Chapter 7

Seabirds

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7.1 INTRODUCTION

Seabirds are among the most numerous and widespread of the marine megafauna (Croxall et al., 2012). A single species can range from pole to pole with some having the longest known migrations to date. Seabirds are also among the most threatened groups of birds (Croxall et al., 2012), and many populations are in decline (Paleczny et al., 2015), making this group particularly important to consider when examining marine ecosystem structure and condition.

Here, we consider seabirds as defined by Gaston (2004), with some updated classifications as proposed by the International Ornithological Committee World Bird List (http://www.worldbirdnames.org/classification/orders-of-birds-draft-7-1/). This includes gulls, terns, skimmers, skuas, auks, and selected phalaropes (Charadriiformes), tropic birds (Phaethontiformes), penguins (Sphenisciformes), tubenoses (Procellariiformes), cormorants, frigate birds, boobies, gannets (Suliformes), and pelicans (Pelecaniformes) as the main species. While we focus on ecological issues, the issues impacting seabirds often affect other marine bird groups such as loons (Gaviiformes), sea ducks, and mergansers (Anseriformes), which often occupy the same marine habitat.

This chapter includes widespread ecological issues (e.g., sea level rise) and point source anthropogenic perturbations (e.g., interactions with fishing) affecting seabirds. Beyond the importance of seabird populations to biodiversity, human subsistence hunting (Falk, Merkel, Kampp, and Jamieson, 2006), and as ecosystem modifiers through nutrient subsidies (Honig & Mahoney, 2016), they are recognized as indicators of the condition of the marine environment. The use of seabirds to detect changes in marine ecosystems spans from fish stocks to plastic pollution, from changes in food webs to the global transfer of contaminants (Piatt, Sydeman, & Wiese, 2007). We aim to provide context and background on anthropogenic-related issues, such as those where the research community has a history of successfully implementing management strategies (e.g., hunting, incidental bycatch). The use by seabirds of both terrestrial and marine environments makes them vulnerable to many different types of ecological issues, such as the impacts of introduced predators on islands where seabirds breed. For further reading on terrestrial ecological issues, see Jones et al. (2015) and Towns, Byrd, et al. (2011).

7.2 CHANGES IN MARINE PREY

Seabirds feed almost exclusively in the marine environment, intrinsically linking seabird survival, reproduction, and condition to marine biota from phytoplankton to fish. Thus, changes in marine biota have direct impacts on seabird individuals and populations, and changes in seabird populations have been detected both under naturally oscillating regimes and in relation to anthropogenically induced climate change.

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7.2.1 Naturally Oscillating Ocean Regimes

Seabird populations are affected by naturally occurring marine regime shifts such as El Niño/La Niña events in the Pacific Ocean. These influence oceanic productivity in the region, and thus many aspects of seabird life history. During La Niña events, the Pacific Ocean's equatorial temperature is lowered, resulting in offshore winds that drive upwelling along the coasts of western South and North America. During these periods, productivity is increased along the west coast of the Americas (e.g., Jackson & Domeier, 2003). Conversely, during El Niño years, upwelling is reduced, resulting in warmer sea surface temperatures and reduced productivity in the region. These changes can affect seabird diet. For example, both diet composition and prey size differ in Blue-footed Boobies (*Sula nebouxii*) during El Niño years, compared to La Niña oscillations (Ancona, Calixto-Albarrán, & Drummond, 2012). El Niño events are also associated with a number of reproductive metrics in seabirds, including increased propensity of female infidelity (Kiere & Drummond, 2016).

Effects of the El Niño/La Niña are not confined to the Southern Hemisphere, but extend into the North Pacific, affecting seabird breeding ecology as far north as Canada and Alaska (Gaston et al., 2009). In the North Pacific, long-term monitoring of marine birds on Triangle Island, Canada, has shown that ocean warming during El Niño/La Niña events can lead to changes in both marine prey species, and the seabirds that feed on them (Gaston et al., 2009). During El Niño events, when warmer waters are present, there is a reduction of an important copepod species in the diet of Cassin's Auklets (*Ptychoramphus aleuticus*), which has been linked to poor auklet nestling growth (Bertram et al., 2017). These naturally occurring oscillations, coupled with long-term ocean warming, could eventually affect populations of this species in the region (Bertram et al., 2017). Additionally, declines in annual breeding effort of Black Oystercatchers (*Haematopus bachmani*) have been linked with increasing spring sea surface temperature in the same region (Hipfner & Elner, 2013). While the mechanisms for this are still not clear, Black Oystercatchers were less likely to initiate breeding in warmer years, and these authors suggested that ocean warming may affect predator-prey interactions in the intertidal zones. Thus, both pelagic and coastal marine dependent birds can be affected by prey species changing in response to environmental conditions.

7.2.2 Climate Change Induced Alterations in Marine Prey

Increasing sea surface temperature is altering oceanic current regimes and nutrient movement (Lewandowska et al., 2014). In some regions, this is affecting seabird populations as marine prey species may shift, either in time or space, to stay within their ideal temperature conditions (Crawford et al., 2015). In turn, temporal shifts in the peak of prey abundance close to breeding colonies, mediated through timing of phytoplankton blooms and subsequent fish spawning, have negatively influenced seabird chick growth and survival in the North Atlantic and Norwegian Sea (Scott et al., 2006).

With the increased rate of changing climatic conditions, ecosystems in the polar regions will respond the most, mainly due to changes in both sea ice cover and sea surface temperature (AMAP, 2015). This has already led to changes in prey species composition and availability for some seabirds (Provencher, Gaston, O'Hara, & Gilchrist, 2012; Trivelpiece et al., 2011). The Arctic is a particularly good place to examine these relationships as there are few fishery activities, allowing for changes in seabird diet to be directly linked with changes in prey abundance and distributions. Long-term studies in the Canadian Arctic demonstrate how fish and zooplankton may change with environmental conditions, and simultaneously how this influences seabirds. Since the 1970s, the low Arctic regions in Canada have experienced significant declines in summer sea ice cover. During the same period, the dominant prey species of Thick-billed Murres (*Uria lomvia*) has changed from cold-water species (Arctic cod; *Boreogadus saida*) to warmer, more temperate species (capelin; *Mallotus villosus*) (Provencher et al., 2012). This change in diet of the birds has also been associated with decreased chick growth and lower adult body mass (Gaston, Gilchrist, & Hipfner, 2005), suggesting that altered prey species can directly affect seabird condition and reproduction (Elliott, Davoren, & Gaston, 2008). Simultaneously, this shift in prey also influences what seabird species utilize the colony. Razorbills (*Alca torda*) attend Thick-billed Murre colonies in the low Arctic only in years when the warm water prey species (sandlance; *Ammodytes* sp.) is in great abundance (Gaston & Woo, 2008). With prolonged changes in prey distribution, these prospecting Razorbills may establish permanent breeding sites, leading to changes in colony species assemblages.

In the Antarctic, changes in sea ice cover are associated with krill productivity, leading to changes in seabird productivity. In the West Antarctic Peninsula and the adjacent Scotia Sea, mean annual air temperature has risen 4°C since the 1940s, with the area of sea ice in the region decreasing significantly (Trivelpiece et al., 2011). The main pelagic prey species for penguins in the region is krill, which varies annually with sea ice extent and winter ice duration (Clucas et al., 2015). Declines in the number of breeding pairs and recruitment of both Adélie (*Pygoscelis adeliae*) and Chinstrap penguin (*Pygoscelis antarctica*) populations in the South Shetland Islands indicate that decreases in krill are likely causing the declines rather than species-specific affinity for ice covered habitat (Clucas et al., 2015; Trivelpiece et al., 2011).

FIG. 7.1 Owing to changing climatic conditions, seabirds in some areas are experiencing increased predation. Here, a polar bear feeds on thick-billed murres (*Uria lomvia*). The frequency of these events at several colonies in the Canadian Arctic has increased over the past few decades in relation to decreases in summer sea ice which the bears usually use to feed on seals. Photo courtesy of Kyle Elliot.

Importantly, there is variation in how different species respond to changing climatic conditions. While some Antarctic seabird species are arriving more than a week later on their breeding colonies and, on average, laying their eggs a few days later than in the 1950s, this pattern is not universal (Barbraud & Weimerskirch, 2006). Effects of climate change on prey species and seabirds are complex and dependent on species' biology and species interactions.

Climate change induced alterations in prey are not limited to Polar Regions. Warming trends in temperate locations are also linked with changing food webs. In the Benguela ecosystem off southern Africa, a combination of physical changes in the nearshore environment (increased sea surface temperatures, decreased oxygen concentrations) and fishing pressure have caused eastward shifts in the distribution of the prey of the region's endemic seabird species (Blamey et al., 2015). Breeding populations of Cape Gannets (*Morus capensis*), African Penguins (*Spheniscus demersus*), and Cape Cormorants (*Phalacrocorax capensis*) were shown to have responded to an altered availability of adult sardine (*Sardinops sagax*) and anchovy (*Engraulis encrasicolus*) west of Cape Agulhas after the late 1990s (Crawford et al., 2016). Gannets showed variation in survival (Distiller, Altwegg, Crawford, Klages, & Barham, 2012), and increased their reliance on sardine and anchovy in response (Green et al., 2015). Numbers of African Penguins breeding in South Africa declined from ~56,000 pairs in 2001 to ~17,000 pairs by 2013, with adult survival, juvenile survival, chick growth, and chick body condition all decreasing concomitantly (Crawford et al., 2015; Sherley et al., 2014; Sherley, Ludynia, et al., 2017). In addition, increased upwelling and wind variability in the 1990s contributed to an eastward shift in the abundance of west coast rock lobster (*Jasus lalandii*), the main prey of the endangered Bank Cormorant (*Phalacrocorax neglectus*) in South Africa (Blamey, Howard, Agenbag, & Jarre, 2012). The cormorant population responded to the changing lobster abundance between 1993 and 2015, declining at breeding sites north of Cape Town and increasing farther east (Sherley, Botha, et al., 2017). In some cases, spatial shifts in spawning forage fish in southern Africa have put prey species beyond the foraging range of seabirds during breeding (Durant et al., 2010) and even during nonbreeding dispersive phases (Sherley, Ludynia, et al., 2017).

Also of importance for some seabirds is how changes in the environment may lead their predators to change behaviors. For example, Polar Bear (*Ursus maritimus*) predation on seabirds in the Arctic has increased significantly with reduced summer sea ice coverage (Iverson, Gilchrist, Smith, Gaston, & Forbes, 2014) (Fig. 7.1). How such increased predation will affect seabird populations will depend on the intensity and distribution of depredation, but the consequences may be both at the individual and population level.

7.3 SEA LEVEL RISE

Global sea levels are expected to rise nonuniformly across the oceans (Nicholls & Cazenave, 2010), with estimates ranging from an increase of 26 cm up to ~2 m by the end of the century (Church et al., 2013). Some coastal regions are already experiencing inundation of seawater. Groundwater rise is also expected, and the effects will be amplified by storm surges and extreme tides (Reynolds et al., 2015).

Breeding habitat varies among seabird species, from ground or burrow nesting at low elevations, to trees, cliffsides, or high elevation forests, so the impact of sea level rise will be species-specific, based on both exposure of breeding habitats to inundation and the sensitivity of species to the inundation (Hatfield, Reynolds, Seavy, & Krause, 2012; Reynolds, Berkowitz, Courtot, & Krause, 2012). For species breeding in low-lying coastal areas, this and associated climatic changes will lead to salinization and increased land loss from erosion (Wong et al., 2014). More acutely, sudden flooding (or overwash) of breeding colonies can cause direct mortality, leading to lower survival, fecundity, and possibly colony losses and species extirpations. The frequency of sudden flooding is expected to increase, so colony overwash will likely occur more often for species that breed during the storm season, and synchronous colonial breeders are more likely to have catastrophic losses from a single event. Lack of dispersal opportunities to predator-free sites at higher elevations creates an ecological trap scenario on so-called protected low elevation islands (Reynolds et al., 2015).

Efforts to quantify the effects of sea level rise at the island and species scale in the Hawaiian Archipelago suggest that a 1.0 m rise in sea level could leave more than 200,000 tropical seabirds without nesting habitat at French Frigate Shoals, leading to lower carrying capacities of species (Hatfield et al., 2012; Reynolds et al., 2015, 2012). At Midway Atoll, where projections suggest a +2.0 m sea level rise with wave-driven flooding, approximately 60% of albatross and 44% of Bonin Petrel (*Pterodroma hypoleuca*) nests may be flooded with a loss of more than 616,400 breeding birds. The most vulnerable species are those with small ranges, specific habitat niche requirements, that are restricted to low elevation islands (Reynolds et al., 2015). However, opportunities exist to adapt seabird conservation efforts to restore breeding colonies to higher elevation habitat using predator removal with other reintroduction techniques (Reynolds et al., 2015; Spatz et al., 2017). Other approaches include social attraction techniques such as the placement of decoys or vocalization recordings to attract pre-breeders and displaced adults to new colony sites, or natural colonization may occur with species into areas with higher elevation breeding or nesting habitat after invasive predators have been removed.

7.4 PATHOGENS AND PARASITES OF SEABIRDS

7.4.1 Parasites

Seabirds host a diverse range of parasites because of the high densities that often characterize their breeding colonies, and the fact that they tend to be long-lived and highly faithful to their breeding sites and colonies among years. The diversity of endoparasites in seabirds is only partially known due to the difficulty in quantifying parasite species; the host generally needs to be necropsied to perform in-depth analyses. Seabird-hosted parasites include both microparasites (bacteria, viruses, protists) and macroparasites (helminths, insects, ticks, and mites). Parasites are ubiquitous in seabirds, and their impact depends on the nature of the parasite and on interactions with local environmental factors such as food resources, levels of local pollution, habitat quality, presence of predators, competitors, and other parasites. Except under extreme conditions where the presence of a parasite or pathogen causes widespread mortality, little is known about how parasites alter seabird health, reproductive success, population viability, and evolutionary processes.

The life cycle and transmission pathways of endoparasites, such as nematodes, cestodes, and trematodes, are often complex, with both direct (i.e., from parent to chick) and indirect transmission via intermediate hosts or vectors. For example, cestodes and trematodes most often infect seabirds via trophic transmission, where juvenile parasite life stages infect seabird prey items (Galaktionov & Bustnes, 1996). Owing to the difficulty in obtaining reliable estimates of parasite diversity and loads, the impact of these species on seabird health and population dynamics has received relatively little attention. However, experimental removals of gastrointestinal nematode parasites were shown to influence both parental condition and reproductive success in wild European Shags (*Phalacrocorax aristotelis*) (Granroth-Wilding et al., 2015, 2014; Reed et al., 2008). Likewise, Provencher, Forbes, Mallory, Wilson, and Gilchrist (2017) showed a positive effect of parasite removal on the propensity of Common Eiders (*Somateria mollissima*) to breed. However, in both cases, broad spectrum antiparasite treatments were used, so that the precise impact of specific parasites on reproduction could not be directly measured.

The diversity and impact of ectoparasites on seabirds is better documented. Common ectoparasites include fleas, lice, mites, and ticks (Fig. 7.2). The overall diversity of these species on seabirds, particularly for lice and mites, is impressive and largely underestimated (Stefan, Gómez-Díaz, & Mironov, 2013). Ticks and fleas have been more readily studied, but population genetic studies have shown that these organisms repeatedly diverge into local seabird-specific populations (or host races). Therefore, fine scale diversity is important as it can alter infestation probabilities in large multispecific seabird colonies (McCoy, Léger, & Dietrich, 2013). While the overall impact of fleas, lice, and mites on seabird body condition and overall fitness is still a subject of some debate, large populations of ticks within colonies alter seabird body condition (Gauthier-Clerc, Clerquin, & Handrich, 1998), reproductive success (Duffy, 1983), and colony selection

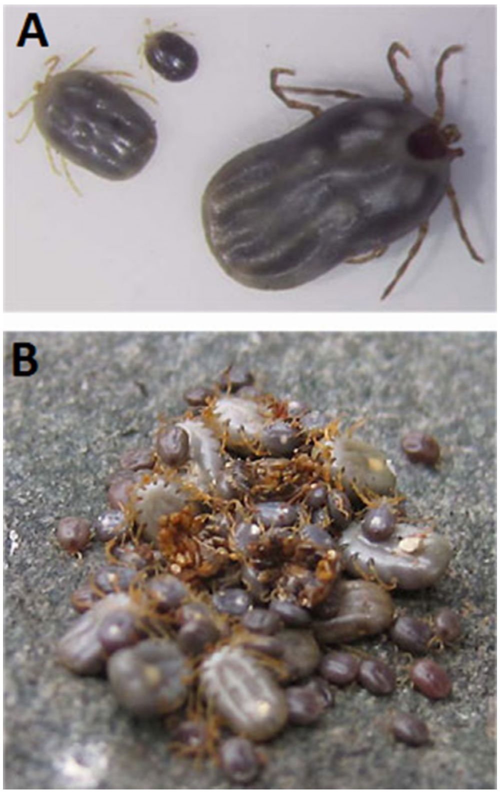

FIG. 7.2 (A) *Ixodes uriae* ticks; an engorged larva, and engorged nymph and an engorged female. (B) engorged tick collected from within a single puffin burrow. Photo credit Karen McCoy.

(Boulinier, McCoy, & Sorci, 2001). This can be due to both direct blood loss and tissue damage inflicted during tick feeding and via the transmission of tick-borne infectious agents. Indeed, seabird ticks are vectors to a large range of bacteria and viruses whose pathogenic effects are largely unknown (Chambert et al., 2012). Thus, macroparasites can indirectly affect seabird populations by transferring pathogenic microparasites.

7.4.2 Diseases

Seabirds are potentially exposed to a wide range of pathogens. Most work has focused on detection and surveillance, with little work aimed at assessing effects on seabird populations. Many diseases found in seabirds are often detected in apparently healthy birds suggesting that, for at least some pathogens, seabirds are carriers or experience sublethal impacts. Work examining potential vectors and transmission routes of infectious agents in seabirds can be particularly informative for understanding both seabird movement and dispersal and for predicting disease emergence (Boulinier et al., 2016; McCoy et al., 2016).

7.4.2.1 Bacterial Pathogens

Avian cholera (caused by *Pasteurella multocida*) has spread from poultry to wild animal populations and has recently reached as far as some Arctic wild bird populations (Iverson, Gilchrist, et al., 2016; Phillips et al., 2016). In seabirds, there has been minimal reporting of this disease except in Common Eiders (Iverson, Forbes, Simard, Soos, & Gilchrist, 2016) (Fig. 7.3), Common Murres (*Uria aalge*) (Österblom, Van Der Jeugd, & Olsson, 2004), Cape Cormorants (Waller & Underhill, 2007), and gulls (Wille et al., 2016). Avian cholera can cause widespread mortality in some populations, especially when the disease is novel (Iverson, Forbes, et al., 2016) or outbreaks coincide with poor conditions (Waller & Underhill, 2007). A decrease in Cape Cormorants breeding in South Africa's Western Cape in the mid-1990s was attributed in part to avian cholera, potentially triggered by concurrent poor prey availability (Waller & Underhill, 2007). While early analysis of Common Eider populations in an avian cholera outbreak estimated potential rapid population declines (Descamps, Jenouvrier, Gilchrist, & Forbes, 2012), more recent work has shown that avian cholera has a high impact on

FIG. 7.3 Dead female common eiders (*Somateria mollissima*) in northern Hudson Bay, Canada after a severe avian cholera outbreak during the breeding season. Photo courtesy of Grant Gilchrist.

the population initially, with a more moderate effect over time (Iverson, Gilchrist, et al., 2016). While avian cholera has become relatively common in seabirds in some regions (e.g., Common Eiders in North America), there are concerns about its potential effects in species that breed in only a limited number of locations and so are vulnerable to potential outbreaks (Phillips et al., 2016). The threat to small and remote populations of seabirds should be considered severe if avian cholera is found in nearby locations.

Borrelia burgdorferi sensu lato, the causative agent of Lyme disease, has been widely detected in temperate and polar seabird colonies (Schramm et al., 2014). These bacteria circulate in seabirds via the tick, *Ixodes uriae*, and can reach high exposure levels in colonies (Gomez-Diaz, Doherty, Duneau, & McCoy, 2010; Staszewski, McCoy, & Boulinier, 2008). Although no direct effect on seabirds has been documented (Chambert et al., 2012), the same strains that circulate in seabirds are found in terrestrial ecosystems, suggesting that seabirds may be important reservoirs for these bacteria and may circulate them at wide spatial scales (Gomez-Diaz et al., 2011). Other *Borrelia* bacteria may be found in tropical seabird populations, notably those responsible for relapsing fever and vectored by soft ticks of the genus *Ornithodoros* (Yabsley, Parsons, Horne, Shock, & Purdee, 2012).

7.4.2.2 Viruses

While avian influenza in birds has been a global concern for species that come in direct contact with humans on a regular basis, such as poultry, there has been less work on seabirds (Lang et al., 2016). Some seabird species appear to carry the avian influenza virus with little effect, such as Southern Giant Petrels (*Macronectes giganteus*) (de Souza Petersen et al., 2017), Thick-billed Murres (Huang, Robertson, Ojkic, Whitney, & Lang, 2014), terns (Lebarbenchon et al., 2015), and gulls (Arnal et al., 2015). Seabirds may therefore represent an important reservoir of influenza diversity. Avipoxviruses are also common in seabird colonies around the globe (Uhart, Gallo, & Quintana, 2017). These double-stranded DNA viruses can be transmitted both directly (via muscosal secretions) or indirectly and primarily affect chicks, often resulting in death. Avipoxviruses have been identified as an important risk factor for the conservation of small and endangered populations, particularly in island species (Gyuranecz et al., 2013). Numerous other viruses have also been documented in seabirds (McCoy et al., 2016; Uhart et al., 2017); a study examining nine species of seabirds from seven islands in the Indian Ocean found that four species regularly tested positive for antibodies against flaviviruses (Jaeger et al., 2016). Flavivirus exposure has also been documented in Yellow-legged Gulls (*Larus michahellis*) in the Mediterranean (Arnal et al., 2014). A flavivirus, an orbivirus, a phlebovirus, and a nairovirus were all isolated from ticks associated with penguins on Macquarie Island (Major et al., 2009). Much work is still required to understand the potential impacts of these viral agents on seabirds.

7.5 HUNTING

Seabird adults, chicks, and eggs are harvested for a variety of reasons, including cultural traditions, food security, recreation, and economy. Hunting takes place both in marine and terrestrial environments, and thus can affect seabirds at different stages of their life history (e.g., Gilliland et al., 2009; Jones et al., 2015). Today, locally harvested seabirds remain important in some regions of the world, as a subsistence harvest (as food or income), or for recreation. The Common Eider harvest by indigenous peoples in the Arctic (Fig. 7.4), the Northern Fulmar (*Fulmarus glacialis*), and Atlantic Puffin (*Fratercula arctica*) harvest on the Faroe Islands, and the shearwater (*Ardenna* spp.) and petrel (*Pterodroma* spp.) chick harvests in Australia and New Zealand are examples of traditional harvests, which are still important as a supplementary source of nourishment or income, and for maintaining traditional lifestyles and cultural identity (AMAP, 2015; Newman et al., 2009; Olsen, 2008). Although the levels of seabird harvest have generally declined recently, often because of more restrictive hunting regulations, urbanization, and occupational changes (Frederiksen, 2010), this is not the case universally (Jones et al., 2015; Lyver et al., 2015).

Seabird harvest in both marine and terrestrial environments is a potentially important pressure on seabird populations, leading to population declines (Spenneman, 1998) or even extinctions (Crofford, 1989). The impacts from present-day harvest can be less obvious (but see Jones et al., 2015; Lyver et al., 2015). Hunting quotas are smaller, and new confounding stressors such as commercial fisheries, bycatch, pollution, and climate change (Croxall et al., 2012) make it challenging to quantify the population-level impact of harvests. International cooperation is needed to assess impacts where migratory seabirds are harvested (e.g., Frederiksen et al., 2016). The impact of harvest has been clearly demonstrated for the Common Eider in Canada and Greenland, and on Gray-faced Petrels (*Pterodroma macroptera gouldi*) in New Zealand. Before 2001, Common Eider hunting pressure was high and unsustainable in Greenland, but after 2001 the open season was shortened and spring hunting was banned, leading to an approximate 70% reduction in birds harvested. Subsequently, the breeding population increased in Greenland and Canada (by >200% in Greenland by 2007), adult survival rate increased, and there was greater recruitment of first-time breeders (Burnham, Johnson, Konkel, & Burnham, 2012; Merkel, 2010). In New Zealand, concerns about declines of Gray-faced Petrels in the 1960s resulted in the cessation of harvesting at some locations. More recently, the combination of traditional knowledge and management practices with demographic modeling has allowed these culturally important traditions to resume (Jones et al., 2015; Lyver et al., 2015).

FIG. 7.4 Inuit hunters near Cape Dorset, Canada hunting common eider ducks (*Somateria mollissima*) during the spring migration. Photo credit Jennifer Provencher.

A similar reduction in hunting pressure on Thick-billed Murres has occurred in Greenland and Canada, the two main hunting regions, but with mixed results (Gaston et al., 2009). While Thick-billed Murre populations in Canada are now stable, the large reduction in hunting pressure did not reverse the declines in Greenland, and nearly two decades later the affected colonies are still declining (Merkel et al., 2014). Other factors may now be equally or more important than harvesting, such as climate-driven oceanographic changes affecting the winter survival of the birds (Merkel et al., 2014). Furthermore, Murre breeding populations mix during the winter in the North Atlantic, and it takes detailed knowledge about migration patterns and timing to evaluate which threats are affecting which populations (Frederiksen et al., 2016). Securing a sustainable harvest for a widely shared migratory species is complex, and working internationally to resolve research questions and management priorities is essential.

A global harvest estimate has not yet been done for seabirds, so the need to identify knowledge gaps and secure sustainable harvest practices is urgent (e.g., Madsen et al., 2015). The nature and extent of illegal harvesting also remains largely unknown (Frederiksen, 2010).

7.6 POLLUTANTS

7.6.1 Oil in the Marine Environment

Seabirds can be vulnerable to oiling events in a number of ways. First, direct oiling of seabirds can cause high levels of mortality. There are limitations for calculating accurate numbers of seabird mortalities from oil spills because the number of beach-driven birds can fluctuate considerably, depending on wind direction and ocean currents (Camphuysen & Heubeck, 2001). Seabird carcasses may be blown offshore, resulting in gross underestimates of the number of birds affected. It is generally accepted that collected birds represent an estimated 3%–30% of total seabird mortality (Piatt & Ford, 1996). Second, birds are impacted by oiling of feathers, which can become clumped and sticky, leading to a decline in the insulative properties of feathers, and leaving seabirds vulnerable to hypothermia. This can in turn lead to seabirds being forced to leave cold waters, which can result in dehydration and mobilization of energy stores, and birds can lose up to 13% of their body mass with a week (Morant, Cooper, & Randall, 1981). Moreover, birds that survive may also try to preen oil off, resulting in significant ingestion and leading to a series of oil-induced diseases, such as aspergillosis, cachexia, hemolytic anemia, ulceration of the stomach, and immunosuppressant effects (Crawford et al., 2000; Haney, Geiger, & Short, 2014).

Three oil spills off South Africa near large colonies of seabirds since 1983 triggered large-scale efforts to relocate and rehabilitate birds, and resulted in a number of studies examining the effects of oil on seabirds. In 1983, 5000 Cape Gannets were affected when the *MT Castillo de Bellver sank;* in 1994 and 2000, 10,000, and 19,000 African Penguins were oiled after the *MV Apollo Sea* and *MV Treasure* sank, respectively. Many were de-oiled, marked, and monitored for subsequent survival and breeding success. Survival rates of de-oiled gannets were *ca.* 2% lower than controls and oiling did not affect their probability of breeding (Altwegg, Crawford, & Underhill, 2008). After the *Treasure* spill, ~19,000 oiled penguins were cleaned and 91% were returned to the wild. Another 19,500 unoiled penguins were relocated to unaffected areas, resulting in only ~2000 individuals dying as a result of the spill. In contrast, approximately 20% of the Bank Cormorants in the region died due to oiling, and the capture and success rate for rehabilitation of this species was low (Crawford et al., 2000). Subsequent survival rates of de-oiled and unoiled penguins were similar in nonbreeding individuals for at least 10 years, but an estimated 27% of penguins oiled by the *Apollo Sea* did not breed again. Additionally, those that bred had survival rates 7%–17% lower than unoiled birds in 2 years immediately following the spill, and penguins breeding after both spills had lower fledging rates and slower chick growth rates (Barham, Underhill, Crawford, & Leshoro, 2007; Wolfaardt, Underhill, Altwegg, & Visagie, 2008).

The ecological effects of the *MV Exxon Valdez* oil spill in Alaska in 1989 have provided useful insights. Immediately following the spill there was a 25% decrease in Harlequin Duck (*Histrionicus histrionicus*) numbers, likely due to direct oiling events (Iverson & Esler, 2010). More importantly, long-term studies found that their survival rates were depressed for 6–9 years following the spill and did not fully recover to rates in unoiled areas for 11–14 years. On the basis of these findings, Iverson and Esler (2010) estimated that the population recovery in the spill regions would take 24 years. Similarly, biochemical indicators of exposure to oil in Barrow's Goldeneyes (*Bucephala islandica*) remained elevated in oiled areas near the *Exxon Valdez* spill for two decades (Esler et al., 2011). These findings suggest that while the direct and acute effects of oil spills on seabirds are critical, oil spills can negatively affect seabird populations for decades (Iverson & Esler, 2010).

The *Deepwater Horizon* spill in the Gulf of Mexico has illustrated how oil platforms can also result in large-scale oil spills. The 2010 oil spill released ~4.9 million barrels of crude oil into the Gulf of Mexico, equivalent to eight *Exxon Valdez* spills (Haney et al., 2014). The slick covered over 100,000 km^2, affected large areas of wetland habitat, and entered the nearshore planktonic food web. Seabird mortality estimates range from 600,000 to 800,000, with the main casualties being

Laughing Gulls (*Leucophaeus atricilla*), Royal Terns (*Thalasseus maximus*), Northern Gannets (*Morus bassanus*), and Brown Pelicans (*Pelecanus occidentalis*) (Haney et al., 2014).

In addition to impacts from the oil, 7.95 million liters of chemical dispersants were mixed with the oil at the surface (5.05 million liters) and at the well head ~1300 m deep (2.9 million liters) (Kujawinski et al., 2011). However, it remains unclear how the mixture of chemical dispersants and oil introduced into marine food webs, such as in the Gulf of Mexico, will manifest in seabird populations.

Oil exploration activities can also pose a threat to seabird populations. Offshore oil platforms provide enhanced foraging opportunities for seabirds, due to artificial light, and the underwater structures attract fish schools (Burke, Montevecchi, & Wiese, 2012). Quantifying patterns of a species' mortality due to oil rig structures, incidental oil pollution, collisions, or flaring is problematic because of the episodic nature of such events (Burke et al., 2012). There is a lack of robust monitoring programs and a reliance on anecdotal evidence, which is often reported by oil rig workers or temporary observers (Burke et al., 2012). However, Russell (2005) reported on a highly standardized multiyear survey of the impacts on seabirds of oil and gas platforms in the Gulf of Mexico. He found that the 4000 oil and gas platforms served as an artificial archipelago for migrating seabirds and nocturnal collisions were a significant source of autumn migration mortalities. Burke et al. (2012) also noted anecdotal reports of storm-petrels and Little Auks (*Alle alle*) flying into oil rig flares. Several thousand individuals are killed annually by offshore oil and gas activities in Canada (Calvert et al., 2013), but there is no global assessment to date.

7.6.2 Anthropogenic Chemicals

It has been known since at least the late 1800s that environmental contaminants (trace elements, various persistent organic pollutants (POPs), and, most recently, plastic debris) could be taken up by wildlife (Hoffman, Rattner, Burton, & Cairns, 2003). Recognition of negative effects of this in seabirds began in the 1960s when POPs were linked to eggshell thinning in some species. Some of the highest levels of polychlorinated biphenyls (PCBs) were found in wild, fish-eating birds, along with plastic debris in albatrosses (Risebrough, Rieche, Peakall, Herman, & Kirven, 1968). Certain contaminants (e.g., POPs, mercury) biomagnify up food chains while others (e.g., cadmium) do not (Campbell et al., 2005). This, combined with the recognition that many contaminants can disperse widely through atmospheric and marine transport processes (Beyer, Mackay, Matthies, Wania, & Webster, 2000), explained why seabirds breeding from the equator to the poles could have high levels of contaminants (Burger & Gochfeld, 2002; Mallory & Braune, 2012).

Research on contaminants in seabirds has progressed along two general lines of investigation. First, studies have examined seabirds for specific impacts such as mortality, deformities, and reduced breeding success directly related to high concentrations of various contaminants (Evers et al., 2008; Verreault, Skaare, Jenssen, & Gabrielsen, 2004), or subtle, behavioral adjustments correlated with higher concentrations of certain contaminants (Bustnes, Bakken, Erikstad, Mehlum, & Skaare, 2001). This work has demonstrated that levels of contamination affect various species quite differently. For example, negative effects of Hg may occur at relatively low concentrations for certain birds, while others tolerate much higher levels (Braune et al., 2012; Shore, Pereira, Walker, & Thompson, 2011). Second, seabirds are exceptionally suitable biomonitors of contaminant levels because they feed at different trophic levels and are often colonial breeders (Fig. 7.5). Consequently, researchers can sample different tissues from different species, often from a single sampling location, and can assess trends in levels of various contaminants in the environment (Elliott & Elliott, 2013; Riget et al., 2011). For broadly distributed seabirds, this has permitted some important regional assessments of marine pollution (Mallory, Braune, Wayland, Gilchrist, & Dickson, 2004; Riget et al., 2011). In other cases, knowledge of seabird natural history, demography, and movement patterns has provided insights into the different seasonal exposure of seabirds to contaminants. For example, Leat et al. (2013) showed that Great Skuas (*Stercorarius skua*) that wintered in different parts of the Atlantic Ocean had markedly different concentrations of POPs in their blood plasma. Monitoring of contaminants in seabird tissues has proven particularly useful for verifying whether international policy implementation on curbing contaminant release has had a direct effect on the environment. For example, Braune, Letcher, Gaston, and Mallory (2015) showed that levels of polybrominated diphenyl ethers (PBDEs) declined dramatically in seabird tissues with a short lag after their North American ban.

There has been evidence of different contaminants causing impairment or mortality of seabird individuals or their eggs, although these often were the result of point-source contamination (e.g., lead shot at key hunting locations; Franson & Pain, 2011) or effects from POPs before their deleterious effects were well understood (Blus, 2011). Early in the 21st century, few studies reported toxic levels and die-offs of seabirds due to contaminants; instead, for many POPs and some trace elements, trends suggest declining environmental concentrations toward background levels, as indexed from seabird tissues (Braune, 2007; Riget, Bignert, Braune, Stow, & Wilson, 2010). However, the possibility of synergistic, deleterious effects from environmentally-relevant "cocktails" of contaminants cannot be ruled out, and increasingly studies show sublethal, negative effects of contaminants associated with seabird reproduction (Provencher et al., 2016), health (Fenstad et al., 2016;

FIG. 7.5 Collection of thick-billed murre (*Uria lomvia*) eggs at Prince Leopold Island, Nunavut, Canada for contaminant analysis. This study site has contributed to the long-term seabird contaminants study managed by Environment and Climate Change Canada and the Northern Contaminants Program. This work has been critical in such international regulations of contaminants as the Stockholm Convention and the Minamata Convention. Photo credit Mark Mallory.

Provencher et al., 2016), and even genomes (Crump et al., 2015). Recent genomic techniques to assess how pollutants may cause the up- or downregulation of genes in seabirds show promise as this allows for a broader approach for assessing cocktails of contaminants than do most traditional methods (Crump et al., 2016, 2015). This is also important when evaluating the impacts of long-term exposure to oil pollution.

Several new areas appear particularly fertile for developments in this field. First, an emerging concern is that contaminants are leached from plastic debris when they are ingested (Lavers, Bond, & Hutton, 2014; Tanaka et al., 2015). These concerns extend to DDT and metals that might adsorb to plastic debris, but additionally to plastic-associated contaminants such as plasticizers (Rani et al., 2015). Second, contaminant levels may vary dramatically in seabird tissues through their annual cycle (Fort, Robertson, Grémillet, Traisnel, & Bustamante, 2014); this may have implications for how we interpret toxicity thresholds if contaminants potentially have differential impacts on birds as their condition and health fluctuate through the annual cycle.

7.6.3 Radioactivity

The magnitude 9.0 earthquake at Tohuku, Japan and the tsunami that followed on March 11, 2011 inflicted immense damage on the Fukushima Dai-ici nuclear power plant (Kitamura, Kumamoto, Kawakami, Cruz, & Fujikura, 2013). One response to prevent further contamination to the surrounding prefects was that the plant's reactor cooling waters were diverted into the Pacific Ocean (Buesseler, Aoyama, & Fukasawa, 2011). This region is a popular foraging area for several seabird species during nonbreeding periods because of its high productivity driven by the Pacific Ocean upwelling currents (Rayner et al., 2011). For example, the trans-equatorial migrant Flesh-footed Shearwater (*Ardenna carneipes*) has been tracked with geolocators and found to feed off the east coast of Japan (Rayner et al., 2011; Reid et al., 2013), within this contaminated marine zone.

The impacts on marine life from radioactive waste are so far unknown (Buesseler et al., 2011). The salty environment may afford some protection by preventing absorption of radioactive ions by marine life (Reardon, 2011). However, other radioactive isotopes, including cesium-137, which has a 30 year half-life, can be incorporated into animals' bodies via the dermis or ingestion and may bioaccumulate in seabird prey (Buesseler et al., 2011; Reardon, 2011). Streaked Shearwater (*Calonectris leucomelas*) chicks from within the affected area had reduced vitamin A levels compared to those breeding at sites unaffected by the accident, though the population effects of the associated radionuclide exposure are unknown (Uematsu, Uematsu, Lavers, & Congdon, 2014).

Measurements of radioactive isotopes in waters off eastern Japan indicate that concentrations are elevated many orders of magnitude above baseline measurements (Buesseler, 2014; Buesseler et al., 2011). Further, radioactive concentrations in marine sediments are highly elevated, providing additional pathways for bioaccumulation by filter feeders and marine herbivores through consumption of contaminated brown seaweeds (Buesseler et al., 2011). If there is wide-scale bioaccumulation of radioactive isotopes, the consequences could include reproductive failure, mutations, and stunted growth in seabirds and other marine species (Reardon, 2011).

7.6.4 Marine Debris

The first publication of a seabird containing human artifacts is from 1838 when Couch (1838) reported part of a candlestick found in the gut of a Wilson's Storm-petrel (*Oceanites oceanicus*). Since the middle of the 20th century, plastic debris has increased (Thompson, 2016). First reported in fish in 1949, accounts of plastic ingestion by seabirds were not made until the 1960s (Provencher, Bond, et al., 2017). Since then, there have been hundreds of reports of seabirds ingesting plastics across the globe (Provencher, Bond, et al., 2017; Wilcox, Van Sebille, & Hardesty, 2015) (Fig. 7.6).

The consensus of *why* seabirds ingest plastics is that they mistake plastics for prey items. Seabirds peck at debris in the same fashion as prey (Cadee, 2002), and plastics can absorb dimethyl sulfide (DMS) from the marine environment, which is an olfactory cue for marine predators (Savoca, Wohlfeil, Ebeler, & Nevitt, 2016). Many species also likely ingest plastic secondarily by consuming plastic-contaminated prey (Hammer, Nager, Johnson, Furness, & Provencher, 2016).

Seabirds are also susceptible to entanglement in plastic debris. Some species such as Northern Gannets incorporate debris into their nests (Bond et al., 2012). Though the reasons for this are less studied, these species often use fibrous organic material when building nests, and for some (e.g., kittiwakes), plastic in the nest does not appear to be detrimental, while others, such as Northern Gannets and Bank Cormorants, experience mortality due to entanglement (Votier, Archibald, Morgan, & Morgan, 2011).

The North Sea is the only region where plastic ingestion is monitored and studied annually in a systematic way (Provencher, Bond, et al., 2017; van Franeker et al., 2011). Beached Northern Fulmars are collected annually and examined using a standard protocol developed specifically for monitoring plastics in the marine environment. This approach has allowed marine plastic pollution to be examined over time and, importantly, allowed for adaptive policy decisions (Provencher, Bond, et al., 2017; van Franeker et al., 2011). The benefit of having such a standardized protocol is demonstrated by the large-scale comparisons that can be made among sites where the North Sea protocol has been adopted (Provencher, Bond, et al., 2017). The data from a dozen countries (the United States, Canada, Iceland, Faroe Islands, Norway, Germany, France, the United Kingdom, Sweden, Denmark, the Netherlands, and Belgium) clearly show spatiotemporal variation in plastic ingestion within a single species.

Despite a significant body of research, the evidence linking plastic ingestion to population-level parameters, though compelling, is largely correlative (Rochman et al., 2016). There are clear links between ingested plastic and both organic and inorganic contaminants in seabirds (Tanaka et al., 2013), and individual cases of physical damage (Carey, 2011), though plastic was unrelated to nestling growth rates in Laysan Albatrosses (*Phoebastria immutabilis)*, despite the high media attention that this species receives (Young et al., 2009).

FIG. 7.6 (A) A Flesh-footed shearwater (*Ardenna carneipes*) fledgling (approximately 90 days old) found on Lord Howe Island, Australia, in May 2012 showing the ingested plastics in its stomach. (B) the ingested plastics from the same individual spread out in the lab. Photos courtesy of Jennifer Lavers.

While ad hoc studies and those reporting basic mensural data have merit, there is a growing push to move beyond reporting to assessing potential impacts and explaining patterns, which need more directed hypothesis-driven studies (Provencher, Bond, et al., 2017; Rochman et al., 2016). These large-scale studies and syntheses, including meta-analyses of broader trends (Wilcox et al., 2015), can be impeded, however, by inconsistent reporting and heretofore unstandardized sampling. Consistent, standard protocols designed to detect plastics explicitly need to be used so that more high-level synthesis research can be undertaken (Provencher, Bond, et al., 2017). Through such efforts, the conservation questions can hopefully be answered, namely, to what degree does plastic ingestion have population-level impacts and through what mechanisms?

7.7 SEABIRD-FISHERIES INTERACTIONS

The footprint of industrial fisheries pervades all ocean ecosystems and is arguably one of the greatest current threats to marine biodiversity (Selig et al., 2014). Seabirds interact with fisheries in all ocean basins (Gianuca, Phillips, Townley, & Votier, 2017). These tend to be dominated by the influences of fisheries on birds (but see Montevecchi, 2002 for examples of the reverse) and include both direct effects such as incidental bycatch (Gianuca et al., 2017), disturbance from or attraction toward fishing vessels (Bodey et al., 2014), and fishery discards as food subsidies (Votier et al., 2004), as well as indirect effects including competition for prey resources (Cury et al., 2011), increases in scavenging and predatory seabirds (Votier et al., 2004), and changes in size-class and/or species-composition of marine communities (Roux et al., 2013). Seabird-fisheries interactions have been reviewed in depth by others, including Furness (2003) and Wagner and Boersma (2011), so this chapter focuses on the three areas of greatest contemporary concern, incidental bycatch, competition for prey, and discards.

7.7.1 Incidental Bycatch of Seabirds in Fisheries

Bycatch has been a documented issue for seabirds since the 1970s (Tull, Germain, & May, 1972), though it was likely widespread since the introduction and adoption of nylon gillnets through the 1950s and 1960s (Munilla, Díez, & Velando, 2007). Gillnet fisheries were among the first to be recognized as problematic for diving seabirds (Tull et al., 1972), and the high levels of bycatch (of several taxa; Northridge, 1991) in high seas drift gillnets resulted in a global moratorium on this gear in international waters (United Nations, 1991) (Fig. 7.7). Bycatch in longline fisheries, and later trawl fisheries, was implicated in the decline of albatrosses in the late 1980s and early 1990s (Brothers, 1991; Weimerskirch, Brothers, & Jouventin, 1997) (Fig. 7.8), though greater research focus on trawl bycatch did not develop until the early 2000s (CCAMLR, 2002; Weimerskirch, Capdeville, & Duhamel, 2000). In recent years, seabird mortality in purse seine fisheries, particularly of shearwaters, has been recorded (Oliveira et al., 2015; Suazo et al., 2014) and is receiving increased attention.

FIG. 7.7 A long-tailed duck (*Clangula hyemalis*) caught as incidental bycatch in a gillnet in Lithuania. Photo courtesy of BirdLife International.

FIG. 7.8 Albatrosses and petrels caught as incidental bycatch during a single set by a pelagic long-liner in Uruguay. Photo courtesy of Martin Abreu.

The impact of bycatch on seabirds is difficult to determine. In Antarctica, the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) demands high levels of observer coverage (CCAMLR, 2016), providing a strong foundation for assessing fisheries impacts (Croxall, 2008). Elsewhere, observer coverage is far lower and understanding the scale of bycatch requires extrapolation from limited data (Richard, Abraham, Richards, & Abraham, 2013). Within Exclusive Economic Zones (EEZs), there is considerable variability among countries and fisheries. This is particularly relevant for gillnet fisheries frequently composed of smaller scale operators that are difficult to monitor (due to the size and geographic extent of fleets and/or small vessel size). The relatively recent discovery of seabird bycatch in purse seine fleets indicates that this gear has been subject to little prior monitoring for birds (Suazo et al., 2014).

There are sufficient data to make global estimates of seabird bycatch for both longline and gillnet fisheries. Anderson et al. (2011) estimated that 160,000–320,000 seabirds were killed in longline fisheries annually, predominantly albatrosses and petrels. This large range in estimates reflects low data reliability from many key fisheries, but the lower estimate is conservative because observed bycatch levels are often underestimated (Gales, 1998). In gillnets, 400,000 seabirds are captured annually, predominantly diving species (Žydelis et al., 2013). There are no global estimates for bycatch in purse seine or trawl fisheries, though observations suggest that bycatch in trawl fisheries, predominantly of albatross and petrels, but also including sulids (Watkins, Petersen, & Ryan, 2008) and penguins (González-Zevallos, Yorio, & Svagelj, 2011), may be of a similar order of magnitude to that in longlines (Bartle, 1991; BirdLife International, 2013).

Trawl and longline fisheries have been the focus of most efforts to tackle bycatch, and have well-established "best practices" in mitigating bycatch through the Seabird Bycatch Working Group of the Agreement on the Conservation of Albatrosses and Petrels (e.g., ACAP, 2016). Measures like bird scaring lines, night setting, and line weighting (for longlines) significantly reduce seabird bycatch (Maree, Wanless, Fairweather, Sullivan, & Yates, 2014). Such measures are built into the strong management of CCAMLR fisheries in the Southern Ocean (CCAMLR, 2016), and are also included in the fisheries regulations of many countries. Beyond national jurisdictions, the five main tuna Regional Fisheries Management Organizations all require vessels to use mitigation measures (Alderman et al., 2011).

Aside from top-down regulatory mechanisms, a bottom-up, grassroots approach can effectively engage industry and reduce bycatch, best exemplified by BirdLife's Albatross Task Force (BirdLife International, 2016). Such an approach has resulted in binding fisheries regulations in several countries (Consejo Federal Pesquero Argentina, 2017; Maree et al., 2014; Republic of Namibia, 2015). These regulations should deliver substantial bycatch reductions if well implemented, a key factor for high seas and national EEZs alike—regulations must be followed by implementation, monitoring, and enforcement. Few fisheries have the competition for licenses that permit strong regulations, as in CCAMLR (CCAMLR, 2016), but novel approaches, including electronic monitoring (Papworth, 2013), may help to reduce the dependence on onboard observers. However, there are still many challenges to implementing best practice mitigation in the riskiest longline and trawl fisheries.

Reliable mitigation measures for gillnets or purse seines do not yet exist, though approaches that seek to increase the visibility of nets to birds show initial promise (Martin & Crawford, 2015). While purse seine bird bycatch is a more recent discovery, highly encouraging results exist from experiments to reduce excess netting in Chilean fisheries (Suazo, pers. comm.[1]). Further efforts to identify solutions for these fisheries are required, particularly for gillnets, the most lethal gear type for seabirds.

7.7.2 Competition for Prey Resources

Seabirds eat ~70–100 million tons (Mt) of food annually, the bulk of which is forage nekton like squid, krill, and small schooling fish (Rountos, Frisk, & Pikitch, 2015) (Fig. 7.9). Forage nekton are also targeted by industrial fisheries which catch ~20 Mt annually, or ~30% of total global landings (Nicol, Foster, & Kawaguchi, 2012). These fisheries overlap with seabirds in space, time, size-classes taken, and trophic level of catch (Rountos et al., 2015). Consequently, most efforts to document seabird-fisheries competition focus around forage nekton and implicitly consider that fisheries outcompete seabirds for limited resources (e.g., Hinke et al., 2017), though other forms of competition exist (Sydeman et al., 2017) and seabirds feeding on benthic organisms may also be affected by fishing (Sherley, Botha, et al., 2017).

Concerns that forage fisheries were competing with seabirds were first raised in the 1930s (Jaques & Murphy, 1936), but began to receive global attention in the 1970s and 1980s. This increased attention occurred when widespread declines in seabird populations were linked to expansions, shifts, and, in some cases, collapse (Roux et al., 2013), of large-scale forage-fish fisheries in several ecosystems (MacCall, 1984; Schaefer, 1970). These fisheries have the potential to reduce prey abundance below levels at which seabirds need to maintain successful reproduction and survival (Cury et al., 2011; Robinson, Butterworth, & Plagányi, 2015), either through localized prey depletion (Bertrand et al., 2012) or by contributing to broad-scale ecosystem change (Sherley, Ludynia, et al., 2017). Populations of forage nekton vary greatly in response to changing environmental conditions, even without fishing (Checkley, Asch, & Rykaczewski, 2017), and seabirds may only react to changes in food resources over long-time scales or at the lowest levels of prey abundance (Crawford, Makhado, Waller, & Whittington, 2014; Piatt, Harding, et al., 2007). Seabird-fisheries interactions are complex, and the evidence that fisheries limit resources for top predators remains equivocal (Hilborn et al., 2017; Smith et al., 2011). Nevertheless, apparent competition between fisheries and threatened marine predators has led to some recent, high-profile conflicts (Cherry, 2014; Mangel, 2010), highlighting the need to better understand when seabird-fisheries competition occurs and how to manage it (Sydeman et al., 2017).

Most studies examining seabird-fisheries competition have used long-term monitoring data to relate changes in fisheries landings and/or abundance to seabird numbers (Crawford, 2007), fecundity (Cury et al., 2011), diet composition (Montevecchi, Birt, & Cairns, 1988), and survival (Robinson et al., 2015). These observational approaches (reviewed by Sydeman et al., 2017) have established important prey abundance thresholds below which seabird performance declines (Cury et al., 2011; Robinson et al., 2015), and provided important baselines for numerical simulations, but they have little capacity to establish causal relationships.

FIG. 7.9 A thick-billed murre (*Uria lomvia*) bringing a capelin (*Mallotus villosus*) back to its chick. Capelin are also a commercially fished species, leading to competition between seabirds and fisheries in some regions for this small forage fish species. Photo courtesy of Kyle Elliot.

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[1] Cristian Suazo, BirdLife Albatross Task Force, Chile.

In recent years, numerical simulations of seabird-fisheries competition have become commonplace (Sydeman et al., 2017). Models used include mass-balance models such as Ecopath and Ecopath with Ecosim (Kaplan et al., 2013; Smith et al., 2011); bioenergetics models of prey consumption (Furness, 1978); individual-based models of foraging ecology and population dynamics (Boyd et al., 2016); system-dynamics models (Weller et al., 2016); end-to-end system models such as Atlantis (Smith et al., 2011); and ecosystem models of intermediate complexity (Punt et al., 2016). While specific conclusions differ, numerical simulations broadly suggest that seabirds can be affected by reductions in prey even in the absence of fishing, that modeled fisheries impacts on seabirds are small relative to natural variation (Kaplan et al., 2013; Smith et al., 2011), but that fishing can accelerate seabird population declines (Punt et al., 2016; Robinson et al., 2015). Several recent studies have also highlighted the importance of the spatial distribution of prey, not just its abundance (Hilborn et al., 2017; Robinson et al., 2015).

Arguably, only field experiments (e.g., time-area fisheries closures) are well placed to demonstrate causal fisheries impacts on seabirds (Sydeman et al., 2017). Field experiments, so far limited to the North Sea and Benguela ecosystems, suggest small but measurable impacts on foraging behavior (Pichegru et al., 2010) and reproductive success (Frederiksen et al., 2008; Sherley et al., 2015), which should translate into population-level impacts over time (Sherley et al., 2015). However, these experiments may need decades to account for changing environmental conditions, and not all species, traits. or sites studied have responded similarly (Ross-Gillespie and Butterworth, 2016), likely because of species-specific behaviors and differences in local prey fields.

Overall, the importance of the underlying environmental conditions in driving forage fish populations (and, thus, seabird responses) over fisheries impacts is clear in most cases. However, international efforts to implement ecosystem-based fisheries management (Sainsbury et al., 2000), coupled with findings that forage fish stock recovery may be slowed or their declines exacerbated by fishing (MacCall et al., 2016), and that seabirds respond most strongly to the abundance of their prey when it is scarce (Cury et al., 2011), have generated calls to suspend fishing when forage stocks reach critical lower thresholds (Pikitch, 2015). Results from the small number of fisheries experiments indicating measurable impacts on top predators (Conn et al., 2014) make it likely that precautionary implementation of Marine Protected Areas will continue around threatened seabird breeding colonies and at important foraging hot spots.

7.7.3 Discards

Fisheries produce huge amounts of waste—of approximately 120 million tons of global landings, 7.3 million tons of discards (8% of total catch) are dumped into the sea each year (Kelleher, 2005). This attracts large numbers of scavengers including benthic invertebrates, fishes, marine mammals, and seabirds (Oro et al., 2013). Discards tend to be relatively predictable in time and space, occur in highly concentrated prey patches, and require less energy to catch. It is perhaps unsurprising therefore that more than half of the seabirds (143 species of 14 families) feed on discards to some degree (Oro et al., 2013), and it is estimated that discards could support approximately 6 million seabirds per annum in the North Sea alone (Garthe et al., 1996). Accordingly, discards can have profound effects on seabirds from individuals to populations.

Many species alter their foraging behavior in response to discarding vessels. For instance, some species alter their movements from super-diffusive to sub-diffusive in the presence of discarding vessels (Bartumeus et al., 2010). Moreover, patterns of activity and distribution strongly influence both the timing and distribution of movements of some species (Cama et al., 2012), with birds responding to the location of vessels at distances of ~11 km (Bodey et al., 2014). Discards can also influence prey choice—with spatial and temporal variation in diet being strongly linked with variation in fishing activity (Votier, Bearhop, Fyfe, and Furness, 2008, Votier et al., 2004). Discard availability may also have important implications for life-history traits. For gulls, fisheries moratoria have led to the production of smaller eggs, lower breeding performance, and poorer over-winter condition in relation to periods when discards are available (Oro et al., 2004, 1996). In contrast, the proportion of discards consumed by northern gannets *M. bassanus* is negatively correlated with adult body condition (Votier et al., 2010), and Cape gannet chicks grow and survive less well when provisioned predominately on discards (Gremillet et al., 2008).

Discards may also have strong indirect effects. Perhaps the most significant is attraction leading to entanglement—this being probably the single biggest threat to seabird populations worldwide. The provision of discards has tended to favor large generalist species, which have increased in number where fisheries waste is abundant. For instance, a range expansion on northern fulmars *Fulmarus glacialis* can, in part, be attributed to the expansion of fishing fleets in the north east Atlantic (Thompson, 2006). Moreover, populations of large gulls and skuas increased during the latter part of the 20th Century, which coincided with fisheries management that led to the production of large quantities of discards. While these changes cannot be solely attributed to fisheries subsidies, circumstantial evidence suggests that they have played an important role. However, these increases have also led to problems. Gulls and skuas are important predators, often close to the apex of marine food webs, which can exert strong top-down forcing. For instance, in Scotland, great skuas *S. skua* feed extensively on discards (Votier, Heubeck, and Furness, 2008), but in years of low discards and forage fish availability they switch to feeding on other seabirds (Votier et al., 2004), leading to declines in some cliff-nesting species (Votier, Heubeck, et al., 2008). Understanding such downstream effects of variation in discard production is challenging and central to an ecosystem based approach to fisheries management.

As fisheries change, it seems likely that discard production will decrease, with potential winners and losers among seabird communities (Bicknell et al., 2013). Of particular concern is the impact of prey-switching by generalist predators (Votier et al., 2004). In the short term, this may cause problems, but in the longer term large populations of generalist predators may diminish, alleviating top-down pressure, but leaving behind very different communities. There may also be negative impacts for large piscivorous species reliant on subsidies. This may impact some components of populations more than others (Patrick et al., 2015), although scavengers do show foraging flexibility (Votier et al., 2013). On the plus side, reducing discards is likely to reduce bycatch in the long term, although possibly following an initial increase (Soriano-Redondo et al., 2016). Future work should monitor how seabird behavior and life-history tactics vary in response to fisheries, and address the many knowledge gaps about discards such as the implications for transfer of pollution and parasites, and the true demography consequences of this subsidy.

7.8 INTERACTIVE AND CUMULATIVE EFFECTS

While some threats result in direct mortality, others are sublethal and more difficult to quantify in terms of impact at population level. Most species are vulnerable to more than one threat (Bertrand et al., 2012; Croxall et al., 2012), and effects can be cumulative and/or interactive. Since seabirds use both marine and terrestrial habitats throughout their lives, they are vulnerable to environmental threats in both ecosystems (Weller et al., 2016).

As a result, many of the world's seabirds are experiencing disproportionate population declines compared to terrestrial avian taxa (Paleczny et al., 2015). Penguins, for example, are among the most threatened groups of seabirds, due to cumulative impacts from marine and terrestrial threats (Weller et al., 2016). Similarly, taxa in the order Procellariiformes (petrels, shearwaters, albatrosses, storm-petrels, and diving petrels) are disproportionately exposed to multiple marine threats (Paleczny et al., 2015). For example, Flesh-footed Shearwater populations are declining due to a combination of fisheries bycatch, plastic ingestion, road strikes, and predation pressure (Priddel, Carlile, Fullagar, et al., 2006; Reid et al., 2013). In New Zealand, annual mortality of this species from commercial and recreational fisheries (Richard et al., 2013) occur at rates high enough to cause a population decline, while this species is also highly vulnerable to plastic ingestion and its associated sublethal effects (Hutton et al., 2008; Lavers et al., 2014). Unfortunately, the consequences of these cumulative, and potentially interactive, effects are unknown for many seabird populations.

In some cases, understanding the relative and cumulative effects of multiple threats can help to manage culturally important species. Wiese et al. (2004) used long-term demographic data for eastern Canadian Arctic populations of Thick-billed Murre to establish that the cumulative impacts of hunting and chronic marine oil pollution were hindering population growth rates and, therefore, increasing the vulnerability of these populations. This information was used to mitigate the illegal behavior of ship operators and thereby reduce chronic oil-related mortality (Lucas et al., 2012).

Quantifying the effect of direct mortality (e.g., through hunting, oil spills, bycatch, or parasite species) is already difficult for wide-ranging populations, but the challenge in evaluating the impacts of sublethal effects at the population level is even greater (Oro, 2014). Further complicating efforts to estimate cumulative and interactive effects of multiple threats are the confounding influences of demographic variables (e.g., body size, breeding phenology) and ecological noise (e.g., foraging guild, at-sea distributions, interference competition, predation, and density dependence factors) on population dynamics (Oro, 2014). To determine relative and cumulative contributions of different pressures on seabird populations, it is usually necessary to have detailed demographic data, as well as information on exposure to each threat. Modeling exercises that use simulation testing, noninformative priors (in a Bayesian context), or expert opinion, such as Models of Intermediate Complexity (MICE; Plagányi et al., 2014) or system-dynamics models (Weller et al., 2016), have been increasingly employed to understand the impact of multiple effects in complex systems.

To meet the ongoing challenge of conserving marine biodiversity, it is important that managers and conservationists remember to consider threats in the context of how they may act synergistically (or antagonistically) with other threats (Weller et al., 2016). Estimates of the relative and cumulative effects of multiple threats could inform decisions about conservation actions. Such decisions need to consider the magnitude of each threat, the cost of mitigating them, and the probability that the mitigation measure would achieve the intended outcome (Carwardine et al., 2012). When pressing conservation action is needed, but there is data scarcity or uncertainty in our understanding of how population dynamics relate to multiple drivers, models and adaptive management can be used to support decision-making (Weller et al., 2016). Ultimately, tactical decision-making requires continued investment in approaches to disentangling individual and population-level impacts from multiple stressors, as well as continued support for long-term population studies to gather these data.

7.9 MARINE PROTECTED AREAS

Seabirds are often found thousands of kilometers from land, as immortalized by the albatross in Coleridge's *Rime of the Ancient Mariner* and more recently quantified with tracking technology. As humans have increasingly exploited resources from the oceans, seabirds have come into contact with a variety of ecological stressors. Consequently, consideration of marine protected areas (MPAs) to protect seabirds and other marine life has gained attention in recent decades (Agardy et al., 2011). As top predators in oceans, seabirds tend to congregate in areas at sea that are productive, often with locally high biodiversity. Identifying hot spots at sea for marine birds generally correlates with identifying key sites for conservation of multiple organisms (Lascelles et al., 2012). However, these locations are also key for human resource extraction, notably fisheries (Brothers, 1991).

As of 2017, there were 13,674 recognized MPAs, representing 2.98% of the global ocean (Marine Conservation Institute, 2017), although these have varying levels of protection. MPAs are mostly focused along coastal and nearshore areas, with far fewer existing in the pelagic zone (Game et al., 2009). Moreover, their size varies greatly, and may not be adequate in many cases to provide suitable protection for seabirds (Thaxter et al., 2012). For MPAs to be effective, they must consider the movement of organisms, not just of top predators but also of the prey that attracts them to these sites. Consequently, there are substantial temporal and spatial aspects to effective MPA design (Agardy et al., 2011). Pelagic seabirds may require large MPAs, or carefully placed aggregates of small MPAs, to protect all of their life stages, while other species may benefit from smaller, well-defined areas that protect important resources (Lascelles et al., 2012; Thaxter et al., 2012). However, proper management of MPAs has a greater impact on effectiveness than simply MPA size (Gill et al., 2017).

Although it can be difficult to measure the benefits of MPAs on seabirds, some studies have shown positive effects of MPAs mostly because of the availability of tracking/telemetry devices. The proliferation and miniaturization of telemetry devices has revolutionized our ability to follow almost any seabird (Hart and Hyrenbach, 2009) and now allows us to locate the key foraging areas, migration stopover sites, and wintering areas of birds from the poles to the tropics (Shaffer et al., 2005). With this increased knowledge and ability to define key marine habitats for different species, we are now able to assess temporal and spatial threats to seabirds when they are at sea, such as development of fisheries (Montevecchi et al., 2012), and thereby provide credible scientific data on the need for MPAs and their benefits. For example, Young et al. (2015) found that two large MPAs in the Pacific (Pacific Remote Islands Marine National Monument and Papahānaumokuākea Marine National Monument) effectively contained the majority of foraging habitats for three species of sulids during incubation and early chick rearing. These authors concluded that pelagic MPAs have strong potential to protect relevant habitat for some tropical pelagic seabirds. In another example, experimental fisheries closures in South Africa suggested that even relatively small, no-take zones may reduce foraging effort (Pichegru et al., 2010) and improve chick survival (Sherley et al., 2015) for penguins relying on pelagic prey, with demographic benefits expected to accrue in the long run (Sherley et al., 2015; Weller et al., 2016). This strongly suggests that protection of well-defined areas known for productivity (e.g., fronts) will benefit top marine predators like seabirds. The much greater challenge is generating the collaborative political will to create MPAs, especially in the pelagic zone, even given the weight of evidence of long-term benefits for conservation and fishing industries (Agardy et al., 2011; Hart and Hyrenbach, 2009).

7.10 COLONY RESTORATION

Colony restoration is a leading method to mitigate seabird population declines. Colony restoration may be defined as any action taken to mitigate population declines at a nesting colony and increase populations of nesting seabirds through expanding available nesting habitat, adult survival, and/or fledging success. Colony restoration can be separated into two broad classes of activities: (1) island restoration (e.g., eradication of introduced predators or non-native invasive species) and (2) active seabird restoration (e.g., translocations, social attraction, and colony enhancement).

Introduction of mammalian predators to islands is a significant threat to insular bird populations (Diamond, 1985; Moors and Atkinson, 1984; Towns, Byrd, et al., 2011) and has been a major focus of colony restoration work. The negative direct and indirect effects of introduced predators on seabirds, particularly rats (*Rattus* spp.) and cats (Jones et al., 2008), and the resultant ecosystem-wide changes that occur when seabird populations are drastically reduced or eliminated (Fukami et al., 2006; Grant-Hoffman et al., 2010) gave rise to the need for island restoration through the eradication of introduced predators. To date, at least 925 islands have had introduced vertebrates successfully eradicated (DIISE, 2017), allowing the islands to revert to a more natural state. The success of these eradication programs can be far-reaching and may result in increased reproductive rates, survival, and recolonization of seabird species (Ismar et al., 2014; Jones et al., 2016; Towns et al., 2016). Further, the eradication of introduced Arctic foxes (*Vulpes lagopus*) from islands within the Alaska Maritime National Wildlife Refuge resulted in population expansion of Whiskered Auklets (*Aethia pygmaea*) (Williams et al., 2003) and the recolonization of Amatignak Island by Ancient Murrelets (*Synthliboramphus antiquus*) (Buxton and Jones, 2012). However, predator eradication alone may not be sufficient to facilitate recovery, and islands may become locked into a "non-seabird" state (Jones, 2010).

The propensity of many seabirds to nest among conspecifics and their high rates of philopatry make recolonization of previously occupied sites unlikely, especially in the short term (but see Brooke et al., 2007; Jones et al., 2016). Further, given that passive recolonization tends to occur when a source population is within 25 km (Borrelle et al., 2015; Buxton et al., 2013) and that most seabirds exhibit high survival and low reproductive rates (Weimerskirch, 2002), for many species it may take many decades for a colony to develop (Kappes and Jones, 2014). Thus, active restoration is often a desired and important addition to restoration projects (Fig. 7.10). The best known, and pioneering, active seabird restoration project is Project Puffin. Started by the National Audubon Society in 1973, its goal was to restore breeding Atlantic Puffins to Eastern Egg Rock Island in the Gulf of Maine (Kress, 1998). This work led the way in developing methods to restore seabirds to historically occupied nesting islands using chick translocation and social attraction, including acoustic attraction methods. More recently, Jones and Kress (2012) reviewed global active seabird restoration programs and found that over 128 seabird restoration projects have been implemented in 14 countries with varying success. These authors note that methods used to restore seabird nesting colonies need to be implemented in response to specific aspects of the species' life history. For example, chick translocations are often not successful for semi-precocial species (where adults lead chicks to the ocean and provision them on the water), but have a reasonable rate of success in species that are fed at the nest and fledge independent of their parents (Jones and Kress, 2012).

Other restoration activities, such as habitat modification and/or creation of artificial/new habitat, are undertaken to mitigate anthropogenic impacts such as harvesting or oil spills that may occur outside of the breeding period or away from a nesting colony, or to safeguard a population that may occur at a limited number of colonies. For example, Priddel, Carlile, Fullagar, et al. (2006) used nest boxes to create an artificial habitat to establish a colony of Gould's Petrels (*Pterodroma leucoptera leucoptera*) on Boondelbah Island in Australia. Prior to this, Gould's Petrels were restricted to one colony on Cabbage Tree Island (Priddel, Carlile, Fullagar, et al., 2006). Predator-proof fences are another technique used to safeguard at-risk colonies against terrestrial predators. The fence built at Ka'ena Point, Hawai'i allowed the eradication and control of introduced species within the fence and permitted record numbers of Wedge-tailed Shearwater (*Ardenna pacifica*) and Laysan Albatross (*P. immutabilis*) chicks to fledge (Young et al., 2013). There are a large number of activities encompassed in colony restoration and the continual improvement of techniques will continue to benefit global seabird populations.

FIG. 7.10 Gray-faced petrel (*Pterodroma macoptera gouldi*) chick being fed at Matakohe Island, New Zealand during a translocation. Photo courtesy of Cathy Mitchell.

7.11 COMMUNITY-BASED SEABIRD CONSERVATION WORK

Community-based outreach and engagement programs are essential components of long-term conservation strategies in places where humans coexist with wildlife. Until relatively recently, most seabird conservation activities, including invasive species eradications, occurred on remote, uninhabited islands (Towns, Aguirre-Muñoz, et al., 2011). These projects have largely been undertaken in the absence of meaningful community involvement (Towns et al., 2013). However, more than half of the islands (55%) identified as having high conservation benefits from invasive mammal eradications also have permanent human communities (Brooke et al., 2007). Additionally, seabirds are valued by many indigenous communities across the globe, with seabird populations important to culture and food security (Mallory et al., 2006; Moller, 2009). In this context, the fundamental need for substantive engagement with local communities becomes readily apparent, and without genuine commitments from and active participation by local communities, conservation initiatives are not likely to succeed in the long term.

In some regions, there is limited literature available on community participation in seabird and island conservation, likely reflecting the short history of seabird conservation management attempts on inhabited islands (Towns, Aguirre-Muñoz, et al., 2011). In other regions, particularly in Canada with the Inuit, and in New Zealand with the Māori, there is more documentation of researchers and indigenous communities working together to direct research questions and achieve successful co-management strategies (Mallory et al., 2003; Moller, 2009). This includes the use of traditional knowledge to identify species in decline (e.g., Mallory et al., 2003; Robertson and Gilchrist, 1998). The importance of cultural inclusion in community-based conservation is a critical component to the long-term sustainability of seabirds and needs to be considered whenever conservation projects are undertaken.

Wider community-based efforts have been successful in many different regions using a variety of different strategies and approaches (Towns, Aguirre-Muñoz, et al., 2011). Following are three examples of successful long-term community-focused programs. First, efforts led by the Grupo de Ecología y Conservación de Islas (http://www.islas.org.mx/) in México provide compelling examples of innovative engagement with indigenous communities, where both the community and the island's ecosystems and species benefited (Aguirre-Muñoz, et al., 2005; Towns, Aguirre-Muñoz, et al., 2011). Second, on Réunion Island in the Indian Ocean, "Life + Petrels" (http://www.petrels.re/?lang=en) has used educational initiatives and worked closely with communities on a stranded petrel rescue program. This program has organized a highly successful "Nights Without Lights" campaign to raise awareness of and engage residents in activities to address the impacts of introduced mammalian predators and light attraction on two endangered endemic petrel species (Pinet and Le Corre pers. comm.). Third, in Chile, Oikonos Ecosystem Knowledge works closely with multiple stakeholder groups, from local communities to federal agencies, and uses innovative programs, such as a Shearwater Cup soccer tournament and festival, creative writing and drawing workshops for children, and environmental education programs for schoolchildren (http://oikonos.org/what-we-do/), to achieve seabird conservation goals.

Prior to initiating any conservation or restoration project, the effect of the proposed action/s on human inhabitants and their activities, cultural needs, and relationships needs to be considered. This requires clearly defined and agreed seabird conservation goals that integrate local and indigenous knowledge and community needs (Oppel et al., 2010; Reed, 2008). To ensure that conservation is socially supported, it is important to work with local communities and other stakeholders from the outset. Such an approach to conservation must ensure that there are fair governance and decision-making processes that equitably represent all stakeholders and rights-holders, respect cultural heritage, and promote transparency and accountability in conservation processes (Bennett et al., 2017). Successful outcomes require a flexible approach that recognizes that communities possess unique social contexts and are contingent on a project design that is appropriate for the particular local environment (Lundquist and Granek, 2005; Towns, Byrd, et al., 2011). Passionate leadership, the demonstration of a genuine commitment to the community, and regular presence of, and engagement by, conservation scientists and facilitators in local communities build trust and credibility but also require more time than external agencies and scientists may realize. Ultimately, effective community conservation requires a transdisciplinary approach that joins conservation science with the needs and aspirations of the indigenous and local community (Towns et al., 2013; Towns, Aguirre-Muñoz, et al., 2011).

7.12 INTERNATIONAL CONSERVATION EFFORTS

Throughout their migrations, seabirds frequently occupy waters within the jurisdictions of multiple countries, as well as Areas Beyond National Jurisdiction (Dias et al., 2017). Threats in one region may impact a species across a much broader area (Iwamura et al., 2013), and so conservation of migratory species requires cooperation by many parties, including national governments, international organizations, and industry regulators. Therefore, seabird conservation poses a challenge to most of the current conservation frameworks as flyway-level study, action, and conservation are often needed.

The widest international focus on seabirds is through the Agreement of Conservation of Petrels and Albatross (ACAP), under the Convention on Migratory Species (CMS), although this agreement covers only a few groups of seabirds. In polar regions, there is a long history of international cooperation between the Arctic Council's Conservation of Arctic Flora and Fauna (CAFF), Circumpolar Seabird group (C-bird), and the Scientific Committee for Antarctic Research's (SCAR) Expert Group on Birds and Marine Mammals (EG-BAMM). These organizations foster knowledge exchange and research leadership in their respective polar regions. Other groups with large trans-boundary interests in seabirds include the African-Eurasian Waterbird Agreement (AEWA; also under CMS), the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), and the East Asian Australasian Flyway Partnership (EAAFP). In addition, there are numerous nongovernmental organizations involved in seabird conservation efforts. International scale management and agreements are inherently complex, and thus it can take time to achieve meaningful conservation outcomes. Importantly, however, these collaborative international initiatives provide a forum for knowledge exchange and scientific cooperation, a cooperative and international approach that is critical to conserving the world's enigmatic seabird species.

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