) need to be revised *and integrated* before they can serve as an easily usable product that would help protect critical seabird areas (or other resources) from further injury. For this reasons we recommend that the Trustee Council provide funds to update, integrate, and publish new versions of the environmental sensitivity index atlases that include seabird breeding and at-sea areas.

6. Monitor all restoration activities. – The only way to determine if a population is naturally recovering or if restoration is required is to monitor the population under consideration and,

in some cases, reference or control[2] populations. In Chapter 7 we outline the monitoring activities that should be implemented as part of a restoration plan. We note here that these monitoring activities are an essential part of the plan and should be emphasized.

7. Fund multi-year projects. – Single-year funding cycles make it difficult to plan and implement cost-effective and scientifically justifiable projects. The workshop recommends that projects be funded on a multi-year basis whenever possible.

Part B: Species-Specific Recommendations

INTRODUCTION

In the following chapters we describe specific restoration techniques and identify their assumptions, advantages, and deficiencies. We also discuss seabird restoration in an ecosystem context and demonstrate how the recovery of seabirds following a perturbation, such as an oil spill, may be enhanced or hampered by large-scale effects. This information is provided (1) to describe what types of restoration activities are available to the Trustee Council, (2) to describe what particular restoration techniques would be favored in Alaska, and (3) to consider community and ecosystem functions when designing a restoration plan. It is in this larger ecosystem framework that the recovery of seabird populations occurs and within which we evaluate particular restoration techniques for common murres, marbled murrelets, and pigeon guillemots (this workshop did not completely address harlequin ducks) in Prince William Sound, the Gulf of Alaska, and the Aleutian Islands.

For each restoration technique we judged the probability of success for a particular species based on the goals of the technique, the current status of that species in Alaska, and the species' life history. We then ranked the restoration techniques using a four-part scale ranging from "do not consider" (= 0) to "best probability of success—should be applied" (= 3) (Table 1).

Comparison of Techniques Among Species

Based on the distribution of scores shown in Table 1, there are more viable restoration options available (techniques receiving scores of either 2 or 3) for common murres than for either pigeon guillemots or marbled murrelets. Discounting restoration techniques Numbers 3 and 5 (see Table 1), the modal and median scores for common murres, pigeon guillemots, and marbled murrelets are 2, 1, and 0, respectively. In other words, more than half the restoration techniques listed in Table 1 (discounting Numbers 3 and 5) can be applied to common murre populations, while most of the techniques are of little use in restoring marbled murrelet populations.

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[2] Restoration activities should be funded only when the restoration plan incorporates appropriate controls or reliable baseline data, so that the success of the restoration can be measured.

With the exception of the techniques that reduce fisheries bycatch and colony disturbance, restoration options that show a relatively high probability of success for common murres also show a high probability of success for pigeon guillemots. There is very little correspondence between these two species and marbled murrelets. The general restoration category that shows the most agreement among these three species is Management of Seabird Habitat. However, even within this "family" of techniques the species differ in that restoration for marbled murrelets should focus on the protection of large tracts of land, while pigeon guillemot restoration should be focused at a smaller spatial scale; even enhancing individual nest sites offers a high probability of success. Common murres may benefit from restoration aimed at either small or large spatial scales, from establishing habitat preserves to protecting nest sites.

Categories of Restoration Techniques That Offer the Highest Probability of Success

Among all the restoration techniques described in Chapter 9, and listed in Tables 1 and 2, the categories that offer the highest probability of success are designed to reduce both the direct and indirect effects of *human disturbance*, rather than to directly manipulate seabird population demographics. In addition, techniques that *reduce mortality of adult birds* show the greatest promise in increasing the rate of growth of a disturbed colony (see also discussion of models in Chapter 11). Two restoration techniques stand out in this regard. First, the removal of introduced exotic predators from seabird colonies (an indirect or lingering form of human disturbance) and the prevention of their introduction or reintroduction are perhaps the techniques with the longest history and highest overall probability of success (see Chapter 9 and references therein). Second, reducing the number of seabirds inadvertently killed in net fisheries (direct disturbance, akin to predation) should have a substantial positive effect on common murre and marbled murrelet populations by increasing the survival of birds in all age classes. *These are broad-based techniques that benefit a suite of species. The removal of introduced exotic species from islands has the potential to restore an entire ecosystem, not just one species of seabird, while the effective management of net fisheries bycatch will benefit all species that are inadvertently taken in fishing nets.*

Categories of Restoration Techniques That Offer the Lowest Probability of Success

The addition of birds to wild populations through captive rearing, translocation, and rehabilitation offers the lowest probability of success of all restoration techniques considered in this workshop, owing to a variety of problems (Table 2). Among the major shortcomings of these techniques are that they are extremely labor intensive, there is a relatively high risk of failure or low level of success, and they are expensive. Furthermore, these techniques are most appropriate when whole colonies have been extirpated or when populations are close to extinction (see Table 2 and Chapter 9 for discussion). Options that involve reduction of human interactions with the resource (Table 2: Management of Human Impacts) may be problematic because they involve alteration of lifestyles, and may therefore receive political or public opposition. Some of these options may pit jobs against the resource.

TABLE 1. RANKING[1] OF POTENTIAL RESTORATION TECHNIQUES FOR COMMON MURRE, PIGEON GUILLEMOT, MARBLED MURRELET, AND HARLEQUIN DUCK POPULATIO 'S AFFECTED BY THE *EXXON VALDEZ* OIL SPILL

RESTORATION TECHNIQUE	COMU	PIGU	MAMU	HADU[2]
1. Management of Predators and Herbivores	**3***	**3**	**1+**	**3**
a) Remove introduced exotics*	3	3*	0	?
b) Remove indigenous species	2	3?	0	?
c) Manage indigenous species	2	3	1	?
2. Management of Human Impacts	**2**	**1**	**3**	**3**
a) Reduce fisheries bycatch	3	1	2+	?
b) Reduce habitat loss	0	0	3	3
c) Reduce colony disturbance	3	0	0	?
d) Reduce at-sea disturbance	1	1	1	1
e) Prevent predator introduction	3	3	0	3*
f) Minimize marine pollution	?	?	?	?
g) Reduce subsistence harvest	0	0	0	1
3. Management of Food	**?**	**?**	**?**	**?**
a) Manage fisheries	?	?	?	?
i) Salmon hatcheries	?	?	?	?
ii) Pollock harvest	?	?	?	?
iii) Herring harvest	?	?	?	?
b) Enhance nearshore habitat	?	?	?	?
i) Sand lance spawning	?	?	?	?
ii) Blennies/sculpins	?	?	?	?
4. Management of Seabird Habitat	**2**	**2**	**3**	**3**
a) Preserve habitat or purchase land	2	1	3	3
b) Improve nest sites	2*	3	0	0
c) Deploy social attractants	2*	1	0	0
d) Reduce predator/competitor interactions	1 (gulls)	1 (puffins)	0	0
e) Create habitat	0	1	0	0
5. Supplement Wild Populations	**0**	**0**	**0**	**1**
a) Release captive-raised juvenile birds	0	0	0	1
b) Translocate juvenile birds	0	0	0	0
c) Rehabilitate injured birds	0	0	0	0

[1] = 0: Do not consider. 1: Likely not to succeed. 2: Appropriate for feasibility studies; moderate level of success. 3: **Best probability of success; should be applied.**

[2] = Workshop did not adequately address harlequin ducks.

* = Outside official "spill area."

TABLE 2. POTENTIAL OR ACTUAL DEFICIENCIES OF SEABIRD RESTORATION TECHNIQUES

	High Financial Costs	Excessive or Extended Logistics or Labor	Enforcement Required	Stakeholder Resistance	Potential Public or Political Opposition	Potential Injury to Source Population	Other
MANAGEMENT OF PREDATORS AND HERBIVORES							
Control predators and herbivores		●			●		Potential for injury to nontarget species
MANAGEMENT OF HUMAN IMPACTS							
Reduce fisheries bycatch	●		●	●	●		Untested techniques
Reduce habitat loss				●			
Reduce disturbance	●		●	●	●		
Reduce predator introductions	●		●				
Reduce chronic pollution			●				Difficult to monitor
Reduce harvest by humans			●		●		
Reduce aquaculture conflicts				●	●		
MANAGEMENT OF FOOD							
Manipulate fisheries, hatcheries, or habitats	●			●		●	Uncertain outcome
MANAGEMENT OF SEABIRD HABITAT							
Acquire habitat preserves or corridors	●				●		Hard to find tracts of land
Create or enhance nest sites		●			●		
Deploy social attractants		●				●	New method
Reduce competitive interactions	●	●					
SUPPLEMENT WILD POPULATIONS							
Release captive-raised juvenile birds	●	●	●				High chance of failure
Translocate juvenile birds	●	●	●			●	High chance of failure
Rehabilitate injured birds	●	●				●	Low returns for effort
NATURAL RECOVERY							
Allow unassisted recovery to occur					●		Potentially of long duration

Enhancing Food

We have proposed several restoration techniques that involve the enhancement of the prey base for seabirds. These techniques are divided into two categories: (1) fisheries management and (2) nearshore habitat enhancement (Tables 1, 2). Enhancing food through altering fisheries practices or improving nearshore forage fish spawning habitat may be very useful; however, not much is known about how these techniques may be implemented or what effect they will have on seabird populations. For example, will further attempts to enhance salmon production in Prince William Sound, which is already at an all-time high level (Francis and Hare 1994), have a detrimental effect on other marine vertebrates, including seabirds, that compete with salmon for forage fish resources? Unfortunately, little research has been done to assess the impacts of large-scale salmon enhancement projects on local marine ecosystems. Conversely, would encouraging a larger harvest of pollock in Prince William Sound have a beneficial effect on seabirds and other marine predators and enhance their recovery? Links between environmental change, pollock abundance, and fisheries and stocks of forage fish have been examined in some detail (e.g., Laevastu 1984, Springer 1992), but a link to seabird populations is missing. As a result, we scored all food enhancement techniques as question marks; but we do consider these techniques promising and recommend that research be conducted to determine not only their feasibility but also their potential effects.

In the following sections, we discuss in detail our recommended restoration techniques for marbled murrelets, common murres, and pigeon guillemots, and provide restoration suggestions for Kittlitz's murrelets, common loons, and double-crested, pelagic, and red-faced cormorants.

Part C: Recommended Marbled Murrelet Restoration Techniques

INTRODUCTION

Most of the world's population of marbled murrelets breeds in Alaska (Mendenhall 1992, Ralph *et al.* 1995), and some of the highest nesting densities of murrelets occur in the area affected by the *Exxon Valdez* oil spill (Piatt and Ford 1993). An estimated 8,400 murrelets were killed by the spill, possibly 7% of the total summer population in the spill area (Kuletz 1996). Marbled murrelets spend most of their lives at sea but breed inland in old-growth forest. Attempts to restore or conserve murrelet populations require that we consider both terrestrial and marine aspects of their biology. Marbled murrelets are widely dispersed and loosely colonial, and concentrations may occur at forested breeding locations and at sea. Their nesting behavior is secretive, except for vocalizations, and their nests are typically widely dispersed and concealed (review in Ralph *et al.* 1995). All these factors make it difficult to census breeding populations or obtain the demographic information needed to develop a restoration plan.

In California, Oregon, and Washington, where the marbled murrelet is listed as threatened under the Endangered Species Act, loss of old-growth forest nesting habitat is considered the primary cause of the population decline (Stein and Miller 1992). Recovery plans in these states have emphasized the protection of nesting habitats. Similarly, habitat preservation has been the main approach to murrelet restoration following the *Exxon Valdez* oil sp although this acts to prevent further injury due to loss of habitat rather than restore the populations. This approach also provides habitat protection for other species.

Following the *Exxon Valdez* oil spill, the first step in murrelet restoration was to identify the characteristics of the birds' nesting habitat, because historical data suggested that it differed from habitats used in southern regions. Prior to 1989, only one tree nest had been discovered in southeast Alaska (Quinlan and Hughes 1990), and six ground nests had been located in south-central and southwest Alaska (Day *et al.* 1983). Between 1991 and 1993, studies funded by the Trustee Council led to the discovery of 22 murrelet nests and the characterization of nesting habitats.

Reducing human impacts on murrelets, in both the terrestrial and marine environments, is likely the best approach to restore murrelet populations (Table 3). On land we can protect large tracts of nesting habitat and, on a smaller scale, minimize the effects of artificially enhanced predator populations. In nearshore habitats, important foraging areas or habitats vital to prey can be protected, and gillnet bycatch of murrelets can be reduced or eliminated by modifying net characteristics or fishing seasons (e.g., Melvin and Conquest 1996). In this section we discuss primary restoration options, consider less viable options, and discuss the benefits of monitoring programs.

TABLE 3. RECOMMENDATIONS FOR RESTORATION OF MARBLED MURRELET POPULATIONS

Primary	*Protect Nesting Habitat:* 1. Acquire and protect prime nesting habitat as suggested by habitat studies. 2. Establish habitat database on a geographic information system, and contribute murrelet data. 3. Survey potential lands available for purchase and rank their value to murrelets. 4. Continue research on use of marine habitat and relationship to terrestrial habitat. 5. Change management and logging practices to minimize impact to murrelets. *Reduce Mortality:* 6. Reduce nest predation by reducing human-caused increase in predators. 7. Institute active predator control in problem areas. 8. Evaluate scope of gillnet mortality and factors influencing bycatch.
Secondary	9. Evaluate human disturbance at critical marine areas and reduce if necessary. 10. Implement efforts to conserve or increase food resources. 11. Continue to research methods that may increase survival of rehabilitated birds. 12. Develop procedures to more effectively aid recovery of injured juvenile birds.

TABLE 3. CONTINUED

Monitoring	13. Monitor populations during summer and winter.
	14. Monitor productivity using at-sea surveys during fledging period.
	15. Monitor inland activity to gauge relative breeding attempts.
	16. Monitor annual mortality: bycatch data, beach census, investigation of events.
	17. Maintain database on birds collected, trapped and released, or rehabilitated.

PRIMARY RECOMMENDATIONS

Protect Nesting Habitat

Because murrelet nesting density increases with forest stand size (Paton and Ralph 1990, Marks *et al.* 1995, Raphael *et al.* 1995) and is highest in major watersheds (Miller and Ralph 1995), conservation of large tracts of suitable habitat is perhaps the most significant method for conserving murrelet populations in Alaska. Current knowledge suggests that prime habitats within the spill zone are composed of old-growth forests with the largest trees in the region, including lands around the heads of bays, and with slopes protected from prevailing summer winds (Kuletz *et al.* 1995a, 1995c; Naslund *et al.* 1995). Although most nests found adjacent to the spill zone were less than 1 kilometer from the ocean, two ground nests were 2 and 6 kilometers inland (Kuletz *et al.* 1995b), and suitable forest habitat exists farther inland along river valleys. The best predictors of murrelet occupation include tree-branch size, potential number of nesting platforms per tree, and epiphyte cover (Kuletz *et al.* 1995a, 1995c; Naslund *et al.* 1995, Hamer 1995).

There are few data that allow us to assess the amount of land needed to preserve a given number of murrelet nests. Nesting density may vary in different habitats or with proximity to prime marine habitats. There is evidence that marbled murrelets are loosely colonial. At Naked Island, Prince William Sound, 7 to 12 pairs of murrelets used a 17.5-hectare stand, and 2 to 3 pairs used a nearby 3 to 6-hectare stand (Naslund *et al.* 1995). Thus, densities ranged from 0.4 to 0.8 pairs per hectare of suitable forest. In one fjord, three radio-tagged birds were nesting in trees less than 1 kilometer from each other at the head of a small bay. In contrast, three ground-nesting birds were separated by 6 to 12 kilometers in different drainages of the main fjord (Kuletz *et al.* 1995b).

Highly fragmented forest may create an "edge effect," resulting in reduced murrelet nesting success due to predation, adverse weather, and tree blowdowns (review in Ralph *et al.* 1995). Conversely, Raphael *et al.* (1995) found that forest patches with more complex edges had higher murrelet activity. However, long, narrow buffer strips along streams or shoreline may not be suitable (Kuletz *et al.* 1995a, Marks *et al.* 1995). Forests of lower quality may provide adequate buffer around high-quality forest patches (Kuletz *et al.* 1995a).

Suitable land parcels should be evaluated for murrelet activity by conducting dawn surveys, and their relative value should be ranked in terms of murrelet occupation. This would assure that the

parcels being considered for purchase are valuable to murrelets. If the potential parcels are too extensive, dispersed, or inaccessible to survey, as is often true in the *Exxon Valdez* spill zone, then habitat-use techniques can predict where optimal nesting habitat may occur.

The best habitat-use studies employ geographic information system databases that incorporate vegetation and landform features. The results of murrelet dawn surveys, which measure nesting activity of murrelets, can then be overlaid with habitat data to ascertain optimal nesting habitat. In the *Exxon Valdez* spill zone, nesting habitat studies for murrelets have compared U.S. Forest Service timber-type databases and on-site measurements to murrelet dawn activity (Kuletz *et al.* 1995a, 1995c; Marks *et al.* 1995). Similar results have been derived in other regions through the use of geographic information system landscape-level databases (Raphael *et al.* 1995) and forest vegetation databases (Grenier and Nelson 1995, Burger 1995). With the geographic information system it is also possible to incorporate the habitat requirements of other species into land purchase decisions. For example, harlequin ducks may nest in valleys that are also important to marbled murrelets.

Murrelet nesting habitat may also be defined by a combination of terrestrial and marine features. In 1993 and 1994, radio-tagged murrelets in Prince William Sound foraged an average of 20 kilometers from their nests (Burns *et al.* 1994, Kuletz *et al.* 1995b), suggesting that good foraging areas are relatively close to nest sites. Murrelets in Prince William Sound and elsewhere can forage up to 120 kilometers (75 miles) from nests if necessary (Kuletz *et al.* 1995b, Hamer and Nelson 1995). Hypothetically, marginal nesting habitat near predictable foraging "hot spots" may be preferred over "optimum" forests far removed from good foraging area. In general, little is known about the relationship between murrelet foraging behavior and nesting habitat selection in Alaska. Further research in this area would be useful.

Finally, public and private lands can be managed to minimize the disturbance to nesting areas and reduce the concentration of predators. We recommend the following forestry practices:

1. Increase the width of buffers along streams and shoreline.
2. Practice selective cutting and, where possible, removal by helicopter. Selective cutting ensures that some older trees will remain available for murrelet nesting.
3. Leave a percentage of large trees during selective cutting, particularly those with large numbers of "platforms" or branches with large moss patches. Older trees with substantial core-rot can be valuable as nest trees.
4. Leave buffers of lower-quality forests around prime nest trees.
5. Minimize the creation of roads that eliminate nest trees and create more edges through and around the stands.
6. Use harvest methods that minimize the spread of disease-carrying insects such as the bark-beetle.

Reduce Mortality

Reduce predation

The breeding plumage and behavior of murrelets appear to be adaptations that minimize predation, but murrelets still experience high losses of eggs, young, and even adults at the small number of nests that have been discovered (Nelson and Hamer 1995). Adult murrelets may be taken at the nest, or in transit to or from the nest, by sharp-shinned hawks (Marks and Naslund 1994) or peregrine falcons (J. Hughes, Alaska Department of Fish and Game, pers. com.). Bald eagles may attack murrelets at sea (K. Kuletz, pers. obs.) and have been observed feeding on murrelet carcasses, but whether they scavenged or killed the birds was uncertain (Burns *et al.* 1994, Kuletz *et al.* 1995b).

Of 32 murrelet nests with known outcomes, 43% were lost to predation (Nelson and Hamer 1995). The most common predators on eggs or chicks are corvids, such as Steller's jays, magpies, northwestern crows, and common ravens. These predators tend to concentrate and expand their population around human habitation. To minimize predation, human trash should be controlled at cleanup sites (in the case of oil spills), campsites, permanent shelters, villages, and coastal towns. The public should also be educated about proper disposal of food waste and discouraged from hand-feeding all predatory species.

Squirrels and small mustelids may also be nest predators (Marzluff *et al.* 1995). As with corvids, squirrels are attracted to human habitation and are best controlled by minimizing human activities that draw them. If concentrations of predators become unmanageable, we recommend predator extermination or translocation in especially important murrelet nesting areas.

Reduce gillnet bycatch

The loss of adults due to natural annual mortality or gillnet bycatch is of much greater consequence to the population than is the loss of juveniles (Beissinger 1995). Murrelets are susceptible to gillnet mortality for several reasons. They forage by diving underwater and usually feed less than 1 kilometer from shore. Both behaviors bring them into contact with salmon gillnets. In addition, murrelets frequently feed in low light conditions, when it may be difficult to see and avoid gillnets. Finally, oceanographic conditions that concentrate commercial fish also attract the forage fish on which murrelets feed, thus increasing encounter rates with gillnets (Carter and Sealy 1984).

Little is known about the importance of gillnet mortality to the Alaska murrelet population, but it is known that elsewhere murrelets are caught in all continental shelf areas with many types of gear (Carter *et al.* 1995). Murrelets were the seabird most commonly caught in salmon gillnets during a bycatch study conducted in Prince William Sound in 1990 and 1991, with an estimated 1,231 and 298 murrelets killed in those years, respectively (Wynne *et al.* 1991, 1992). Extrapolating to other areas of Alaska, Piatt and Naslund (1995) estimated that as many as 3,300 murrelets die annually in Alaska gillnets. This is almost half of the minimum estimated mortality from the *Exxon Valdez* oil spill, and may represent a significant proportion of total adult mortality for this population.

A comprehensive survey of seabird bycatch in gillnets, including set nets, should be conducted to determine which areas have the highest mortality and which factors contribute to high mortality rates (see also Wynne *et al.* 1991, 1992). In British Columbia, murrelet bycatch was found to be highest at night (Carter and Sealy 1984), and commercial fisherman Pete Isleib reported a similar pattern in Prince William Sound (Carter *et al.* 1995). If murrelet bycatch is concentrated temporally or spatially, it may be possible to restrict fishing activity with minimal impact on commercial fisheries. Additionally, experiments could be conducted with different types of fishing gear to determine which gear minimizes seabird bycatch (see Melvin and Conquest 1996 for experiments in Puget Sound, Washington).

SECONDARY RESTORATION TECHNIQUES

The following restoration techniques may be difficult to implement, but are included here to indicate potential options that may become feasible as our knowledge of murrelets and the ecosystem improves or as our ability to manipulate survival of the species increases.

Reduce Human Disturbance

Boat traffic may keep murrelets from critical foraging areas (Kuletz 1996). During the breeding season, limiting boat traffic in key feeding areas may benefit murrelets. Studies should be done to determine if murrelets habituate to some levels or types of traffic. Additionally, low-level pollution associated with boat traffic, particularly small oil and diesel discharges, could be causing habitat degradation or direct mortality. Chronic pollution may directly reduce use of a foraging area if the birds avoid oil, or it can harm birds that ingest oil or suffer reduced insulation from light oiling. Indirectly, pollution can affect murrelets by altering the abundance or distribution of prey. Many of the species on which murrelets depend are intertidal spawners and/or sediment dwellers during periods of their diel cycle (i.e., sand lance). These species are highly susceptible to nearshore pollution (Trasky *et al.* 1977).

Increase Food Resources

Diet studies of marbled murrelets in Prince William Sound (Oakley and Kuletz 1979, Kuletz *et al.* 1996b), like those of pigeon guillemots (Hayes 1996, Hayes and Kuletz 1996), suggest that the prey base has changed since the 1970s. Murrelets' consumption of sand lance, in particular, has decreased as their consumption of gadid species (e.g., pollock, cod) has increased. Because sand lance has been associated with high reproductive success for seabirds (Harris and Hislop 1978, Vermeer 1979, Monaghan *et al.* 1989a, 1989b), murrelets may benefit from increased sand lance availability. The spawning areas and habitat requirements for species like sand lance and capelin are not well known, but should be identified and protected to assure a healthy prey base for all seabirds.

Although ecosystem changes may be responsible for changes in the prey base (Hollowed and Wooster 1995, Piatt and Anderson 1996), studies of key prey such as sand lance and capelin may identify management practices or coastal planning strategies that enhance forage fish abundance. Juvenile clupeids (e.g., herring) and gadid species may also be important in the murrelet diet (Carter 1984, Sealy 1975, Krasnow and Sanger 1986, Sanger 1987); juvenile salmon may be significant as well (Carter and Sealy 1986). These commercial fish species are already a focus of studies funded by the Trustee Council. Because the apparent decline in certain forage fish species was concurrent with the introduction of salmon hatcheries into Prince William Sound, research could examine the effects of hatchery-reared fish on native forage fish abundance.

Currently, hatcheries may provide a temporary and limited resource to murrelets. In 1994, several radio-tagged birds visited the Main Bay hatchery (Kuletz *et al.* 1995b). In 1995, D. Scheel (pers. com.) noted that the number of murrelets at a hatchery increased for four days after release of salmon smolt. Although hatcheries are probably of minimal benefit to murrelets, these observations suggest they may provide a short-term supplement to the murrelet diet.

Rehabilitation

Capture and rehabilitation of oiled murrelets appears to be of little value in enhancing the viability of marbled murrelet populations. In 1989 a relatively small proportion of murrelets were brought to rehabilitation centers (less than 3% of all birds rescued during EVOS), and few birds survived. Only 3 of 33 marbled murrelets (9%) survived (M. Wood, International Bird Rescue, unpubl. data), compared to 51% of the 1,630 birds treated (Wood and Heaphy 1991). However, International Bird Rescue (Berkeley, California) continues to research techniques that would improve the survival rates of small alcids.

The rehabilitation of murrelets under other conditions could be encouraged through public outreach and education. Although oiled adults are not currently good candidates for rehabilitation, temporarily stunned adults, as well as chicks and newly fledged juveniles, have survived to be released. Adults found on the forest floor following unknown injury or downing of their nest tree have been treated and released (G. van Vliet, Alaska Department of Fish and Game, pers. com.; K. Sundet, Alaska Department of Fish and Game, pers. com.). Downy chicks that have fallen from nests and completely feathered juveniles that have not reached the ocean also have been successfully reared and released (Anchorage Bird Treatment and Learning Center, unpubl. data). A secondary benefit from these events has been community involvement and education about this little-known seabird.

MONITORING ACTIVITIES

Estimate Population Trends

It is not practical by conventional means to directly monitor the breeding population of murrelets because they are not colonial, and their nests are difficult to locate. However, their at-sea

populations can be monitored using standard USFWS survey protocols (Klosiewski and Laing 1994, Agler *et al.* 1994). Because murrelets are widely distributed, population estimates can be calculated with relatively narrow confidence intervals, making them good candidates for monitoring population trends at sea (Klosiewski and Laing 1994).

Monitor Murrelet Productivity at Sea

Productivity should be monitored to enable natural resource trustees to respond quickly to a negative trend in the murrelet population. Little is known about the demography of marbled murrelets, but based on their body size, their single-egg clutch, and information extrapolated from other alcids, they probably depend on high adult survival to offset their low reproductive potential (Beissinger 1995).

Because it is not financially practical to measure the reproductive success of large numbers of murrelet nests, a productivity index has been developed (Ralph and Long 1995, Strong *et al.* 1995, Kuletz *et al.* 1996a). This method relies on the ratio of adults to juveniles counted at sea during the fledging period. In south-central Alaska, surveys for juvenile birds can be conducted from late July through August. This period does not coincide with that currently used for the Prince William Sound population surveys and will require a separate effort. Baseline adult-to-juvenile ratios should be obtained for areas of concern and monitored before and after a catastrophic event.

Monitor Murrelet Terrestrial Activity

While at-sea surveys can provide an index of reproductive success, they do not measure reproductive effort. For colonial seabirds, the percentage of birds attempting to breed can be estimated in order to gauge the proportion of breeding birds in the population and annual fluctuations in the size of the breeding population. For marbled murrelets, an analogous survey might be the dawn watch, where inland activity is measured by the number of murrelet detections. There is circumstantial evidence that dawn watches are an index of breeding effort. At Naked Island, Prince William Sound, detections increased from 1989 to 1991, concurrent with a decrease in spill-related disturbance and increasing numbers of juveniles at sea (Kuletz 1996). In Oregon (K. Nelson, pers com.) and British Columbia (Burger 1995), murrelet detections decreased during years with higher than normal sea surface temperatures associated with El Niño. Selected murrelet nesting sites, preferably adjacent to marine areas surveyed for juveniles, could be monitored to determine if birds are visiting nest sites and to detect long-term trends in breeding activity.

Monitor Annual Mortality

The population will not recover even with stable reproductive success if other sources of mortality offset annual recruitment. For example, winter can be a time of food stress, resulting in low overwinter survival. Postfledging survival is normally low for seabirds (Lack 1966) and can

be decreased by reduced food availability in late summer. Other sources of mortality may be identified by periodic and regular monitoring of gillnet bycatch, by conducting beached-bird censuses at selected sites, and by opportunistically obtaining dead or weakened birds.

Part D: Recommended Common Murre Restoration Techniques

INTRODUCTION

The common murre is a circumpolar species of boreal and low Arctic habitats (Nettleship and Evans 1985). On the Pacific coast of North America, common murres breed in dense colonies from mainland northwestern Alaska and the Bering Sea south to central California (American Ornithologists' Union 1983).

About 1.4 million common and thick-billed murres nested in the Gulf of Alaska prior to EVOS, with common murres comprising 80-85% of that total (Sowls *et al.* 1978; but see Erikson 1995). Where both species nest at the same colonies, thick-billed murres prefer narrow nesting ledges, and common murres favor wide nesting ledges and larger, flatter areas (Tuck 1961). About 1.2 million murres nest in the western Gulf of Alaska on the Semidi Islands. Before the spill the largest colonies in the EVOS area were located at the Chiswell Islands, near Seward; at the Barren Islands, at the mouth of Cook Inlet; and in three colonies on the Alaska Peninsula (Sowls *et al.* 1978; see Boersma *et al.* 1995, Erikson 1995, and USFWS unpubl. data for population estimates both before and after EVOS).

Common murres form breeding colonies on seaward-facing cliffs, where they are highly social and lay only one egg (Tuck 1961). Timing of breeding within a breeding group is synchronized, and breeding success is variable, with a maximum of 70-90% of young fledged per breeding pair (Birkhead 1977, Hedgren 1980, Ainley and Boekelheide 1990). Common murres are long-lived, with adult survival averaging over 89% per year (Birkhead 1974, Hudson 1985, Harris and Wanless 1988, Hatchwell and Birkhead 1991, Sydeman 1993); banded murres have lived as long as 32 years.

In spring and summer, common murres are distributed in Alaska mainly over the continental shelf (Gould *et al.* 1982, Harrison 1982). In late fall and winter, they often migrate to protected coastal bays and fjords of the Gulf of Alaska, including the area around Kodiak Island (Forsell and Gould 1981), Prince William Sound (Agler *et al.* 1994), and Cook Inlet (Agler *et al.* 1995b).

In summer, common murres in the Gulf of Alaska forage mainly on fish over the continental shelf (Sanger 1987), while their winter diet also includes euphausids (Krasnow and Sanger 1986). Murres are among the deepest-diving alcids (Piatt and Nettleship 1985), and have been caught in crab pots at 110-130 meters near Kodiak Island (Forsell and Gould 1981).

Murres are particularly vulnerable to floating oil (King and Sanger 1979) and have been determined by respective natural resource trustees to be an injured species in the *Apex Houston*, *Nestucca*, *Tenyo Maru*, and EVOS spills. In fact, common murres comprised 61%, 60%, 73%, and 74% of the total number of seabird carcasses recovered from these spills, respectively (Page *et al.* 1990, Warheit 1996; USFWS, unpubl. data). Piatt *et al.* (1990) estimated that EVOS killed 120,000-134,000 breeders, mostly from the Chiswell Islands and the Barren Islands, while Piatt and Anderson (1996) used a figure of 185,000 for common murre mortality.

PRIMARY RECOMMENDATIONS

Reduce or Prevent Mortality

Restoration activities that reduce or prevent the direct mortality of common murres (juvenile, subadult, or adult birds, but particularly established breeders) were considered by the workshop participants as the most promising of all murre restoration options. We considered five different restoration alternatives designed to reduce or prevent common murre mortality.

Remove introduced predators

Restoration projects designed to *remove introduced* predators from nesting habitats both within and outside the spill areas have the highest potential for succeeding on islands. Releasing nesting populations from predation pressures caused by introduced species should result in an almost immediate increase in population numbers. Furthermore, because the recovery of a colony of common murres within the spill area may result, in part, from immigrants from colonies outside the spill area, we advocate predator removal from colonies outside the spill zone as a potentially effective restoration option for colonies within the spill zone. Programs to remove introduced foxes from Alaskan islands have been conducted successfully by the USFWS for several years (see Bailey 1993), and we recommend that these programs be implemented at islands where common murres are most vulnerable to predation by introduced predators. These programs should also be designed as experiments with adequate postremoval monitoring (e.g., EVOS-sponsored Projects 94041 and 95051). Finally, although predator removal projects are widely applicable to many seabird species, they are effective in restoring murre colonies only where such colonies contain nesting habitats accessible to predators (e.g., flat or less precipitous rubble-type habitats easily accessible to foxes).

Prevent introduction of predators

Because introduced wild or domestic predators negatively affect common murre populations (see above), preventing their introduction helps ensure that a colony remains viable and is a potential source of emigrants. Furthermore, because a small number of predators can result in high seabird mortalities at colonies (e.g., red foxes; see Peterson 1982), it is prudent to design and implement programs that will prevent the introduction of even one individual. As with projects for

removing introduced predators (particularly rats, but also other rodents, canids, mustelids, and felids), projects designed to prevent their introduction have the highest potential for succeeding on islands. Prevention programs are species specific and may employ a variety of methods. For example, programs that prevent the introduction of rats to islands with seabird colonies may include an immediate and organized response to ship-grounding, the placing of poison bait stations, developing/supporting programs to inspect vessels for rats, and educating vessel operators about the dangers of rat introductions to island habitats. Rat response and educational programs have been recently developed by the USFWS for use on the Pribilof Islands, and also can be employed on the Aleutians and in the Gulf of Alaska. Finally, as with the removal of predators, preventing their introduction is an effective mechanism for maintaining viable murre colonies only where such colonies contain nesting habitats accessible to the potential predator.

Reduce gillnet mortality

Little is known about the effects of drift- and set-net gillnet fisheries on common murre populations in Alaska, and there is a great need for research in this area. DeGange *et al.* (1993) summarized the effects of coastal gillnet fisheries on seabirds in Alaska and reported that common murres are among the species most frequently caught. Furthermore, data from other regions, such as California, indicate that coastal gillnet activities can have a drastic effect on common murre populations. For example, between 1983 and 1986, 50-97% of all seabirds killed in gillnets in the Gulf of Farallones and Bodega Bay, California, were common murres, and their estimated mortality in central California from 1979 to 1987 was 70,000-75,000 birds and included the extirpation of one colony (Takekawa *et al.* 1990). The resulting decline in the central California population may have been as high as 52.6% (Takekawa *et al.* 1990, reported in DeGange *et al.* 1993). Wynne *et al.* (1992) estimated that 432 common murres died in gillnets in the Prince William Sound and Copper River fishing districts in May and June 1991. If common murres are being caught in high numbers in gillnets in coastal Prince William Sound, Cook Inlet, and the Gulf of Alaska, the effects on their populations may be severe. We recommend that research be conducted to determine the effects of gillnet bycatch on common murre populations in the extended EVOS area, and that programs be developed and implemented to reduce or eliminate the drowning of common murres in gillnets. Furthermore, we recommend that partnerships be developed among state and federal agencies, fishing associations, and native corporations to modify fishing gear or the timing and location of gillnet activities in the vicinity of nesting colonies and foraging areas.

Reduce Human Disturbance

Humans can disturb seabird colonies unintentionally through such activities as recreation (e.g., hiking, hunting, kayaking, boating) and aircraft overflights, and this disturbance may negatively affect common murre recruitment and productivity. We recommend that projects be designed and implemented to reduce or eliminate this type of disturbance at and near common murre colonies.

The responses of common murres to human disturbance are difficult to quantify because reactions depend on a series of potentially confounding variables. First, responses may depend on the stage of breeding when the birds are disturbed (e.g., prelaying, laying, incubation, hatching, chick-rearing). Second, responses may vary markedly among individuals and colonies depending on local conditions and circumstances. Birds exposed to regular, ongoing disturbing activities may react differently from birds exposed to the same activities on an intermittent basis (e.g., birds at colonies with histories of close-flying aircraft may respond differently from individuals in populations where this form of disturbance is rare or nonexistent). Third, older, more experienced breeders may tolerate disturbance better than younger, less experienced breeders; individuals incubating eggs or brooding chicks may tolerate events better than roosting off-duty mates or nonbreeders (e.g., Denlinger *et al.* 1994). Also, the effects of disturbance may be cumulative over time (e.g., several years); however, these types of effects are extremely difficult to measure because local abundance and breeding phenology may differ among years as a result of differing environmental conditions.

Population size and local habitat conditions (e.g., configuration and stability of nesting substrates) should be considered when designing programs that eliminate or reduce the negative effects of human disturbance. For example, protecting small colonies where negative effects are likely to be proportionally larger may be of greater value than protecting large colonies where the same human activity may not be a disturbance. Furthermore, the disturbance may be of little or no consequence to large populations before an oil spill, but the same level of disturbance may become biologically important if the populations are markedly reduced in size or are under stress by the event. Finally, preventing or reducing specific forms of disturbance, such as noise and vibration from low-flying aircraft, is likely to be more beneficial at colonies with unstable nesting substrates or densely packed concentrations of nesting birds than at colonies with more stable or less densely populated nesting substrates. In designing projects to reduce human disturbance at nesting colonies, we offer the following recommendations:

- Projects should be site-specific and tailored to address local circumstances and needs.
- Projects should be developed in close cooperation with local user groups (e.g., sport, commercial, and subsistence hunters and fishermen; charter vessel and aircraft companies and associations; guiding and tourism businesses and associations), and appropriate state and federal agencies (e.g., Alaska Department of Fish and Game, U.S. Fish and Wildlife Service, National Marine Fisheries Service, National Park Service, U.S. Forest Service, Bureau of Land Management). Specific concerns regarding vessel and aircraft activities should be discussed with the U.S. Coast Guard and the Federal Aviation Agency, respectively.

Conduct Research on Fish and Fisheries Management Practices

The workshop identified at least four management areas where there were insufficient data to determine if common murres are being negatively affected by fisheries activities. In particular, we recommend that research be conducted on what effects particular fisheries management practices may have on common murre productivity and survival. Research may be directed

toward a variety of issues on an as-needed basis, but the following topics should be given priority:

Hatchery-raised salmon

Investigate and evaluate the effects that large-scale releases of hatchery-raised salmon may be having on marine food webs. During the past 10 to 15 years, hundreds of millions of salmon fry have been raised and released into western Pacific marine ecosystems annually by private and government-sponsored hatchery programs. These programs, for the most part, have been developed to support, maintain, and enhance local and regional commercial fishing industries and are particularly well developed in Alaska. Hatchery-reared salmon present a twofold problem that may ultimately depress food resources for common murres: competition with, and then predation on, forage fish. In Alaska there are concerns that *young* hatchery-reared fish may be competing for zooplankton stocks needed to support and sustain forage fish populations (e.g., sand lance, capelin) important to fish-eating seabirds and marine mammals (e.g., common murres, young seals and sea lions). In addition, as these hatchery-reared fish grow they no longer compete with the forage fish but become their predators, and the artificially inflated at-sea populations of released salmon may reduce local and regional availability of forage fish to seabirds and marine mammals. The need to develop and implement studies that can address these concerns appears to be particularly important in Prince William Sound and parts of the northwestern Gulf of Alaska where large-scale hatchery programs are operating annually. We recommend that food-web interactions between hatchery-reared salmon and forage fish and the effects of these interactions on the stocks of forage fish important to common murres be investigated.

Commercial harvest of walleye pollock

Large-scale harvests of walleye pollock in the northern Gulf of Alaska may reduce the numbers of young-of-the-year pollock available to common murres at some colonies in some years. Pollock harvests may also be altering marine food webs in unknown ways. We recommend that research be conducted to investigate the relationships between pollock harvests and seabird productivity.

Nearshore/shore habitats

Sand lance are an important prey item for common murres in the areas affected by EVOS. We recommend that research be conducted on how the nearshore and beach habitats can be *protected or modified* to protect or enhance sand lance spawning. If research determines that modification techniques are feasible, it should also be determined whether it would be too difficult or expensive to modify enough habitat to significantly alter sand lance productivity to the degree that it would benefit murres and other fish-eating seabirds.

Residual oil on forage fish

Residual oil from EVOS is present on certain beaches and may be inhibiting spawning activities of forage fishes such as sand lance and capelin. We recommend that research be conducted to determine (1) if residual oil is present along spawning beaches and (2) if the oil is affecting the productivity of those forage fish that are an important part of common murre diets.

SECONDARY RECOMMENDATIONS

Social Attraction

Social attraction may be a useful technique for assisting recovery of common murres *at certain colonies* both inside and outside the EVOS area, but the workshop determined that this technique should be restricted to sites where the entire nesting population has been eradicated (the cause of the eradication at an individual colony is not important if the purpose in conducting restoration is to return birds to a particular region). Social attraction may also be useful when employed in combination with predator control or removal programs (see above) at sites that no longer support populations of birds (e.g., western Gulf of Alaska and the Aleutian Islands, where colonies have been extirpated by introduced predators). In most cases, though, social attraction techniques may be of little value for at least five reasons:

- Birds still present at injured colonies likely serve as better attractants than any manmade decoys or sound recordings.
- The number of decoys that can be effectively deployed at an injured colony may be limited by available funds and physical factors. That is, placing decoys in many typical cliff-nesting habitats may be costly, time-consuming, and dangerous.
- Decoys placed at injured but nonextirpated colonies will occupy space (i.e., potential nest sites) more appropriately used by the remaining birds or new recruits.
- Attracting birds to one colony may preclude recruitment to others.
- Common murres have shown the ability to find and colonize suitable nesting habitat without human-assisted social attraction.

Enhancement of Existing Nesting Habitats

Habitat modification

Improving nesting habitats has some potential to increase murre productivity at injured colonies by providing areas that may be less susceptible to egg and chick loss. Techniques might include modifying nesting ledges (e.g., altering widths and slopes) to prevent egg loss, shoring up areas to prevent or reduce the number of natural rockfalls, or creating overhangs to provide better shelter for eggs and chicks during inclement weather conditions. To help ensure positive results, programs proposing to use these techniques should be required to evaluate whether certain types of nesting habitats are preferred by the birds, or are measurably superior in terms of increasing

productivity and survival. Also, projects proposing to use these methods should be required to identify if the abundance of any particular habitat is limiting recovery, and then determine if this habitat can be constructed efficiently and cost-effectively by modifying existing habitats or substrates in nearby adjacent areas. Furthermore, the habitat to be modified must not be required by other naturally occurring animals or plants in the region.

Habitat protection

A different class of nesting habitat enhancement is the removal (or the prevention of the introduction of) exotic or domestic species *that have the potential to damage common murre nesting habitats* both within and outside of spill zones (e.g., cattle, sheep, goats). These types of projects have the highest potential for succeeding on islands. They are usually of greater benefit to burrow-nesting species (e.g., puffins, petrels), but they may also be relevant to common murres if birds are nesting (or previously nested) on flat, accessible terrain. One possible method for accomplishing this would be to purchase privately owned land that is currently being affected by the grazing activities of domestic species, and place this land into the public trust.

Captive Management

Captive management (e.g., captive rearing and release of birds) is a technique that should be considered only in extreme cases when all other possibilities have been exhausted and common murre numbers have dropped to the point at which they are endangered over an entire region. There are several problems in using captive management as a restoration tool for common murres:

- Rearing enough chicks to positively influence injured populations would be technically difficult and extremely costly (Fry 1991).
- Postfledging survival of chicks released at injured colonies would require that chicks be adopted and fed by unrelated adult males for the extended period of postfledging care (approximately 60 days) or kept in captivity until adult age is reached (Kress and Carter 1991).

Translocation of Birds

Translocation of common murres is another potential restoration technique that should be considered only in extreme cases when all other possibilities have been exhausted. This method involves the capturing of chicks at noninjured, healthy colonies and releasing them at injured or extirpated colonies. Although this method has not been tried, it suffers from the same problems as captive rearing of common murres. That is, common murres have extended postfledging parental care, and the successful translocation of chicks would require that the chicks be adopted by chickless adults or kept in captivity until independent (Kress and Carter 1991). If chicks are being translocated to colonies that have been extirpated, there will be no adults in the area to adopt the chicks.

Rehabilitation of Oiled Birds

Rehabilitation of oiled seabirds may have intangible benefits in terms of public support for restoration. However, the survival of rehabilitated common murres, once released back to sea, is low, while the cost of rehabilitation is high (Sharp 1996, Fry 1991). Furthermore, the rehabilitation of oiled birds may give the public false perceptions about the impacts of spills and the subsequent probabilities of recovery. In general, we recommend that rehabilitation of oiled birds be used only with small populations of common murres where the survival of individual birds is important to the viability of the population. We also recommend that the public be educated about the fact that the rehabilitation of *most* seabirds, including common murres, is costly and generally not successful. Finally, if rehabilitation is to be used, we recommend that effective triage procedures be developed and employed (see Chapter 9f).

MONITORING ACTIVITIES

Monitoring activities associated with seabird restoration projects are discussed in Chapter 7. We list here important activities associated with monitoring common murre colonies, and emphasize that such studies should be designed for both the target (injured) and reference (uninjured) colonies (see Wiens and Parker 1995, Wiens 1995). We recommend that the following population parameters be monitored.

Productivity

Data on murre productivity (chicks per nesting attempt) should be collected from a series of plots at each colony in an effort to monitor reproductive success. Preferably, monitoring should be conducted annually at several colonies within the affected region until it can be demonstrated that productivity has remained within normal limits for several consecutive years (e.g., four to five consecutive years; see Chapter 6 for other ways of measuring success). These data also can be used to monitor nesting phenology should that be an issue.

Size of Breeding Population

Data on population numbers at breeding colonies should be based on at least five separate counts made on different days during the nesting season at a statistically adequate set of monitoring plots (see Gerrodette 1987, Byrd 1989, Hatch and Hatch 1989, Wanless *et al.* 1982, Harris *et al* 1985). These activities should be conducted annually at several sites within the affected region until significant positive trends are clearly apparent. In the event that numbers show little change for several years (e.g., five to six years), monitoring efforts may be modified to census colonies about every two to three years until trends are evident. To calibrate counts, the diel attendance patterns must be determined in conjunction with total counts. This will show what proportion of

the population is present at a given time of day, thus allowing comparison of counts conducted at different times of day.

Survival

Survival is one of the most difficult population parameters to monitor, and requires repeated observations of banded birds. Therefore, our first recommendation is that studies be implemented to band both common murre adults and chicks at breeding colonies with continued monitoring for resightings of banded birds. High breeding fidelity in common murres allows survival to be monitored by observing the rates of return to the breeding colony.

Additionally, implementing long-term beached bird surveys can provide estimates of "normal" postfledging and winter mortality in a region, and can identify those years and events that result in unusually high mortality. Although this method does not provide an estimate of average survival rates for individual birds, it may help provide data on the demographic impact of unusually high fall and winter mortalities (especially if sex, relative age, and area of origin [via genetic or morphometric markers] are determined for each bird).

Part E: Recommended Pigeon Guillemot Restoration Techniques

The pigeon guillemot is a cavity- or crevice-nesting alcid with a broad geographic range extending from Arctic Alaska south to southern California (American Ornithologists' Union 1983). The species forages in nearshore waters, usually within 5 kilometers of the nest (Drent 1965). The pigeon guillemot breeds solitarily or in loose colonies (as do the black and spectacled guillemots), and the distribution and abundance of breeding pairs is often dependent on nest-site availability (Storer 1952). The typical clutch size is two eggs.

The pigeon guillemot population in Prince William Sound decreased from about 15,000 birds in the 1970s to less than 5,000 in the 1990s (Agler *et al.* 1994, Sanger and Cody 1994). Over 600 pigeon guillemot carcasses were recovered after the spill, and may represent 10-30% of the total mortality resulting from the spill (Piatt *et al.* 1990). Although there is evidence suggesting that the Prince William Sound population was in decline at the time of the spill, relative declines in populations were greater along oiled than unoiled shorelines (Oakley and Kuletz 1996).

Reasons for the decline and lack of recovery are not clear and could be related to changes in prey availability and/or increased predation at the nest. Schooling fishes, particularly sand lance, account for a smaller proportion of food returned to chicks now than before the spill. Also, predation on guillemot eggs and chicks was minimal before the spill but now is a major factor influencing breeding productivity (Hayes 1995, Oakley and Kuletz 1996).

PRIMARY RECOMMENDATIONS

Predator Removal or Control

The most efficacious restoration technique for pigeon guillemots in Prince William Sound is likely to involve the eradication or control of predators on eggs and chicks. Control of terrestrial predators has been shown to benefit guillemot populations in the Aleutian Islands, where populations rebounded dramatically after eradication of foxes (Byrd *et al.* 1994). The species that prey on guillemot eggs and chicks in Prince William Sound are many; they include northwest crow, common raven, black-billed magpie, Steller's jay, gray jay, mink, and river otter. Mink and river otters will also prey upon adults in the nest cavities. Adults and fledglings may be taken by bald eagles and peregrine falcons. There is evidence that predation on guillemots on Naked Island has increased since the late 1970s and early 1980s (Oakley and Kuletz 1996). More than 25% of the nests monitored on Naked Island were depredated in 1994 (Hayes 1995).

Any reduction in the number of predators would almost certainly increase guillemot productivity. Besides negatively affecting productivity, the presence of these predators could be acting to reduce recruitment at the affected colonies. Islands outside the spill zone that have introduced animals should be considered for predator control or eradication. Any colonies in the northern Gulf of Alaska that can be increased through predator eradication may be a source of potential recruits for Prince William Sound colonies. Rates of immigration in pigeon guillemots may be high; recent work in Arctic Alaska has shown that black guillemots will regularly disperse more than 500 kilometers and that over half the recruits at one colony were immigrants (G. Divoky, unpubl. data).

The control or eradication of indigenous predators is more problematic, and is not generally recommended given both the potential ecological effects and public opposition. USFWS has made exceptions, however, and indigenous predators (e.g., gulls) have been eradicated to protect or enhance another species (e.g., Atlantic puffin; Kress and Nettleship 1988). For terrestrial predators, fencing of high-density nesting areas, rather than trapping or poisoning, may be a sufficient predator control measure.

Nest Site Enhancement and Artificial Nest Sites

Guillemots are cavity nesters that can use a variety of nest types; their only nesting requirement is overhead cover (Storer 1952). Artificial nest sites have been used successfully by several burrow-nesting species of seabird (Priddle and Carlile 1995 and references therein). The use of artificial nest sites has been documented for pigeon guillemots in Washington (M. Mahaffy, pers. com.) and on the Farallon Islands (Ainley and Boekelheide 1990), and for its congener, the black guillemot, in Arctic Alaska (Divoky *et al.* 1974). In the latter instance, artificial nest sites increased a population of black guillemots from 15 to 225 pairs over a period of 15 years.

In Prince William Sound guillemots nest in rock crevices in cliffs, in talus piles at the base of cliffs, and beneath cavernous tree-root systems at the edge of cliffs. On Naked Island, and

probably on many other islands in Prince William Sound, suitable cavities are probably not limiting to the population. On Naked Island, many sites used by guillemots in the late 1970s and early 1980s currently are not being used, possibly because of increased predation pressure from corvids and mustelids. However, on Jackpot Island (1.6 hectares with little shoreline), nests may be limiting and only one type of nest site (tree roots) is available. If the abundance or availability of prey is not limiting the numbers of guillemots at this location, creating high-quality nesting cavities might be a viable restoration technique.

Artificial nest sites for pigeon guillemots would need to be designed to exclude predators while appealing to prospecting guillemots. Crows, mink, and magpies can probably enter most openings that allow access to guillemots. However, a tight entrance and several baffles might deter corvids. The location of the nest box, rather than the dimensions of its entrance, would be more important for preventing mink from getting to eggs or chicks. River otters could be excluded by a small entrance. By varying the size and shape of entrances and passageways and monitoring rates of prospecting and occupation, it may be possible to develop a functional and predator-free nest site. Occupation of the sites by breeding birds will increase the sample size of nests for ongoing studies (assuming that they attract nonbreeders and not experienced birds abandoning nearby natural sites) or, at the very least, allow for better monitoring of nesting success and chick growth rates. An alternative to providing nest sites at available islands would be the provision of nest sites on offshore pilings and "dolphins" (a group of pilings, often with a platform) created for the express purpose of providing guillemot nesting habitat. Such structures would lack terrestrial predators and could support a cluster of artificial nest sites with easy access for monitoring.

An alternative to providing entirely artificial nest sites is the enhancement of natural nesting cavities. Some existing crevices might attract guillemots if they were slightly more concealed or simply offered some additional protection from the elements. Enhancement techniques would not require the purchase of any new materials (boulders and flat rocks at the colony could be used), and such work could be done coincidentally with normal field work during nest visits.

Control of Anthropogenic Factors

The effects of human disturbance at seabird colonies are legion; examples come from around the world (see Manuwal 1978, Burger and Gochfeld 1994 for reviews). Because pigeon guillemots generally breed in small, scattered colonies, the potential for catastrophic population effects caused by human disturbance at the colony is not high. Disturbance of birds rafting just offshore from a colony is not likely to harm their breeding efforts unless the disturbance is chronic. However, camping and other on-land activities that disturb breeding birds at the nest could result in abandonment (Drent 1965), a reduction in breeding success (Cairns 1980), decreased recruitment, and increased breeding dispersal.

Guillemots successfully occupy working docks and other locations where human activity occurs daily during the breeding season. Thus the species can habituate to the presence of humans if their nesting cavities offer security to the incubating adults. In Prince William Sound, Jackpot

Island may be most vulnerable to the effects of human disturbance because of the high density of nests there. Colonies of concern should be identified and, when possible, access to these colonies prevented during the breeding season. Alternatively, a public information campaign, targeted at recreational and commercial boaters, could identify the areas and activities to be avoided during the breeding season.

Gillnetting operations in Monterey Bay, California, have drowned large numbers of pigeon guillemots (King 1984). In Alaska, pigeon guillemots are caught in set gillnets (K. Kuletz, pers. com.). A study should be undertaken to identify the magnitude of the guillemot bycatch mortality in Prince William Sound, with the goal of decreasing mortality associated with these fisheries (see discussion of bycatch in Chapter 2c).

SECONDARY RESTORATION TECHNIQUES

Enhancing Food Supplies

Sand lance has declined in the diet of pigeon guillemot chicks at the nest, while apparently lesser-quality prey has increased (Hayes 1995). This apparent change in abundance and availability of a preferred prey may be part of an ecosystem shift and could be a factor in the lack of recovery of pigeon guillemots. However, there may be methods that modify nearshore habitats or shorelines that will increase prey abundance for this species (see Chapter 9d). The lack of known techniques and the uncertainty of the role that prey abundance or composition is playing in the lack of recovery makes this a low-priority restoration option. Studies of the nearshore ecosystem (e.g., the Alaska Predator Ecosystem Experiment, EVOS Restoration Project 95163) may provide some understanding regarding the lack of recovery by pigeon guillemots. In addition to the possibility that an ecosystem shift has occurred, prey populations may still be affected by EVOS. Although it is unlikely that direct ingestion of oil is affecting the birds seven years after the spill, indirect effects of oil might be important. Hemosiderosis has been observed in demersal fish collected from oiled eelgrass beds in Herring Bay, Knight Island; these fish were in poor condition as judged by lipid and glycogen stores (S. Jewett, pers. com.). The incidence of hemosiderosis would likely be less of a factor with time.

Monitoring Activities

As with all nonrecovering species, the monitoring of the population is necessary to provide information that will allow assessment of the need for or success of various restoration techniques. Monitoring should consist of censuses of affected colonies and populations in known oiled locations as well as censuses in unaffected (i.e., unoiled) areas. The latter will provide important reference sites to allow the determination of whether population trends at affected colonies reflect natural regional trends or impacts of the spill, and will also help determine the effects of the restoration effort. Population size, as measured by the number of breeding pairs or, less ideally, total number of birds (breeders and nonbreeders), is the most important parameter. Populations are best sampled before the beginning of egg laying when both

members of a pair are visible during the daily periods of colony attendance (Vermeer *et al.* 1993). General population estimates could be obtained with minimal field time. The percentage of nonbreeders associated with a breeding colony ranges from 0 to 50% (Ewins 1985, Hilden 1994; G. Divoky, unpubl. data). Only at colonies where intensive work is conducted can we obtain accurate estimates of the number of breeding pairs or detect trends in the breeding population. A colony with 50% nonbreeders, for instance, could have its number reduced by half and still have had no change in the breeding population. For those populations where the size of both the breeding and nonbreeding populations can be monitored, changes in the nonbreeding population can act as important indicators of the condition of a population (Klomp and Furness 1991).

In addition to the number of adult birds, the productivity of target and reference colonies should be monitored. For nonrecovering colonies or populations, breeding success and the factors that limit hatching and fledging success should be monitored annually. The possibility of monitoring productivity through the use of nearshore censuses of adult/young ratios, as is being tried with marbled murrelets (Kuletz 1996), should be examined. If successful, the technique would reduce the need to locate nests for productivity studies. Banding of chicks and adults can help elucidate the reasons for a lack of recovery and should be part of a monitoring program. At a minimum, all fledging chicks at target colonies should be banded so that the percentage of fledglings that survive and return to breed at their natal colony is known. Banding at reference colonies or any colony in the northern Gulf of Alaska could provide information on immigration to the target colonies and should be conducted when possible. In northern Alaska, immigrants made up well over half the recruits at a black guillemot colony (G. Divoky, unpubl. data), showing the importance of productivity at adjacent colonies in rates of colony growth.

Banding and individual marking of adults is more logistically complex than banding chicks, but should be done if target colonies fail to recover. If banding shows that adult mortality is a factor contributing to the lack of recovery, then manipulation of the sources of adult mortality (i.e., gillnet bycatch and predation) could provide additional avenues of restoration. Monitoring studies should also include studies of the chick provisioning and growth rates at target and reference colonies. If low fledging weights caused by low-quality prey are contributing to a lack of recovery, then food enhancement restoration techniques may be worth pursuing.

REJECTED TECHNIQUES

The workshop deemed captive breeding, translocation, and social attraction to be last-resort techniques, appropriate for use only when a pigeon guillemot population is on the brink of extinction. Captive breeding and translocation could be employed only where there is little or no possibility of immigration from adjacent colonies. Because guillemot chicks are independent at fledging, young could be released into the wild from captive breeding or after translocation. Additionally, because they lay two eggs and are able to re-lay if the initial clutch is removed early in incubation, it might be possible to obtain eggs from wild populations for raising chicks in captivity without harming source populations.

Part F: Recommendations Regarding Restoration and Monitoring of Other Marine Bird Species

At the time of the workshop, the Trustee Council listed common murres, harlequin ducks, marbled murrelets, and pigeon guillemots as nonrecovering injured species. This judgment was based on the quantified injury resulting from the spill and the status of each species. However, several species of marine birds were not listed as injured because there were no data detailing how the spill affected populations or the status of those populations. For example, Kittlitz's murrelet was not initially considered an injured species, mainly because this species is rare, local, and difficult to study, and few data were available about its abundance and distribution. In the absence of data, the Trustee Council was unable to determine if this and other species were injured.

In 1996, the Trustee Council added Kittlitz's murrelet, common loon, and double-crested, pelagic, and red-faced cormorants to the nonrecovering injured species list (Trustee Council 1996). In this subchapter, we review the status and restoration/research options for these species.

KITTLITZ'S MURRELET

Introduction

Kittlitz's murrelet breeds from northeast Siberia and the Commander Islands to southeast Alaska, with its center of abundance extending from southeast Alaska to Kodiak Island (Harrison 1983). The primary breeding areas for Kittlitz's murrelets are the southern Kenai Peninsula, Prince William Sound, and Glacier Bay in southeast Alaska (Isleib and Kessel 1973; USFWS, unpubl. data). Its population probably numbers in the tens of thousands, but little is known about its abundance or its biology. Kittlitz's murrelet coexists with its more abundant and widespread congener, the marbled murrelet, and is similarly noncolonial, nests inland, and has cryptic breeding plumage. However, unlike the marbled murrelet, Kittlitz's murrelet nests exclusively on the ground, usually at high elevations in barren scree (Day *et al.* 1983).

It is difficult to distinguish between marbled and Kittlitz's murrelets during at-sea surveys, and the two species are frequently combined as *Brachyramphus* murrelets in survey estimates. In Prince William Sound, *Brachyramphus* murrelets have declined 67% since the 1970s (Klosiewski and Laing 1994), and approximately 10% of the identified *Brachyramphus* murrelets in the area were Kittlitz's murrelets (Agler *et al.* 1994). The *Brachyramphus* murrelet population in Prince William Sound is currently estimated at 89,000 to 138,000, which would suggest that Kittlitz's murrelet numbers approximately 9,000 to 14,000.

In general, during the breeding season, Kittlitz's murrelets are found near tidewater glaciers at the heads of bays and fjords in Prince William Sound. Although these areas were not directly oiled, murrelets breeding in these areas were probably affected by oil southwest of Prince William Sound before arriving at their breeding grounds (Kuletz 1996). Only 72 Kittlitz's murrelet carcasses were recovered and identified from EVOS; as a result, this species was not included in the initial list of injured species. However, we know little about the abundance, distribution, and productivity of Kittlitz's murrelet. The actual mortality from the spill may have been considerably higher than the 72 carcasses recovered (Kuletz 1996), perhaps as high as 3% of its total population (van Vliet 1993). Because the spill occurred in the center of Kittlitz's murrelet's range, and because there is a legitimate question as to the status of this species following the spill, the Trustee Council added Kittlitz's murrelet to the list of injured species (Trustee Council 1996). Furthermore, beginning in 1996 the Trustee Council funded a study to investigate the life history of and habitat use by Kittlitz's murrelet.

Research Recommendations

Determine abundance and distribution

There is a need for more precise data on the population size and distribution of Kittlitz's murrelet in the spill zone. Since the local distribution of Kittlitz's murrelets is rather patchy, we recommend that at least one complete shoreline survey of Prince William Sound be conducted to locate all major concentrations. This information should be used to modify the current protocol used to monitor the Prince William Sound population.

Similar surveys could be conducted along the outer coast of the Kenai Peninsula and Kachemak Bay/lower Cook Inlet and compared with historical data in the southern Kenai Peninsula. Sites of particular interest along the southern Kenai include the upper portions of East Nuka, Harris, and Aialik Bays and, in Kachemak Bay, the Grewingk Glacier runoff. Kittlitz's murrelets also occur around Kodiak Island and the Alaska Peninsula, but USFWS surveys suggest that their numbers in these areas are too low to warrant a large census effort.

To estimate the effects of EVOS on Kittlitz's murrelet, comparisons can be made among Prince William Sound, Kenai Fjords, and Kachemak Bay. Although Kachemak Bay is in the designated zone, the inner bay where Kittlitz's murrelets congregate was relatively unoiled, with little apparent effects on *Brachyramphus* murrelets (Kuletz 1996). Kittlitz's murrelets occur in several large fjords in northern Prince William Sound that were not oiled, and comparative studies on long-term population trends within Prince William Sound should be conducted.

Investigate breeding phenology, habitat use, and diet

There is little information on the marine habitat use, diet, or productivity of Kittlitz's murrelets. Information on seasonal and diel activity patterns would improve monitoring protocols. These types of intensive studies are best done at multiple sites. The breeding phenology of Kittlitz's murrelets is not well known, but observations suggest that they arrive at breeding areas later and

leave earlier than marbled murrelets (K. Kuletz, unpubl. data). Replicate surveys from April to September would document dates of arrival and departure from the breeding area. Late summer counts of fledglings at sea could define the fledging period and provide an index of productivity.

Kittlitz's murrelets are usually found near tidewater glaciers and glacial runoff, and therefore use less true marine habitat than marbled murrelets. Because Kittlitz's murrelets forage near glaciers, they may depend on physical and biological properties associated with tidewater glaciers, such as upwelling and turbulence where macroplankton productivity is high. Kittlitz's murrelets feed on the same fish as marbled murrelets, but may also eat more crustacea and euphausids (Krasnow and Sanger 1986). However, a chick monitored by video camera inland of Kachemak Bay was fed exclusively Pacific sand lance, capelin, and other forage fish (Naslund *et al.* 1994). Diet should be recorded by observations of adults with fish, by stomach samples, or by stable isotope analysis.

Investigate the food limitation hypothesis

Because both species of *Brachyramphus* murrelets have declined, Kittlitz's murrelets should be included in studies investigating the effects of prey resources on seabird populations. Changes in fjord or glacial regimes might impact the productivity and abundance of Kittlitz's murrelets, either positively or negatively. The highly localized occurrences of Kittlitz's murrelets could promote studies that compare the abundance of fish with the abundance and productivity of Kittlitz's murrelets.

Define nesting habitat

Few Kittlitz's murrelet nests have been found, and little is known about their nesting habitat or behavior, conspecific associations, or foraging range. As with the marbled murrelet, radio-telemetry is probably the best method of discovering Kittlitz's murrelet nests in an unbiased manner. Tagged birds would also provide data on the distances between nesting and feeding areas. Ground searches in potential nesting habitat could be conducted in association with dawn watches, although a protocol for surveying upland activity of Kittlitz's murrelets needs to be developed. Kittlitz's murrelets do not appear to be vocal during dawn flights to the nest (Naslund *et al.* 1994), but several nests have been found by sighting a departing bird (Day *et al.* 1983, Day 1995, Naslund *et al.* 1994). Once a nest is found, time-lapse cameras can provide information on nesting behavior and fledging success (Naslund *et al.* 1994).

Restoration Recommendations

Minimize disturbance at nest sites

Most Kittlitz's murrelet nests have been found above 300 meters, and all have been found in unforested habitat. Therefore, the nesting habitat of Kittlitz's murrelet would not likely be affected by logging; however, mining operations, construction of roads or power lines, or similar activities may negatively affect breeding activities. Until more is known about the nesting

habitat of Kittlitz's murrelet and about the distribution of nests in a breeding area, we cannot make specific recommendations.

Reduce disturbance at foraging sites

There are no data on the effects of boat traffic or noise on the foraging activities of Kittlitz's murrelets. However, the birds' association with tidewater glaciers makes them susceptible to disturbance from tour boats and glacial ice harvest. Both operations occasionally use horns or explosives to cause glacial calving. Once areas of Kittlitz's murrelet activity are identified, disturbance should be minimized or restricted during the breeding season. We recommend a study on the effects of boats and noise on Kittlitz's murrelets and their potential to habituate to disturbance.

Investigate and reduce gillnet mortality

The few data on the gillnet mortality of Kittlitz's murrelets suggest that mortality associated with their incidental bycatch is a serious problem. In Prince William Sound, Kittlitz's murrelets constituted 5% of the total identified murrelets killed in gillnets in 1990 (Wynne *et al.* 1991). However, in 1991 they accounted for approximately 30% of murrelet bycatch (Wynne *et al.* 1992). By extrapolating from net permits and data from Wynne *et al.* (1991), Piatt and Naslund (1995) estimated that *Brachyramphus* annual mortality was between 813 and 2,043 murrelets (±95% confidence intervals) in Prince William Sound and 1,100 murrelets in lower Cook Inlet. Because Kittlitz's murrelets averaged 16% of the total murrelet bycatch, between 130 and 323 Kittlitz's murrelets may be killed annually in Prince William Sound. An additional 100 Kittlitz's murrelets may be taken in lower Cook Inlet (based on the 9% proportion of the Kittlitz's murrelet in the *Brachyramphus* population).

In Prince William Sound, the estimated annual bycatch of the Kittlitz's murrelet is approximately 2% of the population. Although most murrelets were caught in the Copper River district, three northern fishery districts—Coghill, Unakwik, and Eshamy—overlap with areas of very high Kittlitz's murrelet densities. Fishing in Unakwik and Coghill in particular may have affected local Kittlitz's murrelet populations since the 1970s.

Further study is needed to determine the extent of gillnet bycatch and factors affecting bycatch rate. The effect of bycatch in the Coghill and Unakwik areas can be determined by surveying specifically for Kittlitz's murrelets and focusing bycatch studies in those areas. Limited data suggest that Kittlitz's murrelets may leave Prince William Sound by early August, which may preclude them from being caught in nets located in the northern districts. However, nothing is known about their postbreeding dispersal, and the late summer fishery in the Copper River district may overlap with postbreeding congregations of murrelets.

COMMON LOON

At least 216 of the 395 oiled loons recovered in Prince William Sound following EVOS were common loons (J. Piatt, unpubl. data). The Trustee Council recently placed the common loon on the injured species list (Trustee Council 1996).

Subadult and adult birds from one or more unidentified breeding populations were killed by EVOS. Efforts to restore the injured populations would first require the identification of nesting geography of common loons found in Prince William Sound in mid-March. Evers *et al.* (1996) have captured and color leg-banded 867 common loons in Prince William Sound and sighted 148 of these birds on their breeding grounds using a night spot-lighting technique. Recent efforts in wintering areas indicate that this technique can be adapted for identifying individual common loons in coastal wintering groups (D. Evers, pers. com.). Although color marking research is feasible, it may not result in a large-scale determination of common loon breeding grounds. Satellite-tracking of radio-tagged individuals appears to be a better technique to identify the breeding areas of wintering or migratory loons. A pilot project using the adapted capture technique on winter coastal waters and experimental implantation of satellite telemetry appears promising.

Once identified, an injured population should be assessed for restoration needs. Primary criteria used to identify the need for restoration are population density and productivity rates (chicks fledged per territorial loon pair). These data could be compared with densities in adjacent areas or similar habitats and with reported productivity rates such as those summarized by McIntyre (1988). Monitoring is discussed below. Nesting frequency may also reflect injury (Field *et al.* 1993, McIntyre 1992).

Restoration efforts for a common loon breeding population may be direct or indirect. Indirect restoration techniques include those designed to enhance the productivity of the remaining breeding loon pairs by reducing the effects of limiting factors on nesting success and chick survival. Factors limiting the productivity of North American common loons include human disturbance, direct killing or harvest, egg and chick predation, habitat loss, water-level fluctuations, fishing line and net entanglement, fishhook and lead sinker ingestion, and environmental contamination (McIntyre 1988, Loon Preservation Committee 1990, Evers *et al.* 1996). Management and mitigation techniques specific to individual factors include management of gillnetting, public education, warning signs, employment of artificial nesting islands, nest covers, breeding habitat protection, and improvement of water-level management regimes (e.g., Sutcliffe 1979, Loon Preservation Committee 1990, Fair and Poirier 1993). Other mitigation techniques may include reduction of illegal hunting, additional protection from predators, and alteration of fish harvest techniques and management.

Successful indirect restoration efforts require significant return of subadults to natal areas. D. Evers (pers. com.) reports a 5-30% return of subadults to natal lakes; dispersal and mortality rates of nonreturning loons are unknown. However, intensive indirect restoration management appears to successfully enhance common loon populations. The threatened New Hampshire common loon population has doubled in number during two decades of intense management

(Loon Preservation Committee 1995). Common loon populations of multiple-pair reservoirs have increased after 5 to 14 years of intensive management (Fair and Poirier 1993). Because natural colonization is slow, translocation is potentially valuable in areas where loons have been extirpated. This technique, however, has not yet been attempted. Translocation of independent nonfledged juveniles appears feasible, and experimental development of this reintroduction technique has been proposed in the Anchorage area (Evers *et al.* 1995). This technique assumes that subadults return to transplanted fledging areas. Return data from approximately 350 juvenile common loons color-banded from 1989 through 1995 will more accurately indicate return rates over the next several years (D. Evers, pers. com.).

In the context of restoration, common loon populations are monitored on the breeding grounds to determine significance of injury, investigate limiting factors of production, and assess effects of restoration efforts. Loon monitoring techniques on the breeding grounds (Belant *et al.* 1993, Lanctot and Quang 1994) and on wintering areas (Jodice 1992) have been described and evaluated. Determination of injury may begin with less intensive ground or aerial surveys of adult populations, nesting frequency, and possibly chick production. Determination of limiting factors and effects of restoration efforts require more intensive monitoring (e.g., Loon Preservation Committee 1990, Evers *et al.* 1996).

Preventive efforts include development of techniques to determine the geography of injured common loon breeding populations, development of direct restoration techniques, and improvement of oil transport systems to reduce wildlife injuries caused by oil spills.

PELAGIC, DOUBLE-CRESTED, AND RED-FACED CORMORANTS

Three species of cormorants nest throughout the spill area, with the exception of the inside waters of Prince William Sound (USFWS 1996). In the Gulf of Alaska, cormorants generally nest in relatively small colonies of less than 100 nests (Baird *et al.* 1983). Cormorants eat bottom-dwelling and midwater-schooling fishes (Ainley *et al.* 1981). In the Gulf of Alaska, schooling fishes such as Pacific sand lance and capelin are important prey species (Ainley *et al.* 1981, Baird *et al.* 1983).

Injury to cormorants resulting from the spill was documented for nonbreeding birds that spend their summer in Prince William Sound (Klosiewski and Laing 1994, Day *et al.* 1995) and for birds breeding along the south coast of the Kenai Peninsula (Vequist 1990, Day *et al.* 1995). The number of breeding pelagic cormorants also declined at Gull Island in lower Cook Inlet in 1989 (Slater *et al.* 1995). Although the Trustee Council now lists cormorants as injured, it has funded no studies specific to cormorant restoration. However, biannual marine bird surveys in Prince William Sound have documented the lack of recovery of the nonbreeding cormorants (Agler *et al.* 1994). It is not known if cormorants along the Kenai Peninsula have recovered, because data have not been collected since 1991 (Day *et al.* 1995). Before restoration activities can be designed, the nonrecovering populations must be identified. Recommendations for restoration of cormorants would be similar to the recommendations for restoration of murres (see Chapter 2d).

OTHER MARINE BIRD SPECIES

The oil spill affected the abundance and habitats of several marine bird species in Prince William Sound and along the south coast of the Kenai Peninsula that were not included in the Trustee Council's injured species list (Klosiewski and Laing 1994, Day *et al.* 1995). Along the Kenai Peninsula, these species include red-necked phalarope, mew and glaucous-winged gulls, rhinoceros auklet, tufted puffin, and common merganser. In Prince William Sound, these species include Arctic tern, mew and glaucous-winged gulls, scoters, horned and red-necked grebes, Barrow's goldeneye, bufflehead, and common and red-breasted mergansers. Based on marine bird surveys in Prince William Sound, none of these species seem to have recovered significantly. In addition, Irons (1996) demonstrated that the oil spill affected the productivity of black-legged kittiwakes in Prince William Sound and that their productivity had not recovered by 1995; further, goldeneyes and mergansers are not increasing in the oiled area as fast as they are increasing in the unoiled area, which may indicate an oil spill effect (Agler *et al.* 1994, Agler *et al.* 1996). Day *et al.* (1995) concluded that by 1991 some of these species were using oiled habitat no differently than they used unoiled habitat, while other species continued to avoid the oiled areas (see also Day *et al.* 1997).

FOOD AS A LIMITING FACTOR

Food may be an important factor limiting seabird populations (Ashmole 1963, Birt *et al.* 1987, Cairns 1992). There is evidence that the recovery of injured piscivorous marine birds in the Gulf of Alaska and Prince William Sound may be limited by food (Duffy 1996). Population sizes of several piscivorous marine birds in Prince William Sound and along the Kenai Peninsula coast had declined before the oil spill (Nishimoto and Rice 1987, Klosiewski and Laing 1994, Agler *et al.* 1995a). Diets of some seabirds in Prince William Sound and the Gulf of Alaska have shifted during the last two decades from energy-rich prey (e.g., Pacific sand lance and capelin) to lower-energy prey (e.g., gadid species) (Hayes 1996, Hayes and Kuletz 1996, Piatt and Anderson 1996), and evidence of food stress has been noted in some birds (Piatt and Anderson 1996).

MONITORING

Following an oil spill, the abundance and productivity of marine bird populations need to be monitored first, to ascertain which populations have been injured, and second, to determine the degree to which the injured populations have recovered. The Trustee Council has funded specific studies designed to measure the abundance and productivity of species identified by the Trustee Council as being injured. However, for species only recently identified as being injured, and for those species whose status is unknown, monitoring activities have not been extensive. Monitoring studies can include colony monitoring for some species and at-sea monitoring for all species. Currently the Trustee Council is funding an at-sea survey that assesses the status of all

species in Prince William Sound every two years. These surveys have been conducted since the spill and are being compared to prespill surveys. In addition, the Trustee Council has funded sustained monitoring of common murre colony attendance and productivity in the Barren Islands every year since 1990. Other areas in the spill region have not been monitored since 1990 or 1991.

We recommend that, in addition to common murre sites on the Barren Islands, index colonies and at-sea areas outside Prince William Sound be monitored to determine marine bird population trends throughout the spill area. Areas that are monitored should be selected based on historical data so that recovery can be quantified; one such area is the south side of the Kenai Peninsula. We also recommend that forage fish abundance be monitored regularly in index areas throughout the spill area.

CHAPTER 3

ASPECTS OF SEABIRD BIOLOGY THAT BEAR ON VULNERABILITY TO AND RECOVERY FROM DISASTERS

Part A: Seabird Populations and Genetics

POPULATION DEFINED

The word "population" is commonly used to refer to any group of organisms, whether the group is part of a species inhabiting a local area, all members of a particular species, or all members of all species within a region. The biological definition of a population, however, involves a group of organisms that actually interbreed and share a common gene pool. A population differs from a species, which (according to biological definitions) is a group of organisms with the potential to interbreed. If dispersal of individuals from their place of birth to a breeding site is restricted, a population will include only the residents of a local area, but if dispersal is more widespread, the population may include all members of the species. Each population in turn may consist of two or more subpopulations or demes—groups of individuals that reside in a local area and that interbreed with members of other such groups. For seabirds, colonies and regions may or may not constitute separate populations, depending on levels of gene flow among sites. For example, colonies of thick-billed murres within the North Atlantic appear to constitute a single population and apparently are genetically isolated from colonies in the north Pacific (Birt-Friesen *et al.* 1992).

Some species appear to comprise "metapopulations"—networks of subpopulations that become extinct and are recolonized by immigrants from other sites over time periods ranging from a few generations to tens of thousands of years. Generally, subpopulations of a metapopulation are geographically isolated but exchange migrants on either a regular or intermittent basis (Levins 1969). The rate at which subpopulations disappear depends on conditions within the site as well as stochastic (random) processes, whereas the rate of colonization of new sites and recolonization of previous sites depends on dispersal rates. For example, subpopulations of muskoxen thrive, grow, and disappear over periods of a few generations, only to form again due to immigration of animals from neighboring sites (P. deGroot, Queen's University, pers. com.). Many species of gulls also appear to represent metapopulations.

In such species, at one point in time, individual subpopulations may constitute "sources" of immigrants or "sinks" for immigrants. The degree to which productivity exceeds mortality in certain subpopulations, which will then act as exporters or "sources" of breeders to other sites, will depend on the current state of the environment for these subpopulations. Some subpopulations will, at one point in time, not produce sufficient recruits to offset annual

mortality and will act as demographic "sinks," requiring immigration to prevent extinction. Whether a subpopulation represents a source or a sink at one point in time will depend on its demographic characteristics and on the current and local state of its environment. The status of a subpopulation is independent of population size or density: in theory, as little as 10% of a metapopulation can act as a source and still maintain up to 90% of the population in temporary sinks (see Pulliam 1994). For Pulliam (1994) many species could function as a network of source and sink populations. We emphasize that a given subpopulation can alternatively act as a source and a sink depending on fluctuations in the quality of the local environment.

In such a context, the goal of ecological restoration is to find ways to shift local subpopulations with demographic "deficits" to a state of demographic stability. For example, the control of the introduced raccoon on the colony of ancient murrelets on Limestone Island, British Columbia, has shifted the status of that particular colony from a sink, maintained by the neighboring predator-free Reef Island colony, to a colony that is again self-sustained (J-L. Martin, pers. com.).

IMPORTANCE OF DELIMITING POPULATIONS

For many reasons, understanding the dynamics and geographic limits of populations is essential for management and conservation:

- Species that consist of numerous localized populations may not naturally recolonize areas from which they are extirpated, either because levels of dispersal are too low (Cairns and Elliot 1987) or because migrants lack key adaptations to local conditions and thus do not survive or reproduce. In such species, populations that are decimated or extirpated through natural or anthropogenic disturbances may require human assistance for recovery (see Chapter 4). For example, common murres have failed to repopulate colonies in southern Quebec from which they were extirpated by egging and shooting in the late 1800s and early 1900s (e.g., Tuck 1961). In species that are essentially panmictic (populations characterized by random breeding) or that constitute metapopulations, subpopulations may recover from disturbance relatively quickly and without assistance. For example, double-crested cormorants have recolonized many sites from which they were exterminated by pesticides and human predation in the 1950s and 1960s (see Nettleship and Duffy 1995).
- Protection of healthy populations (i.e., current sources of immigrants) is critical to the longevity of species. Protection of populations that currently act as demographic sinks will be efficient only if we are able to identify and act upon the causes of the reproductive deficit. The removal of the human-caused perturbation(s) to the population (e.g., introduced predators, habitat destruction) is an efficient means of restoration and protection of such sink populations. However, if a population is currently acting as a sink, but with no apparent factor explaining its demographic deficit, the decision to restore that population will require additional, sometimes subjective, information. For example, the decision may depend on our ability to estimate the value of such a population as a potential demographic source in the future, or as a stepping stone in maintaining the functional integrity of the metapopulation.

- Knowledge about the geographic limits of populations is also important for determining the impact of natural and anthropogenic mortality. For example, a small, localized oil spill may have little impact on a large, geographically widespread population, such as the North Atlantic population of thick-billed murres, but may have a catastrophic effect on a small, demographically isolated population, such as red-legged kittiwakes on Buldir Island.
- Population data are required to determine the effective size of a population. The effective size is the number of individuals that actually contribute to the gene pool of the population, and may be one or two orders of magnitude lower than the census size due to unequal breeding success and population bottlenecks. For example, the North Atlantic population of thick-billed murres consists of approximately 2.5 million breeding pairs (Nettleship and Evans 1985), but appears to have an effective size of only ~10,000 females (Friesen *et al.* 1997).

Thus, although seabird colonies are attractive units for conservation and restoration due to their generally well-defined geographic limits, the population is the more appropriate unit toward which effort should be applied.

METHODS OF DELIMITING POPULATIONS

The geographic limits of a population can be delineated using one or more of four basic approaches.

Demographic Data

The direction and extent of gene flow among local populations can be approximated from demographic data such as dispersal information (e.g., Rockwell and Barrowclough 1987). Such information provides estimates of the geographic limits of a population, the extent to which it represents a metapopulation, and the identity of source and sink populations. Although dispersal data provide direct measures of contemporary movements, accurate estimates of gene flow also require information about lifetime fitness (the contribution of recruits to the next generation) of both resident individuals and migrants. Unfortunately, generation of the required data involves long-term mark-and-recapture studies (such as banding data) and is extremely labor-intensive, especially for seabirds that have secretive nesting habits, such as marbled murrelets. Furthermore, demographic data do not account for historical gene flow, which may be one of the most important forces defining populations, especially in species occurring at high latitudes because of the effects of Pleistocene glaciers. This is especially true of metapopulations: subpopulations may exchange few or no migrants over human lifespans, but may be connected by infrequent, mass movements of individuals. For example, band returns suggest that little or no dispersal has occurred among colonies of thick-billed murres in the North Atlantic during the past 100 years, but intensive hunting and egging at the colony at Ydre Kitsigsut (Greenland) appeared to result in movement of thousands of individuals to a neighboring site at Arsuk Fjord (Nettleship and Evans 1985). Furthermore, genetic data suggest that Atlantic colonies of thick-

billed murres were founded by large numbers of birds from one ancestral population following recession of the Pleistocene glaciers (Friesen *et al.* 1997).

Morphometrics

Morphological differences among animals from different subpopulations can provide a suggestion of the extent to which they represent genetically isolated populations. For example, Warheit (1996) was able to identify breeding populations of common murres from the north Pacific based on morphometric variation of skeletal elements. This approach has the advantage of being relatively quick and inexpensive, but rigorous analysis requires that birds be killed. Furthermore, it provides only an indirect measure of the amount of gene flow, may be confounded by environmental forces, and does not provide an indication of the extent to which a population represents a metapopulation or consists of sources and sinks.

Traditional Genetic Methods

Protein electrophoresis may also be used to estimate the geographic limits of populations, the extent to which a population represents a metapopulation, and the identity of source and sink sites. This approach has the advantage of being relatively quick and inexpensive, but it often necessitates that birds be killed and requires highly trained personnel and specialized laboratory facilities. Also, protein electrophoresis often is not suitable for measuring genetic subdivision in populations that breed at high latitudes due to low levels of variability (Evans 1987); because most of these populations were established following recession of the Pleistocene glaciers, insufficient time has elapsed for evolution of population-specific protein markers. Furthermore, due to the low mutation rates at most protein loci, classical electrophoresis measures rates and directions of evolutionary gene flow, which may be very different from contemporary values. Thus, populations may have been genetically isolated for tens of thousands to hundreds of thousands of years, but may have very similar electrophoretic profiles due to historical association. For example, protein data suggest that the North Atlantic population of thick-billed murres is essentially panmictic (Friesen 1992), even though band returns indicate that very little gene flow occurs among colonies today.

Recent Molecular Methods

Recent innovations in molecular and theoretical genetics, especially the polymerase chain reaction (PCR, or DNA amplification), provide potentially accurate and sensitive methods of measuring the direction and magnitude of gene flow among populations. PCR uses a modification of the fundamental cellular process that replicates DNA to generate millions of copies of specific target genes. The gene that is amplified is determined by the choice of primers—short pieces of DNA that match regions flanking the gene of interest and that provide initiation sites for DNA replication. Thus, PCR enables researchers to focus on genes with high levels of variability and has several advantages over previous methods of genetic analysis. Most importantly for the present purposes, PCR enables variation in DNA to be compared directly

among individuals from different sites (e.g., Kocher et al. 1989, Birt-Friesen et al. 1992, Quinn 1992, Wenink et al. 1994). Furthermore, it allows researchers to focus their attention on genes that have slower or faster mutation rates and that, therefore, provide measures of historical and recent levels of gene flow, respectively. Unfortunately, many existing PCR-based protocols are slow and laborious (e.g., analysis of DNA sequences or microsatellite loci), produce results that are not reproducible and are difficult to interpret (e.g., analysis of randomly amplified polymorphic DNA [RAPDs]), or provide data for one gene only (often mitochondrial DNA, which is not typical of the rest of the genome; Wilson et al. 1985). However, recent technical developments, such as denaturing gradient gel electrophoresis (DGGE) and the analysis of single-stranded conformational polymorphisms (SSCPs; Lessa and Applebaum 1993), provide rapid, inexpensive, and sensitive methods of comparing genetic variation among individuals. For example, use of single-stranded conformational polymorphisms or denaturing gradient gel electrophoresis in conjunction with targeted amplification of nuclear genes is a powerful new technique that combines the strengths of classical protein electrophoresis with those of cutting-edge DNA-based techniques (Palumbi and Baker 1994). However, even this method does not provide a measure of gene flow within the last few generations.

The most powerful approaches to delineating populations involve use of contemporary molecular techniques in combination with dispersal and demographic information, enabling accurate estimation of both gene flow and population structure. However, few such studies exist, and none on seabirds have been published.

OTHER USES OF POPULATION MARKERS

Morphometric and genetic markers have several applications for wildlife management and conservation, in addition to their uses for defining populations.

Preservation of Genetic Diversity

As a population declines, its genetic resources become depleted (Allendorf and Leary 1986, Gilpin and Soulé 1986). Initially this depletion involves loss of rare variants (alleles) from the population, but ultimately it includes loss of individual variation (heterozygosity). Both these effects decrease the species' ability to cope with environmental perturbations, such as climatic changes and disease epidemics (e.g., O'Brien and Evermann 1988, Vrijenhoek 1994). Eventually, a declining population may reach a threshold size below which inbreeding, deleterious alleles, and stochastic events may result in extinction. Loss of a population will result in reduction of overall genetic diversity, which may compromise the species' longevity. Thus, genetically differentiated populations must be managed as independent units; in contrast, if a species is essentially panmictic, protection of individual subpopulations may be less critical.

Environmental Impact Assessment

Morphometric and genetic variation can provide markers for monitoring the impact of human activities on sensitive or remote ecosystems, such as marine systems and the high Arctic. They also can enable the identification of breeding populations of animals killed during migration or winter. For example, many seabirds killed by oil spills are migrating or wintering; the "affected" zone, or the population of seabirds that was affected by an oil spill and that requires a restoration effort, may be very different and geographically distant from the actual spill zone (see Chapters 1 and 4).

Environmental Monitoring

Knowledge of the geographic limits of a population is required to identify appropriate reference or "control" sites from which to obtain baseline data for monitoring, restoration, and modeling (e.g., to determine if a seabird colony has recovered "normal" functioning following an oil spill; see Chapter 7). Demographic parameters may be very different for genetically isolated populations, even if they occur in ecologically similar areas.

Captive Management and Translocation

Delineation of populations is also essential for captive breeding and translocation, to prevent both inbreeding and crosses between genetically incompatible individuals (e.g., Hansen and Loeschcke 1994; see Chapter 9). For example, after a captive breeding program was designed to restore the dusky seaside sparrow by hybridizing the last remaining males with females of the morphologically similar Scott's seaside sparrow, genetic analyses indicated that Scott's seaside sparrow was not the most closely related subspecies to the dusky seaside sparrow. Therefore, Scott's seaside sparrow was not the most appropriate choice for captive breeding (Avise and Nelson 1989).

Determination of Population Uniqueness and Identification of Cryptic Species

Population markers can be used to determine if a colony is unique (e.g., endemic or genetically distinct), information that may then be used to rank conservation and restoration efforts (see Chapter 4). Most importantly, genetic data can lead to the identification of cryptic species—populations that are similar in appearance but that represent separate, noninterbreeding species. For example, genetic comparisons revealed that North American and Asiatic subspecies of the marbled murrelet actually represent reproductively distinct species that have been genetically isolated for five to six million years (Friesen *et al.* 1996). Therefore, these two taxa must be managed independently.

Improved Basic Knowledge for Management

Finally, demographic, morphometric, and genetic data can lead to improved general understanding of the dynamics of small, potentially endangered populations. For example, a correlation between genetic variation and disease resistance in small populations has been postulated but not directly demonstrated (O'Brien and Evermann 1988).

Part B: Seabird Ecology

INTRODUCTION

Seabirds are important and visible components of marine ecosystems worldwide. They are highly mobile animals capable of long-distance movements and are often found thousands of kilometers from land (Harrison 1983). They tend to breed on inaccessible coastal habitats, often in large, dense aggregations, and are often highly conspicuous victims of oil spills and other environmental disasters. These factors generate considerable public interest, thereby placing seabirds at the forefront of marine conservation issues.

In this section, we (1) evaluate aspects of the ecology and natural history of seabirds that make individuals and populations vulnerable to human impacts and (2) describe the utility of seabird life history studies in designing, implementing, and evaluating seabird restoration programs. It is beyond the scope of this chapter to review the relative importance of numerous conservation problems facing seabirds. For a recent synopsis of management issues for seabirds, see Duffy and Nettleship (1992) and references therein.

SEABIRD BIOLOGY AND VULNERABILITY

As marine organisms, seabirds appear to be more vulnerable to a variety of human anthropogenic factors than do other forms of marine life that have been studied. We use the term "vulnerability" to indicate both the number of individuals impacted and the capacity for populations to recover from perturbation. A number of biological characteristics relate to the susceptibility of seabirds to human impacts.

Sociality

Many seabirds are highly gregarious, often breeding in large colonies, some numbering in the hundreds of thousands (e.g., Wittenburger and Hunt 1985, Hunt *et al.* 1986). Sociality influences both the number of affected individuals and the capacity for seabird populations to recover. For example, coloniality and behavioral mechanisms promoting grouping behavior place large numbers

of individuals at risk in the event of an oil spill or other anthropogenic impact. From the standpoint of recovery, some seabird species require other individuals or a minimum group size to stimulate reproductive activities (the "Allee effect," Allee *et al.* 1949). Reed and Dobson (1993) review another phenomenon, known as "conspecific attraction," that relates to the recruitment of birds to colony sites occupied by individuals of the same species. Conspecific attraction in relation to foraging also places large numbers of birds at risk in small areas. Lastly, some seabird colonies may serve as "information centers" (Ward and Zahavi 1973, Wittenburger and Hunt 1985, Clode 1993). If colonies of certain species function as information centers, this phenomenon may limit the capacity of small groups to successfully find and exploit available resources.

Foraging Ecology

Most seabirds represent mid- to upper-trophic-level predators in marine food webs. Although seabird diet varies substantially by species, location, and time (by day, season, year, and decade), seabirds largely feed on moderate-sized marine zooplankton (e.g., free-floating copepods and euphausids), schooling pelagic fish, and ages 0 and 1 demersal fish. These prey are often patchily distributed in oceanic environments (Hunt and Schneider 1987). In response, seabirds concentrate on prey patches where they may be more (or less) susceptible to mortality factors. Seabirds are also visual-pursuit predators. Many species swim (or fly) through the water column in search of prey. This type of foraging behavior, demonstrated by alcids, penguins, and cormorants, places birds at risk of contact with both surface oil and fishing nets at depth.

Demographic Parameters

Knowledge of demographic traits of seabirds is essential for evaluating the vulnerability of seabird populations, rather than individuals, to anthropogenic impacts (cf. Wiens *et al.* 1984) and for planning and implementing restoration programs. Most seabird species are characterized by high adult survival probabilities (often greater than 80% per year), low levels of productivity (often less than 0.5 young/year per adult), delayed maturity and age at first breeding (often greater than five years), low recruitment probabilities (often less than 35%), variable annual breeding probabilities (often less than 100%), and low levels of dispersal. Combined, these life history traits predict a low rate of intrinsic increase and population recovery. Additionally, these characteristics indicate that if an impact increases the mortality rate of breeding adults or subadults, effects at the population level will be long-lasting and the time needed for recovery will be substantial. Conversely, if an impact affects reproductive success, effects on the population may be minimal, requiring little or no recovery (unless the impact is chronic). Moreover, although the number of individuals suffering mortality may be estimated (e.g., Piatt *et al.* 1990, Page *et al.* 1990), it is impossible to measure the effects of mortality on the population without prior information on the size, structure, productivity, and dispersal characteristics of the population in question.

SEABIRD BIOLOGY AND RESTORATION

Many of the factors that make seabirds vulnerable to anthropogenic impacts should also be considered when planning and implementing restoration programs. Below we consider some of the same aspects of seabird biology that have been mentioned above, but from the standpoint of population recovery.

Demographic Parameters

In addition to assessing vulnerability, demographic data provide a strong biological basis for planning seabird restoration projects. While we are not suggesting that demographic and life history data are a prerequisite for each and every restoration project, knowledge of life history and demographic parameters can vastly improve the design and evaluation of restoration programs. In addition, long-term (i.e., greater than ten years) life history studies, if available, provide information on (1) the range of parameter values that may be witnessed during restoration, (2) interdependencies of life history traits that may influence the outcome of restoration efforts, and (3) the traits that are most likely to promote population growth and persistence—i.e., the traits that, if manipulated, would have the greatest value as a restoration tool. To adequately investigate the factors that are most likely to influence population growth and recovery, data from demographic studies must be synthesized via stochastic population dynamics modeling and sensitivity analyses (e.g., Burgman *et al.* 1993; see Chapter 11 for discussion of modeling).

Below we review the seabird demographic parameters that are likely to be influential in seabird population dynamics, hence restoration. Our hope is to provide a "shopping list" of desired inputs for planning restoration via demographic analyses. As an introductory comment, we remind readers that seabird populations are age-structured (Furness and Monaghan 1987). By this we mean that when estimating and evaluating demographic parameters, one must consider how parameter values change with age. Gaston *et al.* (1994) provide a recent example for thick-billed murres. A review of age-specific life history traits is beyond the scope of this chapter, but recent reviews of this topic are available (Wooller *et al.* 1992, Forslund and Part 1995). Demographic parameters that should be considered when designing a restoration program include the following.

Adult survival (from breeding age to death)

Early views considered this a time-constant parameter. There is now considerable evidence that adult survival in seabirds varies from year to year and decade to decade, and, furthermore, that changes in adult survival are associated with corresponding population dynamics (Coulson and Thomas 1985, Harris 1991, Croxall and Rothery 1991, Hatchwell and Birkhead 1991, Sydeman 1993). Given the generally high survival of most seabirds, enhancing adult survival may be difficult to accomplish, but if possible might provide an effective means of promoting population growth and recovery. However, aside from managing food resources (by limiting fisheries), controlling predators, and reducing net fisheries bycatch of adult birds, techniques to enhance this parameter have yet to be developed.

Juvenile and subadult survival (from fledging to age 1, age 1 to breeding age)

Knowledge of these life history parameters is fragmentary, at best, for nearly all seabird species and, where known, often involves species (i.e., large larids) that are not in need of restoration efforts. Little is known about whether juvenile or subadult survival varies from year to year, if temporal fluctuations are as great (or greater) than variation in adult survival, and if there is a strong relationship between fluctuations in adult and subadult survival. Moreover, estimates of juvenile and subadult survival based on capture/recapture (or capture/resighting) methods are subject to biases due to dispersal (this may be less of a problem for studies of adult survival because of high breeding philopatry). Nonetheless, indications are that both juvenile and subadult survival have important, if not critical, roles in the population dynamics of many seabirds (Buckley and Downer 1992, Nur *et al.* 1994). As an example, Hatchwell and Birkhead (1991) concluded, albeit indirectly, that a change in juvenile or subadult survival must have been the major demographic factor explaining why the Skomer Island common murre population grew in the 1980s but not in the 1970s. As suggested above, enhancing survival could provide an effective means of restoring seabird populations, although techniques for such an undertaking for adults, let alone juveniles and subadults (which spend less time at a colony site), have yet to be developed. However, given that subadult and juvenile survival is often considerably lower than adult survival, there is greater room for improvement, which may then promote population recovery.

Reproductive success

Substantial data are available on this parameter, although it may be one of the less important parameters in relation to understanding population dynamics and planning restoration for seabirds. It is well established that reproductive success varies from year to year and from decade to decade, and that much of this variation is related to marine climate and food availability in some systems (e.g., Furness and Monaghan 1987, Ainley *et al.* 1995b). For example, a decline in North Sea herring stocks was associated with declines in black-legged kittiwake reproductive success and population growth rates (Coulson and Thomas 1985). Whereas a major change in reproductive success may presage population growth or decay, it does not follow that all fluctuations in reproductive success are similarly influential. For example, consider species with single-egg clutches (e.g., procellariiforms and many alcids): reproductive success is high relative to the species' capacity for productivity—i.e., generally 65-80% of all eggs result in free-flying fledglings in these species. Consequently, boosting reproductive success beyond levels that are already relatively high may be difficult and will not be an effective means of restoring populations. Conversely, when reproductive success is low relative to the potential success rate, restoration efforts focused on enhancing productivity will be more effective. Another consideration in relation to enhancing reproductive success might be that for seabird populations, one or two strong cohorts may sustain recruitment for many years (e.g., Ainley *et al.* 1990). In this case, improving reproductive success might again accomplish little with respect to population growth.

Breeding probability

This parameter is important to restoration because it may be more easily manipulated. Overall breeding probabilities may be considered as two separate components: (1) the probability of breeding among experienced breeders and (2) the probability of breeding among individuals entering the reproductive population for the first time. Good information on these parameters usually requires monitoring banded individuals through time; thus, it is generally scarce.

The breeding probability of experienced breeders appears to vary substantially among species and even within species. "Skipping" (i.e., nonbreeding among experienced breeders) reflects both individuals present at the colony but not attempting to breed and individuals absent from the colony. Because skipping birds are absent or inconspicuous, the extent of skipping is probably underestimated. In the short-tailed shearwater, 12% of adults did not attend the colony in a given year and 19% maintained burrows but did not lay an egg (Wooller *et al.* 1989). Similarly, in the Manx shearwater, breeding probability was estimated to be 80% (Brooke 1990). Aebischer (1986) attributed a population crash in the European shag on the Isle of May, Scotland, to extensive lack of breeding by experienced adults.

The probability that a sexually mature bird will enter the reproductive population for the first time also varies greatly among and within species. When competition for space or mates is intense, breeding probability among potential new recruits may be low. For example, few western gull females on Farallon Island, California, start breeding at age 4, when they are physiologically capable of producing eggs. Most start at ages 5, 6, or 7, when they are more competitive for territory-holding males (Spear *et al.* 1995a). Moreover, 4- and 5-year-old male western gulls are more likely to recruit in years when food is abundant (Spear *et al.* 1995a). A similarly wide range of age at first breeding has been reported for common murres by Halley and Harris (1993) and other species (Bradley and Wooller 1991). Because variability in the age at first breeding is high, we consider activities that alter the factors that affect the age at first breeding to be a potentially powerful restoration tool. For many seabird species, a pool of nonbreeders provides a potential source of recruits to be tapped. As an example, a catastrophic red tide mortality of breeding shags on the Farne Islands, England, allowed many new individuals to be recruited in subsequent years (Potts *et al.* 1980). Similarly, if territories, nest sites, or mates are made available through restoration activities, population growth and recovery may be facilitated (see Chapter 9).

Age of first breeding

The distribution of age at first breeding is not normal; most individuals initiate breeding earlier in life, and few breed for the first time in the various older age classes. For this reason we are more interested in minimum or modal age of first breeding rather than average age of first breeding; the latter reflects the tail end of the distribution (individuals who recruit only late in life), and factors influencing this tail have been discussed above. Certainly there is interspecific variation in age of first breeding, for example, with respect to body size (Croxall and Gaston 1988, Gaillard *et al.* 1989, Nur 1993) and longevity (Bradley and Wooller 1991). In addition, there appears to be variation within species as well. For example, common murres on the Isle of Canna, Scotland, were observed breeding for the first time at ages 3 and 4 (Swann and Ramsay 1983). Only a single

individual murre from the Farallon Islands, California, has been observed breeding at age 3, with most recruiting at ages 5 to 7 (W. Sydeman, unpubl. data). At Skomer Island, Wales, common murres bred at four to six years (Birkhead and Hudson 1977). As with breeding probabilities, age of first breeding is likely to reflect reproductive opportunities. For example, the colony on the Isle of Canna was a fast-growing colony, presumably with many available breeding sites.

Immigration and emigration

We have very little good information on this parameter for seabirds. Emigration is inherently difficult to study because, by definition, individuals are leaving the study colony, and death is hard to distinguish from emigration. The number of immigrants can, in some cases, be quantified, but the pool from which they come is much harder to identify. Nonetheless, a review of population recovery of marine birds indicated that immigration has played a key role in many growing or recovering populations (Nur and Ainley 1992). Immigration can play a role in restoration in several ways: when establishing a new colony (or re-establishing an extirpated colony), all individuals are, at first, immigrants; and among growing colonies, immigration will often reinforce population growth. On the other hand, the establishment of a new colony may siphon off individuals from an established colony, thereby leading to no net change in the larger metapopulation (see discussion below and in Chapter 3a).

The majority of seabirds were once thought to be intensely philopatric (Bradley and Wooller 1991), but more recent studies have indicated that this may not be a general pattern. For example, Porter and Coulson (1987) published an accounting of factors affecting philopatry and recruitment in kittiwakes. They found that about 11% of each cohort return to breed at their natal colony and noted that this proportion was time-constant (1952-84). Coulson and de Mévergnies (1992), in a regional survey of kittiwake colonies in Britain, indicated that roughly 35% were philopatric, while 45% emigrated. In Atlantic puffins, Harris and Wanless (1991) suggested that approximately 50% of young emigrated, revising earlier conclusions (Harris 1984) that the majority of young were philopatric. Halley and Harris (1993) showed that during the prospecting period, immature common murres visited colonies close to their natal colony more frequently than colonies farther away. Finally, Bradley and Wooller (1991) concluded that recruitment and philopatry in long-lived birds were influenced by many factors including sex, age, food and nest site availability, population size and density, and expected longevity. In conclusion, if intercolony movement and recruitment occurs rarely, this behavior will not have major demographic implications. Nevertheless, immigration and emigration rates should be accurately estimated because this parameter may have substantial implications for restoration of small, incipient seabird colonies.

Coloniality and Density-Dependence

As discussed above, many seabirds are gregarious, reproducing in large colonies. The relationship between coloniality, density, social behavior, and life history parameters is fundamental to seabird restoration. Colonial breeding in seabirds may or may not act to constrain or promote population growth and recovery if density-dependent population-regulating mechanisms are operating. By

this, we mean that fecundity, survival, or recruitment is a function of population size (or an equivalent such as population density), either negatively (i.e., increasing population density reduces survival, fecundity, or recruitment) or positively. The concept of negative density-dependence is ubiquitous in both the scientific and lay literature, and yet the evidence for negative density-dependence in seabirds is not robust. This is not to say there is no evidence (reviews are provided by Birkhead and Furness 1985, Croxall and Rothery 1991), but rather that direct evidence is often lacking. An example of one prevalent view is provided by Baker *et al.* (1990), who argued that catastrophic mortality of marine birds (with specific reference to the *Exxon Valdez* oil spill) was actually "good" for these species, as it served to reduce intraspecific competition.

One example of possible negative density-dependent reproductive success is provided by Hunt *et al.* (1986) on five seabird species nesting on the Pribilof Islands, Alaska. They compared two colonies, one very large (2.5 million seabirds) and one large (250,000 seabirds). In four species, chick growth at the very large colony was reduced compared to the large colony. However, there was no significant difference in reproductive success between the colonies for any of the species. These results may suggest the operation of negative density-dependence at very high population levels, but they do not demonstrate the action of similar density-dependence at intermediate or low levels of density.

On the other hand, a positive correlation between reproductive success and breeding density may be important in some species or populations, especially at low densities. For these species or populations a technique that increases breeding density would be a worthwhile restoration tool. Allee *et al.* (1949) first recognized that the population dynamics of social species may be positively density-dependent when population size is low. They postulated that mating success, reproductive success, and recruitment may be limited by a critical density that must be exceeded before a resource (habitat or prey) can be properly exploited. With respect to seabird restoration, this implies that a threshold group size is needed to establish productive colonies. An example of the Allee effect is provided by wedge-capped capuchin monkeys: as group size increased from 5 to 30 individuals, so too did per capita production of young (Robinson 1988). In the common murre, there is good evidence that reproductive success increases with density at the colony (Birkhead 1977), apparently due to better protection from predators. Hudson (1985) also considered the implications of positive density-dependence for murre population dynamics; he modeled a scenario in which an oil spill (or similar catastrophic mortality) could lead to long-term population decline, eventually resulting in population extinction. Whether Allee effects occur in other seabird populations is not well known, but it seems likely. Thus, in planning seabird restoration, one would not want to disturb colonies in which density was close to or below an Allee threshold. Furthermore, as minimum colony size and density (numbers per unit area—i.e., the Allee threshold) have not been established for most seabird species, a review of available data would be enlightening with respect to designing recovery programs.

Other density-dependent aspects of coloniality in seabirds (e.g., breeding phenology and synchrony, productivity) are also important and bear upon issues of colony establishment and population growth. First, we distinguish between spatial and temporal aspects of coloniality. Coloniality itself represents differences in spatial dispersion. In contrast, chronology and synchrony are reflective of temporal clustering of pairs within colonies. Information on dispersion within seabird colonies is

an important element of seabird breeding ecology. For example, data on the spatial configuration of murre colonies may be useful in deploying decoys that simulate colony structure. For some species, this may be important to minimize predation. Furthermore, Coulson (1968) recognized the importance of spatial dispersion in relation to center-edge effects and productivity in kittiwakes. Others (e.g., Birkhead 1977) have shown how birds at the edge of colonies were more likely to suffer predation than birds in the interior. Spear (1993) also demonstrated that when the spatial structure of a murre colony fails (in this case due to El Niño), all pairs were likely to suffer predation by gulls. Restoration activities that attract birds to the colony center, while simultaneously providing some degree of protection on the edge, might be most effective if predation is limiting population recovery. Unfortunately, little is known about mechanisms that might attract birds to one area or another within a colony.

Metapopulation Dynamics: Sources and Sinks

The importance of "sources" and "sinks" in relation to vertebrate population dynamics has only recently been recognized (Pulliam 1988, Pulliam and Dunning 1994). Buckley and Downer (1992) have investigated some aspects of this phenomenon in seabirds. A sink population is one in which the current local production of recruits is less than the mortality of established individuals, and therefore the population is not currently self-sustaining; it can be sustained only by immigration from other, currently more productive populations. A source population is productive enough so that an excess of potential recruits is produced relative to mortality. This can lead to growth of the source population and emigration of recruits to other, possibly sink, populations. A network of source and sink populations can be joined by immigrants and emigrants; this may be referred to as a "metapopulation" or "network of populations." An important implication of the source-sink paradigm is that population dynamics cannot be understood at the level of a single population or colony (which may be either a source or sink) but rather must be approached at the level of the entire network or landscape of populations. Pulliam and colleagues further demonstrated that a single source (i.e., "mother") population can effectively maintain a large number of sink populations; in fact, most of the individuals in a metapopulation may be breeding in sink populations, and yet the overall network of source-sink populations may be self-sustaining. In relation to conservation and restoration efforts for seabirds, projects should be directed at (1) maintaining the dynamics of source populations and (2) altering the dynamics of sink populations. If sink populations can be manipulated to the point where they also produce excess recruits, the overall stability and persistence of seabird metapopulations will be improved. Moreover, efforts directed at current sink populations without regard for the current local source population may be doomed to failure.

Habitat Selection

Seabirds select colony sites based upon a number of factors: climatic characteristics, oceanographic conditions of local foraging grounds, and habitat features (reviewed by Buckley and Buckley 1980 and Kaiser and Forbes 1992). Much information on suitability of nesting and foraging habitat can

be obtained from the presence and activities of conspecifics and other breeding seabirds (Kharitonov and Siegel-Causey 1988, Reed and Dobson 1993).

Once a colony has been formed, population growth and recovery may be facilitated by providing artificial nest sites (e.g., nest boxes or nesting ledges). This type of effort will be most effective if nest sites are limited and other factors (e.g., food availability) are not currently limiting the population. In general, seabird populations are not usually limited by a lack of available nest sites (Furness and Birkhead 1984, Birkhead and Furness 1985, Cairns 1989, Croxall and Rothery 1991), although in some species evidence in support of a "habitat saturation" hypothesis of population regulation is compelling (Manuwal 1974a, Potts *et al.* 1980, Duffy 1983, Porter and Coulson 1987, Coulson 1991). Aside from placing nest boxes for cavity- and burrow-dwelling seabirds, habitat manipulations have not been regularly attempted in the Northern Hemisphere, presumably because it is expensive, except where direct economic benefits have been realized (e.g., nesting platforms constructed for guano harvest in Peru and Africa). Furthermore, this type of restoration requires data on species-specific habitat use (e.g., Bédard 1969b, Grant and Nettleship 1971, Nettleship 1972, Manuwal 1974a, Birkhead 1977, Vermeer *et al.* 1979, Gaston and Nettleship 1981, Birkhead and Nettleship 1987). However, detailed habitat information is exceedingly rare. Moreover, the value of artificial habitat, including nest boxes, as a restoration tool has not been adequately evaluated. Although nest boxes will be used by a variety of crevice-nesting seabirds, including procellariids and alcids (Ainley and Boekelheide 1990, Hester and Sydeman 1995, Podolsky and Kress 1989b), an evaluation of artificial habitat use through time and comparisons of the demography of pairs nesting in or on artificial structures versus natural ones is needed.

CONCLUSIONS

In general, a great deal more basic life history research is needed to answer some fundamental questions pertaining to seabird ecology and restoration. Life history and demographic analyses have high priority in terms of seabird restoration, as well as in assessing the impacts of oil pollution on seabird populations (Wiens *et al.* 1984). In particular, analyses of demographic parameters can be used to understand population dynamics and quantify the potential results of differing restoration options (Burgman *et al.* 1993). We recommend increased attention to life history characteristics in seabirds that appear to be crucial for understanding population dynamics. *Demographic parameters that appear to be most important in promoting population growth include adult survival, juvenile and subadult survival, and the breeding probability of first-time breeders.*

The status of a population and the success of restoration efforts ultimately depends upon the subadult, prebreeder population as well as on the size and status of breeding populations. Nonbreeding individuals make up as much as 45% of all seabird populations. Consequently, greater effort to monitor prebreeder population size and to identify dispersal and recruitment characteristics, as critical demographic processes, is recommended. This work will be challenging because nonbreeding individuals are difficult to monitor or study. Nonetheless, some fine examples of this work are available (Harris and Wanless 1991, Coulson and de Méexergnies 1992).

Specific questions concerning recruitment that require investigation in relation to many, if not most, restoration programs include the following.

(1) What proportion of young recruit into natal colonies/locations?

(2) If emigration occurs, what is the typical range of gene flow?

(3) Which factors are responsible for differences in philopatry and emigration, and are there individual-, cohort-, year-, and colony-specific effects?

(4) How do young birds select the colony sites to which they emigrate?

(5) What behaviors by conspecifics are attractive to recruits, and what behaviors are practiced by recruits during dispersal?

Molecular genetic (e.g., mtDNA and microsatellite) analyses may be an effective means of addressing some of these issues, which, in the past, have been addressed only through long-term banding and monitoring studies (see Chapter 3a). Meanwhile, it is also important to recognize that individual colonies and populations may not be isolated, and that information about metapopulation and source-sink dynamics is needed in order to understand and predict the dynamics of seabirds and to plan restoration. Many population models have neglected the important effects of emigration (Dauchin and Monnat 1992, Nur *et al.* 1994, Beissinger 1995, but see Buckley and Downer 1992). In order to succeed, restoration programs must evaluate and incorporate dispersal.

From the standpoint of both life history and behavioral ecology, a great deal more could be learned concerning the role of density-dependence. A review of the minimum group size required to establish a breeding colony (as well as the size needed for successful reproduction) would be an excellent way to initiate such investigations. Additionally, a related question that needs to be answered is: what are the minimum viable population sizes for colonial seabird species? One way to investigate this question would be to consider the effect of density on reproductive success and evaluate whether the relationship appears to be monotonic, a step-function, or a parabola. With regard to the Allee effect, some factors that appear to affect surface-nesting seabirds (e.g., predation) would appear to promote threshold or step-function colony-size relationships. However, even seabirds that are protected from aerial predators may require a critical mass in order to stimulate mating and territorial behavior. For example, nocturnal alcids and petrels may use vocalizations to communicate between conspecifics (Podolsky and Kress 1989b). Restoration ecology of seabirds would benefit from empirical data establishing the conditions when these density-dependent effects may occur. Moreover, minimum viable population or colony size represents a metric for evaluating the effectiveness of restoration activities. Certainly, any restoration project that fails to generate a minimum viable population size for a given population or colony should be considered unsuccessful.

CHAPTER 4

IDENTIFICATION OF SPECIES OR POPULATIONS REQUIRING RESTORATION

INTRODUCTION

The purpose of this workshop was to identify and to discuss oil spill-related *restoration options* for seabirds injured by EVOS. Our intent was not to discuss or to evaluate the procedures used by the Trustee Council (or any other oil spill trustee council) to identify which seabird species or populations were injured by the spill and, therefore, may require restoration efforts. However, the workshop steering committee decided that to address the concept of restoration *goals*, the workshop needed to discuss the kinds of data that should be collected to adequately assess spill-related injury. Furthermore, the committee found that clearly stated and objective *a priori* criteria to identify which injured seabird species or populations required direct restoration were not established by the Trustee Council. This chapter suggests criteria or guidelines to establish oil spill-related seabird injury. In presenting these guidelines, we also point out that certain kinds of baseline data need to be available prior to a spill and certain kinds of data need to be collected during a spill to adequately evaluate injury.

TRUSTEE COUNCIL'S CRITERIA FOR INJURY AND RESTORATION

The Trustee Council (1994b) listed three types of injury to biological resources: (1) mortality, (2) sublethal effects, including effects to gametes and larvae, and (3) habitat degradation. Although the Trustee Council stated that the most serious injuries result in "large population declines" (Trustee Council 1994b:29), a spill-related effect does not always have to produce a *measurable* decline to a population to be considered an injury. The Trustee Council listed four reasons why an injury may not result in a population decline, with only one reason related to the severity of the injury (i.e., the injury was not severe enough to produce a population decline; Trustee Council 1994b:30). Finally, although the Trustee Council stated that any injured resource can be considered for restoration, it focused on those species or services that have not recovered (recovery based on monitoring activities). It also decided that "priority will be given to restoring resources and services which have economic, cultural and subsistence value to people living in the oil spill area" (Trustee Council 1994b:13).

WORKSHOP RECOMMENDATIONS

The process by which species are identified as candidates for restoration activities following oil spills should include (1) an assessment of the immediate injury and compilation of baseline data,

(2) the use of *explicitly stated criteria* to determine if the population's injury is significant, and (3) the use of *explicitly stated criteria* to determine if an injured population should qualify for restoration. In other words, in order for a population to be a candidate for restoration, it must pass through injury and restoration "filters." We provide the following lists: (1) essential baseline data needed to evaluate injury, (2) initial injury assessment activities, (3) criteria to define significant injury, and (4) criteria to establish if a population requires restoration. We list issues in approximate descending order of priority. For example, total population size (pre- and postspill) and total mortality from the spill are the most important data to use in assessing injury to populations and the biological significance of mortality.

Biological Baseline Data Prior to Spill

Total population size

We emphasize that an estimate of total population size includes information on the relative proportion of the population that is at sea and at colonies or breeding sites at any given time, and includes all age classes. Furthermore, this information is essential in estimating injury and can be used for setting restoration goals, although achieving a prespill population level may not be the best goal for a restoration project (see Chapter 6).

Index plots

Index plots should be established in areas with a relatively high potential for oil spills or other disasters (this relates to the sensitivity maps discussed in Chapter 2a). Ideally, long-term and ongoing phenology and productivity, annual adult mortality, recruitment, and dietary information should be available as baseline data, to which spill effects can be compared and evaluated. The workshop recognized, however, that collecting such data can be constrained both logistically and financially. Estimating annual adult mortality is especially difficult in that it requires a long-term and concentrated banding effort and data on (or assumptions about) emigration. Because of this difficulty, we recommend that sources of mortality (e.g., gillnet bycatch) be identified as part of baseline data.

At-sea areas

To adequately estimate total population size and to help determine which populations may be affected by an oil spill, it is important to have a general idea of where individuals from specific populations forage and to determine if these foraging areas are age- and/or sex-specific. These data should be available for both the breeding and nonbreeding periods. As with long-term index-plot data discussed above, these data may be difficult to collect due to logistical, methodological, and financial constraints.

Hydrocarbon levels and blood parameters

Seabirds affected by oil spills may also have been exposed to background hydrocarbon contamination and, prior to the spill, may have experienced physiological stress affecting their blood chemistry. To adequately evaluate the sublethal or chronic effects of an oil spill, data on background hydrocarbon levels and blood chemistry parameters (e.g., total protein, packed cell volume) should be available.

Biological Data Collected During Spill

Total population size

As with the baseline data, total population size after a spill is a function of the numbers of birds at colonies or breeding sites and the numbers at sea. Attendance or status at index plots should be assessed as soon after the spill as logistics permit. Data on the total population size (and proportion of that population at sea) will help assess total mortality from the spill by modeling spill trajectory, at-sea distribution, and number (and location) of recovered oiled carcasses.

Beached bird surveys and modeling

The number and composition of dead and oiled beached birds recovered by spill responders are among the most essential data collected during the injury assessment phase of spill response. The number of birds recovered for each species, the geographic area from which they were recovered, and the date of recovery, along with information on the spill trajectory, distribution of birds at sea, and real-time drift experiments, are used to estimate the total mortality from the spill (see Ford *et al.* 1996 for EVOS example). Other data that should be collected from the carcasses are (1) age and sex composition of each species, to estimate the *demographic impact* of the spill; (2) genetic and/or morphometric analyses, to help determine the *origin* of birds; (3) the degree of oiling; and (4) the stomach contents, to determine *diet*.

Hydrocarbon levels and blood parameters

Hydrocarbon levels of dead birds and hydrocarbon levels and blood parameters of injured and recovered birds will help evaluate the sublethal or chronic effects from the spill.

Seabird species list

The seabird species list will help guide the injury assessment by determining which species should be emphasized in data collection activities. Furthermore, the vulnerability of each of these populations can be assessed using historical data on their distribution and real-time oil spill trajectory data.

Criteria for Injury Determination

The criteria used to determine whether a species or population has been injured should be similar to the biological criteria used in the Ramsar Convention to identify wetlands of international importance to water birds. Under this approach, a species or population that meets any *one* of the following criteria[3] should be "flagged" for concern about possible injury. This filter is conservative, and seeks only to identify all species and populations that merit further consideration for possible restoration activities. It can help focus restoration planning for seabirds because it will remove many species from further consideration while highlighting those that may have suffered serious injury.

Species and population criteria

- Species is threatened, endangered, or globally rare.
- One percent (or some other percentage, determined *a priori*) of the breeding population in the general area was killed by the spill.
- A colony or population was extirpated by the spill.
- A population's range was reduced by the spill (i.e., extirpation of colony at edge of range).
- Species has a significant biological or ecological role (e.g., "keystone" species).
- Species is of socioeconomic importance (e.g., tourism, food supply).
- Breeding success or some other population parameter is depressed (the absolute change in the affected parameter must be beyond the 95% confidence interval for a control or baseline population).
- One percent (or some other percentage, determined *a priori*) of the population's food supply or habitat was injured by the spill.
- The spill caused a significant hydrocarbon load affecting productivity or survival (the absolute change must be beyond the 95% confidence interval for a control or baseline population).

Finally, a species that has a high vulnerability index (*sensu* King and Sanger 1979) and is present in the spill area should be included. Thus, natural resource trustees should assume at this stage of the analysis that such a population experienced injury even if no carcasses were recovered. Such a population might be removed from further consideration in the steps we recommend below.

[3] Some of the criteria assume that prespill baseline data are available. The workshop did not address the issue of what should be done in the absence of such data, although this clearly represents a critical issue in the oil spill damage assessment process. We emphasize the importance of collecting prespill baseline data and developing contingency plans for assessing injury to those resources for which there is little or no potential for collecting baseline data.

Criteria for Restoration Determination

Once a species or population has been flagged as a possible candidate for direct restoration activities using the criteria listed above, it can be established as a priority candidate for restoration if it meets the following criteria. These criteria can be used to establish a priority list of seabirds for restoration activities.

1. The loss was biologically significant.

or

The injured population had high socioeconomic or public interest value.

and

2. Restoration is practical in terms of methods, logistics, and cost.

Biological significance

To determine if a specific mortality is of biological significance, resource managers and trustee councils should consider, at a minimum, data collected during the injury assessment, the quality of those data, whether a significant percentage of a species was killed during the spill, demographic information about the populations that were affected (e.g., sex and age), and the population's "distinctiveness" (i.e., endemic, rare, genetically distinct).

Socioeconomic or political value

If the injury to a species or population is not biologically significant, a secondary factor to consider is the population's socioeconomic value. Colonies that are regularly visited by tourists (i.e., colonies with high educational value), are important research sites, or are used by indigenous people for subsistence, for example, should be considered as candidates for restoration.

Feasibility

If a species or population is identified as a candidate for restoration because it is biologically significant or has high socioeconomic or public interest value, restoration managers must still consider the practicalities of restoration. Each species or population may have only a small set of possible restoration options or techniques available. These options or techniques must be logistically feasible, affordable (based on the terms of the settlement and the policies set forth by

the Trustee Council), and, most importantly, have a high probability of producing a positive effect for the population. In considering the practical aspects of restoration, trustee councils must also determine the probability that a population is likely to recover naturally without implementation of hands-on activities.

CHAPTER 5

RESTORATION DEFINED

INTRODUCTION

In the following chapter we describe the various definitions of restoration and rehabilitation, and discuss the categories of activities considered by the U.S. government as being part of restoration (restoration, rehabilitation, replacement, enhancement, and acquisition of the equivalent resources). We also outline what we believe to be the most appropriate ecological units (e.g., colony, population, metapopulation) on which to conduct restoration and describe the potential time frames restoration might require.

DEFINITIONS

There is confusion in the literature over the definitions of restoration and rehabilitation. This confusion stems in part from regulations governing damage assessment and "restoration" activities in the United States, in which the definitions used are slightly different from those in the ecological literature (cf. National Research Council 1992). Below we discuss restoration and rehabilitation, and define them for the purpose of this report.

Restoration

The Society for Ecological Restoration defines restoration as "the process of altering a site intentionally to establish a defined, indigenous, historic system. The goal of this process is to emulate the structure, functioning, diversity, and dynamics of the specified ecosystem." A similar goal was suggested by Simberloff (1990:40), who proposed that restoration would be successful "if it produces a system whose structure and function cannot be shown to be outside the bounds generated by normal dynamic processes and ecosystems."

In the United States since 1987, restoration of seabird populations injured[4] by oil spills has been guided by the Natural Resource Damage Assessment (NRDA) regulations (43 CFR Part 11) of the Department of Interior. These regulations were enacted as a requirement of the Comprehensive Environmental Response Compensation and Liability Act (CERCLA)and applied to natural resource injuries resulting from spills of oil and hazardous substances, generally including injuries resulting from the *Exxon Valdez* oil spill.

[4] Injury can be defined in reference to individual organisms, populations, or any other higher-order ecological unit. A resource (e.g., individual, population, species) is injured when it has been detrimentally affected. For example, an individual bird is injured when it becomes oiled. In this case, an extreme injury would be death. A population is injured when some aspect of its demographics, abundance, distribution, or genetic variance is altered. An extreme injury here would be local extinction.

The National Oceanic and Atmospheric Administration has recently released Natural Resource Damage Assessment regulations (see *Federal Register* Vol. 61, No. 4, January 5, 1996) under the Oil Pollution Act of 1990 that replace the Department of Interior regulations for natural resource injuries that result from oil spills (NOAA 1996). The Department of Interior regulations are still applicable to natural resource injuries resulting from spills of other toxic substances. Both sets of regulations recommend approaches for use by natural resource trustees in assessing and quantifying injuries, estimating associated damages, and selecting restoration alternatives. The definition of restoration is essentially the same in both the Department of Interior and NOAA regulations (NOAA 1996:505):

> Restoration means any action (or alternative), or combination of actions (or alternatives), to restore, rehabilitate, replace, or acquire the equivalent of injured natural resources and services. Restoration includes: (1) primary restoration, which is any action, including natural recovery, that returns injured natural resources and services to baseline; and (2) compensatory restoration, which is any action taken to compensate for interim losses of natural resources and services that occur from the date of the incident until recovery.

In other words, the regulatory definition of restoration is broad, using "restoration" as a general term for restoration, rehabilitation, replacement, and acquisition of the equivalent of the injured natural resources and the services they provide. Trustees are given considerable flexibility to select appropriate restoration alternatives from the four restoration categories, depending on the location of the spill, the nature and extent of the injuries, the species involved, and the amount of trust funds available. Thus, direct restoration of an extirpated colony may be selected as the most appropriate way to deal with seabird injuries at one location, whereas another technique (e.g., removal of exotic predators or herbivores from an island to enhance nest success) may be selected as most appropriate to restore the same seabird species under a different set of circumstances at another spill location.

The Trustee Council adopted a definition of restoration that is essentially the same as the NOAA/Department of Interior definition, except that it makes reference to "pre-spill condition" (Trustee Council 1994a, 1994b) rather than baseline (see below). However, while replacement and acquisition are not defined in the NOAA rule, the Trustee Council (1994b:4) uses these terms synonymously, and explicitly defines them as "compensation for an injured, lost or destroyed resource by substituting another resource that provides the same or substantially similar services as the injured resource."

For the purpose of this report, we define restoration as any action taken directly or indirectly to manipulate a system for the repair or recovery of injured populations, colonies, or communities. *We emphasize that restoration is an action taken by humans; the natural recovery of an injured resource without some form of human input is not here regarded as restoration.* That is, restoration is something that people do for seabirds, not what seabirds do for themselves.

Rehabilitation

Atkinson (1994) defines rehabilitation as the removal of the affecting or disturbing agent, without the direct manipulation of any population demographic factor. Conducted by itself, rehabilitation promotes natural regeneration (recovery). In other words, if the population decline at a seabird colony was the result of an oil spill or gillnet activity near that colony, rehabilitation would involve the removal of the oil or gillnets. Likewise, according to this definition, if seabird populations have declined at a colony as the result of predation from exotic predators, the removal of these predators would constitute rehabilitation. If, following removal of the predators, the colonies eventually returned to a baseline or "predisturbance" condition, the system would have recovered. Namely, the system was restored through a process of natural recovery aided by rehabilitation. However, if colonies of the original species did not return despite removal of predators, and no further action was taken, the system would not have recovered and restoration would not have progressed beyond rehabilitation. In other words, rehabilitation may constitute an incomplete form of restoration if natural recovery does not occur.

The U.S. agencies dealing with oil spills generally view rehabilitation as the procedure by which injured individuals animals are treated, nursed back to health, and released into the population. This process involves retrieving live oiled animals (in particular, seabirds) and then washing them, feeding them, and, if necessary, conducting medical procedures on them. Although these activities may seem very different from removing exotic predators from islands, they can be viewed as a different stage in a similar process. With the removal of oil, the focus is on the individual animal, whereas in the ecological definition the focus is on a group of animals such as a seabird colony, population, or community. The ecological effects of the two kinds of rehabilitation may be identical. Just as the removal of a predator may not by itself restore all of the original extirpated species, the removal of oil residues from individual birds is unlikely to restore the original population; many oiled birds will not be recovered, and others may die during or soon after rehabilitation attempts (see Chapter 9f).

If the intent of restoration is to repair an injured population, then removing oil and treating individual animals (rehabilitation) may be a first step toward that repair or recovery (see Chapter 9f for comments about this method). Likewise, removing exotic predators from islands is also an activity that can repair an injured system. In this report, we generally do not distinguish rehabilitation of individuals from that of populations since both can contribute to the process of restoration; however, our discussion of rehabilitation in Chapter 9f does concern only the rehabilitation of individuals (i.e., the rescue and cleaning of oiled birds).

Baseline

The goal of restoration is to return a population (or colony, metapopulation, etc.) to a predetermined level that existed before a defined disturbance event. In Chapter 6 we discuss restoration goals and define the point at which restoration is no longer needed (i.e., recovery). However, it is important to outline here what is meant by baseline.

The concept of baseline is an important part of U.S. Natural Resource Damage Assessment regulations. Baseline refers to the condition of natural resources and services that would have existed had the spill not occurred. Implicit in this concept is an understanding that biotic and abiotic factors can cause the carrying capacity of environments to change with time, so that restoration of populations to prespill numbers may not always be feasible or appropriate (see Chapter 12). Restoration of ecosystem function is always considered appropriate, and trustees are encouraged to use their best judgment and knowledge of local ecological processes and trends to select feasible restoration alternatives and criteria for evaluating project success (e.g., Trustee Council 1994b:35).

The Trustee Council also emphasized an ecosystem approach in conducting restoration activities. In fact, Mission Statements 1 and 2 note that "[r]estoration should contribute to a healthy, productive and biologically diverse ecosystem within the spill area . . . [and] restoration will take an ecosystem approach to better understand what factors control the populations of injured resources" (Trustee Council 1994b:12).

WHAT IS RESTORED?

Restoration of Populations

Mitigation of population declines of seabirds that forage over vast distances must inevitably be site based; there is little chance of restoring the marine systems that provide resources to seabirds, except on a very limited spatial scale. Indeed, the kind of management undertaken will depend in large part on the biology of the birds (see Chapter 3). For example, species with mobile colonies (such as some gulls and terns) may require only *rehabilitation* of destroyed nest sites to provide the full range of locations available to the identified metapopulation. Even if the rehabilitated sites are then recolonized rapidly, there is no guarantee that the birds will remain indefinitely if their distribution is a reflection of the quality and location of a shifting food resource, rather than availability of nest sites. However, where food resources are more stable, such as in estuarine environments along the California coast, protection of least tern nests from human disturbance and predation has enabled colonies to recover and persist for many years (i.e., rehabilitation in the form of removing the human and predator disturbance).

Rehabilitation of nesting sites for highly philopatric species may not always be sufficient. For these species, a concerted *restoration* program may be required. For example, in the period between their introduction in 1890 and eradication in 1964, feral house cats devastated the burrowing seabird assemblage on Cuvier Island off New Zealand. In the 30 years following removal of the cats (rehabilitation), two species returned unaided: sooty shearwaters and grey-faced petrels. But common diving petrels, little shearwaters, fluttering shearwaters, and Pycroft's petrel, although abundant on some islands nearby, have not returned naturally (Bellingham *et al.* 1981). The re-establishment of species with moderate levels of philopatry may require the development of innovative translocation or social facilitation methods (e.g., Kress 1983, Podolsky 1990, Bell 1994; see also Chapter 9). As a further complication, the site-

specific reproductive behavior of highly philopatric species could be reflected in genetically distinctive populations.

Consequently, depending on the kind of disturbance faced by seabird populations and the biology of the birds, we may need to consider restoration of individual colonies as demes (genetic isolates), restoration of geographic range, and rehabilitation of metapopulations. For some metapopulations, the solution may be site-specific rehabilitation, but other populations may require intensive manipulation (restoration)

Restoration of Habitats

The restoration of islands for seabirds is a recent concept. The most comprehensive restoration project to date anywhere involves revegetation and the re-establishment of three species of birds (including cahow), marine turtles, and a land snail on the 6-hectare Nonsuch Island in Bermuda (Wingate 1985). This is not a large pool of experience from which to extrapolate the large-scale effects of oil spills. However, there are situations in which oil spills may so drastically affect terrestrial environments that significant habitat restoration could be required. If an incident like the *Braer* spill off the Shetland Islands, United Kingdom, in 1993 (Ritchie and O'Sullivan 1994) were repeated during extreme weather off seabird islands in New Zealand, for instance, wind-driven petroleum products could have a devastating effect on vegetation, poison the soils, and thereby destroy large areas used by burrowing seabirds (P. Irving, pers. com.).

Restoration of Communities and Ecosystems

Fortunately, there have not yet been any catastrophic oil spills combined with extreme weather around the seabird islands of New Zealand. However, because of the keystone role of seabirds in coastal and island ecosystems, even minor disturbance events may have long-term effects on some components of terrestrial ecosystems (see Chapter 13). For example, scurvy grasses (cresses: *Lepidium* spp.) are coastal plants that may be extremely abundant around seabird colonies. If seabird numbers are reduced, the soil chemistry changes and the scurvy grasses may disappear (Norton *et al.* 1997). An oil spill with an apparent population effect on seabirds could thus have successional effects on plant communities. In such circumstances the focus may need to be shifted from rehabilitation of a seabird population capable of only slow natural recovery to an accelerated restoration campaign designed to overcome the community effects of low bird numbers.

RESTORATION TIME SCALES

Given all the potential effects on seabird populations, the time scales required to complete their restoration will be equally varied. However, an overriding consideration is the potential productivity of the species involved. Species with high dispersal rates can show rates of population increase that exceed 20% per annum (see Nur and Ainley 1992; Chapter 12). These

may be the least likely candidates for restoration because the probability of natural recovery may be high. It is more likely that restoration will be required for species with low dispersal rates and low probability of natural recovery; many such species have rates of population increase of less than 5% per annum. Furthermore, some of these species remain at sea for five years or more after fledging. Therefore, if techniques are used that involve translocation of nearly fledged chicks (e.g., Bell 1994), no returns can be expected until several years after commencement of the program. Even if first-year breeders do return to the translocation site, because of their low productivity many years of monitoring will be required before we can be sure that a self-sustaining population is established. Consequently, the recovery of dense colonies of some seabirds following restoration may take many decades.

CHAPTER 6

RESTORATION GOALS

Distinct populations, species, communities, or ecosystems may differ in their undisturbed states and in their responses to human perturbations. Consequently, it is inappropriate to use the same restoration goals or the same criteria to evaluate if these goals have been met for each seabird population or species that may be restored. However, it is possible to define a single *conceptual goal*: to achieve a healthy and normally structured ecosystem, operating within the bounds of normal functions and processes, such as might have existed before the perturbation. Implicit in this conceptual goal is an understanding of (1) constraints to natural or human-assisted recovery of ecosystems and their constituent populations and (2) processes that may lead to extinction or extirpation of populations. (See Chapter 12 for a discussion of potential difficulties in achieving any of the following goals.)

DEMOGRAPHIC PARAMETERS

Unfortunately this conceptual goal requires a deeper understanding of the normal structure and function of perturbed ecosystems than is typically available. Within populations, we seldom have robust data on the three key demographic parameters—reproduction, survival, and migration (see Chapter 3b). Even less frequently can we quantify how these parameters vary in relation to natural abiotic and biotic changes in the environment. Operational goals in seabird restoration tend to focus on individual populations and measure numbers of birds or reproductive attributes. Interpreting censuses or information on reproduction can be problematic.

Numbers of individuals counted in an area in a particular year (i.e., attendance), such as a set of study plots within a colony, typically represent a variable and unknown proportion of the seabirds using the area. Delayed maturity typifies all seabird species. Subadults often travel widely and may briefly visit a colony or regularly visit particular areas within a colony for much of the breeding season (Halley *et al.* 1995). Adults whose reproductive attempts fail may attend a colony sporadically. Generally, breeding pairs alternate attending the nest, with off-duty mates spending little time on the colony. Thus, a survey will include all on-duty mates of breeding pairs, but only variable numbers of subadults, failed breeders, and off-duty mates. Because reproductive activities may vary markedly from year to year, annual counts may be highly variable even in the absence of perturbation. Thus, if researchers use only single counts at colonies, it may be difficult to quantify with any accuracy or precision the impact of a perturbation or the progress of recovery following a perturbation. Moreover, at some colonies it is difficult to select plots to study that are representative of the colony.

Setting restoration goals in relation to reproductive performance is also problematic. All phases of Alaskan seabird reproduction may change greatly from year to year. The number of adult

pairs that attempt to breed, number of females producing clutches, number of eggs laid, hatching success, and fledging success may all vary substantially in the absence of human perturbation. As with colony attendance data, reproductive variability obscures both effects of and recovery from perturbations.

Many demographic measures of seabird populations, such as survival of juveniles, subadults, or adults and age at first breeding, have rarely been quantified. The factors affecting the variability of immigration and emigration are poorly understood for most seabirds and may contribute greatly to local changes in numbers.

Given this daunting background in relation to the conceptual goal, it is nonetheless necessary to establish *operational goals* to measure progress toward and achievement of restoration of injured seabird populations. Each operational goal outlined below focuses at the population level of individual species. Choice of a particular operational goal will be dictated by knowledge about the population before the perturbation and the degree to which both the effect of the perturbation on the population and recovery of the population can be quantified. In many restoration programs it may be appropriate to include more than one operational goal or to combine elements from several goals to evaluate recovery.

OPERATIONAL GOALS

1. Population Returns to Pre-Oil Spill Level

Although we can quantitatively evaluate whether the population has returned to prespill levels (i.e., attendance), this goal will be unsuitable in situations where populations have large natural fluctuations, or when prespill data are insufficient to provide needed information. Where a population had been increasing or decreasing before the perturbation, it may be impossible to evaluate whether this goal has been attained. This constraint can be overcome if there are sufficient preperturbation data to establish the *trend of the baseline population* and thus predict changes that would have occurred in the absence of a perturbation. However, predictive models can result in highly variable outcomes even when there are satisfactory explanations to account for baseline variability and trends. Only a few years of baseline data would be needed if the population has been stable, but many years of population data would be necessary if variability is high. Unfortunately, in order to determine if the population is stable, data must be collected for many years to fully appreciate baseline variability.

Any of three quantitative criteria can satisfy this goal. First, mean attendance at the recovering colony, calculated from censuses taken during the entire breeding season, must exceed, for three consecutive years, that colony's prespill mean attendance minus one standard deviation. Second, attendance must exceed, for five consecutive years, the prespill mean minus two standard deviations. Third, a similar criterion, defined at the time of the perturbation, can be identified; it should be selected so that it will be difficult to meet if the population remains depressed after the perturbation, but relatively easy to meet if the population has recovered. For example, for either unaffected or fully recovered populations, the probability that annual counts would exceed the

mean of previous annual counts for three consecutive years would be 0.125 (i.e., 0.5^3), assuming that the frequency distribution of counts is symmetric. If the frequency distribution of annual counts is normal, 84% of the values will exceed the mean minus one standard deviation. That is, there would be an 84% chance that the mean count in a particular year would exceed the mean minus one standard deviation. Thus, for either unaffected or fully recovered populations, annual counts could be expected to exceed the mean minus one standard deviation for three consecutive years with a probability of 0.59 (i.e., 0.84^3). Clearly, if the population has not recovered, the probability of exceeding the mean minus one standard deviation would be lower, and a population that is far from recovery would be highly unlikely to satisfy this criterion.

2. Population Functions Normally

If baseline data on reproductive success and survival are available, they may serve as criteria to determine if the population is functioning normally. A population would be considered to be functioning normally if reproductive success or survival were fluctuating within bounds predicted by baseline values. Alternatively, the parameter could be compared to trends at nearby reference sites. The goal would be achieved if the parameter exceeded a particular threshold value for a specified period.

This operational goal is useful if baseline attendance data for the injured population are not available or are highly variable, thus precluding comparisons of current populations to prespill levels. Data can be compared to concurrent values at nearby reference sites or to baseline values at the affected site. This approach assumes that conditions are identical, except for the effect of the spill, across both space and time.

3. Population Fluctuates in Parallel with Environmental Change or with Reference Populations

If the injured population fluctuates in tandem with environmental changes or with reference populations, we can conclude that it is no longer constrained by the effects of the spill. Evaluating the achievement of this goal requires not only extensive (probably decadal) baseline data but also long-term monitoring of the population after the event. There are now several Alaskan colonies where the population status of one or more species of seabirds has been monitored for a decade or more, and where fluctuations in numbers and breeding performance have been associated strongly with fluctuations in environmental conditions. Even at colonies where reproduction has varied markedly among years, long-term data provide a suitable baseline and the foundation for an effective analytical approach to evaluate postevent counts and other parameters.

The influence of immigration/emigration among sites and the possible lack of concordance even among nearby sites must be considered in establishing this operational criterion. For example, population numbers may decline more at one colony during a regional population decline or increase more slowly during a regional expansion (see Chapter 3). Given local site differences,

changes in numbers may not always occur in parallel, and reference sites to evaluate recovery must be chosen carefully.

4. Population Ceases to Perform Better Than Before the Spill or Than Reference Populations

This goal assumes that restoration activities in some way augment the population so that it is "artificially" enhanced beyond "normal" until it reaches density-dependence. The goal is reached when the affected population ceases to perform better for some specified period (i.e., density-dependent adjustment is complete). We did not consider this an appropriate goal because we know little about the role of density-dependence for most populations (e.g., population numbers at a particular colony could be kept in a nonequilibrium state by repeated natural disturbances). Furthermore, restoration activities may increase the carrying capacity for this colony, whereby density-dependent effects would occur at a population size greater than "control" populations. In either case, it would be extremely difficult to measure density-dependence both before and after an oil spill.

5. Population Achieves a Level Predicted by Prespill Trend

To implement this operational goal, it must be possible to predict population trends following the spill based either on long-term documentation of trends and environmental fluctuations before the spill or on models using demographic data collected before the spill. Although the demographic approach uses well-established modeling protocols, it may be limited by the quality of the data used for input.

To model the recovery, values of demographic variables that would have occurred in the absence of the spill must be estimated and compared to actual postspill values. Recovery can be defined as either (1) that point at which the actual trajectory based on postspill conditions intersects the predicted trajectory based on prespill conditions or (2) the point at which the trajectories are parallel. Parallel trajectories, with postspill populations at lower levels than prespill populations, imply that the postspill population is functioning "normally," but at a lower level. This lower value can be the result of many different factors, including the fact that the spill has altered the ecosystem such that it has lowered the overall carrying capacity.

This goal is preferable to Operational Goal 1 if the population is cycling or changing monotonically up or down, but requires more data and a modeling approach, which may in turn require the incorporation of untestable assumptions.

6. Population Achieves Relative Former Size Compared with Control Colonies (i.e., it reaches its prespill percentage of a regional or global population)

This may be a good operational goal for common murres and can be applied not only to population numbers but also to areas or habitats that are occupied. It would not be a useful criterion for colonies at the edge of a species' geographical range or near other ecologically set

limits, where environmental factors may naturally alter relative percentages. As with Operational Goal 3, this criterion should be used with caution because populations of adjacent or nearby colonies often do not change in parallel.

7. *Replacement Birds are Demonstrated*

The goal of this approach would be to replace, for example, 500 seabirds killed by a spill with 500 birds either at the colony from which the birds were killed or at another site, perhaps away from the spill zone. This strategy was used in British Columbia following the *Nestucca* oil spill. It would likely be effective in the Aleutian Islands if exotic Arctic foxes were removed from islands that once had much more numerous populations of murres or other affected species. This goal may be easy to achieve and evaluate if introduced predators or other habitat alterations can be manipulated.

8. *Restoration Is Evaluated Using At-Sea Populations*

There are circumstances where restoration must be evaluated at sea instead of at colonies. Species such as *Brachyramphus* murrelets and *Cepphus* guillemots, and probably all petrel species, cannot be monitored readily on land because they nest in cavities, singly or in small, scattered colonies, or are nocturnal (e.g., Piatt and Ford 1993, Spear *et al.* 1995b). In addition, spills that occur between late summer and the following spring will injure birds in wintering assemblages, not necessarily near breeding colonies. The seabirds in these assemblages may include birds from several breeding populations over a wide area as well as species that nest inland and cannot easily be monitored there, such as grebes, loons, and ducks.

Operational Goals 1, 3, 5, 6, and possibly 7 are suitable for evaluating at-sea populations. Methods of monitoring birds on the water have been well developed, but surveys are labor-intensive. Because variability is high, large areas must be sampled within a short period, and sampling must be designed rigorously so that results can be extrapolated to the entire water body.

CONCLUSIONS

The ability of seabird populations to recover from catastrophic mortality and our ability to recognize recovery is both species- and locality-specific. It depends on the demographic characteristics of the population and the environmental context or ecosystem stability.

Because seabirds are part of ecosystems that vary both spatially and temporally, an evaluation of the success of restoration efforts must incorporate consideration of this variability. Variability on short time scales can be quantified by multi-year monitoring. Likewise, spatial variability can be quantified by the study of multiple affected sites (if more than one site is affected) and multiple reference sites. Selection of colonies or sites with a similar level of injury or colonies with a gradient of injury provides a far more effective framework for evaluating recovery than does

selection of a single affected colony or site. Single affected sites and single control or reference sites should be avoided if possible, because it is difficult to distinguish the effects of restoration programs from intrinsic differences between the two locations. Several reference sites that were unaffected by the event within the same biogeographical region in which the event occurred should be established. Replication is critical both within affected areas and in control sites. Affected and reference populations may be linked by immigration, thus forming subpopulations within a larger population rather than a series of distinct and demographically independent populations. In many seabirds, dispersal of subadults before first breeding may occur, blurring distinctions between affected and control populations even when there is a lingering effect of the spill. In general we lack critical information on movement among colonies, but genetic variance can be partitioned within and among colonies to estimate movement rates, and such a study should be incorporated in evaluating the recovery of affected populations.

CHAPTER 7

MONITORING ACTIVITIES DURING RESTORATION

Following the establishment of practical restoration goals (see Chapter 6), monitoring is needed to determine if goals are being met. The monitoring program must produce data that are accurate enough to measure definitively the responses of target populations. Factors to consider in designing monitoring programs include (1) the restoration goals, (2) defining the target populations (i.e., species and sites) to be monitored, including appropriate "control" or "reference" populations for comparison, (3) selecting appropriate parameters to measure, (4) quantifying objectives for desired minimum detectable differences, and (5) choosing the required sampling intervals and methods. A properly designed monitoring program should test hypotheses about patterns of change over time. For restoration monitoring of seabird populations following oil spills, it is important to monitor not only target populations but also resources that affect their survival and reproductive success. An understanding of ecosystem processes is necessary to try to sort out reasons for changes observed during monitoring (see Chapters 12 and 13).

TARGET POPULATIONS

Species

Injured species are the most obvious targets for restoration monitoring, but it may also be important to monitor other species that indicate important ecosystem processes affecting recovery rates. For example, if restoration goals call for common murre reproductive success to return to a self-sustaining level, it would also be important to monitor the reproductive success of other piscivorous seabirds breeding at the same sites, particularly if they were not injured by the spill, to evaluate whether environmental conditions, including prey availability, are conducive to normal reproductive success.

Sites

Multiple sites need to be monitored both within ("experimental") and outside ("control" or "reference") restoration areas to measure geographic variation. Ideally, comparisons would be made along an injury gradient from "heavily injured" to "not injured." Comparisons among experimental areas and between experimental and reference sites permit a more powerful evaluation of restoration efforts than do single-site comparisons. Selection of experimental and reference sites for which prespill data are available is desirable, because these data will be used to define "normal" or baseline conditions, including how populations were trending at the time of the spill.

Selection of reference sites requires careful evaluation of ecological similarity to experimental sites with respect to important comparative variables. For example, oceanographic conditions at experimental and reference sites should be similar enough so that trends in monitored parameters are similar (in the absence of injury). If possible, reference sites should be selected for which there are available historical data about ongoing trends and normal patterns of variation in parameters of interest. Furthermore, efforts should be made to select reference sites where target populations are not linked to experimental sites through dispersal to or from affected areas. Otherwise, reference sites may not provide independent "controls."

Another important consideration in selecting monitoring sites is the feasibility of collecting data safely and efficiently. For example, many seabird breeding colonies do not lend themselves to the rigorous collection of data on productivity because of difficult access.

PARAMETERS

Hatch *et al.* (1994) list the types of parameters that are normally measured in seabird monitoring programs. Data on the demographic parameters that may have been affected by an oil spill and are being targeted for restoration provide a basis for selecting which parameters to measure in restoration monitoring (see Chapter 3b). Since data on the number of individuals killed and the demographic profile of that population is usually the basis for initiating restoration activities, population trends are clearly important to measure. Nevertheless, population increases do not occur rapidly for long-lived seabirds having relatively low reproductive capacity (Nur and Ainley 1992). Therefore, other parameters frequently provide more sensitive indications of responses to restoration programs. Various components of productivity, survival, timing of nesting events, behavior, diet, and energetics are all potential candidates. Data also need to be gathered on environmental conditions that affect factors like prey availability and are independent of perturbations caused by oil spills.

Populations

Depending on restoration objectives, it may be necessary to estimate total populations of target species, but frequently, abundance indices will be monitored instead. For example, for most seabirds it is very difficult and expensive to derive overall population estimates. For this reason, replicate counts of birds or nests on a series of systematically selected plots provide the basis for estimating trends.

Productivity

One or more from a set of variables could be measured to provide an indication of productivity for target populations. For most seabirds, these would need to be measured at breeding sites (but see specific recommendations for marbled murrelet and pigeon guillemot). The list of potential measures includes laying success (percentage of nests in which eggs are laid), clutch size,

hatching success, and fledging success. Causes of loss of reproductive potential (e.g., predation) should be evaluated to try to separate direct and indirect effects of oil spills from normal mortality.

Survival

Characteristically, annual adult survival tends to be relatively high for most species of seabirds. For at least some species (e.g., murres; Sydeman 1993), this parameter can vary between years in response to changes in food availability. It would also be expected to change following perturbations like oil spills if large numbers of adults were killed. Measures of survival need to be fairly accurate (e.g., within 2-3% for some species), because even small changes can have substantial population effects. Survival monitoring will involve banding adequate numbers of birds with markers that allow individual recognition.

Timing of Nesting Events

For many species of seabirds, egg laying is timed so that the increased energy demands of reproduction coincide with periods of relatively high food availability (Lack 1968). Substantial shifts in timing of laying in response to environmental perturbations such as oil spills or oceanographic events (e.g., El Niño Southern Oscillations) can result in reduced productivity. Restoration monitoring programs in cases where timing was disrupted should include some measure of nesting chronology (e.g., laying, hatching, or fledging dates). Synchrony of egg laying may also be an important variable that is sensitive to perturbations like oil spills.

Behavior

Disruptions in some aspects of breeding behavior may occur following oil spills, and parameters that may be important in restoration monitoring programs include colony attendance patterns, feeding rates, and foraging trip lengths. For many species of seabirds, "normal ranges" in these parameters are available for comparison, but within-season variability needs to be considered in designing monitoring programs.

Diet

Restoration monitoring will frequently need to include some measure of seabird diets, because food availability has a major influence on most of the other parameters monitored. Shifts in the composition of diets may cause fluctuations in reproductive parameters that are independent of oil spill or restoration effects.

Energetics

Several seabird life history or population parameters have been used as indicators of temporal changes in the marine food web (see Boersma 1978, Cairns 1987, Montevecchi 1993, Ainley *et al.* 1995b). These parameters include adult survivorship, breeding success, chick growth rates, colony attendance, and adult activity budgets, and have been reviewed by Cairns (1987). The most appropriate parameter(s) that would indicate temporal changes in the marine food web will depend on species and location. However, in any case, data on diet composition and energy content of major prey must also be available to provide a link between these parameters and an understanding of how ecosystem processes affecting seabird energetics cause both long- and short-term fluctuations in seabird populations. We recommend that aspects of seabird energetics be monitored using the most appropriate set of parameters.

MINIMUM DETECTABLE DIFFERENCES

A major consideration in selecting parameters appropriate for restoration monitoring is defining the level of change that researchers need to be able to detect. The minimum differences that can be detected are based on variability within the target populations, desired confidence levels, and sample sizes. For restoration monitoring programs in field settings, it may be too costly to gather adequate samples for some parameters. Prior to instituting restoration monitoring, minimum detectable differences need to be set for each parameter, and necessary sample sizes need to be determined and evaluated for feasibility.

SAMPLING INTERVALS AND METHODS

The usual objective of restoration monitoring programs is to compare postevent time-series data for selected parameters with restoration target levels. For example, a restoration target may be the return of reproductive success to "normal" (with "normal" being defined as, for instance, between 0.5 and 0.7 fledglings per nest). Furthermore, part of the objective may be to examine correlations between environmental factors and patterns of change in reproductive success. Parameters that are sensitive to environmental change on an annual time scale, such as reproductive success, need to be measured annually to increase the probability of understanding ecosystem processes. Parameters that may not change rapidly for long-lived species with relatively low annual recruitment rates, such as population levels, may not need to be monitored annually. Power analysis (Gerrodette 1987) can help to select the sampling interval needed to meet restoration monitoring objectives.

Recommended monitoring methods have been published for many species of seabirds (e.g., Nettleship 1976, Walsh *et al.* 1995), and it is important to use standard methods in restoration monitoring for selected species. Very specific protocols need to be developed so that results for different sites may be readily compared. Even apparently obvious terms need to be defined so that all observers are recording data in the same way. For example, it is not sufficient to state

that kittiwake nests should be counted; the word "nest" needs to be defined (e.g., a structure to which vegetation has obviously been added during the current year).

CHAPTER 8

RESTORATION TECHNIQUES: PREREQUISITES

The development and implementation of *all* seabird restoration plans involve general information needs or requirements. A plan's suitability and chances of success will increase to the extent that these requirements or information needs are satisfied. We discuss these general requirements in temporal sequence.

BASELINE AND HISTORIC DATA

Baseline data on population trends, demographic parameters, and factors that may limit population growth are essential for identifying spill-related injuries (see Chapter 4), helping to determine causes of population trends (see Chapters 3, 4, and 12), designing and implementing restoration plans (including setting restoration goals; see Chapter 6), and evaluating the need for direct restoration (human intervention) activities. Where these data are not currently available for populations that are at risk to oil spills, we recommend that natural resource agencies initiate baseline studies. Baseline time-series data will help demonstrate the variability of a population over time, evaluate injury, and provide an indication of the probability of natural recovery.

Demographic information is also needed to assess the probability of natural recovery as well as the probability of success for a particular restoration project. Nevertheless, prespill demographic information for many populations will be unavailable. Parameters from conspecific populations can be substituted, but we caution that these values may not be appropriate for the environment under consideration. Finally, information on which resources or demographic parameters may be limiting population growth (see Chapters 3b, 12) is essential in designing a restoration plan. If population growth is constrained by a limiting resource (e.g., food near a colony) or demographic parameter (e.g., low breeding population resulting from gillnet bycatch), restoration plans must address these factors to be successful.

INJURY ASSESSMENT

An accurate identification of oiled birds and an estimate of the number of birds killed directly by a spill are essential to estimate injury to populations and to set restoration goals. Total mortality will be estimated by extrapolating from carcass counts using models that include a spill trajectory and data on the at-sea distribution of the affected species. The demographic composition (i.e., age and sex) of the carcasses should be determined and used with demographic models to estimate effects at the population level and the probability that the birds will recover without human intervention. Finally, genetic, morphometric, or plumage analyses of the

carcasses may provide data on the area of origin of the birds killed by the spill. These data are required to help identify the geographic areas affected by the spill (which for seabirds may extend well beyond the spill zone defined by the physical extent of the spilled oil) and, therefore, the areas most appropriate for direct restoration and monitoring activities.

POSTSPILL, PRERESTORATION MONITORING

Identifying the need for direct restoration and the appropriate techniques, if restoration is warranted, requires postspill monitoring of abundance and demographic parameters (in particular, breeding productivity and survivorship). If a population is not recovering from a decline associated with a spill, the reasons for the lack of recovery need to be determined or at least estimated. The natural and anthropogenic effects that may be hampering recovery need to be ascertained so that the restoration plan can attempt to address these effects.

EVALUATION OF RESTORATION TECHNIQUES

If a population is not recovering (based on postspill monitoring) and the reasons for the lack of recovery have been identified or estimated, potential restoration techniques need to be evaluated. The evaluation procedure should include (1) the development of models to test the potential effect of each technique on the population, (2) a cost-benefit analysis to determine which technique promises the most benefits to the population given the biological, social, and financial costs, (3) the development of a suite of strategies, including deciding whether the technique(s) will be implemented singly, sequentially, or in combination, and 4) consideration of site-specific issues (e.g., native subsistence, tourism).

CONTINUING MONITORING

Following the implementation of the restoration plan, abundance and demographic parameters at target and reference populations need to be monitored for the duration of the plan. These data will help evaluate the success of the restoration technique and may indicate the need for alternate or additional restoration efforts.

CHAPTER 9

RESTORATION TECHNIQUES: DESCRIPTIONS

Part A: Introduction

This section describes a variety of restoration techniques that managers might consider in designing and implementing a seabird restoration program. As discussed elsewhere, each technique must be evaluated in light of the specific conditions at colonies and former colonies near an oil spill. Seabirds are migratory, with many species undertaking extensive migrations and spending much of the year distant from their natal or breeding colony. Thus, individuals impacted by an oil spill (especially one during the nonbreeding season) could come from colonies hundreds or thousands of kilometers distant (see Chapter 2a). Additionally, rates and distance of natal dispersal for seabirds can be high (Halley and Harris 1993, Harris and Wanless 1991), and population growth or recovery can involve immigrants from distant colonies or regions. Some of the techniques described below might be employed to the benefit of the injured population far from the colony.

Part B: Management of Predators, Herbivores, and Vegetation

BIOLOGICAL EFFECTS OF INTRODUCED PREDATORS AND HERBIVORES

Predator, herbivore, and vegetation management can, in many circumstances, be very effective techniques for restoring injured seabird populations. These techniques can enhance recruitment, productivity, and survivorship of seabirds. Management can be directed toward introduced exotic species, indigenous species introduced by humans from other sites in the same region, or species that are indigenous to the colony. The techniques available for managing each category of species are similar, but managers and the public are often more willing to use more severe measures to remove exotic species than to remove indigenous species.

Exotic and indigenous predators throughout the world have had profound adverse effects on seabird populations (Nelson 1979, Burger and Gochfeld 1994), and managing predators can reduce the take of eggs and chicks and mortality of adults. While habitat destruction and human exploitation and disturbance have also been important, the widespread introduction of mammals, both deliberate and accidental, has dramatically reduced the natural biodiversity of island ecosystems (Moors and Atkinson 1984). Predation by alien mammals and other pests is

probably the single most significant factor influencing the decline of, or maintaining the small size of, many seabird populations today.

Seabirds are particularly vulnerable to predation by alien predators. Their low annual productivity and their general lack of effective antipredator behavior against mammals makes most seabird populations in previously predator-free environments vulnerable to introduced predators (Moors and Atkinson 1984). In addition, most seabirds are colonial breeders, nesting in large, conspicuous colonies on islands or in islandlike situations. Nests are typically on the ground or in shallow burrows. Some species, such as the small procellariiforms, auklets, and terns, are especially vulnerable because of their small size and the fact that they leave their chicks unattended for extended periods while they make long-range feeding flights. In combination, these features mean that predation by alien predatory mammals frequently results in annual mortality in seabird populations that consistently exceeds annual recruitment. Predation impacts may be masked by the large size of some breeding colonies, the long breeding life of some species, immigration from other breeding colonies, and the characteristics of particular sites. Nevertheless, such predation almost invariably leads to dramatic population declines and, in some cases, extirpation (Moors and Atkinson 1984, Harrison 1990).

Such predators as carnivores and rodents have reduced populations or extirpated colonies of virtually every seabird taxon (Moors and Atkinson 1984). Foxes, mongoose, mink, and cats will eat nesting adults, chicks, or eggs and can eliminate a colony rapidly. Introductions of foxes in Alaska have had a devastating effect on seabird populations, especially burrow-nesting species (Bailey 1993). While foxes were originally absent from most Alaskan islands in the north Pacific, Russians began introducing Arctic and red foxes on Aleutian Islands in the 18th century for fur farming. By the 1930s, more than 450 islands had been stocked, and fox trappers regarded seabirds as free feed (Bailey 1993). At a 600-hectare island off Newfoundland, 12 foxes consumed 31,000 Leach's storm-petrels during a single breeding season (Skepkovych 1986).

Mongoose have severely restricted the range of all ground-nesting birds on four of Hawaii's main islands. For example, wedge-tailed shearwaters are restricted to breeding on mongoose-free bluffs on Maui (Harrison 1990). Mink are currently spreading in western Scotland and ravaging its ground-nesting and cavity-nesting seabirds, such as gulls, terns, cormorants, eiders, and black guillemots (Craik 1993). Ghost seabird colonies are becoming common in Scotland, where a single mink can seize and cache as many as 100 eggs or chicks. Introduced cats and pigs have reduced the grey-faced petrel population to very low numbers at Tuhua, New Zealand (A. Saunders, pers. com.).

Often a predator will not extirpate a colony but will diminish recruitment, productivity, and survivorship. For example, on Terui Island, Japan, predation by feral cats on adults and chicks seems to contribute to a declining population of black-tailed gulls (Watanuki and Terasawa 1995). Other factors, such as vegetation cover, would also be involved, but controlling or eradicating cats at Terui Island probably would increase the gull population.

The introduction of rats and other rodents at colonies has been catastrophic for many seabirds and has caused local extinctions of populations (Atkinson 1985). Rats tend to eat eggs and young instead of adults, although they have attacked adult birds as large as Laysan albatross (Harrison 1990). In Hawaii, black rats introduced during 1943 on the Midway Islands have extirpated storm-petrels and Bulwer's petrels, have depleted Bonin petrel populations, and may even have affected large seabirds such as red-footed boobies (Harrison 1990). Black rats have caused breeding failure of populations of Cory's shearwater in the Corsican Islands, Mediterranean Sea, and introduced Polynesian rats have severely reduced the breeding success of gadfly petrels by eating chicks at Henderson Island, Pitcairn group (Brooke 1995). The U.S. Fish and Wildlife Service in Alaska has implemented a program to prevent the introduction of rats within the Alaska Maritime National Wildlife Refuge and on the Pribilof Islands (A. Sowls, pers. com.). Norway rats and ground squirrels have been introduced at several colonies in Alaska.

Introduced herbivores can also damage seabird colonies. In a classic example of ecological catastrophe, the manager of a guano mine set loose guinea pigs and two types of rabbits on Laysan Island, Hawaii, in 1903. Within six years the rabbits had overrun the island, consumed the vegetation, created desert conditions, caused the extinction of three endemic land birds, and destroyed the habitat for seabirds that nest on or under vegetation (Harrison 1990). Competition for cavities between seabirds and the introduced European rabbit was thought to be responsible, in part, for the decline of tufted puffin and rhinoceros auklet populations on Southeast Farallon Island; puffin and auklet populations increased following the removal of the rabbits in 1974 (Ainley and Boekelheide 1990).

ERADICATION OF INTRODUCED PESTS AND RESTORATION OF SEABIRD COLONIES

Sites where introduced predators have extirpated colonies or depressed populations present a great opportunity for seabird restoration. While frequently the colony was extirpated or greatly reduced by factors unrelated to an oil spill, these sites usually offer suitable nest sites and foraging conditions to allow for re-establishment of the colony once the perturbing factor (e.g., exotic predators) has been eliminated. There is little doubt that removal of alien predators or herbivores from breeding colonies can allow the restoration of the natural biodiversity, including the recovery of depleted seabird populations (Moors and Atkinson 1984). Removal of indigenous predators, however, is controversial in areas where their populations are not increasing.

Seabird managers have developed and employed cost-effective techniques for predator removal in diverse locations around the globe. In many cases the removal of exotic predators has quickly allowed the re-establishment of the former colonies.

Nizki and Alaid Islands in the western Aleutian Islands, with a combined area of 1,200 hectares, are often joined by a sandbar. Arctic foxes were introduced in 1911, and by 1937 nesting birds had been drastically reduced or extirpated (Byrd *et al.* 1994). Managers killed the foxes in the

mid-1970s and subsequently made periodic counts. By 1990, the population of breeding birds had tripled to about 14,000 birds. Byrd *et al.* (1994) found particularly impressive increases of the populations of red-throated loons, pelagic cormorants, common eider, glaucous-winged gulls, and tufted puffins. These increases are probably continuing.

Kaligagan Island, in the eastern Aleutians, was stocked with foxes in 1921. In the 1930s, its seabird population plunged so low that the renowned Alaska naturalist Olaus Murie recommended that it continue as a fox farm because it seemed to hold little promise as a seabird colony. In the early 1980s, after foxes had died out without human assistance, Kaligagan had 125,000 burrowing seabirds (Bailey 1993). Bailey (1993) estimates that there are about 46 islands in Alaska where foxes continue to survive by depredating seabird colonies.

There have been similar success stories in many parts of the world. The removal of cats from Jarvis Island, in the central Pacific, enabled blue-gray noddies and Christmas shearwaters to recolonize and populations of other species to increase dramatically (M. Rauzon, pers. com.). When rabbits were eliminated from Laysan Island, native vegetation recovered and seabird populations recovered dramatically (Harrison 1990). The cessation of human predation has enabled Manana Island, Hawaii, to support the largest seabird colony in the main Hawaiian Islands despite being devoid of seabirds at the turn of the 20th century (Harrison 1990).

Because New Zealand's Department of Conservation considers introduced pests to be the most significant remaining threat to New Zealand's biodiversity (Clout and Saunders 1995), it has implemented an extensive program that has eliminated 12 predatory mammals and one predatory bird from 60 islands (Veitch and Bell 1990). The eradication of feral pigs in 1936 from Aorangi Island, New Zealand, increased Buller's shearwaters from about 100 pairs in 1938 to over 200,000 pairs in 1981 (Harper 1986). The survival of grey-faced petrel chicks increased dramatically immediately following the eradication of Norway rats in 1986 at Motuhoura (Harrison 1992). At Marotiri, New Zealand, 85% of the chicks of little shearwaters survived following the eradication of Polynesian rats from one island in 1993, while only 5% of the chicks survived on an adjacent island where rats remained (R. Pierce, pers. com.).

The Canadian Wildlife Service is currently using funds from the *Nestucca* oil spill to remove introduced raccoons, which are colonizing new islands, in the Queen Charlotte Archipelago, British Columbia. This area supports more than 1.5 million breeding seabirds, including about one-half of the world population of ancient murrelets. Raccoons have tremendous destructive potential to burrow-nesting seabirds such as ancient murrelets. The Canadian Wildlife Service is also using oil spill funds to remove rats from Langara Island. While this site is distant from the spill, the removal of the rats is likely to increase seabird populations in the Queen Charlotte Archipelago.

It is not possible to estimate with any precision the increase in seabird populations when constraints on growth from predators or herbivores are reduced or eliminated. Evidently, increases *per island* can be substantial. The seabird population on the Nizki and Alaid Islands has increased by about 10,000, and the Kaligagan Island population seems to have increased by

100,000 or more. It is possible that a few decades following predator removal, a colony of one million or more birds might be re-established.

METHODS

Removal of introduced predators or herbivores can usually be accomplished cost-effectively if managers are allowed to use the most effective tools available. In virtually every situation, eradication of the target species is the preferred option over merely a sustained reduction in numbers (Veitch and Bell 1990). In most situations, obtaining approval to use toxicants (e.g., M-44s or Compound 1080) is necessary, although trapping and shooting programs are feasible on smaller islands. The responsible agency must firmly commit to defending the decision to use toxicants and to countering any adverse publicity that may be generated by opponents of this approach. Veitch and Bell (1990) recommend asking opponents to suggest another viable alternative.

Because there is often opposition to the use of toxicants on nontarget species, any toxicant program must be planned carefully. It is usually possible to choose locations and design a program that reduces or eliminates the risk of affecting nontarget species. In recent years important advances have been made in the development and refinement of techniques to effectively eradicate mammal pests from larger islands. In particular, the aerial application of second-generation anticoagulant rodenticides has allowed for successful rodent eradication operations on islands greater than 200 hectares where ground-based approaches were impractical (Towns *et al.* 1994). In New Zealand, plans are well advanced to use aerial application techniques to eradicate rodents from islands greater than 1,000 hectares in size.

CONTROL OF INDIGENOUS SPECIES

The issue of controlling native species may arise most frequently with avian predation. On Terui Island, Japan, slaty-backed gulls and crows eat eggs and chicks of common murres (Watanuki and Terasawa 1995). In Prince William Sound, Alaska, black-billed magpies and northwestern crows eat the eggs and chicks of pigeon guillemots (L. Hayes, pers. com.), a species that the Trustee Council has determined was injured by EVOS. Northwestern crows and black-billed magpies, foraging opportunistically with bald eagles, have dramatically reduced the reproductive success of black-legged kittiwakes in Prince William Sound (Irons 1992). Bald eagles have also had a drastic effect on the colony attendance, stability, and reproductive success of common murres on Tatoosh Island, Washington, and Triangle Island, British Columbia (Parrish 1995, 1996) and on common murre breeding phenology on Shag Rock, Oregon (R. Lowe, pers. com.) In the Gulf of Maine, a few Atlantic puffins have re-established colonies, assisted in part by the poisoning of more than 3,800 native herring gulls and great black-backed gulls (S. Kress, pers. com.). The circumstances under which it is appropriate to control one native species to increase the population of another native species is a question of overall management philosophy that this report does not address.

CONTROL OF VEGETATION

Control and strategic use of exotic vegetation can be a useful technique for enhancing recruitment and productivity of seabirds. Laysan albatross began nesting on coastal bluffs at Kilauea National Wildlife Refuge, Hawaii, in the 1980s in part because refuge managers removed exotic shrubbery and created an exotic lawn that attracted adult pairs. The success of this project can also be attributed to the fact that the lawn was completely fenced so that feral dogs could not kill albatross. It is recognized that the expansion of the native *Calamagrostis* at Teuri Island favors an increase of rhinoceros auklet burrows, while black-tailed gulls appear to avoid dense *Calamagrostis* areas (Watanuki and Terasawa 1995). The question of whether to manipulate vegetation in such a situation is ultimately a question of defining the management goals.

Part C: Management of Human Impacts

Seabirds are long-lived, have low reproductive rates, tend to breed in large numbers on predator-free islands, and feed in relatively small areas of high biological productivity (Duffy and Nettleship 1992). These characteristics make seabirds vulnerable to human activity, especially since activities of humans also are often concentrated in areas of high biological productivity (e.g., fisheries).

Human activities near seabird nesting and foraging areas may negatively or, in some cases, positively affect productivity. Activities that negatively affect seabird reproduction include disturbance or destruction of nesting locations and egg collection, causing decreased productivity; introduction of predators near nesting areas; hunting; pollution; and mortality associated with fisheries bycatch. Future problems may include increased human disturbance and competition with humans for marine resources (Duffy and Nettleship 1992).

Recently, resource agencies have placed much emphasis on multispecies or ecosystem management, but managers often have limited information on how ecosystems function. Lacking sufficient information on ecosystem function it may be simplest to address anthropogenic effects known to be harmful, such as disturbance, feral animals, unsustainable exploitation by humans, and pollution (Duffy 1994).

The workshop divided human disturbances to seabirds into six categories: (1) colony disturbance, (2) at-sea disturbance (primarily at foraging areas), (3) incidental mortality associated with net fisheries, (4) predator introduction at nesting locations, (5) habitat loss, and (6) marine pollution. We discuss managing the effects of human disturbance for seabirds in general, and include a summary of the probability of success if applied to the four species listed

as injured by the Trustee Council (common murre, marbled murrelet, pigeon guillemot, harlequin duck).

COLONY DISTURBANCE

Sources of human disturbance at colonies include low-flying aircraft, boats near shore, and people present on colonies. Within the designated EVOS zone, aircraft is a common means of transportation and poses great potential for colony disturbance. Fishing operations and nature tours have the greatest potential for nearshore boat activity causing disturbance to seabird colonies, especially since seabirds, fishery operations, and nature tours all seek areas of marine productivity. Other than anecdotal observations, however, minimal data exist that could help determine the extent of disturbance by humans and its impact on colonies within the designated EVOS zone.

Two primary methods of reducing colony disturbance by humans include posting "no access" signs and public education. Posting signs is feasible for seabird nesting locations that are accessible from shore, where signs can be posted in an obvious location at a safe distance from the colony. This would be a valuable approach to reducing disturbance in an area with many pedestrians (e.g., a site near a town, or a site that is regularly visited by nature viewers). Signs are more difficult to place effectively near colonies accessible by boat, however, and are not practical to place near colonies accessible by aircraft, although signs can be placed at boat ramps, harbors, and local airstrips. As has been done at Farallon Island National Wildlife Refuge, for instance, the Federal Aviation Administration should mark important seabird colonies on aerial maps and state that it is a violation of law or refuge/sanctuary policy to approach below a certain altitude. Obeying restrictions posted on signs requires voluntary compliance because enforcement would likely be a low priority for enforcement officials. Particularly in remote sites in Alaska, educating people who travel near seabird colonies is likely to be the most effective means of long-term management.

For species affected by EVOS, reduction or elimination of disturbance at colonies would have the highest probability of success for common murres, little or no success for pigeon guillemots or marbled murrelets, and uncertain success for harlequin ducks.

AT-SEA DISTURBANCE

At-sea disturbance is more difficult to measure than colony disturbance. Boat traffic at or near foraging areas may disturb birds and interrupt feeding. Disturbance to seabirds at foraging areas primarily results from fishing activities or persistent vessel traffic (near ports and shipping lanes, and in areas of recreational boating). At-sea disturbance will have varying degrees of impact on different species. Gulls and kittiwakes are unlikely to be disturbed by boat traffic (as long as the patch of prey on which they are feeding is not disrupted), but marbled and Kittlitz's murrelets

may be susceptible to disturbance by boats. Because boat-traffic restrictions may be difficult to implement or enforce, public education may be the best method of reducing at-sea disturbance.

For species affected by EVOS, reduction or elimination of at-sea disturbance would have a low probability of increasing productivity for common murres, pigeon guillemots, marbled murrelets, and harlequin ducks due to the difficulty of identifying specific foraging areas that need protection, and of implementing and enforcing regulations.

NET FISHERIES

Gillnet and longline fisheries have a great potential for impacting seabirds (e.g., DeGange *et al.* 1993). Seabird bycatch can be monitored by implementing observer programs, but the effect on seabird populations cannot be determined unless it is known what colony, metapopulation, or population the birds belong to. This information is easily obtained if fisheries occur near nesting locations, or if the species has a restricted breeding range. In most cases of at-sea mortality resulting from fishing activities, however, the impact on breeding populations of seabirds is unknown because the source population is unknown (Schneider *et al.* 1992). Jones and DeGange (1988) reported that common murre mortality in gillnets in California was large enough to affect local colonies (see also Takekawa *et al.* 1990).

For species affected by EVOS, reduction or elimination of fisheries bycatch would have a high probability of benefiting populations of common murres and marbled murrelets, but would have uncertain benefits for pigeon guillemots and harlequin ducks.

PREDATOR INTRODUCTION

Among the many environmental challenges faced by conservation scientists and managers in the coming decades, managing the inexorable invasion of alien species from distant lands and waters and between previously isolated regions may be the most difficult (Soulé 1990). The introduction of exotic or indigenous predators or pests on islands where they do not occur naturally has resulted in dramatic decline and local extinction of nesting seabirds (Atkinson 1985, Moors and Atkinson 1984, Bailey 1993, Byrd *et al.* 1994).

It is essential that introductions and reintroductions do not occur on islands without introduced predators or where introduced exotic or indigenous predators have been successfully removed. This can be achieved by (1) public education highlighting the potential threats to the local biological diversity posed by the introduction of predators or pests, (2) limiting access to islands with seabird populations that are vulnerable to introduced predators or pests in an effort to avoid accidental colonization, and (3) on islands with permanent human settlements, encouraging people to manage pets and domestic animals to minimize the risk of invasion of natural habitats (e.g., permit only sterile pets on such islands; limit garbage or sources of food that could

maintain or facilitate the establishment of rat and mice populations; use poison to control rodent outbreaks).

At seabird nesting locations with nearby human inhabitants, it is important to avoid enhancing populations of indigenous predators beyond the natural carrying capacity. Human activity may provide alternative or additional food resources for indigenous predators (e.g., garbage or fish processors attracting eagles) and may result in larger populations than the habitat would naturally support. An increase in predation rates may follow from an increase in predator numbers. Controlling indigenous predator populations through nonlethal means is preferred over eradication programs because indigenous predators are part of the natural system.

The deliberate introduction of large animals is now illegal in New Zealand, but the accidental introduction of smaller animals remains a problem (Veitch and Bell 1990). Permanent bait stations and traps can be placed around the shores of islands prone to reinvasion (Veitch and Bell 1990), but this requires a long-term commitment because traps and baits will need to be checked regularly. Alternatives include regular inspections of vessels traveling to islands, but this is very costly and logistically difficult. One method of combating the problem of unintentional introduction or reintroduction is through public education. Veitch and Bell (1990) consider it essential that any eradication program be discussed with the appropriate people and agencies from the beginning of planning in an attempt to reduce misunderstandings and undesirable or ill-informed publicity.

Preventing predator introduction or enhancement to nesting areas within the EVOS zone would have the highest probability of benefiting common murres, pigeon guillemots, and possibly harlequin ducks (although in Alaska most harlequin ducks nest on the mainland or on large islands), but will likely have little effect on the productivity of marbled murrelets (but see Chapter 2c).

HABITAT LOSS

Foraging areas for seabirds are often defined by oceanographic processes, and identifying specific foraging areas is difficult. Foraging areas may consist of a small estuary near a nest site, or locations within large geographic regions that change over time. The scope of management, therefore, is limited to managing human activities that conflict with the foraging of seabirds (primarily fisheries or vessel traffic). Seabird biologists need to work with the people involved in those activities on a regional or international level. On a large scale, the Department of Interior currently does not enforce the U.S. Migratory Bird Treaty Act beyond the 12-mile territorial sea. Seabird biologists could press for national marine reserves or sanctuaries to protect foraging areas for seabirds and other marine animals within the 200-mile Exclusive Economic Zone or international reserves beyond the 200-mile limit. They also could help develop multinational treaties that set common standards of behavior toward seabirds among nations (Duffy and Nettleship 1992).

Breeding habitat can be created by constructing artificial burrows (Byrd 1979, Priddle and Carlile 1995), modifying vegetation, or creating nesting islands and platforms. Seabirds also may be attracted (Kress 1983) or translocated (Kress and Nettleship 1988, Towns *et al.* 1990b) to alternate nesting locations (see Chapters 9e and 9f). However, if a large number of adult birds die as a result of an oil spill, many previously occupied nest sites would become available, and there would be no need to create additional nest sites. Therefore, simply protecting available habitat and potentially enhancing the nesting area through vegetation or predator control or attracting birds back to the site may be all that is needed.

Reduction or elimination of breeding habitat loss within the EVOS zone would be very beneficial for marbled murrelets (old-growth forest) and harlequin ducks (streamside habitat), but would have little benefit for common murres or pigeon guillemots, since there is little threat to their nesting habitat.

SUBSISTENCE HARVEST

In locations where subsistence harvest can significantly affect the local population of seabirds, it is important to have an estimate of annual take and life history models to determine what, if any, management actions are necessary. Life history models provide managers with different management schemes. For example, is it better to exploit eggs or adults, or to harvest one egg of each three-egg clutch throughout the colony, or to harvest all eggs in one-third of the colony (Duffy 1994)?

Enforcement of laws and regulations on private land has been a primary method of protecting seabirds in Northern Hemisphere countries (Doughty 1975). Such methods, however, have not been effective in isolated areas where traditional subsistence hunting occurs (Blanchard 1994). We recommend regulation of subsistence harvest by implementing education programs that emphasize minimum take and self-regulation. In locations where seabirds are an important part of a local culture or where seabirds are linked, directly or indirectly, to the local economic base, it is important that seabird managers work with local cultures and economic goals rather than against them (Blanchard and Nettleship 1992).

Within the EVOS zone, subsistence harvest of common murres, pigeon guillemots, and marbled murrelets likely is minimal and does not affect recovery of populations. Subsistence harvest of harlequin ducks is probably greater than that of alcids, with an unknown effect on recovery of populations.

MANAGEMENT OF MARINE POLLUTION

Pollutants enter the marine environment through point sources (e.g., oil spills and industrial discharges) and non-point sources (e.g., persistent organochlorine, plastic fragments, and small-scale oil discharge from vessels; Fry 1992). Organochlorine pollutants have become globally

distributed from direct point-source inputs (Tanabe *et al.* 1984, Brun *et al.* 1991), and, although discharge of organochlorine pesticides has been eliminated or greatly reduced in North America and Europe, it has taken 15 to 20 years for residues to fall below reproductively harmful levels (Fry 1992). Only about 10% of oil pollution, however, is caused by the few massive point-source disasters such as *Torrey Canyon*, *Exxon Valdez*, and the Persian Gulf spills. Small spills occur more frequently and may be much more damaging to seabirds over the longer term (M'Gonigle and Zacher 1979, Fry 1992, Gandini *et al.* 1994, Nur *et al.* 1995). These conclusions emphasize the importance of working not only to prevent tanker spills, but also to reduce the occurrence of chronic oiling from point and non-point sources.

Given the importance of oil transportation at sea, it is necessary to identify important seabird colony sites on land (e.g., Lloyd 1984) and foraging areas at sea (Tasker *et al.* 1990). These colonies and foraging areas should be marked on navigational charts to encourage navigators to avoid such areas altogether or to take increased precautions in their vicinity (Duffy and Nettleship 1992). Although management of tanker traffic is a complex issue, seabird biologists can play an important role in working with oil shipping companies and the U.S. Coast Guard to help provide protective measures through voluntary compliance or regulations (see the discussion of sensitivity maps in Chapter 2a).

Plastic debris is another increasingly common form of pollution in the ocean. More than 80 species of seabirds throughout the world have been reported as ingesting plastic fragments (Day *et al.* 1985, Ryan 1988, Fry 1992). In the subarctic north Pacific, Robards *et al.* (1995) reported that ingestion of plastic particles by seabirds significantly increased between studies collected in 1969-77 and 1988-90. Enforcement of regulations may be an efficient means of treating pollution, but education promoting voluntary compliance may be the most feasible method of dealing with pollution.

There is insufficient information on the extent of marine pollution and its effects on seabirds within the EVOS zone. However, a program to reduce marine pollution would not negatively affect seabirds, and may be a positive influence.

CONCLUSION

To achieve long-term conservation goals, we feel that education is of paramount importance in the management of human activities affecting seabirds. The focus of such education should be to promote the importance of seabirds in maintaining a healthy ecosystem and to let people know that voluntary conservation efforts will make regulation unnecessary. Although imposing regulations may secure the desired results more rapidly in the short term, education promoting voluntary conservation efforts (using an educational approach that instills a vested interest in resource users) will be more effective over the long term and in remote areas where enforcement of regulations is difficult or impossible. Blanchard and Nettleship (1992) point out that where conservation policies remain at unresolved odds with local economic and cultural imperatives, enforcement of those policies may end in failure and backlash from the culture. This has been the case with spring shooting of waterfowl by Alaskan Natives (Raveling 1984, Pamplin 1986,

Blanchard 1987) and efforts to manage the hunting of thick-billed murres in Newfoundland and Labrador (Nettleship 1977, Brown and Nettleship 1984, Elliot 1991, Elliot *et al.* 1991).

Conservation efforts can succeed only with the support of the public. Craig and Veitch (1990) found a need for three methods of improving public perception of their seabird conservation work: (1) greater public access to islands, (2) increased protection of island ecosystems, and (3) more information for the public. It is important that scientists, managers, environmental educators, and resource users work together to produce effective conservation programs.

Part D: Management of Seabird Food Resources

The likelihood that seabird populations will recover from losses caused by oil pollution or other factors may depend strongly on the availability of suitable food (e.g., forage fish, euphausids) during the first few years following the initial mortality event. Factors influencing the annual abundance and availability of food include natural changes to the marine environment that may directly affect growth and recruitment of prey species (see Chapter 12); competition; and predation by natural marine predators, including seabirds, marine mammals, and large predatory fishes such as cod, pollock, and salmon. In addition, abundance can be affected by a range of anthropogenic effects, including overfishing (Springer 1992).

OVERVIEW OF RELATIONSHIP BETWEEN FOOD RESOURCES AND SEABIRD POPULATIONS

There is good evidence to suggest that over short time periods, seabird foraging success and population dynamics may be tightly coupled with local food abundance (Ashmole 1963, Birt *et al.* 1987, Uttley *et al.* 1989; see also Chapter 12). Seabird populations in some ecosystems may use a significant fraction of local forage fish stocks (Furness 1990, Duffy and Schneider 1994), and seabird breeding parameters may track long-term changes at lower trophic levels (Aebischer *et al.* 1990). In Alaska, dramatic shifts in forage and predatory fish populations apparently have occurred in the Gulf of Alaska and Bering Sea, which were accompanied by marked changes in diet composition and population declines of several seabird and marine mammal species (Springer 1992, Hatch *et al.* 1993, Piatt and Anderson 1996).

While some seabird and marine mammal populations have shown increasing signs of stress (i.e., decreases) during the past decade in the north Pacific, populations of large predatory fishes, such as pollock and salmon, have increased dramatically (Springer 1992), and herring have decreased (Wespestad and Fried 1983). There is strong evidence that these changes have occurred in response to, or simultaneously with, decadal-scale shifts in the marine ecosystems of the north Pacific (Duffy 1993). Predatory fishes may be directly consuming forage fish used by seabirds

and/or outcompeting forage species for shared lower trophic-level food resources during early life stages.

Interspecific competition is suggested by the negative relationship between herring and pollock biomass in the Bering Sea (Wespestad and Fried 1983). Data collected by the Alaska Predator Ecosystem Experiment project in Prince William Sound provide further support for the competition hypothesis: pollock and herring school together during the fall (L. Halderson, University of Alaska Fairbanks, unpubl. data) and their diets overlap (M. Sturdevant, National Marine Fisheries Service, unpubl. data). Furthermore, the Alaska Predator Ecosystem Experiment also determined that piscivorous seabirds select herring and other forage species over pollock (W. Ostrand, USFWS, unpubl. data).

EFFECTS OF MANAGING FOOD RESOURCES

Restoration techniques that increase the abundance of prey available to a seabird population would act primarily to increase productivity and the survivorship of adults and young. However, as stated in Chapter 13, altering the environment for the benefit of seabird populations may, at the same time, negatively affect populations of other species. Before implementing projects that would enhance the productivity of seabird food resources, managers must seriously consider whole ecosystem consequences of such activities.

Manipulating the seabird food environment by augmenting forage fish populations or reducing competition by large predatory fish might be an effective strategy in some circumstances for facilitating the recovery of injured seabird populations. In the following we identify some conditions under which such efforts are most likely to be successful:

1. Smaller-scale interventions are more likely to be effective and have predictable outcomes than are larger-scale, ecosystem-level interventions. For example, the creation or maintenance of local nearshore spawning habitat for forage fish or the management of a specific forage fish hatchery is a small-scale operation that may increase local prey abundance. The deliberate overfishing of a predatory fish competitor of a seabird species is a larger-scale community- or ecosystem-level technique that may result in unpredictable or undetectable results, due to unexpected trophic interactions or time lags (Butterworth *et al.* 1988).
2. The probability of designing effective food-related restoration projects for seabirds is directly related to our knowledge of the trophic structure within the local marine ecosystem. Understanding the causes of the variability in both seabird and forage fish populations, and determining if and how these populations are limited, will determine what types of manipulations might be employed. However, we stress that understanding systems does not necessarily indicate that restoration techniques are readily available, and the use of a known technique can lead to unsatisfactory results if the systems are not well known. For example, trophic-related restoration activities for the Aleutian tern, whose biology is poorly known, may owe any success to luck. Moreover, different species may

respond differently to food manipulation. If factors other than food availability, such as predation on adults or nest contents, are limiting recovery, manipulation of food may be futile.

3. The economic and biological value of intervention is likely to vary with the ecology of the species of birds and prey involved, the scale of the spill, and the rarity of the birds. For example, a small spill at the single colony of a rare species might merit extreme, intensive measures such as feeding young by hand or establishing feeding pools for adults. A larger spill affecting more common species might merit more extensive efforts, such as the establishment of intertidal spawning habitats or commercial fishing restrictions or modifications.

METHODS OF MANAGING FOOD RESOURCES

There are few data on the methods for, or the effectiveness of, managing food to facilitate restoration following oil spills or other perturbations. We offer the following as suggestions and strongly recommend that each of these ideas, as well as any others under consideration, be thoroughly researched before they are implemented.

Increasing Food Availability Through Reduction of Competing Consumers

Ecosystem-level interventions by humans are extremely common in marine systems, but are usually unplanned and have accidental consequences (May 1984). Interventions designed to increase food for seabirds by manipulating their competitors may also have unanticipated outcomes, so they should be used with caution and only where the marine system is reasonably well understood. Where there is competition between forage fish species (e.g., herring and pollock), it may be productive to harvest the species that is less desirable from the birds' point of view (pollock) in order to obtain an increase of the preferred forage species (herring). However, where the renewal rates of the competing forage fishes vary, one possible outcome is the extinction of both species (e.g., Tilman 1982, 1986). Without sufficient information about forage fishes and their competitors to enable us to understand the ecological processes, it is difficult to predict whether this technique would be efficacious or even detrimental.

Nevertheless, directed fisheries or culls of seabird competitors (e.g., pollock) for forage fish may be appropriate in certain conditions. For example, an increased pollock fishery might reduce predation on forage fish or reduce competition of young pollock with such species. Similarly, because hatchery-raised salmon may reduce the seabird prey base through competition (i.e., juvenile salmon) or predation (i.e., adult salmon), it might be appropriate to cut back on salmon hatchery operations or locate hatcheries where the salmon will not conflict with forage fish used by seabirds. Finally, reduction in human fisheries for forage fish such as herring or the exclusion of fisheries from the foraging zones of seabirds may be required in certain circumstances (Duffy and Schneider 1994), although at the current time there are no forage fish species, other than herring, that are commercially exploited in Alaska.

Modeling of the existing system and the effects of intervention on birds, the targeted fish species, and the ecosystem as a whole is a critical step before action is taken. For example, a simple and preliminary carrying-capacity model suggests that *juvenile* salmon are unlikely to compete with other forage fish in Prince William Sound (Cooney 1993), so that managing salmon hatcheries *to reduce competition with herring, capelin, or sand lance* may not be an effective strategy. However, if direct evidence or models indicate that hatchery-raised *adult salmon are competing with seabirds* for forage fish, managing salmon hatcheries may be an effective strategy in restoring certain seabird populations.

Improving Fish Habitat

Several high-latitude forage fish species, such as herring, capelin, and sand lance, spawn in the subtidal and intertidal areas that are highly susceptible to oil spills and their persistent effects (Pinto *et al.* 1984). Thorough cleaning of known or likely spawning sites may be critical, especially in deep, loose sediments, such as gravel or sand, where oil may persist. In addition, new intertidal spawning areas could be created (probably at considerable expense) near seabird colonies or in unoiled areas by placement of appropriate substrates, to increase surviving fish populations. In Alaska the herring roe fishery places tree branches and kelp in the water to facilitate harvest. If such measures increase spawning success, they also might be used as a management tool to increase herring populations. Finally, and most importantly, if existing and productive spawning beaches can be protected from development and the harmful effects of pollution, populations of forage fish may be maintained at levels sufficient for local seabird populations.

Reducing Bycatch of Nontarget Fish

Many commercial fisheries, especially net fisheries, take large quantities of nontarget species that are not marketable or desirable. In most instances, the nontarget species do not survive and are simply dumped overboard. Restoration planners should sponsor research to develop commercial fishing techniques that would decrease the bycatch of prey populations that are important to seabirds. Such techniques might include gear changes or modifications of the time of day that nets or longlines are set.

CONCLUSIONS

The science of managing forage fish resources for seabirds is still in its infancy, but it may prove in time to be an effective approach to restoring injured seabird populations. However, because these techniques are expensive and may produce unexpected results, they should be used with extreme caution. These techniques may be appropriate to use in restoring rare or endangered species or critical colony sites, when "heroic" efforts are warranted.

Directed research could provide the scientific underpinnings for managing food resources to enhance recovery. Experimental interventions in noncrisis situations and models of their effects on seabird populations (Migot 1992, Spendelow *et al.* 1995) could provide objective guidelines on use of such techniques, and we strongly recommend that this research be pursued.

Part E: Management of Seabird Habitats

SOCIAL ATTRACTION

Social attraction as a seabird restoration technique consists of the use of decoys and sound recordings of a particular species to attract recruits to a specific location or habitat. Most seabird species nest colonially, and the majority of recruitment occurs at locations where conspecifics are breeding or at least present. Social attraction lures potential recruits by mimicking a breeding colony or aggregation. It attempts to assist restoration by increasing immigration or directing recruitment to target locations. The technique assumes that an increase in immigration will not be deleterious to a source population or colony and that the probability and rate of natural colony formation are low.

The technique was first described as a management tool by Kress (1978), who used Atlantic puffin decoys in conjunction with translocation of chicks to re-establish puffins to Eastern Egg Rock, an historic nesting island in the Gulf of Maine. Social attraction (without translocation of chicks) was also used to establish a mixed colony of more than 1,000 pairs of common terns, Arctic terns, and roseate terns at Eastern Egg Rock (Kress 1983).

A variation of the technique (sound and artificial burrows) was used to attract Leach's storm-petrels to several historic nesting islands in Maine (Podolsky and Kress 1989b) and to attract endangered dark-rumped petrels to artificial burrows in the Galapagos Islands (Podolsky and Kress 1989a). As a result of these studies, puffins, terns, and storm-petrels successfully re-established breeding colonies that continue to grow or have remained stable more than 10 years after recolonization.

Social attraction offers managers the possibility of reducing risks to seabird populations that are concentrated on one or a few islands. While the technique works by recruiting immigrants, immigrants are typically inexperienced birds or, rarely, experienced birds relocating due to disturbance (Kress and Nettleship 1988, Divoky and Horton 1995). The technique requires a pool of potential recruits that are not philopatric. Seabirds vary in their level of philopatry, and within a species philopatry can be expected to vary with local conditions (Waser 1985). In general, the success of social attraction programs can be expected to be proportional to the species' level of dispersal. Terns, which display high levels of breeding and natal dispersal, have been the subject of several social attraction programs (e.g., Kress 1983, Hall 1995). For example, social attraction, typically employed in conjunction with other restoration techniques,

has been used to successfully restore colonies of several species of terns in various locations: (1) common terns in Upper St. Lawrence River, Ontario (H. Blokpoel, pers. com.) and in Duluth-Superior Harbor on Lake Superior (Matteson 1986); (2) roseate terns at Ram Island, Massachusetts (Harlow 1995); and (3) least terns at many sites, including California (Rigney and Emery 1980, Anderson 1981) and New Jersey (Kotliar and Burger 1984).

While decoys and recordings have been used most often to establish waterbird colonies at historic nesting islands, they may also be used to improve productivity through shifting recruitment to potentially higher-quality breeding habitat. This is the rationale behind an effort to encourage the endangered short-tailed albatross to shift nest sites from the side of an active volcano on Torishima Island, Japan, where albatross chicks and eggs are often buried in ash, to level habitats away from the volcano. After decoys and recorded sound had been used for four years, the first pair of short-tailed albatross nested among the decoys on the more secure habitat in 1996 (H. Hasegawa, pers. com.). Presumably, over the long term, pairs that nest away from the loose-ash substrate will have greater productivity than those that nest on the ash.

Social attraction can be facilitated through the concurrent use of other restoration techniques such as predator control, translocation, or reducing human disturbance (Kress 1983, Kress and Nettleship 1988, Hall 1995). The need for social attraction techniques in conjunction with other techniques may depend on specific biological or political circumstances. Habitat or nest-site creation and predator removal in a region where a species is common might result in relatively rapid establishment or growth of a colony without the need for social attraction. This has been the case in Alaska at islands where predators have been removed (Bailey 1993). In areas where densities of prospecting birds are low, social attraction may be needed for a species to find and prospect a given location. In addition, social attraction can be used to help establish a colony within a time period acceptable to project managers. The success of a social attraction program for terns, or the time period before recolonization occurs once a program has been established, may be related to the time interval between the abandonment or extirpation of the colony and the implementation of the program. This relationship may be a function of the presence of individuals that had bred previously at the site.

Social attraction programs generally require recruits from pre-breeding age cohorts. However, populations classified as nonrecovering could *not* be expected to have large pools of recruits resulting from immigration, recent high breeding productivity, or increased postfledging survival. Thus, in nonrecovering populations, social attraction would have to be used only for a specific purpose (such as re-establishing a high-profile colony), not for the goal of increasing numbers of the *total breeding population*.

Social attraction would probably not be appropriate where recruitment and productivity are already sufficient and there is no reason to believe that seabirds would benefit from additional colonies or subcolonies. This appears to be the case with the depleted common murre colonies on islands in the northern Gulf of Alaska, where productivity for surviving birds remains high. In addition, social attraction would not be practical for noncolonial nesting species with dispersed breeding, such as loons, harlequin ducks, and scoters.

The technique may be detrimental if used to establish birds on locations or habitats where recruits would experience increased risk of predation or disturbance. In summary, social attraction offers managers an opportunity to establish or re-establish colonial waterbirds at additional sites. New colonies may benefit the population by spreading the risk of catastrophic events and by expanding ranges back to earlier limits. However, natural colony formation occurs regularly in many seabird species without the need for social attraction (Divoky and Horton 1995).

NEST SITE IMPROVEMENT OR PROVISION

Nest site improvement techniques (e.g., vegetation control, nest box provision) are best used at colonies where nest sites are limited or where the quality of breeding habitat is suboptimal (and thus thought to be reducing recruitment or productivity). These techniques act to increase the recruitment of birds by creating attractive recruitment opportunities and to increase productivity by providing sites and habitats that, due to increased cover for eggs and chicks, support increased breeding success. Because breeding dispersal is low for most seabird species, the majority of individuals occupying new sites would be expected to be first-time breeders.

The utility of nest site improvement and provisioning for nonrecovering populations is unclear. Populations that are not recovering from oil spill mortality would be expected to already have an excess of sites or habitat available to them (the sites and habitat previously occupied by the part of the population removed by the spill). In such situations nesting habitat improvement or creation may serve mainly to increase the breeding productivity of individuals remaining after the mortality.

Vegetation control has been conducted in order to improve breeding habitat of surface nesting species, including common and least terns (S. Schubel, pers. com.), roseate terns (M. Tasker, pers. com.), and burrow-nesting species. Control of dense ground cover that precludes burrowing by ground-nesting species can free additional areas that otherwise would not be used (J. Takekawa, pers. com.). Vegetation control can be conducted using a variety of methods, including mechanical techniques (e.g., handpulling, hand tools), chemical agents, and limited-controlled grazing (if habitat allows).

Nest boxes have been used by a variety of cavity-nesting species (Priddle and Carlile 1995), including rhinoceros auklet (Wilson and Manuwal 1986), Cassin's auklet, tufted puffin, pigeon guillemot, ashy storm-petrel (Ainley and Boekelheide 1990), horned puffin (Divoky 1982), and black guillemot (Divoky *et al.* 1974). Nest boxes can increase productivity by providing protection from native predators (e.g., gulls and crows) or preventing the collapse of burrows due to erosion, marine mammals, or human activity. In addition, nest boxes can provide nesting habitat in areas where nest sites are limited. The provision of nest boxes in Arctic Alaska, where natural sites are extremely limited, resulted in the increase of a black guillemot colony from 15 pairs to 210 pairs in less than a decade (G. Divoky, pers. com.).

The construction and design of nest boxes can be varied to produce attractive and productive sites. The diameter of the entrance can be modified to select for certain species, and baffles and angled entryways can be used to reduce predation. Nest boxes can be designed to allow investigators easy access to nesting cavities for species that typically nest in inaccessible situations. This can provide access to nest contents and breeding adults that would facilitate the assessment of a range of demographic parameters important in postspill monitoring.

The creation of nesting ledges to increase recruitment is less common. The nests of a black-legged kittiwake colony in Great Britain that was the subject of a long-term study (Aebischer and Coulson 1990) were on the window ledges of a dockside warehouse. Recent research on black-legged and red-legged kittiwakes on the Pribilof Islands showed that both species readily occupied manmade wooden ledges attached to rock cliffs (D. Kildaw, pers. com.).

HABITAT CREATION

The creation of nesting platform islands is a technique used when habitat availability is limited or increased recruitment is desired. In general, nesting platform islands are constructed and designed to meet the habitat requirements of individual species. Artificial nesting platforms built on posts made of wood and metal have been used by double-crested cormorants (USFWS 1983). Also, certain seabird species (western gulls, pelagic cormorants, double-crested cormorants, Brandt's cormorants, least terns, and pigeon guillemots) will nest opportunistically on such structures as abandoned lighthouses, navigational lights, bridges, abandoned piers, duck blinds, and electrical transmission towers (Carter *et al.* 1992). Nesting islands created from dredge-spoil deposits have also been used by a number of species including least terns, ring-billed gulls, Forster's terns, and Caspian terns (J. Hainline, pers. com.; J. Albertson, pers. com.).

HABITAT PRESERVES

The designation of an area as a habitat preserve serves to safeguard habitats or areas that are of critical importance to seabirds. For seabirds this almost always involves the preservation of nesting sites, although coastal roosting sites are frequently included in wildlife refuges. While the designation of an area as a preserve does not directly lead to the increase of a nonrecovering population, it does ensure the availability of essential breeding habitat in the future. Most U.S. habitat preserves for seabirds are owned by the federal government and are in the Department of Interior's National Wildlife Refuge system. There are, however, examples of private preserves (e.g., The Nature Conservancy). Aside from ensuring that the habitat protected will be available and undisturbed in the future, the designation of an area as a reserve allows the restriction of human activities that could disturb breeding (e.g., hiking, motor vehicle activity). Habitat preserves also facilitate the management and monitoring of seabird populations.

LAND PURCHASE

The purchase of land from private owners is considered to be a viable restoration option. Land purchased through restoration funds can benefit seabird populations in the same way as designation of habitat preserves can.

Part F: Supplement Wild Populations

CAPTIVE REARING

Captive rearing has proven an effective technique for reintroducing peregrine falcons and other raptors (Cade and Temple 1977) and has been attempted for a variety of other rare birds, such as *Amazona* parrots (Nichols 1977) and whooping cranes (Erickson 1976). It is generally agreed to be a technique of last resort due to high cost, potential genetic problems, and the complicated reintroduction procedures that it may require.

With the exception of the publicly appealing penguins and puffins, seabirds are notably absent from most zoos because of the expense necessary to maintain aquatic exhibits with ample size, water circulation, and temperature controls. Penguins are the best-represented seabird in zoos, but the number of zoos and aquaria with the specialized and expensive habitats necessary for alcids are increasing (e.g., Oregon Coast Aquarium, Newport, Oregon; Hubbs-Sea World, San Diego, California). In a 1991 review of the status of alcids in captivity, Gunther (1991) found that only eight of the world's 22 species of alcids were represented in the collections of the nine largest aquaria in the United States. To date there has been relatively little success at breeding auks in captivity. Tufted puffins are the notable exception and are considered a routine breeder at most aquaria that keep this species.

The release of captive-raised birds may directly increase the pool of potential recruits for a nonrecovering population, but there are a number of drawbacks to raising seabirds in captivity. The cost per released individual is excessive, and, except for a population that has been reduced to a few birds, the release of captive birds is likely to be insignificant compared to local production. Because the captive-raised birds would be released as young, they could be expected to suffer the high first-year mortality characteristic of seabirds and might experience higher than normal mortality.

Captive breeding would be most practical for species having no postfledging parental care; it seems of limited value for species with precocial young or species whose young have extended parental feeding and care. For example, common murre chicks typically leave the colony at 21 days and remain with the male parent for 60 to 85 days. Therefore, murres are an unlikely candidate for captive breeding. However, Fry (1991) and Kress and Carter (1991) have proposed a method of raising and releasing common murre young to the wild. Finally, captive-raised birds

may also show philopatry to the breeding facilities. At present, there are no seabird breeding facilities large enough to cost-effectively produce birds for release.

TRANSLOCATION

Translocation (the relocation of nestling seabirds) is similar to captive breeding except that the young released to the wild are the progeny of seabirds breeding in the wild rather than in captivity. Potential limitations of this technique include the need to feed young for extended periods, provide ample food supplements (e.g., vitamins, minerals), transport them from collection to release sites, provide quarantine and rearing facilities, and prevent undue human contact that could lead to young birds imprinting on humans. These same limitations would apply to captive breeding. Compared to most other techniques (except captive breeding) translocation is expensive, frequently involving long-distance transportation of chicks, high labor and logistics costs, and long-term monitoring (10-plus years) to determine the outcome of the project. Because mortality is especially high for the fledging period, survival to breeding is low, with less than 50% of most cohorts reaching breeding age. This high mortality necessitates moving large numbers of young. This is especially important since survival of young to breeding age varies from one year to the next due to marine conditions, such as available food and weather, that are beyond the influence of seabird managers.

There has been few long-term translocation of seabird chicks. Fisher (1971) relocated fledgling Laysan albatross and found that many returned to their original hatching location as they approached reproductive age. Likewise, Serventy (1967) moved fledgling short-tailed shearwaters, but these also eventually returned to their hatching place. In both experiments, chicks moved at earlier ages tended to return to the release site rather than to their hatching locality.

More recent work with translocation demonstrates that young Atlantic puffins, translocated as downy chicks, will eventually return to nest at their release site. In this experiment, chicks were translocated from Great Island, Newfoundland, to Eastern Egg Rock and Seal Island National Wildlife Refuges in Maine (Kress and Nettleship 1988). Returning translocated chicks recolonized both islands eight years after commencement of translocation, and currently both colonies have stable or increasing populations.

In summary, the life histories of many seabirds preclude translocation as a potential restoration option. For those species whose life history allows translocation, the technique is most appropriate where natural recovery is unlikely or where less intense techniques, such as social attraction, are not feasible. Translocation should not be attempted unless a large source population is available that could contribute ample young without impacting the source. This technique requires long-term funding and adequate effort dedicated to monitoring in order to reliably evaluate the outcome of the project.

REHABILITATION

Rehabilitation of seabirds that have been oiled by a spill or by chronic pollution is conducted routinely. With some exceptions, the low success rate of rehabilitation and the low survival rate of released birds make it primarily a humanitarian rather than a restoration technique. Live birds are collected, cleaned, fed, and, when they are healthy (as determined by a veterinarian), released back into the wild. The percentage of individuals that survive to release varies depending upon the species, time of year, and type of oil encountered. Sharp (1996) showed that the survival rate for rehabilitated Northern Hemisphere alcids is low. Most importantly, there is no information indicating that the few birds that do survive are able to enter or return to the breeding populations. A study conducted on postrelease brown pelicans resulted in no individuals returning to the breeding population (Anderson *et al.* 1996). In contrast with the relatively poor success rate for alcids and pelicans, 50% of the rehabilitated African penguins oiled during the *Apollo Sea* spill of June 1994 were found breeding (i.e., adults with eggs or chicks) on Dassen Island, South Africa, in 1995 (T. Williams, pers. com.). Furthermore, some rehabilitated cormorants have successfully bred following the Persian Gulf oil spill (P. Symens, pers. com.). It appears that the success of rehabilitation may be species- and locality-dependent; the technique may be useful in some situations for some birds.

To date, rehabilitation is generally not accepted as a viable restoration technique for alcids because few of the oiled birds brought to rehabilitation centers will ever return to the breeding population, and the rehabilitation technique (collecting, cleaning, feeding, etc.) is expensive (but is usually paid for by the party responsible for the oil spill). Rehabilitation has been proposed as a restoration technique in California (Fry 1991). In extreme cases (e.g., endangered species) this technique may be used as a last resort with rehabilitated animals placed in captive-rearing programs. Additional research on rehabilitation techniques is currently under way and may increase survival rates (and decrease costs), making this a viable restoration alternative in the future.

CHAPTER 10

RESTORATION TECHNIQUES: ASSUMPTIONS AND DEFICIENCIES

ASSUMPTIONS

All seabird restoration techniques are based on assumptions about seabird biology or the supporting marine and terrestrial environments. Managers using any restoration technique must assume that demographic parameters can be manipulated though human intervention and that such interventions will benefit the population. *Each technique requires the manager to assume that constraints on the demographic factor being manipulated are restricting the population's rate of growth and that manipulations removing the constraints will significantly increase population growth.* The attractiveness (i.e., potential for success) of any technique to seabird managers is related to the degree to which the underlying assumptions can be supported by scientific evidence or the technique's history of success.

DEFICIENCIES

When judging the utility of any restoration technique or when comparing techniques, deficiencies associated with each technique need to be considered. Deficiencies that will act to lessen the appeal of any technique include:

1. high financial costs
2. excessive or extended labor or logistics
3. continuing enforcement
4. stakeholder resistance
5. potential public or political opposition
6. potential negative impacts on ecosystem or nontarget species
7. low probability of success

Other deficiencies include uncertainty about the utility of a technique, difficulty in determining the success of the technique, and minimal changes in the parameter being modified. The deficiencies listed above are typically situation-specific. For instance, predator control might meet major opposition in an urban area but could be conducted with little opposition in less populated areas. However, if predator control were exceedingly costly or logistically unfeasible, the technique might be abandoned regardless of location.

TECHNIQUE-SPECIFIC ASSUMPTIONS AND DEFICIENCIES

There are three demographic parameters that can be manipulated to restore seabird populations: recruitment of nonbreeders into the breeding population, breeding productivity, and survival. We discuss below the assumptions and deficiencies associated with manipulating each of these parameters. Deficiencies associated with specific techniques are presented in Table 2 (in Chapter 2) and in those sections of this report discussing techniques.

Recruitment-Enhancing Techniques

Assumptions

Recruitment-enhancing techniques presuppose the existence of a pool of nonbreeders, some of which would be recruited only if biologists employ the restoration techniques. They also assume that these birds will be of greater benefit to the population if recruited to the target location than if recruited elsewhere. Deferred maturity is common in seabirds, with age at first breeding for a species dependent on evolutionary, local, or annual conditions (Ashmole 1971, Lack 1967). Techniques that encourage recruitment could act to influence birds to breed at a younger than normal age. Additionally, encouraging recruitment could also increase rates of immigration since immature birds disperse more widely than adults and since nonbreeders commonly visit nonnatal colonies (Harris 1983, Halley and Harris 1993). These techniques also require managers to assume that prey availability or other factors are not restricting recruitment.

Deficiencies

As a group, recruitment-enhancing techniques have a major problem: a pool of surplus nonbreeders is not always present. Populations depleted by an oil spill and not naturally recovering, or having a low probability of recovering, may not have a large pool of nonbreeders needing encouragement to reproduce. Timing of the spill-related mortality could affect the size of the pool of potential recruits. If the mortality affected all age classes (as might be the case for a fall or winter spill), the pool of nonbreeders would also be reduced. If the mortality involved primarily breeding adults (as might be the case for a spring or summer spill) and breeding productivity in the years immediately preceding the spill was high, a substantial pool of nonbreeders could be expected to be present. In this latter case, however, postspill recruitment would be expected to be naturally high, especially if the prespill population had been nest-site limited.

Recruitment-enhancing techniques may also entice birds to attempt to reproduce at locations that may not be able to support a population or a larger population. For example, recruitment-enhancing techniques would not be effective, or may even be detrimental to a population, if they are attempted at breeding sites that are already food or nest-site limited. Recruitment-enhancing

techniques may also reduce numbers at source colonies if the birds recruited to the restoration constitute a significant fraction of nearby healthy colonies.

Productivity-Enhancing Techniques

Assumptions

Restoration techniques that increase breeding productivity have the benefit of acting on birds remaining in the population after a spill-related mortality has occurred. They assume, however, that human activities can have a measurable and substantial effect on hatching and fledging success. There is little information to support this assumption for seabirds. Use of restoration techniques that attempt to increase productivity assumes that factors affecting postfledging survival are not density-dependent and that postfledging mortality will not increase with increased fledging success.

However, several techniques whose principal effect is to act on recruitment or survivorship can also secondarily increase breeding success. Thus, the principal effect of predator eradication may be to increase recruitment, but a secondary effect is to increase the breeding productivity of the entire population.

Deficiencies

The principal deficiency associated with productivity-enhancing techniques is the lack of evidence that humans can influence breeding success sufficiently to achieve significant population restoration. It is difficult to establish the success of these techniques. Annual variation in breeding productivity can be high and may be influenced by various factors (such as a warm-water event or other oceanographic or climatic conditions) that could mask the effects of any restoration technique. Additionally, low survival of prebreeding birds can offset any changes in fledging success.

Mortality-Reducing Techniques

Assumptions

When sources of mortality for adult birds are known, especially if the sources are anthropogenic, managers can reduce or control them as part of a restoration plan. Techniques that attempt to reduce mortality assume that the mortality is additive and that the population is not nest-site or food limited. Mortality-reducing techniques have an advantage over other techniques because they act directly on the most critical segment of the population.

Deficiencies

Mortality-reducing techniques have fewer generic deficiencies than either recruitment- or productivity-enhancing techniques. If successful, they act directly on breeding population size. However, some techniques may meet with stakeholder or public resistance regardless of efficacy; these include reducing subsistence harvests of seabirds and removing introduced species (e.g., foxes). Some techniques, such as reducing chronic pollution, are hard to monitor.

SITUATION-SPECIFIC CIRCUMSTANCES

The potential utility of a technique in restoring a population depends on the status of the injured population, the nature and magnitude of the injury, the reason for the lack of natural recovery, and the carrying-capacity limitations of the ecosystem. Prespill research, including baseline monitoring and studies of breeding and feeding ecology, may provide information on which factors have limited population growth in the past and the ability of the population to recover from decreases. This information is important, not only in ascertaining the need for active restoration, but also in selecting the class of restoration techniques that is most appropriate. For example, if breeding success was low in the years immediately preceding a spill-related mortality, the use of techniques that seek to increase recruitment may be impractical. Similarly, if a population was nest-site limited in the past and oil spill mortality reduced the breeding population, providing nest sites at the depleted colonies, by itself, would be unnecessary because many prespill nest sites could be expected to be vacant.

The quality of the pre- and postspill data and the rigor of the analysis of those data needs to be considered. Preliminary data analysis suggested that delayed phenology and lack of synchrony occurred in the three breeding seasons following EVOS. Although common murre breeding did appear delayed in 1989-91, compared with prespill years, by 1993 at colonies on the Barren Islands and Puale Bay, breeding phenology and productivity appeared to have returned to normal (Boersma *et al.* 1995, Piatt and Anderson 1996). Any hasty attempts to influence synchrony and phenology at these colonies would have been a waste of funds, and the Trustee Council was correct in rejecting such attempts.

CHAPTER 11

RESTORATION TECHNIQUES: MODELS

INTRODUCTION

Models can be useful tools in designing restoration programs because they can often detect signals in noisy systems. Properly used, they can assist in understanding the fundamental processes of seabird biology, in evaluating the progress of a restoration program, and in eliminating management or restoration activities that would result in biologically improbable scenarios. For example, analysis by Nur *et al.* (1993) revealed that adult survival of Brandt's cormorants was positively correlated with a well-established index of prey availability. This information was then incorporated into a population-dynamic model of this species by Nur *et al.* (1994), who demonstrated that if food availability could be improved, both adult survival and reproductive success might be enhanced.

Population models have the advantage of not relying on intuition, which can lead a restoration planner astray. For example, increasing the reproductive success of breeding birds may seem like a useful restoration action, but in many cases the effect will be negligible because other factors have overriding effects on population growth. A population model can explicitly demonstrate and quantify this point.

There are several caveats that must be expressed with regard to the use of models. Because models are an attempt to describe and account for the interrelationships in extraordinarily complex systems that have no physical boundaries, there will be instances in which predictions from models are incorrect. It is appropriate to trust model results in order to understand fundamental processes and to gain insights into how ecosystems operate. However, when results and predictions are used for management decisions, they should be verified or "ground-truthed" whenever possible with direct observations. In addition, any model used to design restoration programs should be readily available so that the public can make independent analyses. This should not pose a problem. There are several age-structured population models, for example, that are readily available either commercially or as shareware or freeware, such as RAMAS/age (Ferson and Akcakaya 1990) and RAMAS/metapop (Akcakaya 1994).

One use of models is to compare a variety of restoration techniques to help select the most cost-effective restoration option. With any specific set of parameters, models can assist in evaluating the relative "restorative value" of possible approaches. By running models on competing options, managers can evaluate potential projects and avoid those with little promise. For example, some models suggest that survival (adult, subadult, and/or juvenile) may be the key to population regulation, thus focusing restoration projects on increasing survival.

Models can guide research to help form testable hypotheses, identify areas where more data would be useful, and interpret results. Models can also assist in identifying what information is needed, although experienced restoration managers often will not need model results for this purpose.

PREREQUISITES FOR USING MODELS

In order for a model to be useful, it must be well designed, with adequate data used as model inputs. For many seabird species injured by EVOS, essential data from that vicinity are limited or unavailable. Occasionally a colony is difficult to access, so that obtaining high-quality data from that location is impossible. This obstacle can be overcome by using biological information from other regions or conducting a study at a convenient location within the spill area. For example, the life history and demographics of double-crested cormorants are well known from locations in Washington and California, even although they are poorly studied in Alaska.

The common murre is one of the few EVOS-impacted species of seabirds for which sufficient data seem to be available to run demographic models. However, this species, in many respects, has been poorly studied at the EVOS area. The drawback of using substituted data is that demographic characteristics may vary significantly with distance from the injured population. Thus, the validity of the model's results may ultimately hinge on the assumption that different colonies have similar demographic profiles. The sensitivity of conclusions to the assumptions being made can itself be, and should be, explicitly modeled. With respect to common murres, the available data from distant colonies are quite variable. Thus, demographic models may not be useful in assessing how the EVOS spill affected common murres because demographic data for common murres from the region are sparse, and we may not have confidence in results derived from substituted data.

For many species, density-dependence (i.e., demographic parameters, such as fecundity, survival, or recruitment, are functions of population size or density) must be incorporated into the model in order to make accurate predictions. For example, where nest-site availability is limited, this limitation acts as a negative density-dependent factor and reduces recruitment (Birkhead and Furness 1985). High population density may result in saturation of high-quality sites, thus forcing new recruits to take up lower-quality sites where reproductive success is reduced. Positive density-dependence is an important factor in a wide range of species at low population densities. In the common murre, reproductive success increases with density at the colony (Birkhead 1977), apparently due to better protection from predators at high density than at low density. Thus, density-dependence is an important factor that should be incorporated into a model.

ASSUMPTIONS UNDERLYING USE OF A MODEL

Models require many assumptions. Central to the use of any model is the assumption that the model inputs are applicable to the situation. As noted above, using demographic data for common murres from the Farallon Islands or Scotland in a model for the EVOS area assumes that those data

are appropriate for murres in the Gulf of Alaska. The modeler must constantly be aware of the maxim "garbage in garbage out," although models lend themselves to evaluation of assumptions.

It is also important to use the correct model. In recent years population models that incorporate stochasticity (i.e., variation due to chance effects) have become common, resulting in a model that is probabilistic rather than deterministic. The reasons for developing stochastic, probabilistic models are manifold. The first is that nature is stochastic; not only is the environment unpredictable, but so are demographic responses to the environment. More realistic and accurate predictions can be made if stochasticity is incorporated. A second reason is that without a probabilistic framework, no sense of variability of outcome is possible.

A modeler should always state explicitly the assumptions in the model. This allows others to evaluate the reasonableness of the assumptions and to test the sensitivity of predictions to changes in model assumptions. For example, a model may assume the presence (or absence) of density-dependence. A different type of assumption may relate to the efficacy of restoration action. For example, following an oil spill, even if one has good estimates regarding the number of oiled birds that are treated and released alive, little is known about the subsequent fate of these "rehabilitated" birds. One can compare predictions regarding impact and subsequent recovery from a spill assuming that (1) some or all rehabilitated birds die within a specified period, (2) a fraction of rehabilitated birds survive but never successfully breed, (3) a fraction of rehabilitated birds survive, and their breeding is impaired only for the immediate breeding season, or (4) some combination of these possibilities. Finally, for some oil spills (e.g., *Apex Houston, Nestucca, Exxon Valdez*), models have been developed to estimate total injury to seabird populations resulting from the spilled oil. Because restoration activities may be based on these estimated or modeled injuries, any evaluation of the efficacy of a particular restoration activity should also include an evaluation of the model's assumptions.

TYPES OF MODELS

Restoration managers can select many types of models as tools in restoration planning or implementation. This section briefly discusses (1) demographic models to predict population changes, (2) stochastic/deterministic/probabilistic models, and (3) sensitivity analysis. Several other types of models could be used depending on the specific situation, including economic models (cost-effective approaches to demographics), statistical models, models of marine trophic systems, and conceptual models.

Demographic Models to Predict Population Growth or Decline

Population dynamics takes a central role in formulating and evaluating a restoration plan. For restoration to succeed, some change in population dynamics must be achieved, either at the level of an entire population or at the level of a subpopulation. Consequently, demographic models can be a useful tool for predicting population growth or decline.

The number of adults is a function of seven population parameters or variables: (1) adult survival, (2) subadult survival, (3) juvenile survival, (4) reproductive success per breeder, (5) probability that an adult will breed, (6) age of first breeding, and (7) net immigration. The significance of this formulation for restoration is that a program will attempt to increase one or more of these parameters, except age at first breeding (which might be decreased). The program can then be judged by considering which parameter or combination of parameters is targeted and the efficacy of the program in altering that parameter.

One difficulty with this approach is that knowledge of subadult and juvenile survival is fragmentary for most seabird species. There is little information about whether subadult survival varies from year to year, and whether there is a correlation between adult and subadult survival. Estimates of survival based on capture and recapture-resighting are biased because of dispersal. Because subadults and breeding adults are usually found in separate areas, different mortality influences may be at work. Juvenile survival seems to vary greatly among populations. Four different population estimates for first-year survival in common murres varied from 0.47 to 0.67 (Nur 1993). In addition, Hudson (1985) provided a list of survival-to-breeding age for Atlantic common murres that ranged from 0.17 to 0.41. Such a large variation in first-year survival or survival-to-breeding age will impact population growth trajectories. Population modeling for common murres on the Farallon Islands showed that 40% juvenile survival results in average population growth of 1.1%, whereas 60% juvenile survival results in a population rapidly growing at the rate of 8% per year (N. Nur, unpubl. data). Hatchwell and Birkhead (1991) modeled the Skomer population of common murres and concluded that a change in juvenile or subadult survival was the major factor explaining why the population grew in the 1980s but not in the 1970s.

Population growth of seabird colonies is undoubtedly influenced by immigration and emigration. Models of single populations have de-emphasized the role of immigration and emigration because it is difficult to incorporate into the usual age-structured or unstructured models. In contrast, immigration and emigration are an explicit part of metapopulation models, so these parameters cannot be ignored (Burgman *et al.* 1993). Emigration is difficult to study because individuals are leaving the study colony and death is hard to distinguish from emigration. The number of immigrants can sometimes be quantified, but the pool from which they come is much harder to identify. A review of population recovery of marine birds shows that immigration played a role in many growing populations (Nur and Ainley 1992).

Species vary in the tendency of young and adults to disperse. Terns and cormorants, for example, show much dispersal, even among breeding adults, while storm-petrels and fulmars have high degrees of philopatry. Within a species the possibilities include (1) a complete absence of interchange between neighboring colonies, (2) wide dispersal so that all colonies in a geographical area are completely mixed and function as a single population, and (3) an intermediate situation between the two extremes. In order to define the boundaries of a population, analysis of DNA or of morphometric characteristics of a population may be feasible (see Chapter 3a). For some species, color phases can help define the boundaries between populations.

In sum, demographic models may be useful because they can set bounds around possible outcomes when population parameters are manipulated and can provide insight into fundamental

demographic relationships. However, demographic models are useful only when input parameters are sufficiently well known to allow reasonable bounds to be set for possible outcomes. Unfortunately, for many seabird populations, such input parameters are not sufficiently well known.

Stochastic/Deterministic/Probabilistic Models

Population dynamics models consider all population parameters together. A variety of such models are available (Emlen 1984, Caswell and Macdonald 1993). Whether to use a deterministic model, which ignores chance effects, or to include stochasticity to develop a model that is probabilistic depends on the question that is being asked. One problem with deterministic models is that they do *not* accurately predict average response. Instead, greater environmental and demographic variability depresses population growth rates (Boyce 1992). Moreover, the deleterious effects of random events are strongest for the smallest populations, such as those colonies that are just forming, have been decimated, or are being restored. Where a large population is being modeled, the additional complications of including genetic and random demographic events in a model may not be warranted.

The stochastic population model for the Farallon common murre by Nur *et al.* (1994) predicted that on average the population would grow by 1.1% per year. However, as a result of "error" associated with the stochasticity, the model also predicted that in the face of a very variable, unpredictable environment, there was a 10% chance the population would shrink by 21% or more, or would grow by 53% or more, after 10 years.

The effects of stochasticity on populations have been categorized in four parts (Shaffer 1981, Lande 1993). First, even in identical environments the genetic makeup of two populations will differ due to genetic drift and founder effects, which affect vital rates. Second, random demographic events will affect the number of adults surviving in a finite population from year to year. Random events can skew any population, especially small ones. Third, demographic parameters can vary in any one year due to environmental fluctuation such as excellent or poor feeding conditions. Fourth, environmental catastrophes, while rare and drastic, can occur.

Probabilistic analyses that incorporate demographic and environmental stochasticity form the basis for population viability analysis (Boyce 1992). These analyses can be useful tools for a restoration manager.

Sensitivity Analysis

Sensitivity analysis should be conducted in most modeling exercises. When the modeler varies the assumptions in a model to learn the effect on the model's predictions, it will become apparent which assumptions are critical to the outcome of the model. This evaluation allows the modeler to decide whether particular assumptions drive the model results and helps to evaluate the efficacy of the model.

USING A DEMOGRAPHIC MODEL TO COMPARE RESTORATION TECHNIQUES

This section provides an example of how demographic models can be used to compare restoration techniques. Under this approach, a restoration manager can select between two projects. For example, the first project, if successful, will increase the reproductive success of common murres by 10%. The second project will increase the survival rate of adult common murres by 1%.

A population model for common murres developed by Nur *et al.* (1994) determined that a 2.8% increase in common murre survival produced population growth of about 1.0-1.1%. In contrast, a 10% increase in murre reproductive success resulted in a population growth of only 0.8%. In this instance, the model's comparison of the efficacy of the two techniques would lead a restoration manager to concentrate efforts on small improvements in adult survival instead of moderate improvements in reproductive success. However, the restoration manager should also be aware of the fact that the model is more sensitive to changes in adult survival than to changes in reproductive success. That is, a relatively small error in estimating survival (e.g., survival is actually 0.93 instead of 0.94) will cause a wider range in the estimated population growth than a larger error in estimating reproductive success (e.g., 0.75 instead of 0.85). Comparisons such as this will vary depending on the relative importance of the demographic parameter being compared and the life history of the species whose demography is being modeled.

Where models show that adult survival is the key factor in restoring an injured seabird species, restoration managers can then focus on how to increase adult survival. Depending upon the specific circumstances, fisheries management practices could be changed to lessen bycatch of the injured population. For example, state or federal regulations could restrict the type, location, or season available to gillnet gear or longlines. In addition, Congress could establish a marine sanctuary that might close key areas to activities that decrease adult survival of an injured species.

CHAPTER 12

ECOSYSTEM-LEVEL FACTORS THAT MAY AFFECT RESTORATION OF SEABIRD POPULATIONS

INTRODUCTION

Some marine bird populations have been reduced by specific and persistent factors: the introduction of pesticides into the food web (e.g., DDT and the brown pelican, Anderson and Gress 1983); oil pollution (e.g., common eiders in the Rhine River, Camphuysen 1989; common murres on the Farallon Islands, Ainley and Lewis 1974); hunting (e.g., thick-billed murres in the eastern North Atlantic, Gaston and Elliot 1991); or the introduction of mammalian predators to breeding islands (e.g., rats, cats, mongooses, and humans and the dark-rumped petrel on Hawaii, Olson and James 1982, Harrison 1990). Other populations have been reduced by more systemic factors, such as curtailment of prey availability (e.g., the Peruvian guano birds, Murphy 1981, Tovar *et al.* 1987a).

The question is, can depleted seabird populations be guided back to their former state?

We argue herein that establishing restoration goals relative to a baseline defined by historical, or even the immediate, pre-impact population size is, at best, an illusory concept in marine systems. Because seabirds are positioned high on the trophic pyramid, they integrate and are sensitive to the ecological and food-web processes in the systems "beneath" them (e.g., Furness *et al.* 1993). Unfortunately for the purposes of restoration, marine systems are characterized by marked decadal variation (e.g., Longhurst *et al.* 1972, McGowan 1990), and may be too big or affected too greatly by large-scale physical processes (e.g., Sherman and Alexander 1985, Sherman *et al.* 1990, 1993) for humans to guide specific restorative processes toward a predictable goal. Thus, factors at play in the larger system most likely will dilute any isolated, local factors that may have affected trends in a seabird colony or population.

GENERAL CONSIDERATIONS

Factors That Limit Population Size: Is Competition Involved?

The size of breeding populations of seabirds is thought to be controlled by either space for nesting, the amount of prey available to foraging parents, or the amount of prey available to the population during the time of year when food is least abundant or accessible, usually the winter (Lack 1966, Furness and Birkhead 1984, Cairns 1992). These factors are further discussed below

Limitation by nesting space

In highly productive areas where there are few nesting islands, such as eastern boundary currents, the availability of nesting space is clearly the factor limiting the size of breeding numbers (but see Chapter 3b for different view). Such seabird communities are structured by competition for nesting habitat (Duffy 1983, Duffy *et al.* 1984, Ainley and Boekelheide 1990). In these situations, "floating populations" develop and are composed of birds that are capable of breeding but that lack breeding space (Manuwal 1974b, Ainley and Boekelheide 1990, Migot 1992). Any growth in numbers is limited to this nonbreeding portion of the population; the breeding population remains stable, being constantly sustained by the pool of waiting nonbreeders. Ultimately, the size of the total population (breeders and nonbreeders) may be limited by food or by a series of catastrophic mortalities resulting from periodic food shortage (e.g., mediated by repeated El Niños; Nelson 1968, Schreiber and Schreiber 1989).

Where nesting space is limited, breeding populations increase only by the provision of new nesting sites. In the Benguela Current, vast nesting platforms were constructed to increase the numbers of guano-producing Cape cormorants and Cape gannets, and provision of artificial burrows has increased numbers of breeding African penguins (Crawford and Shelton 1978, Crawford *et al.* 1995). In the Peru Current, natural predators were eliminated or excluded by fencing to open new nesting areas. After these areas were colonized, there was a manyfold increase in the numbers of guano-producing seabirds—Peruvian pelican, piquero and guanay (Duffy *et al.* 1984). In the California Current, artificial nesting cavities were used readily by several species of seabirds—especially pigeon guillemot and Cassin's auklet—during a period of population expansion (Ainley and Boekelheide 1990). More recently, as the carrying capacity of that environment has decreased (Hill 1995, Roemmich and McGowan 1995: see below), the use of artificial nesting habitat has also decreased (Point Reyes Bird Observatory [PRBO], unpubl. data). Another example exists in Prince William Sound, where a glacier receded in the 1960s, exposing a rocky island; by 1985 more than 5,000 seabirds were nesting there (Hogan and Irons 1988).

Limitation by food during breeding

Where breeding sites are numerous but food is not especially abundant, such as in the waters around the British Isles, the size of breeding populations is controlled by the amount of food available to parents fending for chicks (Furness and Birkhead 1984, Cairns 1992). These seabird communities appear to be structured by the competition for food. The mechanism by which this structuring could come about is the reduction in reproductive capacity owing to the local depletion of prey (e.g., Birt *et al.* 1987) or interference competition (Gaston *et al.* 1983, Hunt *et al.* 1986). In a "natural experiment" in the North Sea, seabird populations declined as stocks of their usual prey (sandeels) declined, possibly related to the heavy commercial fishing pressure (Furness 1982, 1984a, 1984b, 1989; Hammer *et al.* 1991, Phillips *et al.* 1996).

The availability of food to foraging parents may also limit the size of breeding populations of tropical, oceanic species (Ashmole 1963). There, and in other situations, the size of a breeding population of a given species relative to population size of others in the breeding assemblage is

in proportion to the area of foraging habitat available: species feeding inshore have small populations compared to the large populations of species feeding in the practically limitless offshore habitats (Diamond 1978) The major limit to the amount of ocean area available is the flight range of nesting birds (e.g., Pennycuick *et al.* 1984) and the capacity of chicks to forgo frequent feedings.

In polar regions, where food is very abundant for a short period and where nesting space seems unlimited as well, the physical severity of the climate may be an overriding factor in population regulation. At the least, the severe climate often masks the ultimate regulatory factors because, owing to birds' need to escape from particularly inclement weather, breeding areas are greatly separated from wintering areas. For example, a lessening of pack-ice cover—in a sense an indirect change in the accessibility of food—may have mediated changes in populations of nesting penguins and other species in the normally icebound areas of Antarctica (Ainley and Sanders 1989, Taylor and Wilson 1990, Fraser *et al.* 1992). On the other hand, a change in food supply thousands of kilometers away on the ice-free wintering grounds may have mediated changes in breeding population size among murres in the northeastern Bering Sea (Murphy *et al.* 1985).

Limitation by food during winter

Seabird populations may also be regulated by food in the season when it is least abundant, which is usually the nonbreeding season, or winter (Lack 1966). Direct evidence for this is sparse because, as indicated in the above paragraph for murres in the Bering Sea, wintering grounds of seabirds are often far away from breeding sites. Nevertheless, during winter (1) numerous "wrecks" or mass mortalities of seabirds have been documented (e.g., Richdale 1957, Bailey and Davenport 1972, Birkhead and Hudson 1977, Piatt and van Pelt 1993); (2) the most difficult time for immatures occurs (e.g., Harris 1983, and virtually any long-term demographic study); and (3) elevated levels of adult mortality are detected (e.g., penguins, Richdale 1957; gulls, Bergman 1982, Spear *et al.* 1987; alcids, Hudson 1985).

Vader *et al.* (1990) documented a sudden decline of common murres at colonies in northern Norway in 1987, and attributed this decline to a food shortage resulting from a collapse of capelin stocks in the Barents Sea. They hypothesized that either the breeding population of common murres died from starvation during the winter of 1986-87, or that adults were unable to build up enough energy during the winter and spring, and abandoned all breeding attempts. An unusual late winter wreck of shags in eastern Britain was caused by a prolong period of onshore winds, possibly resulting in a food shortage (Harris and Wanless 1996). This wreck was composed equally of adults and immatures, and affected all colonies along at least a 100-kilometer stretch of the coast. Harris and Wanless (1996) reported that this wreck was unprecedented in the number of adult birds killed, causing a reduction in annual survival rates from a normal 88% to 13%; the Isle of May shag population may require 10 years to recover from the decline.

The role of competition

Ultimately then, competition, as a result of resource limitation, could well affect the size of a population. However, in a depleted population—a candidate being considered for restoration—*intraspecific* competition should not be a problem (although under stable conditions, competition should be most intense within a species; Birkhead and Furness 1985). *Interspecific* competition for food among seabird species has rarely been demonstrated, except possibly interference competition on the part of sooty shearwaters and other species (Hoffman *et al.* 1981), interference competition between lesser black-backed gulls and herring gulls (Noordhuis and Spaans 1992), and shifts in foraging behavior of ducks (Pöysä 1986). Interspecific competition for food between seabirds and other organisms, such as whales (in the southern oceans, Beddington and May 1982, and the Bering Sea, Springer and Roseneau 1985), may be another matter, but in recent decades whales have been depleted and, therefore, should have no negative effect on seabird food availability. Seabirds may also compete with upper-trophic-level predatory fish in certain systems, (e.g., the Bering Sea, Springer 1992, or the North Sea, Furness 1984a). When the exploited fish is a competitor, its reduction in prevalence increases the availability of prey to seabirds (e.g., Bering Sea auklets that feed on the zooplankton eaten by pollock). Conversely, when the exploited fish are seabird prey, the seabirds are negatively affected (e.g., Bering Sea murres that feed on juvenile pollock; Springer 1992).

In the case of nesting space, competition among cavity-nesting seabirds has been mediated through body size (e.g., Bédard 1969a, Ainley and Boekelheide 1990), but seabird species tend to nest where others are already nesting, with little evidence for outright exclusion (e.g., Whittam and Siegel-Causey 1981). On the Farallon Islands, as common murre and Brandt's cormorant populations recovered during the 1970s, the high density of murre nesting groups allowed them to move the larger cormorants, but the cormorants then displaced western gulls, who did not nest densely enough to cope with the cormorants' greater body size (Ainley and Boekelheide 1990). In the northwestern Hawaiian Islands, if chicks of the smaller, winter-breeding Bonin petrel remain late, they are displaced by the larger wedge-tailed shearwaters when the latter return to initiate breeding in the same cavities (Harrison 1990). In the Azores, the smaller little shearwater breeds during the winter, possibly to avoid competition for nesting space with the larger Cory's shearwater, which nests during the summer. For the same reasons, morphologically distinct forms of the band-rumped storm-petrel breed in the summer and fall (Monteiro *et al.* 1996), and morphologically and taxonomically distinct forms of the Leach's storm-petrel breed in opposite seasons on Guadalupe Island (Ainley 1980).

One other aspect of the competition for food and space concerns the concept of source versus sink populations (see Chapter 3). Where there is plenty of food but limited nesting space, colonies become source populations because the easiest avenue by which individuals can recruit into the breeding population requires emigration elsewhere. On the other hand, where limited food or too much predation exists but there is plenty of space, such colonies may become sinks. Individuals recruit to them, but breeding success is low and the colony cannot be sustained without immigration from source populations. The latter is the case among South Polar skuas at Cape Crozier (Ainley *et al.* 1990). The colony there is the largest for this species in the world,

but it is sustained by immigrants attracted to plentiful food. Breeding success, however, is almost nil at Crozier, owing to intense storms that blow away eggs and chicks.

In addition to structuring nesting dispersion on the temporal scale, competition can also structure the spatial dispersion of nesting. Around the British Isles (Furness and Birkhead 1984) and in Antarctica (Ainley *et al.* 1995a), competition for food has been identified as a factor that prohibits large colonies of certain species from occurring in proximity to one another. Instead, only small colonies, often in a cluster around the larger colony, can exist nearby. Emigrants from the large (source) colony initially colonize these peripheral localities, and the entire cluster of colonies must be viewed as the "population" (or metapopulation; see Chapter 3).

Whether a colony is a source or a sink—or, for that matter, whether a population is food or space limited—is not necessarily static. Over an extended period, colonies may switch from being a source to a sink or vice versa. For instance, the sudden availability of nesting space (retreat of a glacier, construction of a dredged materials island, etc.), which is then colonized, leads to a sink initially. If the site is especially favorable (e.g., good feeding opportunities, free of predators, free from disease, etc.), it could grow to become a source colony within a few generations. For instance, not long ago in northern Europe and in North America the herring gull was food limited, but with the growth of human refuse dumps and the increased availability of fishing offal, the population has become space limited (e.g., Migot 1992). As other examples, populations of ring-billed gulls in the Great Lakes region and guano birds in Peru grew in response to increased food until they became so dense that disease (botulism and parasites, respectively) began to increase mortality (Blokpoel and Scharf 1990, Duffy 1991).

FACTORS THAT LIMIT OR ENHANCE RECOVERY

Temporal and Spatial Scale of Perturbation

Scale has much to do with the relationship of populations to their resources (Schneider 1994). Where short-term, localized mortality has affected a satellite colony in a larger metapopulation, colonies often recover their former size in a few years, owing to adequate resources more widely distributed than the perturbation and to recruits from the source colony. A classic example is provided by European shags on Farne Island. In 1968, they experienced a red tide and the population crashed from 350 to 75 pairs (Potts *et al.* 1980). However, within six years, the numbers recovered completely as a result of both immigration from nearby colonies (not affected by red tide) and recruitment of individuals who were otherwise not breeding due to a prior lack of breeding space.

Another example is that of the Peruvian guano birds. Before the 1970s, there was a large "floating" population composed of individuals precluded from breeding by lack of a breeding site in spite of periodic large-scale mortality due to lack of food (caused by El Niño). Once the food web was re-established, *breeding* populations recovered too quickly to be the result of renewed breeding success (Murphy 1936, Tovar *et al.* 1987a; although D. Duffy, pers. com., indicates that recovery might be the result of high productivity and not a large "floating"

nonbreeding segment of the population). A third example is offered by Stowe (1982), who noted a decrease in numbers of murres at colonies immediately following an oil spill nearby. Within a year, however, the colonies returned to or even exceeded prespill numbers. The adults who had not been breeding, plus recruits from juveniles who had recently matured, accounted for the increase. The same can be said of rapid recovery from a short-term mortality experienced by rhinoceros auklets on the Farallon Islands.

Conversely, where there is long-term, pervasive mortality that affects all generations of a metapopulation, recovery requires decades to complete. For example, the Farne Islands shag population, following release from a long period of persecution, grew for 45 years beginning with a decade of virtually no growth. Another example is offered by king penguins on Macquarie Island. They had been hunted to the point of near-extinction, and recovery took about 80 years (Rounsevell and Copson 1982). In the first 20 years following the cessation of hunting, the population showed no growth. This pattern was due to the fact that Macquarie Island is very isolated, and little if any immigration was possible from other colonies or metapopulations. Therefore, all growth had to be intrinsic. In the case of the slow-maturing wandering albatross (it first breeds at 12 years of age), which has been declining at many disjunct breeding colonies for several decades owing to mortality of females from entanglement in fishing gear (Weimerskirch and Jouventin 1987, Croxall *et al.* 1990), prospects for a fast recovery are similarly low should the fishing mortality be curtailed. Finally, a population can persist in a depleted state due to the cumulative effects of impacts that occur one after another, as is the case of murres on the Farallon Islands. In the late 1800s, uncontrolled commercial egging reduced the population from an estimated 400,000 birds to 40,000, followed by several decades of oil pollution and other disturbance, which reduced the population still further, to 6,000, by the 1960s. Control of impacts allowed the population to grow to almost 100,000 by the early 1980s, but gillnetting and oil spills then reduced the population again to 40,000 birds (Ainley and Lewis 1974, Ainley and Boekelheide 1990). A similar example is offered by penguins in southern Africa. First diminished by disturbance (mining of guano) and exploitation for eggs, penguin populations, upon protection from these impacts, were reduced still further by a series of oil spills and finally overfishing of their prey by commercial fisheries (Crawford *et al.* 1995).

Food Availability

The population dynamics of seabirds track food availability more than any other ecological factor (Furness and Monaghan 1987, Cairns 1987, Montevecchi 1993, and others). For example, when predatory fish were heavily fished in the North Sea, their prey (sand lance) bloomed and seabird numbers increased, but when numbers of sand lance crashed at Shetland, so too did the reproductive success of many seabirds and in some cases their population size (Furness 1982, 1984a, 1989; Monaghan *et al.* 1989a, 1989b; Hamer *et al.* 1991, Phillips *et al.* 1996). Decline in North Sea herring stocks was associated with a decline in kittiwake reproductive success (chicks fledged per pair) and a decline in population growth rate (Coulson and Thomas 1985). A similar seesawing of prey availability as a function of fishery pressure is perhaps also being played out in the Bering Sea with respect to pollock. Seabirds that feed on small pollock are declining (e.g.,

murres), but those that feed on the zooplankton prey of pollock are increasing (e.g., auklets; Springer *et al.* 1986, Springer 1992).

The Peruvian guano birds represent another and perhaps better example: overfishing of anchoveta, the mainstay in the diet of many predators, in conjunction with environmental stress, caused a crash in the fish stocks and, in turn, a dramatic decline in numbers of boobies and cormorants and in the ability of these seabirds to recover from periodic El Niños (Tovar *et al.* 1987a). El Niño in 1957 (and earlier episodes) caused crashes in guano bird populations, followed shortly by full recovery. Then, beginning with the onset of overfishing of the anchoveta, following each subsequent El Niño (1965, 1972) each seabird population recovery was weaker than the preceding one—that is, the carrying capacity of the environment began to change.

Other examples of how food availability influences recovery are provided by common murres. On the Farallon Islands, the murre population partially recovered during the 1970s and 1980s, having been released from the effects of disturbance and chronic oil pollution, as noted above (Ainley and Lewis 1974, Ainley and Boekelheide 1990). Between 1982 and 1986, however, the murres in the entire central California metapopulation were subjected to heavy mortality from gillnets, a severe El Niño, and two oil spills (Takekawa *et al.* 1990). As a result, the populations of all colonies crashed. Although these perturbations did not extend beyond 1986, none of the colonies in the metapopulation have shown substantial recovery since, in spite of high breeding success (Ainley *et al.* 1994). Coincidentally, many of the murres' prey species are being fished intensively, with fishing pressure increasing dramatically during the 1970s and 1980s (Ainley *et al.* 1994). Lack of recruitment in the fish populations owing to changed oceanographic conditions, too, is likely involved (see below).

The lessening of carrying capacity may also explain why the Farallon murre populations in the 20th century have never come close to recovering their mid-19th-century size, which was 400,000 birds (cf. Ainley and Lewis 1974, Takekawa *et al.* 1990, Ainley *et al.* 1994). As pointed out by Roemmich and McGowan (1995), major changes in the California Current after the mid-1970s resulted in a dramatic reduction in zooplankton biomass. At the same time, fisheries were growing dramatically (Ainley *et al.* 1994). As a result, the trophic environment is different now compared to that of former years. Not only breeding species, but nonbreeders such as the sooty shearwater, have demonstrated depressed populations in response to the poorer feeding conditions (Ainley *et al.* 1995c, Veit *et al.* 1997). Similarly, the "recovery" of the brown pelican in California is stalled—that is, it is stable and self-perpetuating but below the level reached during pre-DDT years (reviewed in Ainley and Hunt 1990). If sustained, a resurgence of the sardine in California (Barnes *et al.* 1992, Wolf 1992), formerly the primary prey of pelicans in California (MacCall 1984), may encourage a period of renewed growth in the pelican population.

The recovery potential of a seabird community can also be affected by changes in the quality, or species composition, of the prey base. The classic example is provided by seabirds in southern Africa (Crawford and Shelton 1978). There, fisheries caused regional shifts in the distribution of various fish species and, in turn, the breeding distributions of some seabird species. The same type of shift has been documented in Peru and Chile beginning in the 1970s. To some degree,

the depleted anchoveta were replaced by sardines, but only in an area poleward of the previous anchoveta concentration (Bakun and Parrish 1982, Parrish *et al.* 1982). Seabird populations shifted southward too, although not in the same numerical mix of species that occurred to the north (Tovar *et al.* 1987b). This shift southward repeated the same pattern that occurred between the 16th and 19th centuries (Hutchinson 1950). Similarly, the northward shift of northern gannets in Newfoundland was correlated with a warming of the Labrador Current (and a northward shift as well in mackerel, the birds' main prey; Kirkham and Montevecchi 1982).

Influence of Disturbance on the Breeding Effort

Unnatural disturbance can also negatively affect the recovery potential of seabird breeding colonies. Often, an effective management practice promoting recovery of seabirds, and perhaps one of the few practically available, is protection of breeding colonies from egging, hunting, and disturbance. Scores of species have benefited from protection, including king penguins on Macquarie Island (Rounsevell and Copson 1982), Laysan albatross on Midway Island (Rice and Kenyon 1962), northern gannets in Newfoundland (Brown and Nettleship 1984), cormorants in eastern North America (Buckley and Buckley 1984), common murres on the Farallon Islands (Ainley and Boekelheide 1990), and Atlantic puffins in Europe (Harris and Wanless 1991).

The disturbance does not have to be direct, as indicated by cases in which breeding colonies are abandoned owing to the threat of predation by mammals. Even if few or no eggs, chicks, or adults are taken, adults refuse to begin their breeding efforts once a mammalian intruder is spotted. An example is abandonment of nesting islands in Mono Lake, California, by California gulls when landbridges allowed the entry of coyotes onto the breeding islands (Winkler and Shuford 1988; W. Shuford, pers. com.). Another example is abandonment by murres when weasels and foxes make their way over sea ice to offshore islands in Newfoundland (Birkhead 1993, Birkhead and Nettleship 1995). Finally, growing populations of pinnipeds, if present on islands during the seabird breeding season, can disrupt the breeding efforts of the birds (Shaughnessy 1984, Warheit *et al.* 1984, Warheit and Lindberg 1988, Ainley and Boekelheide 1990). Recently in southern Africa, pinnipeds have been fenced from the breeding areas of African penguins to allow recovery of penguin populations depleted by oil spills (Crawford *et al.* 1995).

The response of seabirds to disturbance and their recovery once disturbance is controlled is not always a simple matter. For instance, the number of Adélie penguins at the small colony at Cape Royds, Antarctica, decreased during the period 1956-63, ostensibly due to too-frequent visits by tourists (Thomson 1977). When human visitation was reduced, the penguin population began to increase by 3-4% through 1973. Interestingly, the period of decline at Cape Royds coincided with a decline of Adélie penguin numbers at the region's major (source?) colony, Cape Crozier, where tourist visits were not occurring (cf. Ainley *et al.* 1983, Taylor *et al.* 1990). In more recent years, the rate of increase at Cape Royds has grown to 5% per year, an adjacent colony has been re-established after a hiatus of over 80 years, and all colonies in the region have been increasing, including the one at Cape Crozier (Taylor and Wilson 1990). More penguins now breed at Cape Royds than in any other time in recorded history In retrospect, it is now obvious

that the period of decline at Cape Royds coincided not only with uncontrolled disturbance from human visitation but also with a period of cold weather and more dense pack-ice on the sea (thus increasing the energy cost of reproduction by limiting access to open water). The population increase was aided by a period of warming during which time the source colony of the metapopulation was also expanding rapidly (Taylor and Wilson 1990).

Effects of Predators and Kleptoparasitism on Population Growth

Interactions among seabird species, specifically the predation of small species by large ones (e.g., gulls, skuas) or the kleptoparasitism of food by large species (gulls, skuas, frigatebirds) against smaller ones, have led to adaptations in life history patterns. For instance, a species may be diurnal or nocturnal on the breeding grounds, with most small species being nocturnal; or a species may nest in the open or in cavities, with most small species being cavity nesters (e.g., Lack 1968). Defense against larid kleptoparasites likely also influences the dense coloniality of seabirds, attaining its extreme in the murres (e.g., Wittenberger and Hunt 1985, Burger and Gochfeld 1990). Spear (1993) has shown how annual variation in nesting density, among other things, in murres and cormorants can allow greater kleptoparasitism by gulls. Buckley and Buckley (1984) and others for eastern North America, and Mathiasson (1980), Parslow (1967) and others for Europe, chronicle the depressing effect that large populations of large-bodied gulls have on smaller larids, terns, and puffins.

Paine *et al.* (1990) have shown the potential direct and indirect effects of avian predators on the relative species composition of a breeding seabird community. On Tatoosh Island, Washington, as peregrine falcon populations recovered (from decimation owing to DDT pollution), those authors surmised that predation by the falcons led to decreased numbers of Cassin's auklets and rhinoceros auklets, two important prey species of the falcon. Thus, the auklet populations may have been artificially high for a few decades owing to the absence of an important predator. However, another important prey of the peregrine is the northwestern crow. The resurgent falcons reduced the crow populations as well, and this contributed to increased numbers of black oystercatchers, murres, and cormorants, which were parasitized by the crows. In this case, the size of the oystercatcher, murre, and cormorant populations may have been artificially low owing to the short-term absence of a biological control—predation by falcons—on kleptoparasitic crows.

Control of avian predators was a necessity in restoring tern and puffin populations in the Gulf of Maine (summarized in Buckley and Buckley 1984). Gull predation and kleptoparasitism was an important factor that limited the occurrence of tern and puffin breeding colonies. Well documented was the fact that the populations of common, roseate, and Arctic terns had decreased in New England during this century as the populations of gulls had increased. As the gulls were removed from islands (on the order of 10,000 birds) through poisoning and other means, the terns and puffins returned, aided by social attractants (Kress 1982, 1983; see Chapter 9e).

Changes in Climate

It has been claimed repeatedly that seabirds occur where their prey species are (e.g., Ashmole 1971), and as reviewed above, changes in prey composition can lead to regional changes in seabird distribution and numbers. In most cases, however, the situation may be more complex. In spite of decreased availability of nesting habitat (coastal wetlands) and increased disturbance from humans, colonies of elegant and royal terns, as well as of black skimmers, have been founded and numbers have increased in southern California during recent years. The recent invasion of royal terns, in fact, represents a return to their status of several decades previously (cf. Grinnell and Miller 1944, Ainley and Hunt 1990). The black skimmers have continued to spread northward and now nest in San Francisco Bay for the first time in recorded history (S. Terrill, pers. com.). Interestingly, black skimmers on the eastern coast of North America also have been invading (re-establishing) northward (Buckley and Buckley 1984).

Coincident with these northward shifts of southern species in California, the distributions of northern species have been retreating northward. Tufted puffins and common murres no longer breed in the southern California Channel Islands (one or two pairs at best for the puffin), although they did so in appreciable numbers near the turn of the century (cf. Hunt *et al.* 1980, Carter *et al.* 1992). Human-caused factors (e.g., oil spills) may have exterminated the colonies, but these colonies might have been remnants of previously larger colonies, as the old breeding sites have not been recolonized even after control of the degrading influence. During the past few decades, the ocean off California has been warming (Roemmich and McGowan 1995), and this factor may be involved in the faunal shifts in California. Of course, such a process has been played out on geologic time scales as well (e.g., Warheit 1992).

In Antarctica, as the environment has warmed and pack-ice cover has diminished in the past few decades, faunal shifts have occurred. In the Antarctic Peninsula, populations of chinstrap penguins and blue-eyed shags and, perhaps, brown skuas (along with sub-Antarctic elephant seals) have been increasing and colonizing areas southward; southerly species have been retreating (Ainley and Sanders 1989, Fraser *et al.* 1992, Ainley *et al.* 1995a). On the other hand, in the most southerly reaches of Antarctic seas (e.g., the Ross Sea), populations of the southerly breeding Adélie penguin have been increasing and spreading southward (Taylor and Wilson 1990), conceivably as diminishing pack-ice cover has provided greater access to food resources.

Ultimately, climate change also alters the availability of food in a region (e.g., Roemmich and McGowan 1995). Where this factor is involved, it is not so much a faunal shift that results, but rather a depression or enhancement of seabird population growth. Ainley and Lewis (1974) noted decadal changes in Cassin's auklet population size on the Farallon Islands, 1880s to 1970s, in response to climate-related changes in food availability—a pattern that is recurring in recent times (Ainley *et al.* 1994, Hill 1995). Aebischer *et al.* (1990) noted decadal changes in the environment, food, and seabirds in the North Atlantic, and Veit *et al.* (1997) have noted even more closely coupled changes in these factors in the California Current.

The decadal changes noted by Roemmich and McGowan (1995) are part of a shift in climate that has occurred since the mid-1970s and has produced changes in marine communities and seabird numbers throughout the north Pacific region, from north Pacific central waters (Venrick *et al.*

1987, Polovina *et al.* 1994) to the California Current (Ainley *et al.* 1995b, Veit *et al.* 1997), Gulf of Alaska, and Bering Sea (Springer 1992, Hollowed and Wooster 1995). All levels of the marine ecosystem have been affected. The importance of this shift is that restoration of seabird populations to levels attained before the mid-1970s is not possible under existing conditions.

IMPLICATIONS FOR THE RESTORATION OF SEABIRD POPULATIONS

In defining restoration or choosing restoration goals, when applied to seabirds (see Chapters 6, 10), it is important to consider whether or not the species' capacity to increase or at least maintain its numbers is a viable one. Then, if and when natural environmental factors cycle back to an earlier condition, population growth of the seabird population is likely. *Therefore, what needs to be restored as a minimum is the capacity of a population to respond to improved trophic and other environmental conditions* (see Chapter 6 for discussion of restoration goals). In that sense, then, the brown pelican, currently on the U.S. Endangered Species List, could be considered restored, as the population is no longer severely depressed and breeding success again fluctuates with natural fluctuations in food availability (Ainley and Hunt 1990).

Most important, as this review has revealed, food availability (and ultimately climate change) is the dominant factor in affecting long-term growth of depleted seabird populations. Managing for food availability, however, is a very difficult, problematic, and highly political alternative because the only practical way to do so is to curtail commercial fisheries. To date, only if seabirds on the Endangered Species List are at risk has attention been given to management of marine living resources to protect the endangered species (e.g., the anchovy management plans *vis à vis* brown pelicans; U.S. Department of Commerce 1978).

Much easier and better demonstrated is the option of managing predators and kleptoparasitic species to encourage recovery of seabird populations (e.g., Kress 1982, 1983). Breeding habitat can be restored (e.g., for albatross on Midway Island; Rice and Kenyon 1962), and disturbance to breeding islands can be controlled. These management tools require just the decision to implement them, with relatively minor political or social issues to consider.

Restoration of seabird populations to levels attained earlier this century, at least in the north Pacific, is likely impossible under present conditions. Climate change and dramatic changes in prey resources now negate such a goal. In isolated cases, even restoration of individual populations may not be attainable because of climate change (and effects on prey availability) after the population was decimated (e.g., restoring brown pelican breeding colonies to Monterey, California, or restoring common murre breeding colonies to the California Channel Islands). Each restoration project has to be evaluated with this possibility in mind.

CHAPTER 13

ECOSYSTEM CONSIDERATIONS IN SEABIRD RESTORATION

The successful restoration of a seabird species requires not only information on population structure and demographics (see Chapters 2 and 3) but also data on how ecosystem-level factors may affect restoration (see Chapter 12). In addition, because seabirds are part of both marine and terrestrial ecosystems, changes in their local abundance associated with anthropogenic effects (such as oil spills) and with restoration activities stemming from these effects may have unanticipated consequences for the local environment. The linkages between seabirds and their marine and terrestrial ecosystems need to be more fully understood before biologists can assess how the *significant* reduction in the local abundance of seabirds, and the associated recovery of that population, will affect local ecosystems. The fact that seabirds may affect the abundance and distribution of terrestrial or marine biota is important in understanding the role of seabirds in community or ecosystem structure, and we must be mindful of these potential effects when designing oil spill restoration plans for seabirds.

In the following, we *briefly* summarize how seabirds may affect their local marine or terrestrial ecosystems. This précis is not meant to be comprehensive and is included only to ensure that such issues are considered in the design and implementation of restoration plans. We emphasize here the terrestrial component of seabirds. This is done not to minimize the importance of seabirds in marine ecosystems; on the contrary, much of this report focuses on seabirds as marine animals. Rather, there is little consideration in this report on how changes in seabird abundance can affect terrestrial ecosystems, and we are making a minimal attempt here to remedy this.

MARINE ECOSYSTEMS

Food Web

It is unclear how likely it is that changes in marine bird populations will disrupt or alter other elements of the marine food web. On the one hand, the proportion of fish stocks that seabirds are consuming is likely to be comparatively low (e.g., in the range of 20-30% of annual pelagic fish production; Furness 1978, 1984a, 1984b; Wiens and Scott 1975), and the effects of bird predation on a fish population may be less than that of particular fisheries (Furness 1984a, Cairns *et al.* 1991). However, there probably is competition for a potentially limited fish resource (Furness 1984a, MacCall 1984) among such apex predators as seabirds, large predatory fish, marine mammals, and humans, and fish consumption by seabirds may limit the consumption by other food-web components.

The degree to which seabirds affect fish stocks and the degree to which they compete with predatory fish, marine mammals, or humans are functions of at least geography and the species and age-classes of fish consumed by seabirds (MacCall 1984, Cairns *et al.* 1991, Cairns 1992).

In addition, Birt *et al.* (1987) have shown that at a relatively small temporal and spatial scale, double-crested cormorants can significantly deplete prey populations. Therefore, changes in seabird abundance resulting from oil spills or their subsequent recovery can affect food-web components, although it is not clear to what extent this actually occurs and at what temporal and spatial scale the effects are best seen.

Nutrient Enrichment

Seabirds affect the coastal marine and estuarine environments through their production of guano. Guano is rich in marine nitrogen and phosphorus and is an important fertilizer. Approximately 10^4-10^5 tons of phosphorus are produced annually by seabirds worldwide (in Polis and Hurd 1996), enriching plankton, intertidal, and terrestrial systems. Zelickman and Golovkin (1972) reported that primary production around bird colonies was increased and the composition and structure of neritic plankton communities were affected by guano-induced marine enrichment. In South Africa, nutrients introduced into intertidal systems by surface runoff from guano-covered islands or by direct deposit of guano from roosting seabirds enhanced macroalgal growth, producing cascading effects to intertidal community composition (Bosman *et al.* 1986, Bosman and Hockey 1988, Branch *et al.* 1987). Finally, MacCall (1984) hypothesized that phosphate laden runoff from the torrential rains that accompany El Niño in Peru may compensate partly for the reduced nutrients available owing to disrupted oceanic upwelling.

The systemwide effects of marine and intertidal fertilization by guano vary depending on local conditions and community interactions, and the increase or decrease in seabird densities may affect these systems through changes in guano production. Polis and Hurd (1996:412) argued that an increase in seabird-related fertilization "reticulates throughout the food web, producing bottom-up effects beyond increased primary productivity: consumers grow faster, to larger sizes, and increase their population biomass and density." However, Bosman *et al.* (1986) suggested that complex trophic interactions involving oystercatcher predation on limpets and enrichment of intertidal and nearshore waters by seabirds act to produce permanent and thick algal mats immune from grazing, thereby reducing the abundance of a primary consumer (i.e., intertidal limpet). Both Polis and Hurd (1996) and Bosman *et al.* (1986) agree that through the production of guano, seabirds can profoundly affect nearshore, intertidal, and terrestrial communities.

TERRESTRIAL ECOSYSTEMS

Vegetation

The effects of guano production on vegetative communities extend beyond the marine environment. Although they visit land only for a short period and generally remain close to the shoreline, the very dense concentrations of birds at some seabird colonies have an appreciable effect on local soils and vegetation (Furness 1991). In the Arctic, manuring by thick-billed murres, auklets, and dovekies creates local pockets of vegetation in otherwise barren areas. The effects of dovekies on terrestrial vegetation in Spitzbergen extend away from the colonies in

peripheral areas where nonbreeders circle and along the route taken by breeders to and from the sea (Stempniewicz 1990). Growth of vegetation as a result of manuring at the colony may make some breeding sites less attractive, resulting in a gradual shifting of the colony and further extending the nutrient-enriched area.

The moss *Dicranum groenlandicum* is a principal component of peat, laid down over thousands of years, on flat ground above thick-billed murre colonies on Coats, Digges, and Akpatok Islands in Hudson Strait (Gaston and Donaldson 1995). The peat provides a foothold for plants, such as cotton grass *Eriophorum*, that are otherwise absent from the clifftops. Such peat does not occur on clifftops elsewhere in the region, and apparently the flat ground above the murre colonies is enriched by excrement, food, etc., blowing up from the cliffs. On the colony cliffs themselves, where the instability of the rocks does not encourage the formation of peat, scurvy grass often forms luxuriant clumps up to 40 centimeters in diameter. On the colony at the Minarets, Baffin Island, it grows up to an altitude of 800 meters. Away from seabird colonies, scurvy grass seldom forms clumps more than 10 centimeters across and does not grow above 200 meters elevation in the Minarets area (A. Gaston, pers. obs.). In Greenland, several vascular plants occur only in the vicinity of seabird colonies at the northern limit of their range (Salomonsen 1979).

Seabird islands in low-Arctic and boreal waters may also be affected by the manuring and burrowing of auks. In the Queen Charlotte Islands, British Columbia, a distinctive tussock grass understory develops in forests subject to burrowing by rhinoceros and Cassin's auklets, but does not invade forest occupied only by ancient murrelets, which do not defecate in their burrows.

On some New Zealand islands, the burrowing and manuring activities of petrels aerate and fertilize the topsoils, creating favorable conditions for plant regeneration (Towns *et al.* 1990a). This promotes a series of cascading effects benefiting invertebrates, lizards, and the tuatara. Conservation and restoration biologists in New Zealand hope to restore this unique association of seabirds and terrestrial flora and fauna on the Mercury Islands (Towns *et al.* 1990a).

Erosion

The burrowing activities of auks have resulted in the gradual erosion of some colonies, making the islands uninhabitable for the culprits. This seems to have occurred on Grassholm Island, Wales, where a colony of tens of thousands of Atlantic puffins destroyed most of the soil layer sometime in the last century (Lockley 1953). Similar erosion has affected the puffin colony at the Farne Islands, England (Harris 1984).

Food Web

The dense aggregations of some seabirds that form at colonies provide a concentrated food source for predators. Adult birds of practically all species may be preyed on by terrestrial predators while breeding. Red and Arctic foxes are probably the most widespread and important predators. The concentration of Arctic foxes associated with the dovekie colonies in the Thule

District of northwest Greenland forms the basis for an important local fur-trapping industry that would probably not exist without the birds.

In subarctic Alaska and British Columbia, the local race of the peregrine falcon (*Falco p. pealei*) specializes in feeding on marine birds, and the smaller auks are its principal prey. Unlike most peregrines, but like Eleanora's falcon (Walther 1979), Peale's peregrines hunt mainly over the sea, skipping low over the wave tops and snatching ancient murrelets and Cassin's auklets either on the water or in the act of taking off. The social aggregations of ancient murrelets that occur at sea several kilometers from their colonies are much denser and more active on calm nights than when the sea is choppy, perhaps because calm conditions allow them to detect peregrines more easily (Gaston 1992). Peregrines nesting on coastal cliffs in northern Scotland also take many auks, apparently preferring Atlantic puffins (Ratcliffe 1980), while gyrfalcons may specialize in thick-billed murres at certain Arctic colonies (Gaston *et al.* 1985). See Chapter 12 for discussion of how predation by peregrine falcons may structure species composition of mixed-species seabird colonies.

Seabirds also provide links to terrestrial food webs by contributing terrestrial biomass to islands through fish scraps, dead chicks, feathers, eggs, and guano. For example, Polis and Hurd (1996) estimated that chick carcasses of Heerman's gulls and brown pelicans provide 17% and 18% of the terrestrial productivity per meter^2 on Isla Raza and Isla Piojo, respectively. This biomass helps support several trophic categories of arthropods and significantly influences the population dynamics of many terrestrial species (Polis and Hurd 1996).

Conclusions

As a consequence of the relations described above, seabird colonies develop unique associated terrestrial ecosystems of limited area. Seabird impact on plants has been investigated at several sites (e.g., Salomonsen 1979, Gaston and Donaldson 1995, Towns *et al.* 1990a), but their impact on arthropods and other invertebrates has been described at only a few colonies (see Towns *et al.* 1990a, Polis and Hurd 1996) and deserves further study. In any case, there is a need for careful consideration of associated terrestrial ecosystems in designing restoration plans for seabirds.

LIST OF SPECIES

PLANTS:
Scurvy grass (*Cochlearia officinalis*)

FISH:
Anchoveta (*Engraulis ringens*)
Anchovy (*Engraulis mordax*)
Capelin (*Mallotus villosus*)
Mackerel (*Scomber japonicus*)
Sandeel (*Ammodytes marinus*)
Sand lance (*Ammodytes hexapterus*)
Sardine (*Sardinops caerculeus*)
Walleye pollock (*Theragra chalcogramma*)

BIRDS:
African penguin (*Spheniscus demersus*)
Aleutian tern (*Sterna aleutica*)
Ancient murrelet (*Synthliboramphus antiquum*)
Arctic tern (*Sterna paradisaea*)
Ashy storm-petrel (*Oceanodroma homochroa*)
Atlantic puffin (*Fratercula arctica*)
Bald eagle (*Haliaeetus leucocephalus*)
Band-rumped storm-petrel (*Oceanodroma castro*)
Black guillemot (*Cepphus grylle*)
Black oystercatcher (*Haematopus bachmanni*)
Black skimmer (*Rhyncops niger*)
Black-billed magpie (*Pica pica*)
Black-legged kittiwake (*Rissa tridactyla*)
Black-tailed gull (*Larus crassirostris*)
Blue-eyed shag (*Phalacrocorax atriceps*)
Bonin petrel (*Pterodroma hypoleuca*)
Brandt's cormorant (*Phalacrocorax penicillatus*)
Brown pelican (*Pelecanus occidentalis*)
Brown skua (*Catharacta lonnbergi*)
Buller's shearwater (*Puffinus bulleri*)
Cahow (*Pterodroma cahow*)
California gull (*Larus californicus*)
Cape cormorant (*Phalacrocorax capensis*)

BIRDS (cont.):

Cape gannet (*Morus capensis*)
Caspian tern (*Sterna caspia*)
Cassin's auklet (*Ptychoramphus aleuticus*)
Chinstrap penguin (*Pygoscelis antarctica*)
Common diving petrel (*Pelecanoides urinatrix*)
Common eider (*Somateria mollisima*)
Common loon (*Gavia immer*)
Common murre (*Uria aalge*)
Common raven (*Corvus corax*)
Common tern (*Sterna hirundo*)
Cory's shearwater (*Calonectris diomedea*)
Dark-rumped petrel (*Pterodroma phaeopygia*)
Double-crested cormorant (*Phalacrocorax auritus*)
Dovekie (*Alle alle*)
Dusky seaside sparrow (*Ammodramus maritimus nigresens*)
Eleanora's falcon (*Falco eleonorae*)
Elegant tern (*Sterna elegans*)
European shag (*Phalacrocorax aristotelis*)
Fluttering shearwater (*Puffinus gavia*)
Forster's tern (*Sterna forsteri*
Gray jay (*Perisoreus canadensis*)
Great black-backed gull (*Larus marinus*)
Grey-faced petrel (*Pterodroma macroptera*)
Guanay (*Phalacrocorax bougainvillii*)
Gyrfalcon (*Falco rusticolus*)
Harlequin duck (*Histrionicus histrionicus*)
Heerman's gull (*Larus heermanni*)
Herring gull (*Larus argentatus*)
Horned puffin (*Fratercula corniculata*)
King penguin (*Aptenodytes patagonicus*)
Kittlitz's murrelet (*Brachyramphus brevirostris*)
Laysan albatross (*Diomedea immutabilis*)
Leach's storm-petrel (*Oceanodroma leucorhoa*)
Least tern (*Sterna antillarum browni*)
Lesser black-backed gull (*Larus fuscus*)
Little shearwater (*Puffinus assimilis*)
Magpie (*see Black-billed magpie*)
Manx shearwater (*Puffinus puffinus*)
Marbled murrelet (*Brachyramphus marmoratus*)
Northern gannet (*Morus bassana*)
Northwestern crow (*Corvus caurinus*)
Pelagic cormorant (*Phalacrocorax pelagicus*
Peregrine falcon (*Falco peregrinus*)
Peruvian pelican (*Pelecanus occidentalis thagus*)

BIRDS (cont.):
Pigeon guillemot (*Cepphus columba*)
Piquero (*Sula variegata*)
Pycroft's petrel (*Pterodroma pycrofti*)
Red-legged kittiwake (*Rissa brevirostris*)
Rhinoceros auklet (*Cerorhinca monocerata*)
Ring-billed gull (*Larus delawarensis*)
Roseate tern (*Sterna dougallii*)
Royal tern (*Sterna maxima*)
Scott's seaside sparrow (*Ammodramus maritimus peninsulae*)
Sharp-shinned hawk (*Accipiter striatus*)
Short-tailed albatross (*Diomedea albatrus*)
Short-tailed shearwater (*Puffinus tenuirostris*)
Sooty shearwater (*Puffinus griseus*)
South Polar skua (*Catharacta maccormicki*)
Spectacled guillemot (*Cepphus carbo*)
Steller's jay (*Cyanocitta stelleri*)
Thick-billed murre (*Uria lomvia*)
Tufted puffin (*Fratercula cirrhata*)
Wandering albatross (*Diomedea exulans*)
Wedge-tailed shearwater (*Puffinus pacificus*)
Western gull (*Larus occidentalis*)
Whooping crane (*Grus americana*)

MAMMALS:
Arctic fox (*Alopex lagopus*)
Black rat (*Rattus rattus*)
Cat (*Felis felis*)
European rabbit (*Oryctolagus cuniculus*)
Ground squirrel (*Spermophilus undulatus*)
Mink (*Mustela vison*)
Norway rat (*Rattus norvegicus*)
Pig (*Sus scrofa*)
Polynesian rat (*Rattus exulans*)
Raccoon (*Procyon lotor*)
Red fox (*Vulpes vulpes*)
River otter (*Lutra canadensis*)
Sub-Antarctic elephant seal (*Mirounga leonina*)
Wedge-capped capuchin monkey (*Cebus olivaceus*)

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PUBLISHED PROCEEDINGS OF SYMPOSIA OF THE PACIFIC SEABIRD GROUP

At irregular intervals the Pacific Seabird Group holds symposia at its annual meetings. The symposia are often compiled, edited, peer reviewed and published. Published symposia are listed below. Copies can be purchased from the Pacific Seabird Group.

SHOREBIRDS IN MARINE ENVIRONMENTS. Frank A. Pitelka (Editor). Proceedings of an International Symposium of the Pacific Seabird Group, Asilomar, California, January 1977. Published June 1979 in Studies in Avian Biology, Number 2. Out of print.

TROPICAL SEABIRD BIOLOGY. Ralph W. Schreiber (Editor). Proceedings of an International Symposium of the Pacific Seabird Group, Honolulu, Hawaii, December 1982. Published February 1984 in Studies in Avian Biology, Number 8. $12.00.

MARINE BIRDS: THEIR FEEDING ECOLOGY AND COMMERCIAL FISHERIES RELATIONSHIPS. David N. Nettleship, Gerald A. Sanger, and Paul F. Springer (Editors). Proceedings of an International Symposium of the Pacific Seabird Group, Seattle, Washington, January 1982. Published 1984 as Canadian Wildlife Service, Special Publication. Out of print.

ECOLOGY AND BEHAVIOR OF GULLS. Judith L. Hand, William E. Southern, and Kees Vermeer (Editors). Proceedings of an International Symposium of the Colonial Waterbird Society and the Pacific Seabird Group, San Francisco, California, December 1985. Published June 1987 in Studies in Avian Biology, Number 10. $18.50.

AUKS AT SEA. Spencer G. Sealy (Editor). Proceedings of an International Symposium of the Pacific Seabird Group, Pacific Grove, California, December 1987. Published December 1990 in Studies in Avian Biology, Number 14. $16.00.

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THE STATUS, ECOLOGY, AND CONSERVATION OF MARINE BIRDS OF THE NORTH PACIFIC. Kees Vermeer, Kenneth T. Briggs, Ken H. Morgan, and Douglas Siegel-Causey (Editors). Proceedings of a Symposium of the Pacific Seabird Group, Canadian Wildlife Service, and the British Columbia Ministry of Environment, Lands and Parks, Victoria, British Columbia, February 1990. Published 1993 as Canadian Wildlife Service, Special Publication, Ministry of Supply and Services, Canada, Catalog Number CW66-124-1993E. Free. Write: Publications Division, Canadian Wildlife Service, Ottawa, Ontario, K1A OH3, Canada.

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