COMPREHENSIVE SEABIRD MONITORING FOR THE CHARACTERIZATION AND FUTURE EVALUATION OF MARINE PROTECTED AREAS IN CALIFORNIA'S NORTH COAST STUDY REGION

Report to the California Ocean Science Trust
and California Sea Grant

Edited by Richard T. Golightly, Daniel C. Barton, and
Dan Robinette

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EXECUTIVE SUMMARY

Seabirds are long-lived, upper trophic level predators that are integral components of marine ecosystems. During the breeding season, many seabirds nest in colonies. These colonies are usually on rocks or small islands that provide safety from predators. The birds at the colonies forage in the waters adjacent to the colony or at an economical flight distance from the colony, but must return after foraging to their nests to incubate eggs and provide food to their young throughout the day. As such, they have limited flight distances from the colonies during nesting. Marine Protected Areas (MPAs) can provide both direct and indirect benefits to seabirds. Direct benefits involve reducing the direct interactions seabirds have with humans such as from fisheries and recreational activities. Indirect benefits involve reducing competition with humans for prey resources.

The cohesive understanding of marine conditions provided by monitoring seabirds at multiple temporal and spatial scales required a diverse project. We designed a comprehensive seabird monitoring effort to characterize and evaluate MPA performance in the North Coast Study Region (NCSR) of California's Marine Life Protection Act (MLPA) Initiative. Multiple seabird species residing within the NCSR utilize a diverse array of habitats. Thus, it was necessary to focus on some MPA impacts to coastal breeding species of seabirds that forage close to shore while other components of our study used deeper water seabirds as indicators of the regional oceanographic conditions that mediate change within MPAs. The objectives of our monitoring were to 1) document how seabirds use coastal, nearshore habitats in relation to newly established MPAs and the NCSR, 2) develop seabirds as indicators to study change resulting from MPA establishment, and 3) develop seabirds of indicators of oceanographic changes and natural variation in the ecosystem.

METHODS OVERVIEW

We completed four distinct projects to collect baseline data at varying spatial and temporal scales (see Figure 1 for the scope of seabird monitoring). First, we used data from aerial photographic surveys conducted from 1989-2014 to investigate region-wide trends in the populations of Common Murres and Brandt's Cormorants. We used 173 observations of Common Murre colony abundance at 14 colonies and 123 observations of Brandt's Cormorant nest abundance at 10 colonies over the 26-year study period. Additionally, aerial photographic surveys conducted in 2014 were used to document location and abundance of Common Murre, Brandt's Cormorant, and Double-crested Cormorant across the NCSR.

Second, at a more localized scale, we monitored Common Murre reproduction, foraging effort, and diet in 2014 at Castle Rock National Wildlife Refuge using a robotic, remotely-controlled video recording system. Although this is only one of the seabird colonies in the NCSR, it the largest and served to inform our understanding of the mechanisms of population change across the region. Common Murre are very visible and thus ideal for monitoring fine scale patterns in reproduction and changing diet. We measured date of nest initiation, hatching success, fledging success, overall reproductive success, time allocation, provisioning rate, and diet composition via the video. Information gained from these surveys were combined with comparable data from 2007-2013 at Castle Rock to assess baseline condition and variability of these metrics over an 8-year period.

Third, in 2014-2015 we conducted ground-based surveys of coastally breeding seabirds inside and outside of six MPAs to establish a framework for continued MPA monitoring (see Table 1 for list of MPAs). For this, we conducted intensive monitoring of six species likely to benefit from MPA establishment: Pigeon Guillemot, Brandt's Cormorant, Pelagic Cormorant, Double-crested Cormorant, Western Gull, and Black Oystercatcher. We collected data on breeding population size, breeding productivity, foraging rates and rates of human-caused disturbance inside and outside of each MPA. We monitored productivity by following individual nests visible from land and calculated annual breeding productivity as number of fledglings produced per breeding pair. We monitored foraging from land-based observation points, recording all birds foraging within a 1 km radius of an observation point. We calculated foraging rates as number of birds foraging per hour of observation. We recorded all human-caused disturbances observed during any land-based survey and calculated disturbance rates as number of disturbances per hour of observation.

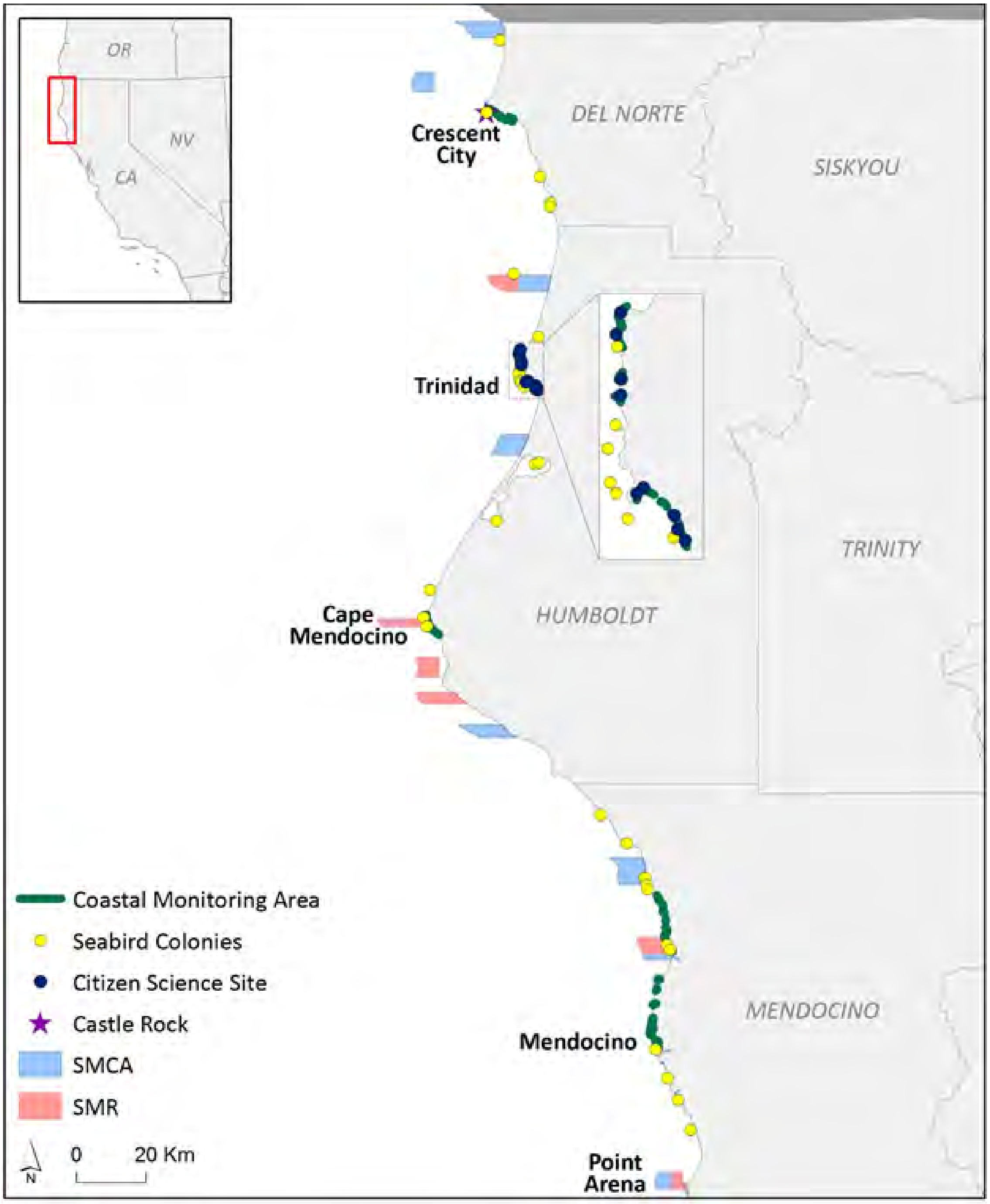

Figure 1. Map of the NCSR showing survey sites for each of the four baseline seabird monitoring projects: Castle Rock National Wildlife Refuge, aerial photographic surveys of seabird colonies, coastal monitoring surveys, and citizen science surveys. The location of all State Marine Conservation Areas (SMCA) and State Marine Reserves (SMR) are also depicted.

Finally, we initiated a citizen scientist program to collect data on coastally breeding seabirds similar to those collected by trained biologists. Citizen scientists collected data at eight sites within the Trinidad Head and Patrick's Point area in Humboldt County. The biologist protocol was tailored for repeatability with trained observers, while the citizen science protocol was tailored for ease of implementation and repeatability with untrained observers and used a photographic datasheet. We compared population data collected by citizen scientists to those collected by trained biologists to assess the feasibility of citizen science in maintaining long-term seabird monitoring within the NCSR.

KEY FINDINGS

1) At the largest scale, we reported a region-wide survey of seabird colonies in the NCSR in 2014 conducted from aircraft and using fine scale photography. Total region-wide counts of three readily-observed fish-eating seabird species were: Common Murre (N = 350,923 attending individuals), Brandt's Cormorant (N = 4,583 nests), and Double-crested Cormorant (N = 1,840 nests) in 2014. These individuals were spread across 32 distinct colonies, including 6 that are now in special closures and 9 that are located in or immediately adjacent to marine reserves or marine conservation areas. These counts were conducted once midway into the nesting season and represent a snapshot of abundance at the colonies.

2) The long-term trend in Common Murre population size showed a pattern of increase within the NCSR (Figure 2). Annual population growth rates were positively related to ocean productivity and appeared to show a shift in patterns of population growth between 1989-1997 and 1998-2014. Common Murre population growth shifted from generally negative to generally positive following the strong 1997-98 El Niño event, and may indicate that oceanographic productivity within the NCSR has generally improved since the 1997-98 El Niño event. Our assessment of initial benefits of MPAs is limited because the only sites with complete recent time series (2010-2014) were within special closures, and not in an MPA, meaning no control sites were available for a strong comparison.

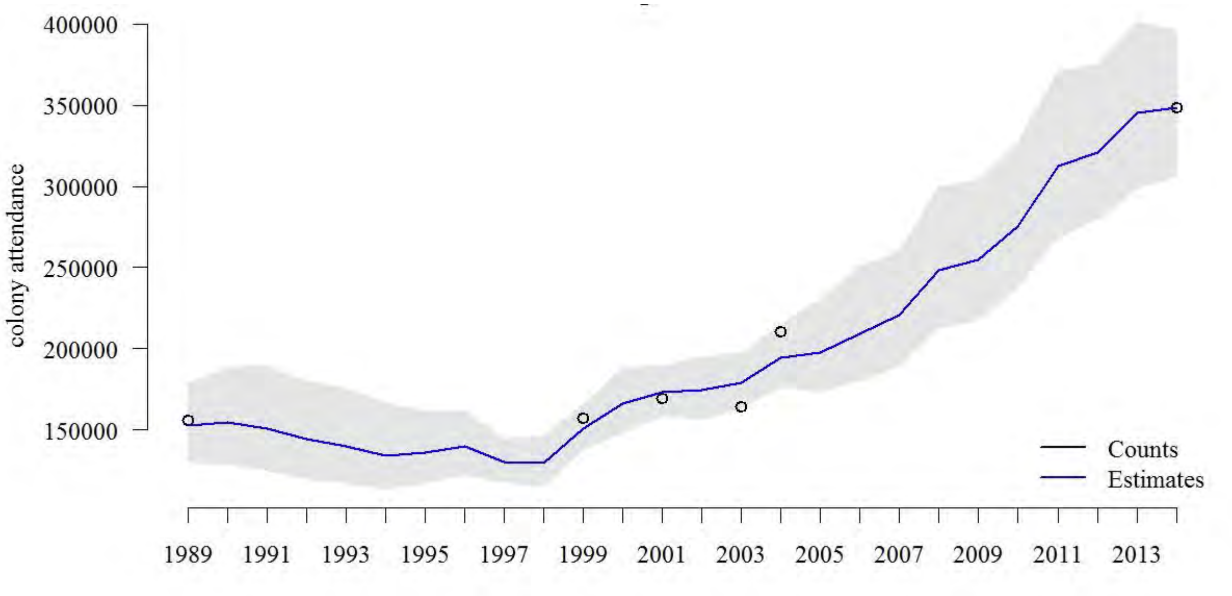

Figure 2. Region-wide Common Murre trend estimates and observed counts, 1989-2014. The symbols shown are the 6 years with complete surveys of all 14 colonies – count data were available from other years, but only at a subset of the colonies. Circles represent total counts.

3) The monitoring of Common Murre (Figure 3) reproductive performance, foraging effort, and chick diet collectively provided valuable information about the health of seabirds nesting in the NCSR. Reproductive success averaged 74.7% over the 8-year study period. From 2007-2014, nest initiation varied annually and was correlated with the timing of the Spring Transition (an index for the seasonal onset of upwelling-favorable conditions and increased marine productivity) in the NCSR. Tracking food delivered to the colony also provided a better understanding of the abundance and composition of prey that murres and other top predators within the NCSR rely on. Smelt and rockfish were the most common prey. This long-term study of Common Murres provides a robust baseline understanding of current conditions of the NCSR. This effort will allow for future alterations in marine condition to be detected by using murres nesting at Castle Rock as a responsive and reliable indicator of short or long term changes.

Figure 3. Representative images of seabirds as viewed through the Castle Rock camera system: Common Murre holding a smelt (upper left), Common Murre and Western Gull with the ocean in the background (upper right), Brandt's Cormorant during incubation (lower left), and Common Murre feeding a shrimp to their chick (lower right).

4) There were few human-caused disturbances observed during the baseline period. In fact, the NCSR had the lowest disturbance rates when compared to the other MLPA Initiative study regions. Furthermore, there were no consistent patterns of disturbance that suggested chronic disturbance was an issue at any study site. We recorded disturbances at the Point Cabrillo SMR and the MacKerricher SMCA in 2014 and at the Ten Mile SMR in 2015. We did not record any disturbances at the Sugarloaf Island Special Closure or the Steamboat Rock Special Closure. Nonetheless, the MPAs and special closures established within the NCSR will play an important role in maintaining low rates of human-caused disturbance.

5) The majority of breeding birds surveyed by the shore-based monitoring element were outside of MPAs, but MPAs provided valuable breeding and foraging habitat to numerous species (Table 1).

Table 1. Marine bird species for which at least small breeding populations and/or moderate foraging habitat (foraging rates are close to average for the species) are being protected by each the six MPAs monitored during the 2014-2015 baseline period. Additionally, special closures within the NCSR include breeding populations of numerous species of seabirds.

MPA	Breeding Population	Foraging Habitat
Pyramid Point SMCA		Double-crested Cormorant Pigeon Guillemot Marbled Murrelet
South Cape Mendocino SMR	Black Oystercatcher Pigeon Guillemot	Common Murre
Ten Mile SMR	Black Oystercatcher Pigeon Guillemot Western Gull	Pigeon Guillemot
MacKerricher SMCA	Black Oystercatcher Pigeon Guillemot	
Point Cabrillo SMR	Black Oystercatcher Pigeon Guillemot Pelagic Cormorant	Pelagic Cormorant Common Murre
Russian Gulch SMCA	Pigeon Guillemot Pelagic Cormorant	

6) Data collected by citizen scientists and trained biologists indexed each other reasonably well, although citizen scientist data appeared to be undercounts. Application of this method more broadly in the NCSR may be an effective technique, but we had difficulty attracting participants, and participation is likely to be further limited by geography as most of the NCSR's MPAs are relatively inaccessible to citizen scientist observers. The NCSR baseline program helped create several partnerships that will continue citizen science in the Trinidad Head and Patrick's Point area as part of a new Seabird Protection Network chapter.

INITIAL CHANGES AND BASELINE CONDITIONS

NCSR MPAs went into effect in 2012, after the beginning of data collection for two of our project elements but before another. We generally expect most changes resulting from MPA establishment to occur over time periods longer than the 3 years post-establishment (through 2015) that we observed. Further, each of the three principal elements of this project established baseline conditions and the potential for initial changes over a different time interval for monitoring metrics with a different time scale of response. The aerial survey element established trends in abundance over the 1989-2014 interval, showed that Common Murre populations grew over this interval while Brandt's cormorant populations did not, and that seabird population growth was correlated with large-scale climatic variation (the El Niño phenomenon). The Castle Rock colony monitoring element established murre reproductive success and diet over the interval 2007-2014 and showed that phenology, diet, and reproductive success varied from year to year and that these differences may be caused by oceanographic variability. The shore-based element established baselines in foraging and roosting distribution, breeding population size, and productivity in a treatment-control (MPA-reference site) study design during 2014-2015. Annual breeding productivity varied by species but was greater in 2014 than 2015 for the nearshore fish foragers (Brandt's, Pelagic, and Double-crested Cormorants) and higher in 2015 than 2014 for intertidal foragers

(Western Gulls and Black Oystercatchers). These results are similar to those observed at sites in central California for the same years.

The 2014-2015 baseline period occurred during an apparent transition in the California Current System from a cooler state to a warmer and less productive state. In addition to this transition, a marine heat wave (referred to as the 'Warm Water Blob') began developing in early 2013 and persisted through 2015 when a strong El Niño event began to develop. Despite these less-productive conditions, upwelling favorable winds were strong in the NCSR during the 2014 spring and summer. This likely contributed to an abundance of juvenile rockfish (an important prey for many seabird species) within the NCSR in 2014. Juvenile rockfish abundance was low by 2015 and likely contributed to the low breeding productivity observed in cormorants that year. Nearshore foraging rates for individuals of each species were highest in 2014 while a greater number of foraging flocks were observed in 2015, indicating perhaps a higher abundance of pelagic prey like anchovies in 2015; though flock foraging was limited mostly to the northern study sites, especially Pyramid Point SMCA. The conditions observed during the baseline period were likely similar to the warmer and less productive states observed during the longer-term aerial survey and Castle Rock elements, which were associated with reduced rate of colony population growth and reproductive success.

Overall, there were few differences among MPA and control sites in our study. Additionally, the differences we observed were not always temporally consistent (i.e. they varied from year to year). We are comfortable with our selection of control sites for the MPAs we investigated and feel that we will be able to detect differences if these MPAs provide benefits at this scale of assessment to seabird communities. It is probable that the establishment of special closures as part of the MPA network reduced disturbance rates at some colonies, yet disturbance rates were already low compared to other regions. Given the among-year variability we observed in our results, it will be important to continue monitoring these sites over the long term in order to detect lasting changes in community metrics due to MPA establishment. The context provided by the longer-term aerial survey and Castle Rock elements helps establish the range of variability for comparison.

SEABIRDS AS INDICATORS WITHIN THE NCSR

Several studies over the past 30 years have shown that seabirds can be reliable indicators of change within marine ecosystems. Additionally, recent studies have shown that seabirds can potentially index recruitment rates of juvenile fish to nearshore habitats. Juvenile fish recruitment is an important factor influencing the rate of response of MPAs to protection, but rates of juvenile recruitment to nearshore habitats vary among years and with geographic location. Thus, not all MPAs are equal in terms of how rapidly we should expect changes to occur. Furthermore, the timing of MPA establishment will influence the rate of response observed within MPAs because management changes implemented during periods of high ocean productivity will show change over a shorter period of time than those during periods of poor ocean productivity. Given the transitional state of the California Current System during the establishment of NCSR MPAs and the response we observed in seabird species, we may expect changes in fish communities to occur more slowly during these initial years of MPA establishment.

Seabirds offer a cost-effective means by which to monitor ocean productivity and track fish recruitment. Seabirds are highly visible and some monitoring can often be easily accomplished from land. Where seabirds nest on rocks or islands, video techniques may allow assessment of reproduction and diet. Aerial photographic techniques may also be a cost-effective means to detect large-scale changes in abundance. Moving forward, seabird monitoring should be used to inform managers in three ways. First, breeding productivity should be integrated with indices of ocean climate (e.g., upwelling, El Niño Southern Oscillation, Pacific Decadal Oscillation) to monitor annual changes in ocean productivity. Second, measures of seabird foraging rates should be integrated with fine-scale maps of ocean currents to track how ocean productivity, including fish larvae, is being delivered to habitats inside and outside of MPAs. Understanding how change in ocean productivity translates into change throughout the NCSR will allow resource managers to establish realistic expectations for the performance of individual MPAs and the NCSR network as a whole. Finally, seabird breeding colonies should continue to be monitored in order to provide regional context for evaluation of finer-scale changes, and to understand the effectiveness of MPAs and special closures in reducing rates of human-caused disturbance.

BASELINE DISCUSSION AND RECOMMENDATIONS

Specific recommendations for long-term monitoring depend on the specific concerns and objectives of the effort. For example, if disturbance rates were a major concern for seabird monitoring, then a technique that includes human observers could have the greatest probability of encountering and recording those events. For efficient cost-effective long-term indicators of change, the aerial photographic surveys can provide sensitive indication of changes in the populations attending the colonies (such as Common Murre and Brandt's Cormorant), and would be an effective indicator of change in the region due to changes in forage in the region (and MPAs). However, aerial photo surveys cannot track seabird species that nest in ways that cannot be detected from the air (e.g. widely spaced, or in burrows or rock crevices), nor do aerial surveys allow assessment of localized causes of change at a colony. Further, understanding the mechanisms of change would require detailed understanding of reproduction and diet, best collected with human observers in near shore coastal habitats or with video for islands or difficult to observe locations. The combination of diet and reproduction with a longer-term population data set can assist in the effort to understand changes in populations of seabirds, and improve the strength and specificity of their use to indicate changes in the marine environment caused by management and biological and physical forcing.

CHAPTER 1: INTRODUCTION

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Marine Protected Areas (MPAs) alter marine ecosystems and food webs directly, by reducing anthropogenic impacts on the survival and reproduction of marine organisms (Halpern and Warner 2002), and indirectly, through changes in food web structure and trophic structure that may in turn influence behavior, distribution, and abundance of other organisms (Weeks et al. 2010). The widespread impacts of fisheries and other consumptive uses on fish populations and marine ecosystem structure and function have motivated the establishment of MPAs around the world in an attempt to conserve fisheries resources and promote ecosystem health (Agardy 1994). Monitored direct and indirect effects of MPA establishment vary from undetectable to dramatic across a range of marine ecosystems worldwide (Halpern and Warner 2002, Lester et al. 2009). Due to the difficulty of observing all the diverse components of marine ecosystems, many monitoring programs have focused on specific readily-observable organisms whose biology provides an indication of harder-to-observe changes in marine ecosystems (Boyd et al. 2006, Piatt et al. 2007, Le Bohec et al. 2013). Organisms that feed at the top of marine food webs (top predators), which include seabirds, marine mammals, and predatory fish, have been widely recognized as indicators of ecosystem change (Boyd et al. 2006, Einoder 2009). Population sizes, foraging behavior and diet, and productivity of top marine predators, which are principally controlled by bottom-up processes, are strongly influenced by prey distribution and abundance which they effectively indicate (e.g. Velarde et al. 2013). Yet, top marine predators may also exert important top-down influences on prey populations (Estes et al. 2011) making them not only indicators but vital components of marine ecosystems (Heithaus et al. 2008, Myers et al. 2007). Further, their economic and cultural value makes them an important utilitarian natural resource (Anderson et al. 1976).
Seabirds and other top predators have generally declined worldwide due to loss of nesting habitat, direct mortality from fisheries, pollution, and reductions in marine ecosystem function (Croxall et al. 2012) and continued loss of these predators may cause dramatic alterations in food web structure and ecosystem function (Heithaus et al. 2008, Myers et al. 2007, Estes et al. 2011).

Robust baseline characterizations of regional marine ecosystems were needed after establishment of California's Marine Life Protection Act in 1999. This baseline characterization will serve as a foundation to assess the initial and long-term responses of marine ecosystems to establishment of MPAs and future monitoring. The North Coast Study Region MPA Baseline Monitoring Program was established to provide this baseline characterization. The North Coast Study Region (NCSR) extends from Point Arena to the Oregon border and includes 27 MPAs (Figure 1). Wide-ranging predators such as seabirds can utilize all or part of this region. We used seabirds as an ideal top predator guild to 1) serve as indicators of change in and around MPAs, 2) monitor the ecosystem effects and benefits of MPAs and 3) monitor the regional oceanographic conditions that influence ecosystem response within the NCSR.

Seabirds may directly and indirectly benefit from MPAs because of decreased fisheries impacts (bycatch, entanglement, light attraction), reduced disturbance to nesting colonies, and increased abundance of prey populations. Seabirds are well studied and they have been widely adopted as indicators of change in marine ecosystems (Diamond and Devlin 2003, Piatt et al. 2007, Einoder 2009). Seabirds are uniquely observable marine predators; they derive food from the marine environment, yet are constrained to the ocean's surface and nest on land (Schreiber and Burger 2001). Given the value of seabirds as indicators and as key members of marine ecosystems, establishing baselines of seabird abundance, reproduction, diet, and distribution within the NCSR to evaluate future changes will provide a valuable tool for quantifying and characterizing the short-term and long-term effects of MPAs on marine ecosystems. Each of these alternative seabird monitoring metrics provides alternative strengths and drawbacks as indicators of ecosystem change (Piatt et al. 2007, Einoder 2009) and as metrics of short-term and long-term seabird population health (Schreiber and Burger 2001). For instance, seabird reproductive phenology, foraging behavior, and diet may rapidly respond to physical and biological changes, sometimes before these changes are detectable via other monitoring metrics (e.g., Mills et al. 2007, Wolf et al. 2010, Velarde et al. 2013). Yet, seabird populations can be buffered from the effects of short-term environmental variation by their generally long life spans and plastic behavioral responses to environmental variation (Cairns 1992). Thus, use of monitoring metrics that respond to environmental changes on both short- and long-term time scales facilitates detection of various ecosystem changes, and may allow determination of the mechanistic basis of change (i.e. altered productivity or foraging success detected in the short-term may cause long-term changes in abundance).

Seabird life history is important to both application of monitoring methods and selection of focal species to be monitored. Traits that facilitate observations and counting are especially important. Species selected for monitoring should also represent the different habitats or regions of interest, with the knowledge that many of these species are capable of fast and far reaching flight. It is also important that specifics of diet and nesting are known, so that population monitoring can be interpreted for inferences about ocean condition, forage availability, and disturbance. In general, seabirds are long-lived species (often living >20 years; Clapp et al. 1982) that annually produce few offspring and provide a large amount of parental care compared to most marine species. During the breeding season, seabirds forage from a central point (the nest, often in colonies), returning to the nest throughout the day to incubate eggs and provision young. Though most "true" seabirds come to land only to breed, many coastal species rely on land throughout the year to rest, dry wetted plumage, and defend breeding sites. Below are biological profiles of the species included in our baseline monitoring for the NCSR. Note that these do not include seabirds that nest underground in burrows or in fine crevices. The Common Murre is one of the most abundant species breeding within the NCSR and because it is so conspicuous it received the most attention in our monitoring efforts. All other species listed below except the California Brown Pelican also breed within the NCSR. The California Brown Pelican breeds in southern California and Mexico, but migrates north in late summer and early fall. The NCSR provides important roosting habitat for this species. The Marbled Murrelet is an endangered species and population monitoring is performed by other programs (Appendix A), and rare species such as Marbled Murrelet are often difficult to use as indicators of large or mesoscale change in marine ecosystems because of their rarity. We therefore only monitored the foraging activity of this species incidental to the overall monitoring effort.

Common Murre (*Uria aalge*). Common Murres breed mainly on offshore rocks and islands as well as coastal cliffs where suitable habitat (inaccessible to terrestrial predators) can be found (Ainley et al. 2002). Most nest on the flatter or more gently sloped portions of offshore rocks/islands, but some nest on ledges of steep cliffs. Nesting is normally in very high densities, with breeding neighbors touching. No nest is built, and the single egg is laid directly on the ground. Colony and nest site fidelity is high. Both the male and female incubate the egg for a period of 26-39 days (average = 32). Young depart the colony with the male parent when only about one-fourth grown (average 23 days old, but as early as 15 days), then are raised to independence at sea. Their diet includes a variety of small schooling fish (e.g, juvenile rockfish *Sebastes* spp., smelt, salmon, northern anchovy (Golightly and Schneider 2016) and invertebrates such as squid and possibly krill.

Marbled Murrelet (*Brachyramphus marmoratus*). Marbled Murrelets nest in mature and old-growth coniferous forests, or forests with old-growth components (Nelson 1997). Adults lay a single egg on a large branch or platform lined with moss, needles and/or other debris (Hebert and Golightly 2006). Adults do not add nesting material and nest consists of a small depression formed by the weight of the incubating adult. Incubation lasts for approximately 28 days and chicks leave the nest after another 28 days. Adults forage primarily in bays and nearshore ocean (within 5 km of shore) in shallow water, usually <60 m deep. Diet is thought to consist of krill or small schooling fishes like northern anchovy, Pacific herring (*Clupea harengus*), and surf smelt (*Hypomesus* sp.) in the summer when rearing chicks. The Marbled Murrelet is federally listed as threatened and listed as endangered in California due to population declines resulting from breeding habitat loss.

Pigeon Guillemot (*Cepphus columba*). Pigeon Guillemots typically breed in rocky crevices in coastal cliffs or offshore rocks/islands (Ewins 1993). This species attempts only one successful brood per season. If the first nesting attempt fails (the egg(s) does not hatch), subsequent nesting attempts may occur. Guillemots typically nest in small colonies. Nests are perennial, with high nest site fidelity. Pigeon Guillemots lay 1-2 eggs (2 is the most common number). Both the male and female incubate the eggs for a period of 25-38 days (with 29 days being average). Young fledge in 29-54 days, with 38 days being the average fledging time. During the breeding season, guillemots form rafts on the water adjacent to their nesting areas. Rafting groups tend to be in the greatest numbers in the early morning hours. Pigeon Guillemots forage mainly among submerged reefs in nearshore waters (Ainley and Boekelheide 1990, Litzow et al. 2000, Robinette et al. 2007). Prey fed to chicks includes a variety of small fish and invertebrates such as juvenile rockfish, sanddabs, sculpins, and octopi.

Double-crested Cormorant (*Phalacrocorax auritus*). Double-crested Cormorants typically breed on ground or cliffs, in trees or shrubs (Dorr et al. 2014). This species typically attempts only one successful brood per season. Second broods have been reported but are extremely rare. If the first nesting attempt fails (the chicks do not survive to fledging), subsequent "relay" nesting attempts may be undergone. Double-crested Cormorants lay 1-7 eggs (5 eggs is most common) during a single nesting attempt. Both sexes incubate the eggs for 25-28 days. Fledging occurs in 40-50 days. Double-crested cormorants typically forage inshore, usually less than 5 km from shore in water less than 10 m deep (Dorr et al. 2014) and take schooling fishes such as silverside smelt (Family Atherinopsidae) and anchovies, as well as non-schooling fishes like croakers (Family Sciaenidae), midshipman (Family Batrachoididae), and surfperches (Family Embiotocidae; Ainley et al. 1981)

Pelagic Cormorant (*Phalacrocorax penicillatus*). Pelagic Cormorants typically breed on steep cliffs along rocky seacoasts and islands (Hobson 1997). This species attempts only one successful brood per season. If the first nesting attempt fails (the eggs do not hatch), subsequent "relay" nesting attempts may be undergone. Relay attempts will take place at the same nest site, usually in the original nest. Nests are located on the ledges of high, steep, inaccessible rocky cliffs facing water. Nests are of the platform type, and are made of seaweed and other marine algae, terrestrial vegetation, or only moss. Pelagic Cormorants lay 3-7 eggs (3-5 eggs is most common) during a single nesting attempt. Both sexes incubate the eggs for 26-35 days. Fledging occurs in about 40-50 days. Like the Pigeon Guillemot, Pelagic Cormorants forage mainly among submerged reefs in nearshore waters (Ainley et al. 1981). Their primary prey in central California includes small fish and invertebrates such as juvenile rockfish, juvenile sculpins, and mysid shrimp (*Spirontocaris* sp.).

Brandt's Cormorant (*Phalacrocorax pelagicus*). Brandt's Cormorants typically breed on the flatter or sloped portions offshore rocks and islands and on mainland cliffs (Wallace and Wallace 1998). This species attempts only one successful brood per season. If the first nesting attempt fails (the eggs do not hatch), subsequent nesting attempts may be undergone. Nests are composed of a variety of seaweed and other marine vegetation as well as terrestrial vegetation. Brandt's Cormorants lay 1-6 eggs (4 eggs is most common). Incubation lasts about 29-30 days. Fledging occurs in about 40-50 days. Brandt's Cormorants forage mainly over soft bottom, continental shelf habitats (Ainley et al. 1981). Their diet in central California includes a wide variety of schooling fish such juvenile rockfish, Northern anchovy, Pacific sand lance (*Ammodytes hexapterus*), and Plainfin midshipman (*Porichthys notatus*).

Black Oystercatcher (*Haematopus bachmani*). Black Oystercatchers typically breed on rocky coasts and islands, although nests have been occasionally found on sandy beaches (Andres and Falxa 1995). This species attempts only one successful brood per season. If the first nesting attempt fails (the chicks do not survive to fledging), subsequent "relay" nesting attempts may be undergone. Black Oystercatchers are monogamous, and have long-term pair bonds. They are also year-round residents that continually defend their feeding territories. Nests are of the scrape form, and are usually built above the high tide line in weedy turf, beach gravel, or rock depressions. Black Oystercatchers lay 1-3 eggs (2 eggs is most common). Incubation lasts 24-29 days. Chicks are precocial at hatching, but highly dependent on their parents for an extended period. Chicks rely on parents to show them food, and to teach them about appropriate food selection. Chicks fledge in approximately 35 days. Black Oystercatchers forage in rocky intertidal areas, where they feed mainly on a variety of intertidal marine invertebrates, particularly bivalves and other mollusks (limpets, whelks, and chitons; Andres and Falxa 1995).

Western Gull (*Larus occidentalis*). Western Gulls typically nest on rocky islets and coastal cliffs (Pierotti and Annett 1995). This species attempts only one successful brood per season. If the first nesting attempt fails (the chicks do not survive to fledging), subsequent "relay" nesting attempts may be undergone. Nests are perennial and are usually located on cliff ledges, grassy hillsides, or sometimes on human built structures. Western Gulls lay 1-5 eggs (3 is the most common number). Western Gulls are colonial and have been known to share nesting sites with other seabirds. Incubation ranges from 25-29 days (26 days is the average length). Chicks fledge in 42-49 days, yet often don't disperse from the colony until after 70 days. Western Gulls have a broad diet that may include subsidies from human landfills and fisheries discards. In central California, Robinette and Howar (2013) found Western Gull diet to be dominated by a variety of rocky intertidal invertebrates and nearshore fishes.

California Brown Pelican (*Pelecanus occidentalis californicus*). California Brown Pelicans breed on the northern Channel Islands (Santa Barbara and Anacapa) and migrate north along the California coast after breeding (Shields 2014). Brown Pelicans breeding in Mexico also migrate north after breeding. This species attempts only one successful brood per season. Ground nests are built steep, rocky slopes using vegetation, including kelp. Brown Pelicans lay 2-4 eggs (3 eggs is most common) during a single nesting attempt. Both sexes incubate the eggs for 29-32 days. Fledging occurs in about 70-81 days. During the post-breeding season, pelicans rely on coastal habitats as important roosting sites. The California Brown Pelican forages primarily on coastal pelagic fishes and has been recognized as an indicator of northern anchovy (*Engraulis mordax*) and Pacific sardine (*Sardinops sagax*) abundance (Anderson and Gress 1984). The California Brown Pelican was state and federally listed as endangered until 2007.

A key challenge in ecosystem-based management and evaluation of the effects of reserve systems such as MPAs is determining whether observed changes in monitored ecosystem processes (such as seabird population growth or diet) are caused by changes in management or by confounding environmental variation (Gerber et al. 2005, McCook et al. 2010). This challenge increases with increased environmental variation, since environmental fluctuations may mask the impacts of management. This challenge also increases with increased disparity between the spatial scale of management and monitored ecosystem processes, since events outside managed areas may influence events inside managed areas, and vice-versa. Thus, establishing a baseline monitoring program of highly mobile marine predators in dynamic environments requires establishing not only the current state, but also requires the characterization of normal spatiotemporal variability in monitoring metrics. We combined a suite of long-term monitoring approaches with short-term before-after control-impact designs to establish both the extent of long-term variability while examining short-term responses.

The cohesive understanding of marine conditions provided by monitoring these ecosystems at multiple temporal and spatial scales using diverse metrics required a diverse but collaborative project. We designed a comprehensive seabird monitoring effort to characterize and evaluate MPA performance in the NCSR. Multiple seabird species residing within the NCSR utilize a diverse array of habitats. Thus, it was necessary to focus on some MPA impacts to coastally-breeding species of seabirds that forage close to shore while other components of our study used more pelagic-oriented seabirds as indicators of the regional oceanographic conditions that mediate change within MPAs. Specifically, we determined the current abundance of seabirds across the region and utilized historical data of seabird abundance to assess trends and variability (Element 1; Chapter 2); examined recent historical data on seabird reproduction and diet at the largest seabird colony in the NCSR (Element 2; Chapter 3); assessed nearshore foraging locations, abundance, and reproduction using land-based monitoring stations (Element 3; Chapter 4); and evaluated the potential for citizen science to be used to assist in acquiring data to meet these goals (Chapter 5). Lastly, we synthesized the pertinent findings of these comprehensive seabird monitoring efforts and provided relevant management recommendations (Chapter 6).

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CHAPTER 2: TRENDS IN SEABIRD ABUNDANCE

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INTRODUCTION

Seabird abundance represents a potentially efficient and valuable monitoring metric of change in marine ecosystems, given that seabirds are readily observable, and are apex predators (Durant et al. 2009; see Introduction). However, the sometimes large spatial and temporal scales of the causes of seabird population change complicate study of the effects of relatively local management changes (Einoder 2009), such as the establishment of California's system of Marine Protected Areas (MPAs). We attempted to address this issue in the context of the North Coast Study Region (NCSR) Baseline Monitoring Program by creating long-term time series of seabird colony size by: 1) counting previously uncounted aerial photographic surveys, 2) compiling previously counted photographic aerial survey data provided by collaborators covering the interval 1989-2014, and 3) conducting and counting a complete region-wide aerial photographic survey in 2014.

The baseline provided by a long-term time series is characterized by two key elements: first, long-term trends in seabird abundance (both region-wide and at individual colonies), and second, the extent, type, and causes of temporal variation in colony sizes. Information on pre-existing trends and temporal variation provides an important means of assessing whether changes following MPA establishment are due to changes in management practices or environmental variability, as well as our future ability to detect changes caused by either. Monitored seabird populations have generally declined worldwide by 70% over the interval 1950-2010 (Paleczny et al. 2015) yet trends in seabird populations varied by region and ecology; thus, establishing trends for monitored seabird populations as well as current population sizes within the NCSR provides important context and a basis for future evaluation of the effects of MPA establishment. Two general types of MPAs were established within the NCSR: special closures around breeding colonies, that are likely to benefit seabirds by reducing disturbance at colonies, and reserves or conservation areas that could benefit seabirds by altering the availability (via effects on both abundance and distribution) of seabird prey items. We monitored colonies within, adjacent to, and distant from both types of MPAs throughout the NCSR, providing data critical in future before-after-control-impact study designs.

We developed and analyzed time series of Common Murre (*Uria aalge*) colony counts at 14 sites representing 13 colonies and Brandt's Cormorant (*Phalacrocorax pencillatus*) nest counts at 10 sites representing 8 focal colonies that span approximately the entire spatial extent of the NCSR (Figure 1) over the interval 1989-2014. In addition to this time series, we established baseline abundance of Common Murre, Brandt's Cormorant, Double-crested Cormorant (*Phalacrocorax auritus*), Pelagic Cormorant (*Phalacrocorax pelagicus*) and Western Gull (*Larus occidentalis*) at colonies throughout the NCSR in 2014 (Figure 1, Table 2). These four focal species are associated with alternative MPA ecosystem features (including Rocky Intertidal, Soft-bottom Intertidal & Beach, Kelp & Shallow Rock, Mid-depth Rock, Soft-bottom Subtidal, and Nearshore Pelagic) and oceanographic conditions (Ainley and Hyrenbach 2010). Common Murre and Brandt's Cormorant are piscivorous seabirds that forage via pursuit-diving (Ainley et al. 1996, Hobson 1997) and their colony attendance and nesting success is thought to reflect foraging conditions at sea (Zador and Piatt 1999, Harding et al. 2007). Further, temporal variation in the growth rates of seabird colonies is thought to reflect a combination of food availability driven by variable oceanographic conditions (Wolf et al. 2010), density-dependence (Lewis et al. 2001), and disturbance history at a colony (Carney and Sydeman 1999). This time series analysis and 2014 region-wide survey thus allows us to characterize current seabird population sizes and trends, and provides a prospectus on the potential to detect future change.

We used state-space models partitioned into process and observation components (e.g. de Valpine and Hastings 2002, Moore and Barlow 2011) to estimate the average annual percentage change in total population size, annual change in individual colony sizes, temporal process variance, and observation variance. We used Bayesian inference (specifically, Gibbs sampling, a Markov Chain Monte Carlo algorithm) to estimate the posterior probability distributions of model parameters. We used a Bayesian approach because of the ease of interpretation provided by posterior probability distributions of estimated parameters, which are interpretable as the probability that a trend occurred at a particular rate (Wade 2000). We report the estimated coefficients of variation of both sampling variance and process variance, and their Bayesian credible intervals, to assess the potential effectiveness of this technique for detecting future changes in population size.

METHODS

Field and Counting Methods: Baseline Year

Surveys

In 2014, all Common Murre, Brandt's Cormorant, and coastal Double-crested Cormorant breeding colonies in the North Coast Study Region were surveyed once with aerial photographs on 5 June (Punta Gorda to Oregon border, including Humboldt Bay) or 7 June (Point Arena to Punta Gorda). These species often breed in intermixed colonies and in colonies with discrete perimeters, such that complete coverage of all breeding areas with aerial photographs is easily obtained. All previously known colonies were surveyed, and other coastal areas were inspected to detect possible new colonies. Complete coverage of Western Gulls and Pelagic Cormorants was obtained only at Castle Rock NWR, where typically all surface area is photographed for complete coverage of Common Murres. Complete coverage of Western Gulls (which have broader and less dense nesting distributions that murres and cormorants) and Pelagic Cormorants (which nest on cliff ledges) typically requires greater aerial survey effort and boat surveys, respectively. Roosting Brown Pelicans also were photographed where they occurred near or adjacent to murre and cormorant breeding colonies.

Surveys were conducted from a California Department of Fish and Wildlife Partenavia aircraft at altitudes of about 650' to 1000' above sea level, utilizing a port in the belly of the aircraft for vertical photography. Photographs were taken between 11:00 and 16:00 hrs, with murre colonies all surveyed by 14:30 hrs. Two biologists photographed each colony, using Canon 60D and 50D cameras equipped with a 200 mm telephoto lens (for the principal photographs used for counting) and 17-85 mm and 70-200 mm zoom lenses (for overview and broader coverage photographs), respectively. A third observer in the co-pilot's seat recorded notes. Aerial photographs (nearly 3,000 images totaling more than 24 GB of storage) were archived by UCSC and USFWS (Don Edwards San Francisco Bay National Wildlife Refuge). These surveys followed methods that have been standardized since 1985, with updates to vertical photography in 1997 and to digital photography in 2007 (Carter et al. 1992, 2001; Capitolo et al. 2014).

Image analysis

The best aerial photographs, considering image quality and extent of colony coverage, were analyzed by manually marking nests, sites, and birds on images, using Image-Pro software developed by Media Cybernetics, and following standardized protocols (Carter et al. 1992, 2001; Capitolo et al. 2014). Multiple images were used to determine complete colony counts, with overlapping areas delineated to prevent double-counting. Categorical counts were tallied automatically by the software. A screen capture of each counted image was saved, and counts from individual screen captures were entered into a relational database and summed to determine whole-colony counts.

For Common Murres, only birds were counted because murres do not build nests. For cormorants, nests were categorized by stage of development, including poorly built nests (before egg-laying), well-built nests with an incubating adult, and nests with visible chicks. Empty (attended by an adult) and abandoned (not attended by an adult) well-built nests also were categorized, though few occurred. Birds in suitable breeding habitat and densities, but with little or no nesting material present, and that sometimes can be seen to be displaying, were categorized as territorial sites. Territorial sites are not included in total numbers of nests for cormorants, though some of these likely become egg-laying sites following aerial surveys. Other cormorants such as attending mates were categorized as birds. Western Gulls were typically categorized by posture, with sitting birds marked as either a site or a nest (if nesting material was seen). Standing gulls were marked simply as birds, unless two standing gulls appeared to be a mated pair, in which case one of the birds was marked as a site. Nests and sites were summed to estimate breeding population size. No correction factors were applied to whole-colony counts (correction factors have been used widely in studies of seabird population size, but because we apply a state-space model with an observation error component, below, we do not). Roosting Brown Pelicans were aged as adults (white heads) or immatures.

Repeated counts over time

In addition to collecting 2014 baseline data, we aimed to complete and analyze trends for long-term count data since 1989 for two colonies, Castle Rock NWR and the Cape Vizcaino/Rockport Rocks Colony Complex, at the northern and southern ends of the North Coast Study region, respectively (Figure 1, Table 2). In addition to the potential for spatial comparison of long-term trends, these colonies were selected because Castle Rock NWR is by far the largest seabird colony in the region and because the Cape Vizcaino/Rockport Rocks time series was nearly complete before this project. For Castle Rock NWR, we determined whole-colony counts from archived aerial photographs for 2012 and 2010. Because of the great effort needed for complete counts of Castle Rock NWR, however, we counted only the East Side from archived photographs from eight other years (i.e., 2013, 2011, 2009, 2008, 2006, 2005, 2002, and 2000). Data from other years, including East Island subtotals, were collated (Carter et al. 1992, Jaques and Strong 2001, Capitolo et al. 2006; USFWS, unpubl. data). For Cape Vizcaino/Rockport Rocks, we only needed to analyze images from 2011-2013 for this report, as photographs from previous years had already been analyzed (Carter et al. 1992, 2001; Capitolo et al. 2006, Parker et al. 2011). For other colonies in the region, data were also available from previous studies for certain years and these were also incorporated in trend analyses (Carter et al. 1992, 2001; Capitolo et al. 2006, 2016; Thibault et al. 2010).

Trend analysis

We estimated the parameters of a state-space model of population growth. The model we fit is a multivariate generalization of state-space models of population growth (de Valpine and Hastings 2002; Paleczny 2015) for time series with more than one series of counts, such as our time series of 14 different Common Murre sites (representing 13 colonies) or 10 different Brandt’s Cormorant sites (representing 8 colonies). We considered several different models with alternative correlation structures between sites and within years, but found few substantive differences in qualitative or quantitative inferences and thus focus on the results of the model described here. We started by assuming the population growth of the *i*th colony population size in time *t*, $N_{i,t}$ is a first-order Markovian process:

$$\log(N_{i,t+1}) = \log(N_{i,t}) + r_{i,t}$$

In this model the estimated parameter $r_{i,t}$ is the intrinsic population growth rate and can be expressed as an annual rate of change for an exponentially-growing population (i.e. multiplying by 100 yields a percentage annual growth rate similar to those yielded by traditional log-linear regression methods). We report the mean and 95% credible intervals of the posterior probability distribution of this (and other) parameters, which are interpretable as the probability that a population increased at a particular rate over time (see Wade 2000).

We incorporated the well-documented effects of the El Niño phenomenon on seabird population growth (e.g. Ribic et al. 1992) into this model by including an environmental covariate, the annual mean of monthly Southern Oscillation Index values (SOI; obtained from the National Climatic Data Center), in our population growth model. We defined the growth rate $r_{i,t}$ at site *i* in time *t* as a linear function of a site-specific intercept and the annual SOI value:

$$r_{i,t} = \beta_{0_{i,t}} + \beta_1 soi_t$$

The intercept of this linear function $\beta_{0_{i,t}}$ was defined as a normally distributed random variable with site-specific process variance σ_i^p which thus makes this model analogous to a "random-intercept" random effects model:

$$\beta_{0_{i,t}} \sim N(\overline{\beta_{0_i}}, \sigma_i^p)$$

The parameter σ_i^p thus represents annual variance in the population growth rate, while β_1 represents the slope of the effect of increasing SOI on the estimated population growth rate, $r_{i,t}$. The observed population counts y were modeled as normally distributed random variables with site-specific observation variance σ_i^o:

$$y_{i,t} \sim N(\log(N_{i,t}), \sigma_i^o)$$

Interpreting this particular model of process variance when applied to whole-colony counts of birds (in the case of Common Murre) and nests (in the case of Brandt's Cormorant) aerial photographic surveys requires several caveats. First, aerial photographic surveys have minimal observation error, because the counts are highly repeatable and the population observed in counts represents a complete census of the individuals present in a given colony on the occasion of observation. Second, colony attendance can be highly variable from year to year while the actual total population size (including unobservable at-sea birds) is likely to be less so (see Chapter 2). This allows interpretation of the estimated observation variance parameter, σ_i^o as representing variance in the proportion of the population that is observed, and the estimates of process variance, σ_i^p, as representing variance in annual population growth rate at each colony over time.

We used minimally-informative priors for all parameters except the first population size estimate for each site $N_{i,1}$ for which we used a moderately-informative prior (following Kéry and Schaub 2011) because this was the first step of the observed time series and Markovian population growth process:

$$N_{i,1} \sim N(y_{i,1}, 100)$$

We sampled the posterior probability distribution of model parameters using a Markov Chain Monte Carlo algorithm, specifically, a Gibbs sampler as implemented in JAGS 4.2.0 (Plummer 2003). We ran 3 independent chains for 50,000 "burn-in" iterations and an additional 50,000 iterations for inference. We selected every 50th iteration for a total of 1,000 samples from each of our 3 chains. We assessed convergence by visually inspecting each chain and using the potential scale reduction factor, or Gelman-Rubin statistic (Gelman and Rubin 1992). The Gelman-Rubin statistic was less than 1.1 for all parameters, suggesting that our Markov Chains had converged on a stationary distribution and were thus a reasonable approximation of the posterior probability distribution, and useful for inference. We report posterior means, standard deviations, and 95% credible intervals from summary statistics of samples from all three chains, and thus n=3000 for each. To allow direct comparison of process and sampling variance components, we calculated the coefficient of variation of each using the posterior probability distributions. We specified the described model in the JAGS language and fit using R 3.3.2, JAGS 4.2.0, and R library R2jags.

RESULTS AND DISCUSSION

Baseline Year Results

Aerial photographic surveys conducted June 5 and June 7 2014 enumerated Common Murre attendance at 19 colonies, Brandt's Cormorants at 20 colonies (with nests at 19), and Double-crested Cormorants at 13 colonies (Table 2). Region-wide counts of each species in 2014 were 350,923 Common Murre, 6,689 Brandt's Cormorant, and 2,589 Double-crested Cormorant (Table 2). Region-wide counts of nests (sometimes doubled to estimate the number of breeding individuals) in 2014 were 4,583 Brandt's Cormorant and 1,840 Double-crested Cormorant (Table 2). Nest counts from aerial surveys are not possible for Common Murre. The large 2014 population size of Common Murre in the North Coast Study Region included the largest single-island seabird colony in California, Castle Rock NWR, which hosted 142,696 murres in 2014. These counts represent an undercount of total seabird population sizes, because colony attendance does not include non-breeding and non-attending individuals, and does not include breeding individuals away from the colony at the time of survey. However, such counts are an index of breeding population size (and change in population size over time) and thus represent an important baseline for assessment of future changes that result from MPA establishment. Further, the surveyed colonies represent an effective sample for future study of the effects of seabird breeding season special closures, now in effect at 6 of the surveyed colonies: Cape Vizcaino, Rockport Rocks, Sugarloaf Rock, Steamboat Rock, False Klamath Rock, and Castle Rock (Figure 1, Table 3). Marine reserves or marine conservation areas are in effect in proximity to or surrounding 9 of the colonies surveyed in 2014 (Figure 1, Table 3). The remaining colonies not included in MPAs may thus be considered as representing a "control" sample for studying future effects of MPAs. Because we do not have data from 2010-13 at most colonies within the region, we are unable to assess possible initial responses to MPAs using this BACI design at this time, but the original photographic surveys for this type of assessment are available.

The same set of aerial photographic surveys also provided counts of Brown Pelicans at 14 roost sites (Table 3), and complete counts of Pelagic Cormorant (303 nests, 25 sites, 356 individuals) and Western Gull (559 nests, 698 individuals) at Castle Rock NWR in 2014.

Trend Analysis

Aerial photographic surveys previously conducted in the region from 2000-2013 at Castle Rock NWR (Table 4) and from 2011-2013 at Rockport Rocks and Cape Vizcaino were counted (Table 5) and compiled into a larger dataset with previously existing data (see methods) for trend analysis.

We estimated parameters of the state-space model for Common Murre using a combined dataset that included 173 observations of colony size at 14 colonies (although 2 were actually sub-colonies, at Castle Rock NWR) over the 26-year study period. Common Murre colony attendance population size estimates and observed counts generally showed a pattern of consistent increase over the study period with the exception of Redding Rock and Goat Island (Figure 4, Figure 5). We estimated parameters of the state-space model for Brandt's Cormorant using a combined dataset that included 123 observations of colony size at 10 colonies (although 3 were sub-colonies at Castle Rock NWR – East, West, or Total) over the 26-year study period. Brandt's Cormorant estimated nesting population size and observed counts were more highly variable and did not show a consistent pattern of growth or decline over the 26-year study period (Figure 6, Figure 7).

Common Murre and Brandt's Cormorant region-wide population size estimates were obtained by summing the posterior probability distributions of each of the individual sites, providing trend estimates and 95% credible intervals of population sizes for each year within the time series (Figure 2, Figure 8). Annual population growth rates of both species were positively related to the annual mean of the southern oscillation index value, as described by the posterior probability distribution of the slope of the relationship between SOI and annual growth rate (Figure 9, Figure 10). Patterns of population growth of

Common Murre colony attendance were variable among colonies but generally showed a consistent pattern of increase with the exception of Redding Rock and Goat Island (Figures 11-24). Patterns of population growth of Brandt's Cormorant nest abundance were variable among colonies and did not show a consistent pattern of growth or decline (Figures 25-32).

Estimated annual population growth rates of Common Murre (Table 6) were variable among years and among sites. The overall estimated mean growth rate for the region was 4.0% (r = 0.040) over the observed interval which stands in contrast to regional declines in Washington and Oregon (Horton 2014). Colonies varied in their mean annual growth rates and the variance of their annual growth rates, perhaps most readily compared by examining the colony-specific coefficient of variation (CV; Table 6). Mean colony-specific growth rate varied from -6.9% (r = -0.069 at Redding Rock, a site which has historically experienced high levels of disturbance; Thibault et al. 2010) to 8.5% (r = 0.085 at Rockport Rocks; Table 6) and 16.9% (r = 0.169 at Goat Island; Table 6). The seemingly improbable estimated growth rates at Rockport Rocks and Goat Island, which are approximately twice and three times the region-wide mean, are likely driven by emigration from other sites (either of first-time breeders or breeding dispersal) given that these growth rates appear larger than possible from local contributions to population growth alone.

The state-space model we used separated observation (or sampling) variance and process variance (i.e. variance in population growth) allowing for a simple comparison of the magnitude of each variance to tentatively assess the effectiveness of this monitoring technique. The CV of observation variance was ca. 20-50 times smaller than the CV of process variance (Table 7) across all colonies, strongly suggesting that this approach is an effective technique for measuring annual variation in population sizes and growth. Limitations of the precision of our estimates of population size and growth are manifest at sites sparsely sampled (see Figures 11-32, shown graphically as very wide credible intervals in years with uncounted photographic aerial surveys. This limitation could be ameliorated by dedicating additional counting effort to improving the sampling within the historic time series. Similarly, our assessment of the initial benefits of MPAs is limited because the only sites with complete recent time series are within special closures, meaning no control sites are currently available for comparison.

Estimated annual population growth rates of Brandt's Cormorants (Table 8) were highly variable among years and among sites. The overall estimated mean growth rate for the region reflects a general lack of long-term change over the observed interval (r = -0.009) which contrasts with regional growth in Brandt's Cormorant populations further south in California (Capitolo et al. 2016). Colonies varied in their mean annual growth rates and the variance of their annual growth rates (Table 9). Mean colony-specific growth rates varied from -4.4% (r = -0.044 at False Cape) to 3.6% (r = 0.036 at Rockport Rocks). The CV of observation variance was again much smaller that the CV of process variance (Table 9), again strongly suggesting that this approach is an effective technique for measuring annual variation in population sizes and growth.

Following estimated population size declines for both species from 1989-1997, population growth of both species appeared to shift to a generally increasing (Common Murre; Table 6) or more variable (Brandt's Cormorant; Table 8) pattern following the 1997-98 interval (Figure 2, Figure 8). The 1997-98 interval had one of the lowest estimated growth rates for both species across years and colonies (Table 6, Table 8), which is fairly typical of the effects of strong El Niño events on seabird population growth. The estimated positive relationship between the annual mean SOI value and Common Murre and Brandt's Cormorant population growth (Figure 9, Figure 10) is likely partly due to this singular event. This interval is characterized as one of the most powerful El Niño events in recorded history (McPhaden 1999). Post-hoc inspection of population growth rates revealed a shift from a period of relative Common Murre population size stability from 1989-1997 During the baseline monitoring period, a powerful El Niño event again occurred during the 2015-16 interval (McClatchie et al. 2016, L'Hereux et al. *in press*), and thus studies of the responses of seabird populations to MPA establishment in this region that incorporate the 2014 baseline data, which were observed immediately before this event, should consider the impacts of this event given the effects of the previous 1997-98 event on seabird population growth.

Long-Term Monitoring Recommendations

The design of long-term monitoring programs to detect the effects of MPAs on seabird abundance or distribution should incorporate several of the points revealed by this analysis and the 2014 baseline survey. First, Common Murre and Brandt's Cormorant population sizes varied widely during the 1989-2014 study period, and appeared to include at least two distinctive patterns of growth (before and after the 1997-1998 interval). Second, estimates of observation variance from the state-space model were very small – the coefficient of observation variance was ≤ 0.1 across both species and all colonies. Third, estimates of process variance were very large – the coefficient of process variance was ≥ 2.57 across both species and all colonies, and was often substantially larger. The coefficient of process variation was generally much more than an order of magnitude (often *two* orders of magnitude) larger than the coefficient of observation variation. The posterior probability distribution of r is directly interpretable as the probability a colony grew at a given rate over an observed interval, obviating the need for 'frequentist' approaches to the estimation of power (Wade 2000) and instead placing the focus on the relative size of process and observation variance components. This conclusion is contingent upon the structural and distributional assumptions made by our state-space model, and inference could be improved by conducting multiple surveys in a single season, which may more effectively characterize observation variance, perhaps by applying binomial mixture models (Kéry et al. 2005). Thus, annual aerial surveys of NCSR colonies appear to yield data useful in detecting change over time, and potentially, in detecting even relatively small effects of MPAs on population growth, contingent upon some of our model-based assumptions. We suggest that the use of aerial photographic surveys is an effective technique for characterizing broad-scale variation in seabird population growth over space and time within the NCSR. The time series developed and 2014 baseline survey provide a key benchmark for monitoring the effectiveness of two distinct types of MPAs: special closures and marine reserves or conservation areas. The complete sample of colonies represented in the 2014 baseline survey and the large number of colonies, including protected and unprotected sites, represented in the long-term trend analysis will provide a wide array of opportunities for future study designs to examine the effectiveness of MPAs. These data will allow strong BACI study designs to examine the future effects of MPAs on seabird populations and to evaluate alternative hypotheses for the mechanism of MPA effects on seabirds (i.e. via reduction of disturbance, increase in forage availability, or both). Aerial photographic surveys appear to be a cost-effective technique for temporally and spatial variance in population growth of seabird populations such as Common Murre and Brandt's Cormorant.

ACKNOWLEDGMENTS

In 2014, aerial surveys were conducted by PJC, Mike Parker, and Allison Fuller, in an aircraft piloted by California Department of Fish and Wildlife Air Services pilots Wayne Burnett and Gavin Woelfel. Counting of aerial photographs was conducted by PJC, Stephanie Schneider, and Crystal Shore.

TABLES & FIGURES

Table 2. Colony attendance (birds), nest, and site counts from 2014 baseline survey for Common Murre, Brandt’s Cormorant, and Double-crested Cormorant at 31 colonies or complexes in the North Coast Study Region. Status as a special closure, or proximity to or location within a marine reserve or conservation area, is also noted for each colony.

			Common Murre	**Brandt’s Cormorant**			**Double-crested Cormorant**		
Colony	**CCN[1]**	**Date**	**Birds**	**Nests**	**Sites**	**Birds**	**Nests**	**Sites**	**Birds**
Prince Island (DN)[3]	DN-414-03	05-Jun-14	1119	311	16	388	312	0	337
Castle Rock NWR[2,3]	DN-414-06	05-Jun-14	142696	1243	187	1593	20	1	23
False Klamath Rock[2]	DN-412-07	05-Jun-14	37416	88	22	146	45	0	50
Radar Station Rocks	DN-412-09	05-Jun-14	-	-	-	-	15	0	15
Flint Rock Head	DN-412-10	05-Jun-14	-	-	-	-	65	0	72
White Rock (DN)	DN-412-11	05-Jun-14	34	209	42	297	10	0	10
Redding Rock[3]	HU-412-01	05-Jun-14	125	73	2	109	-	-	-
Big Lagoon	HU-410-20	05-Jun-14	-	-	-	-	20	0	21
Sea Gull Rock	HU-410-05	05-Jun-14	-	-	-	-	10	0	11
White Rock (HU)	HU-410-08	05-Jun-14	3829				-	-	-
Green Rock	HU-410-09	05-Jun-14	38851	200	19	234	-	-	-
Flatiron Rock	HU-410-13	05-Jun-14	58333	407	56	551	-	-	-
Blank Rock	HU-410-14	05-Jun-14	1212	0	0	0	-	-	-
Pilot Rock	HU-410-16	05-Jun-14	2603	0	0	0	0	0	0
Little River Rock	HU-410-19	05-Jun-14	-	-	-	-	102	0	111
Arcata Bay Sand Islands	HU-404-01	05-Jun-14	-	-	-	-	417	1	705
Old Arcata Wharf	HU-404-02	05-Jun-14	-	-	-	-	31	0	41
Teal Island	HU-404-05	05-Jun-14	-	-	-	-	700	49	1104
False Cape Rocks	HU-402-01	05-Jun-14	25998	87	31	157	0	0	0
Sugarloaf Island[2,3]	HU-402-02	05-Jun-14	-	0	0	0	91	2	101
Steamboat Rock[2,3]	HU-402-03	05-Jun-14	14977	274	26	336	-	-	-
3 Bros. & Hair Seal Rock	HU-400-01	05-Jun-14	184	20	5	106			
High Tip	ME-394-02	07-Jun-14	2	146	3	170	-	-	-
Mistake Point-Big White Rock	ME-394-04	07-Jun-14	0	0	0	0	-	-	-
Soldier Frank Point	ME-394-06	07-Jun-14	17	17	7	35	-	-	-
Rockport Rocks[2,3]	ME-394-07	07-Jun-14	9274	663	63	991	-	-	-
Cape Vizcaino[2,3]	ME-394-08	07-Jun-14	11386	192	139	425	-	-	-
Kibesillah Rock[3]	ME-392-04	07-Jun-14	0	48	4	64	2	0	3
Newport Rocks[3]	ME-392-05	07-Jun-14	0	0	0	79	-	-	-
Goat Island Area	ME-390-01	07-Jun-14	2121	380	122	646	-	-	-
Schoolhouse Creek-Albion River	ME-390-06	07-Jun-14	-	21	1	35	-	-	-
Devils Basin	ME-390-08	07-Jun-14	0	48	1	67	-	-	-
White Rock (ME)	ME-390-13	07-Jun-14	746	156	39	260	-	-	-
TOTAL			**350923**	**4583**	**785**	**6689**	**1840**	**53**	**2589**

[1] California Colony Number following Carter et al. 1992.

[2] Colonies included in a special closure.

[3] Colonies adjacent to or enclosed within a marine reserve or marine conservation area.

Table 3. Roosting bird counts from 2014 baseline survey for Brown Pelican at 14 colonies or complexes in the North Coast Study Region. Note that because Brown Pelicans are a non-breeding in this region, we necessarily assume closure between survey dates when providing totals.

			Brown Pelican (roosting counts)		
Colony	**CCN[1]**	**Date**	**Birds**	**Adult**	**Imm.**
Prince Island (DN)	DN-414-03	05-Jun-14	114	107	7
Rock Rock	DN-412-01	05-Jun-14	20	19	1
False Klamath Rock	DN-412-07	05-Jun-14	28	28	0
White Rock (DN)	DN-412-11	05-Jun-14	167	159	8
High Bluff South	DN-412-12	05-Jun-14	104	101	3
Redding Rock	HU-412-01	05-Jun-14	13	13	0
Green Rock	HU-410-09	05-Jun-14	11	10	1
Flatiron Rock	HU-410-13	05-Jun-14	32	31	1
Little River Rock	HU-410-19	05-Jun-14	31	31	0
3 Bros. & Hair Seal Rocks	HU-400-01	05-Jun-14	8	8	0
High Tip	ME-394-02	07-Jun-14	25	22	3
Mistake Point-Big White Rock	ME-394-04	07-Jun-14	18	18	0
Soldier Frank Point	ME-394-06	07-Jun-14	99	95	4
Rockport Rocks	ME-394-07	07-Jun-14	54	52	2
Newport Rocks	ME-392-05	07-Jun-14	83	62	21
Goat Island Area	ME-390-01	07-Jun-14	91	84	7
White Rock (ME)	ME-390-13	07-Jun-14	162	150	12
TOTAL			**1060**	**990**	**70**

[1] California Colony Number following Carter et al. 1992.

Table 4. Castle Rock NWR 2000-2013 photographic aerial surveys counted as part of the NCSR baseline survey for Common Murre, Brandt's Cormorant, Double-crested Cormorant, Pelagic Cormorant, and Western Gull. Common Murre counts are total adults split into the eastern and western portions of the island. In years when only East Island was counted, reported cormorant counts may be slight undercounts.

	Common Murre			**Brandt's Cormorant**			**Double-crested Cormorant**			**Pelagic Cormorant**			**Western Gull**		
Date	**East**	**West**	**Total**	**Nests**	**Sites**	**Birds**	**Nests**	**Sites**	**Birds**	**Nests**	**Sites**	**Birds**	**Nests**	**Birds**	**Counter**
12-Jun-13	42997			1010	181	1458	31	0	38						CS
14-Jun-12	32819	87787	120606	825	276	1544	20	0	25	93	12	131	473	685	CS
09-Jun-11	33315			1034	256	1514	36	0	55						CS
14-Jun-10	29530	80915	110445	1291	188	2355	32	13	53	226	53	328	592	673	SS
28-May-09	29069			1111	43	1292	41	0	47						CS
05-Jun-08	28716			1043	173	1374	14	0	17						CS
09-Jun-06	25210			859	221	1309	48	0	52						CS
15-Jun-05	15606			1157	79	1387	51	0	60						CS
14-Jun-02	16925			994	80	1267	12	0	17						CS
15-Jun-00	14356			627	49	809	0	0	0						CS

Table 5. Rockport Rocks and Cape Vizcaino 2011-2013 photographic aerial surveys counted as part of the NCSR baseline survey for Common Murre, Brandt's Cormorant, and Brown Pelican.

		Common Murre	**Brandt's Cormorant**			**Brown Pelican**			
Colony	**Date**	**Birds**	**Nests**	**Sites**	**Birds**	**Birds**	**Adults**	**Imms**	**Counter**
Rockport Rocks	08-Jun-11	3775	70	1	89	48	46	2	PJC
Rockport Rocks	12-Jun-12	3958	196	98	332	4	2	2	PJC
Rockport Rocks	12-Jun-13	7052	627	305	1204	101	85	16	PJC
Cape Vizcaino	08-Jun-11	10016	585	341	1166	14	14	0	PJC
Cape Vizcaino	12-Jun-12	10318	187	7	180	0	0	0	PJC
Cape Vizcaino	12-Jun-13	12356	174	4	242	0	0	0	PJC

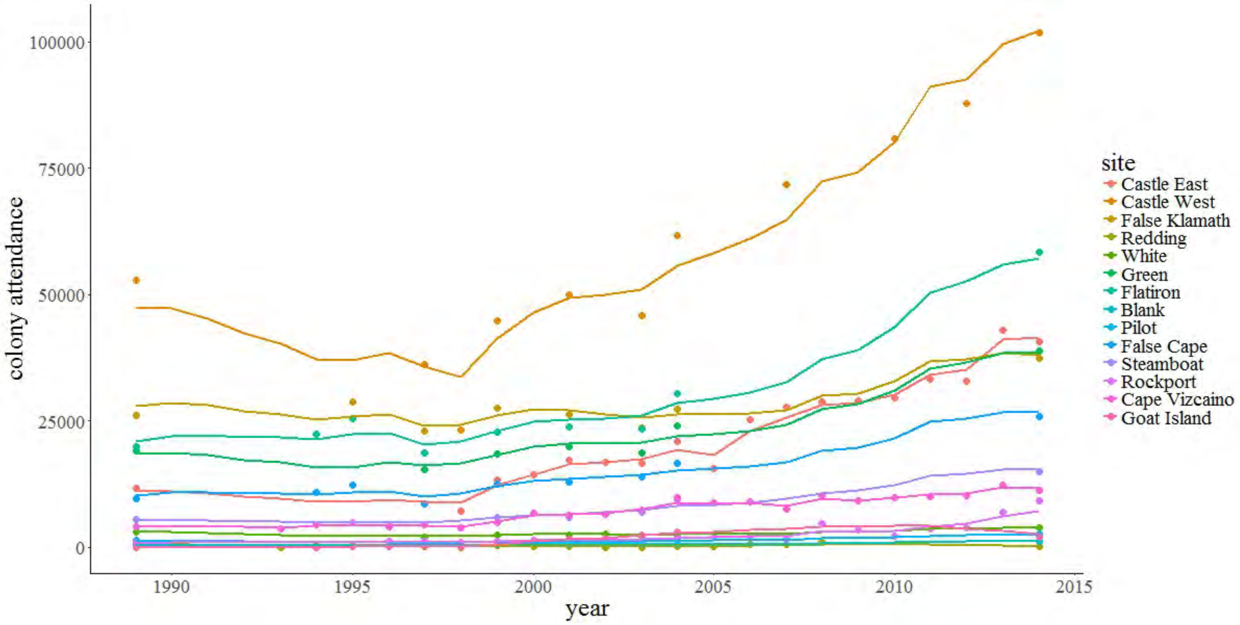

Figure 4. Colony-specific Common Murre trend estimates and observed counts, 1989-2014. Solid symbols are counts of attendance. Solid lines represent colony population size estimates from a state-space model of population growth (see methods).

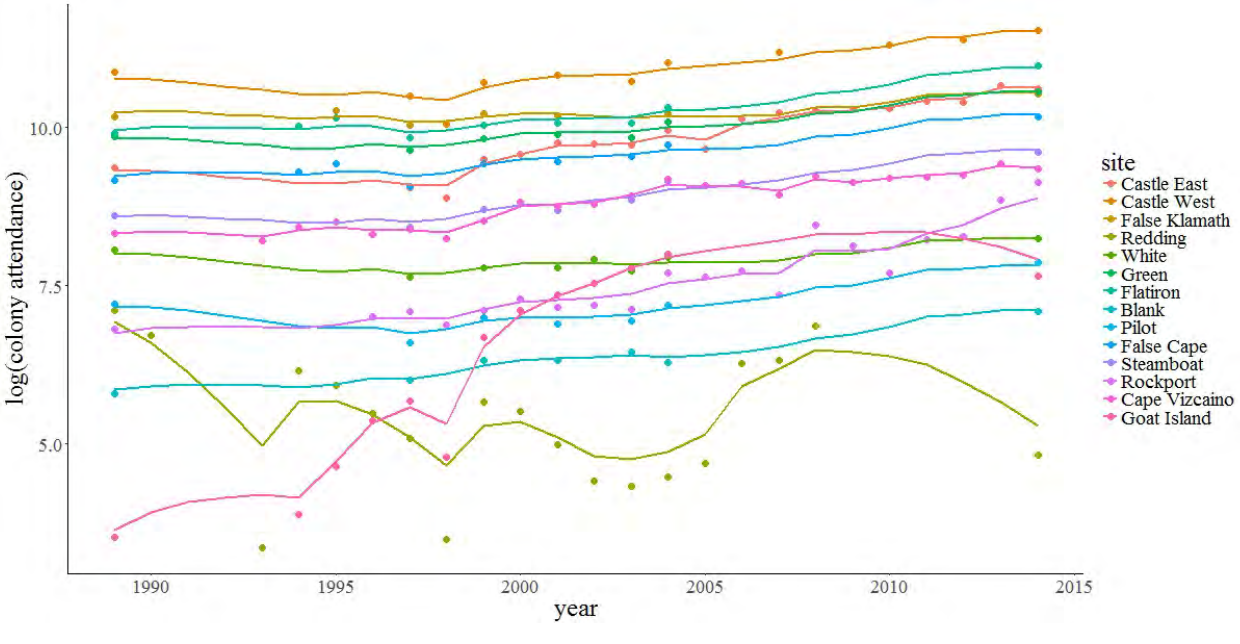

Figure 5. Log-transformed colony-specific Common Murre trend estimates and observed counts, 1989-2014. Solid symbols are counts of attendance. Solid lines represent colony population size estimates from a state-space model of population growth (see methods).

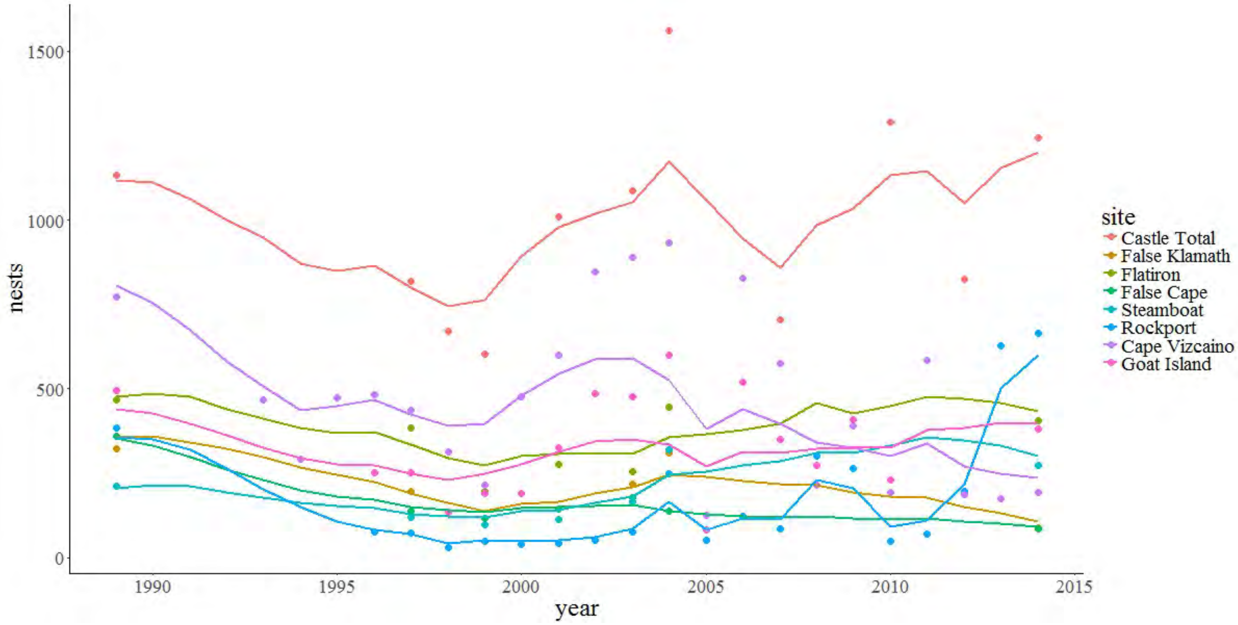

Figure 6. Colony-specific Brandt's Cormorant trend estimates and observed counts, 1989-2014. Solid symbols are counts of nests. Solid lines represent colony population size estimates from a state-space model of population growth (see methods).

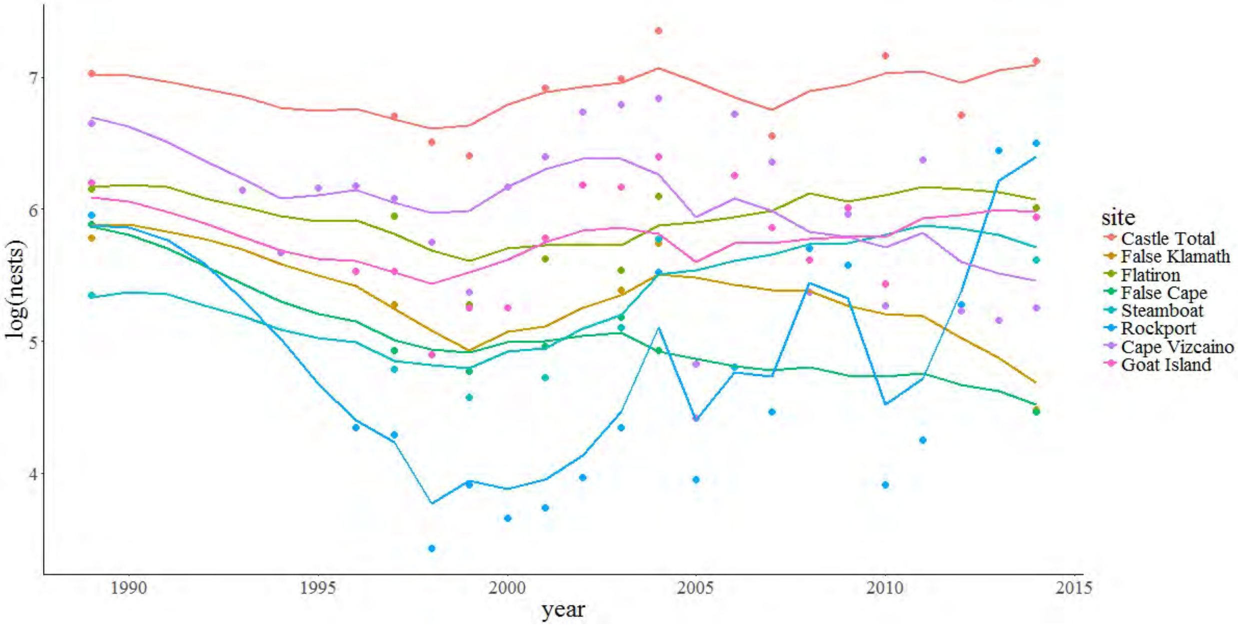

Figure 7. Log-transformed colony-specific Brandt's Cormorant trend estimates and observed counts, 1989-2014. Solid symbols are counts of nests. Solid lines represent colony population size estimates from a state-space model of population growth (see methods).

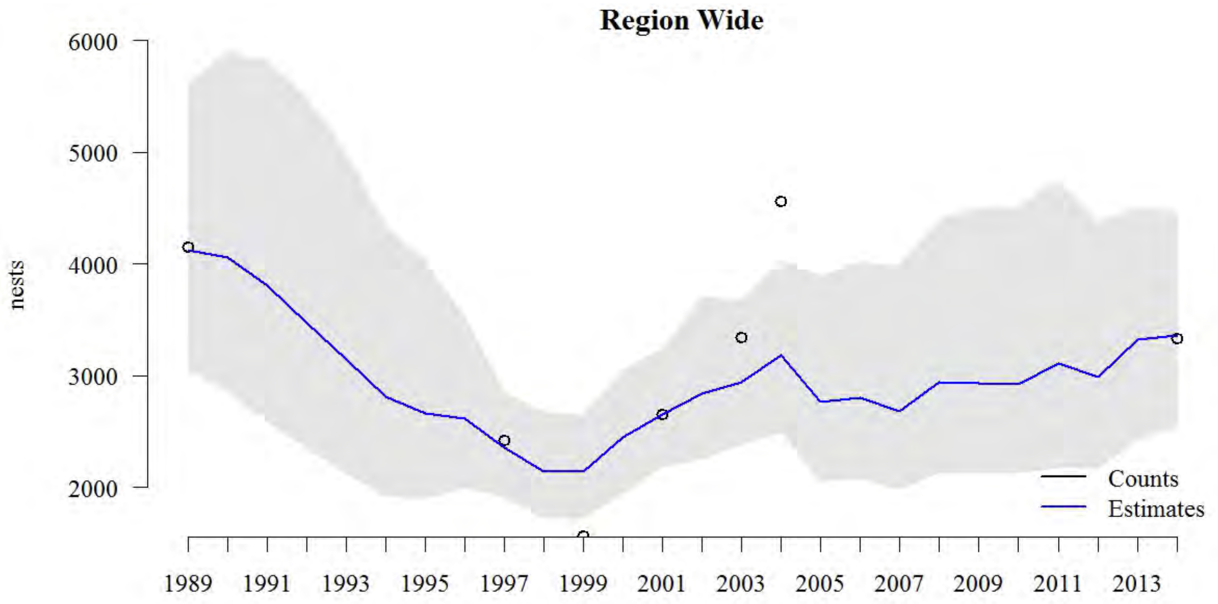

Figure 8. Region-wide Brandt's Cormorant trend estimates and observed counts, 1989-2014. The symbols shown are the 6 years with complete surveys of all 8 colonies – count data were available from other years, but only at a varying subset of the colonies.

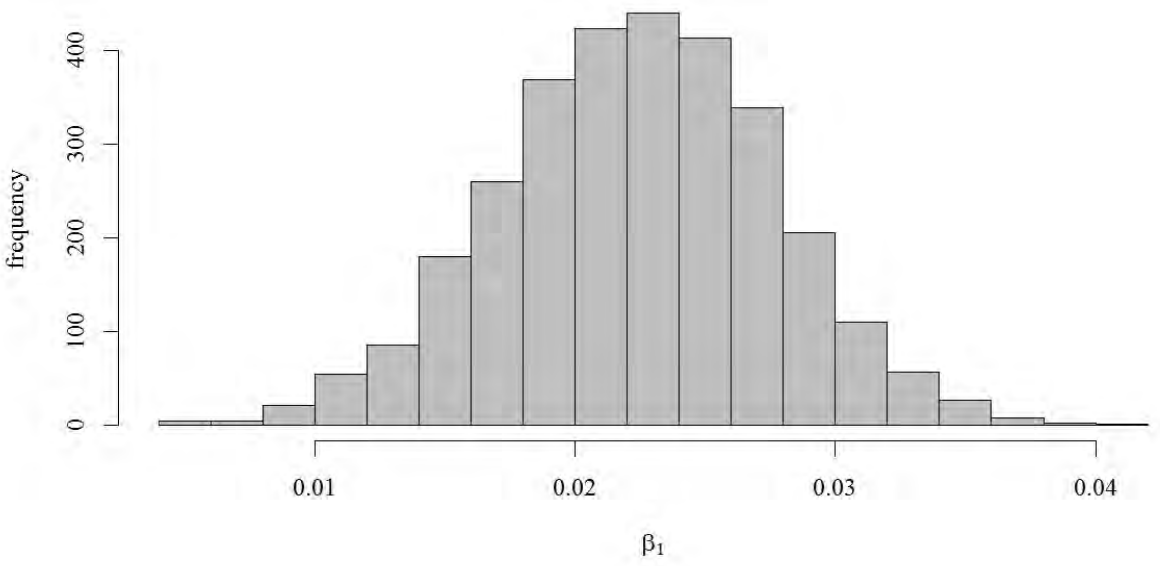

Figure 9. Posterior probability distribution of the slope of the linear effect (β1) of El Niño Southern Oscillation Index (SOI) on population growth rate estimates across North Coast Study Region Common Murre colonies. The mean of the posterior of the β_1 parameter is 0.022, suggesting that for every unit increase in SOI, annual population growth rate increases by 2.2%, and that conversely, growth rate decreases in El Niño years.

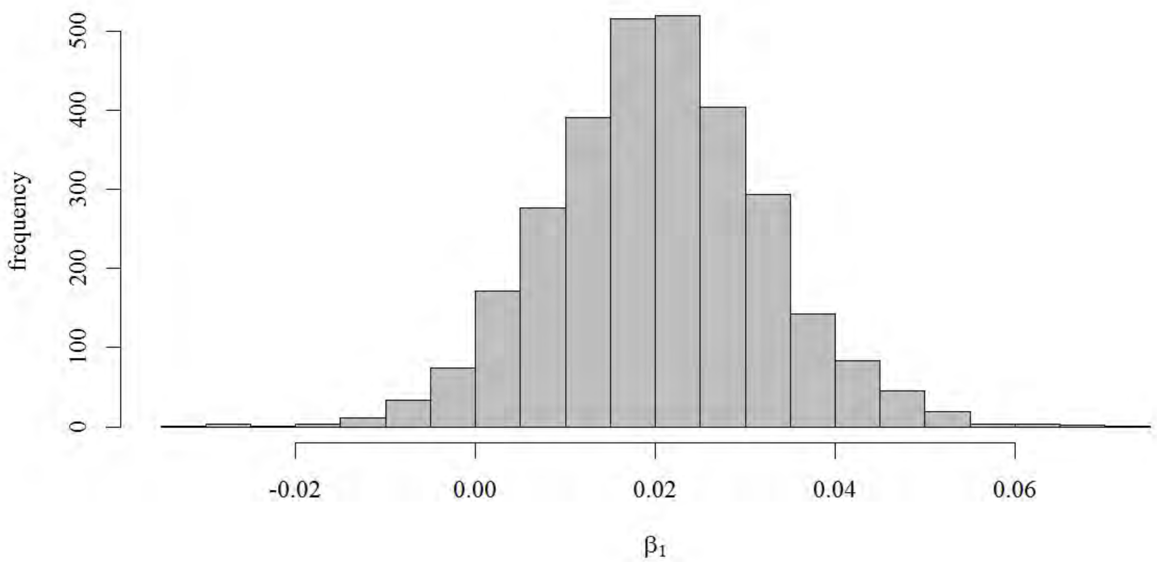

Figure 10. Posterior probability distribution of the slope of the linear effect (β1) of El Niño Southern Oscillation Index (SOI) on population growth rate estimates across North Coast Study Region Brandt's Cormorant colonies. The mean of the posterior of the β_1 parameter is 0.020, suggesting that for every unit increase in SOI, annual population growth rate increases by 2%, and that conversely, growth rate decreases in El Niño years.

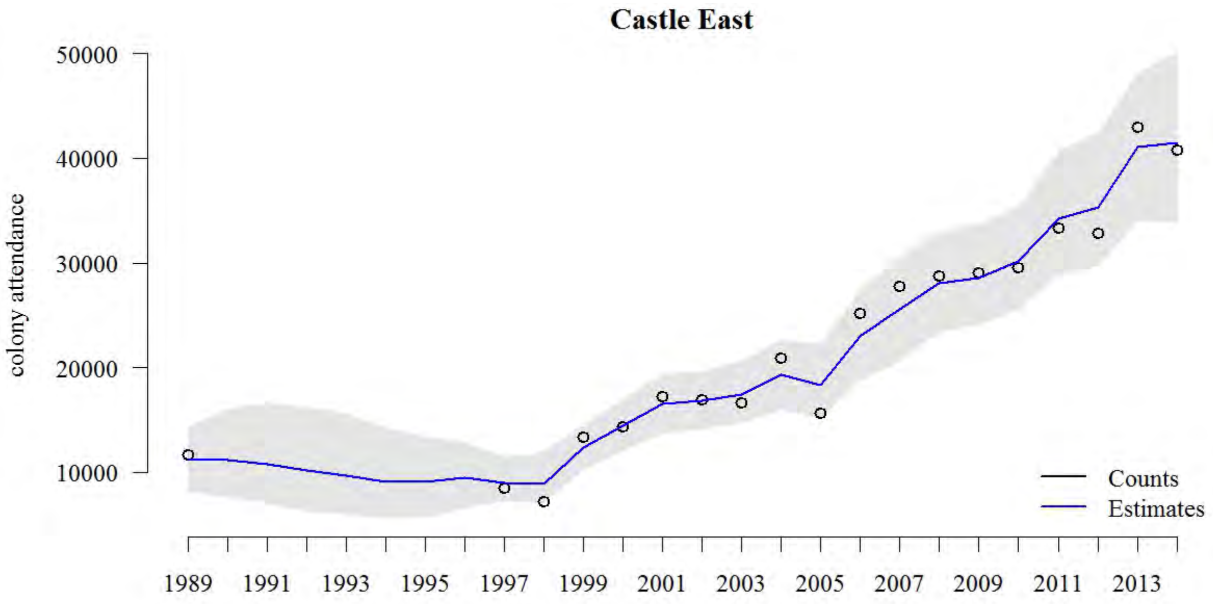

Figure 11. Castle Rock East Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

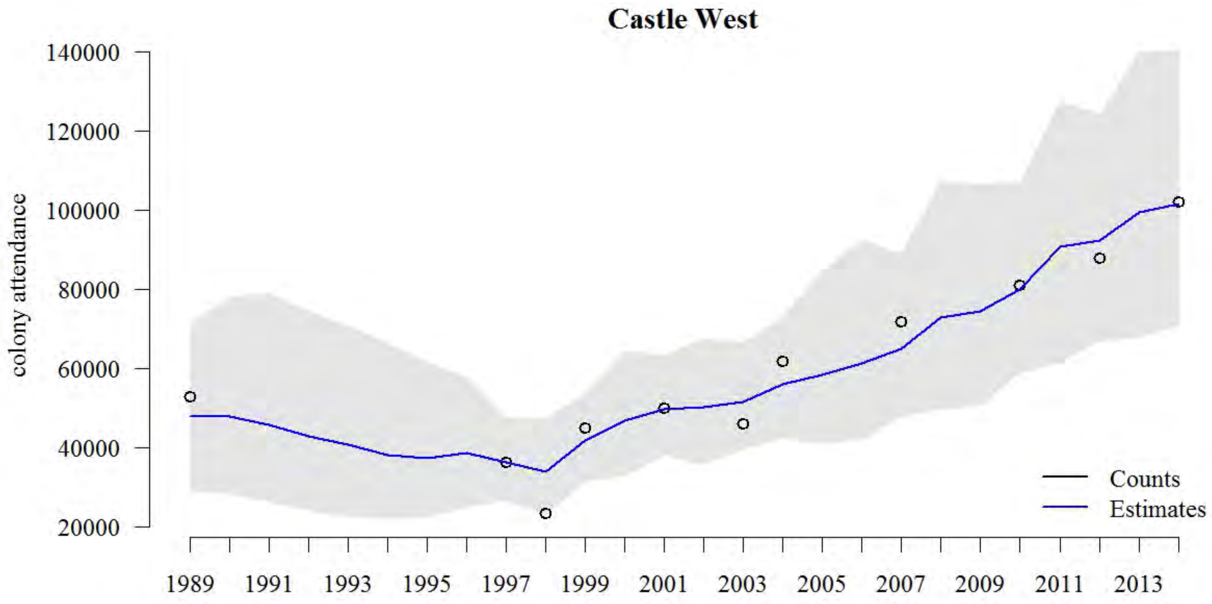

Figure 12. Castle Rock West Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

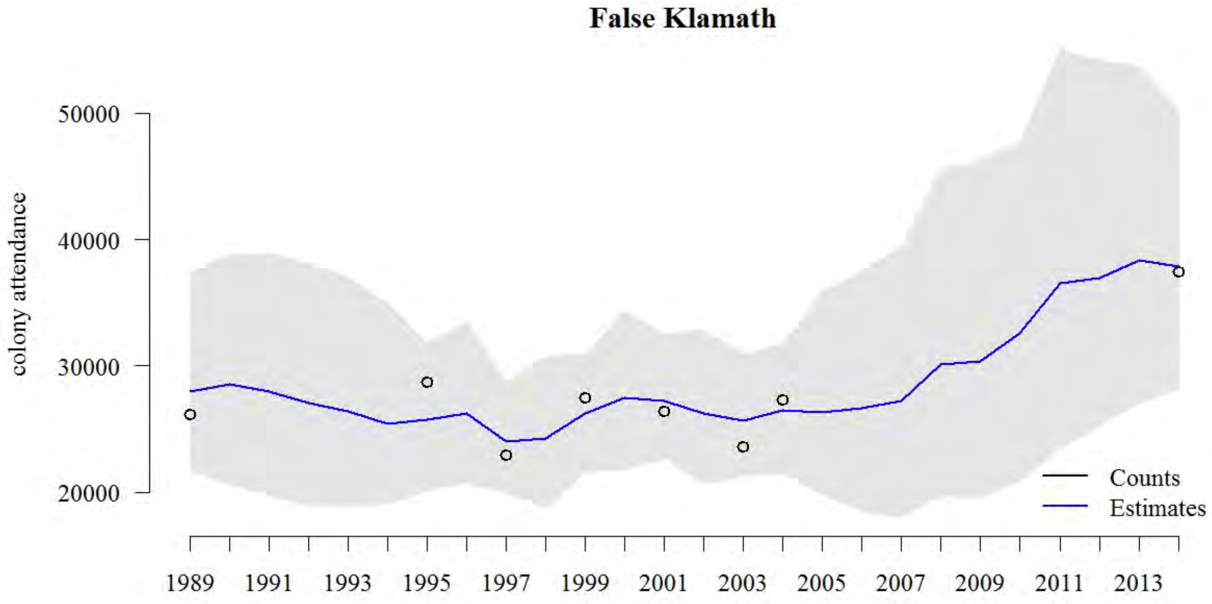

Figure 13. False Klamath Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

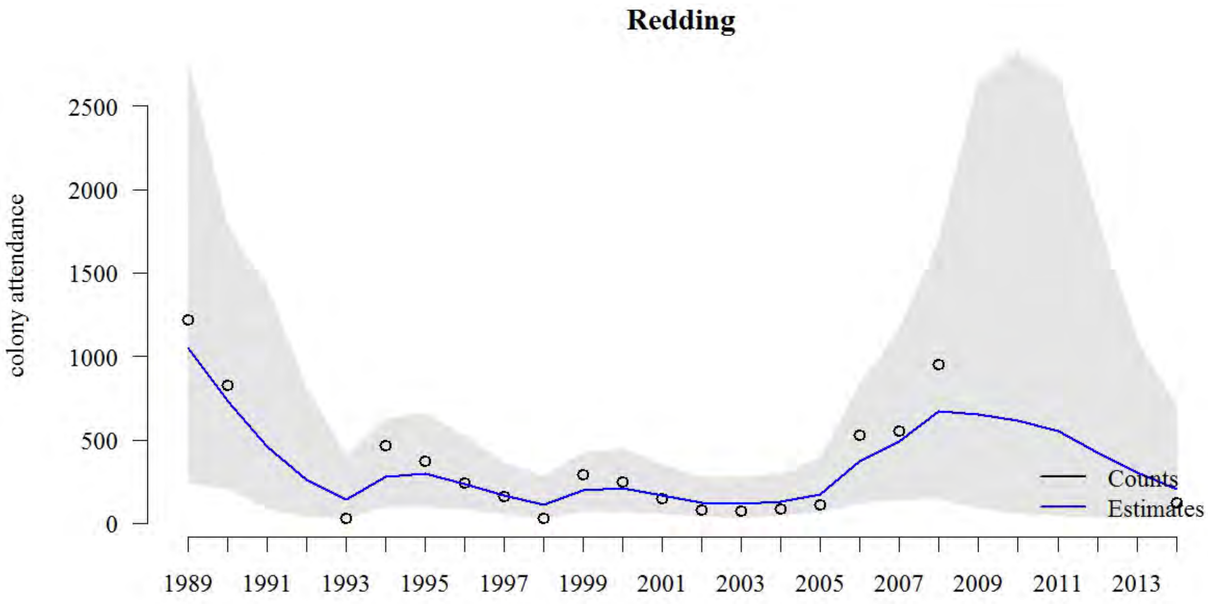

Figure 14. Redding Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

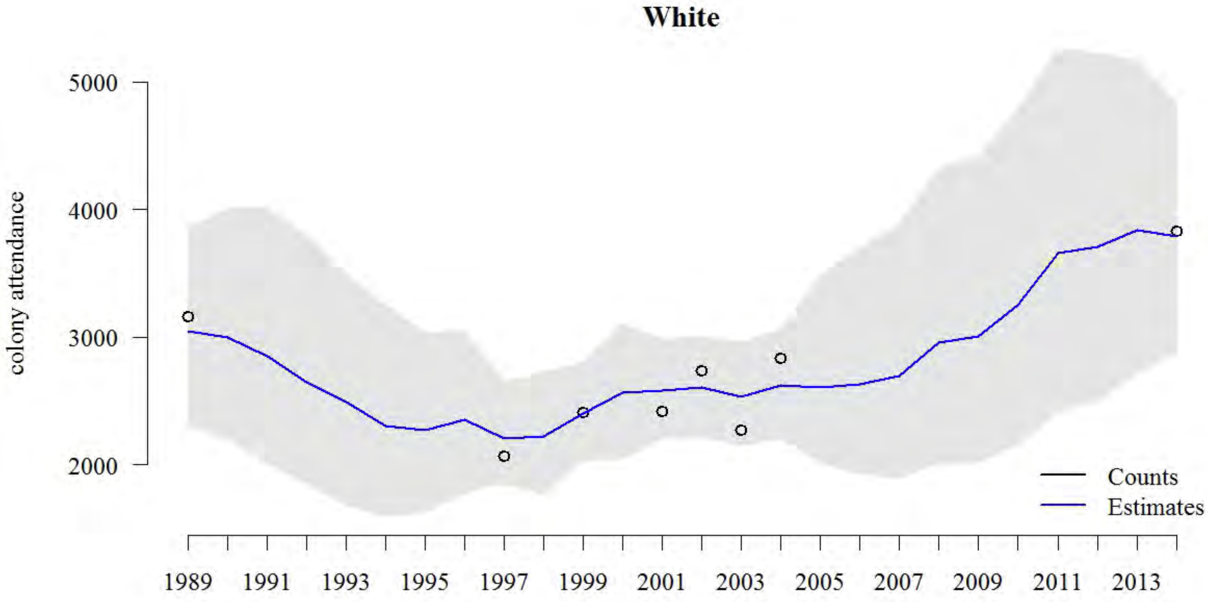

Figure 15. White Rock (Humboldt) Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

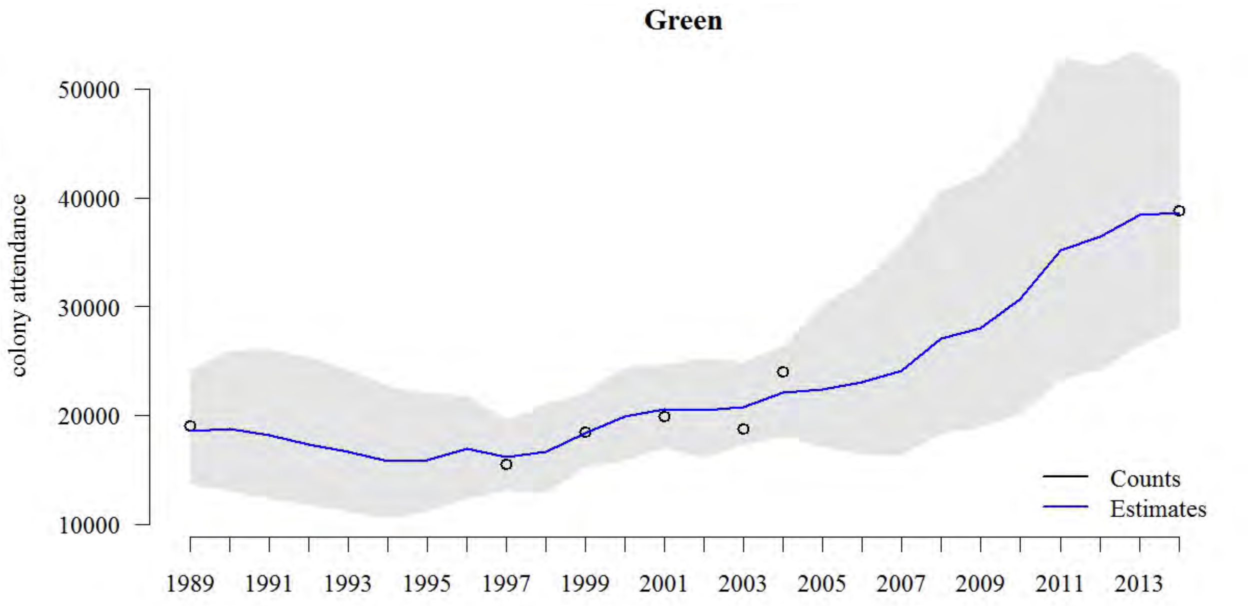

Figure 16. Green Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

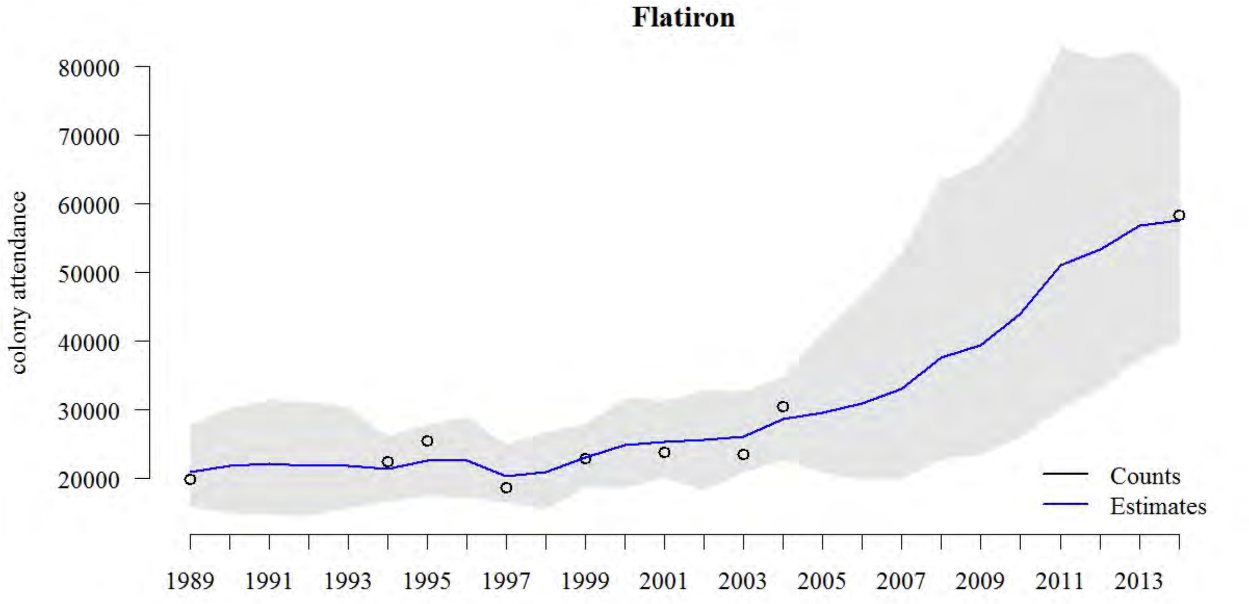

Figure 17. Flatiron Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

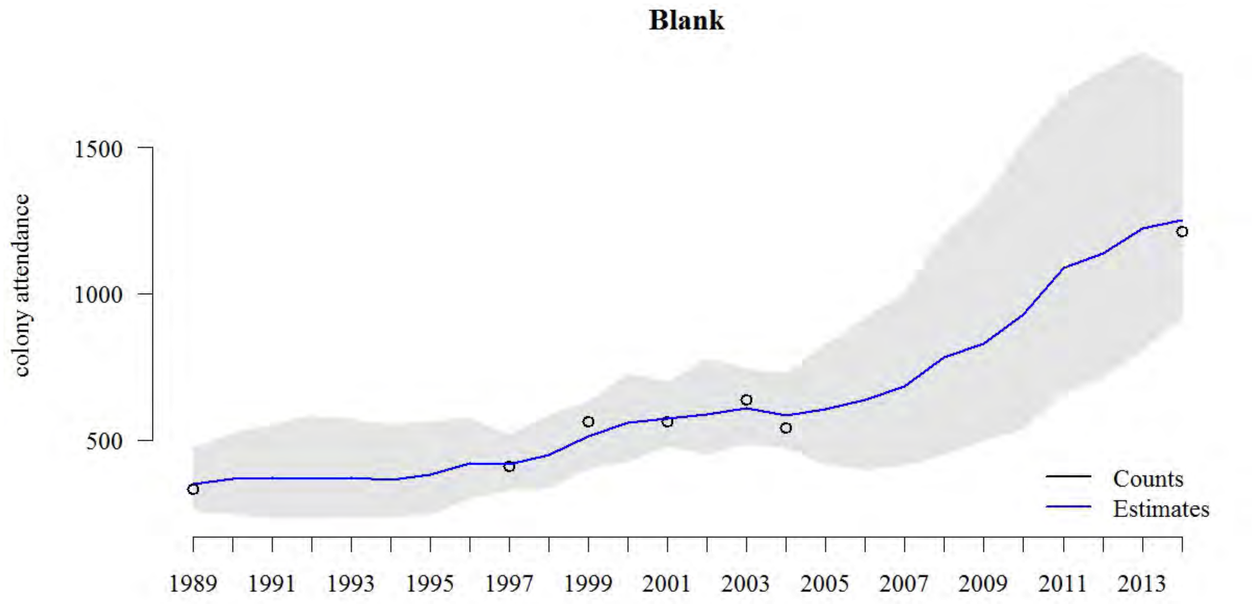

Figure 18. Blank Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

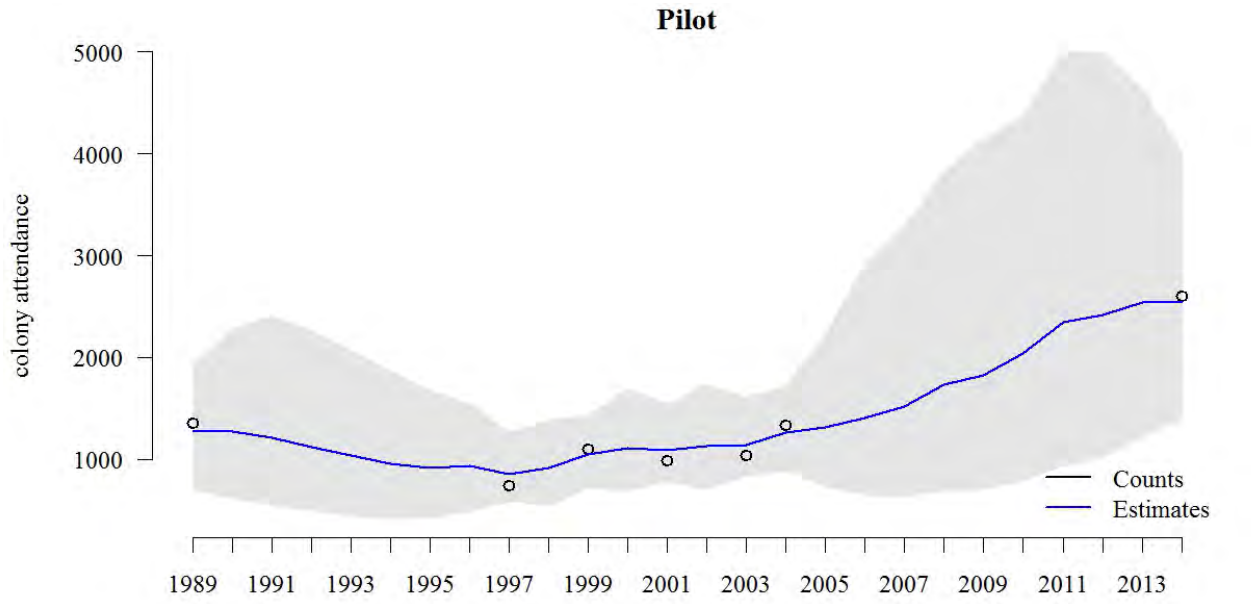

Figure 19. Pilot Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

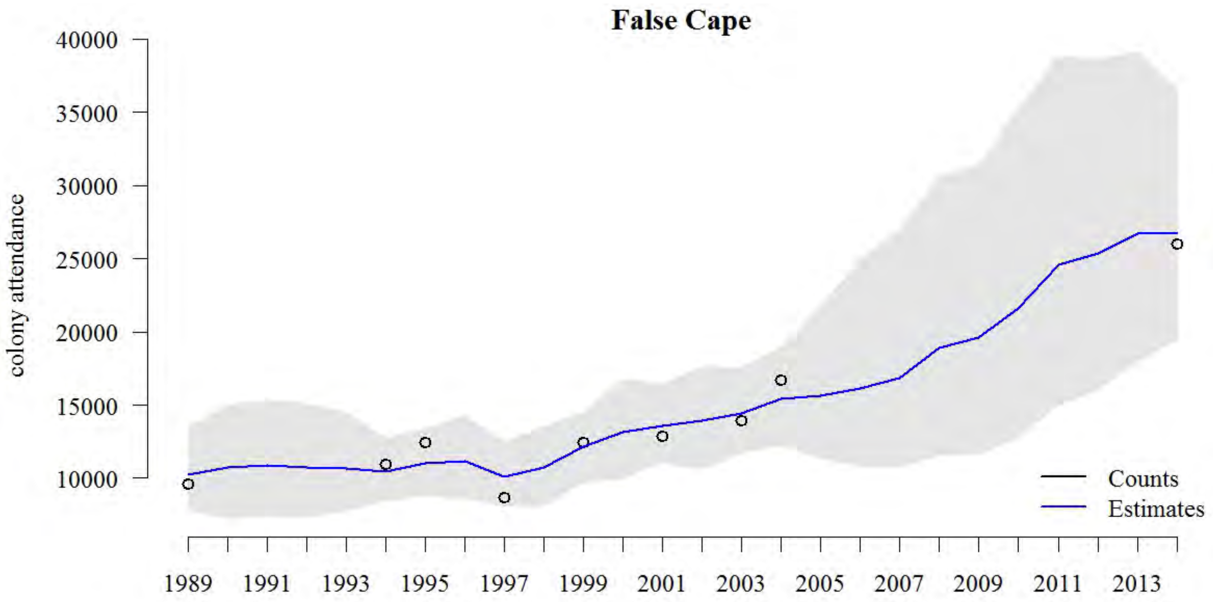

Figure 20. False Cape Rock Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

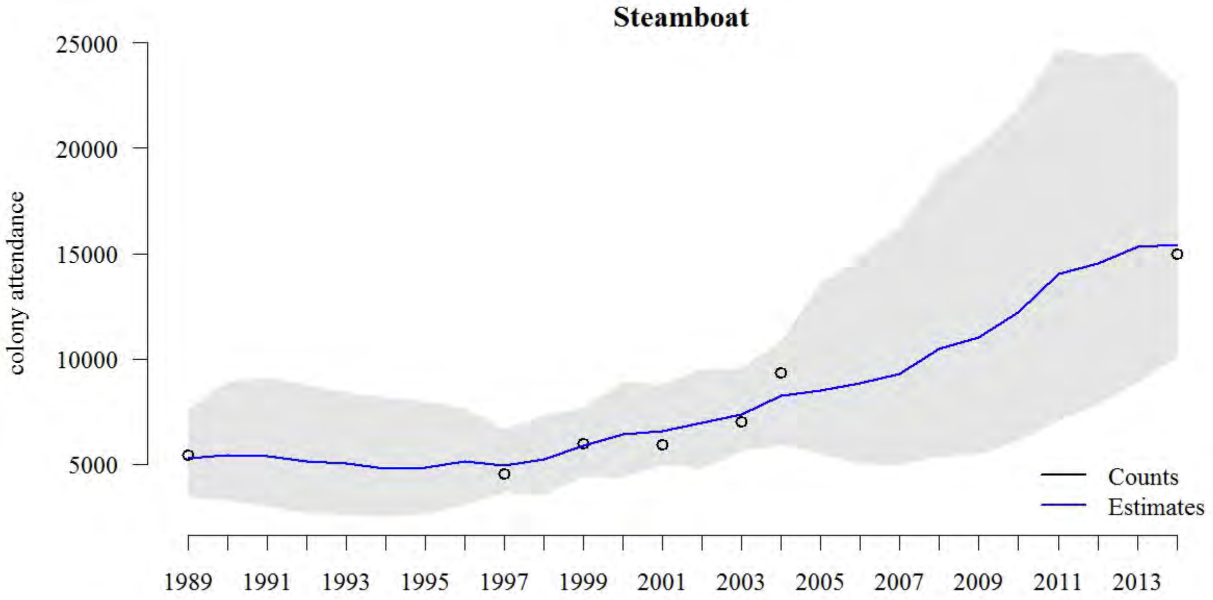

Figure 21. Steamboat Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

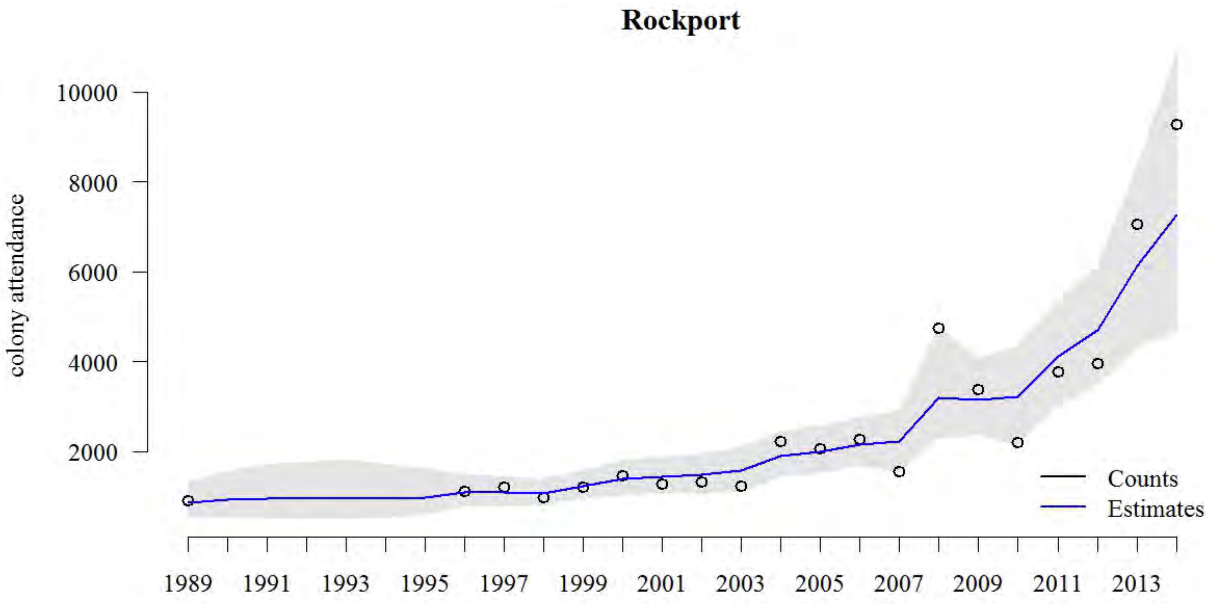

Figure 22. Rockport Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

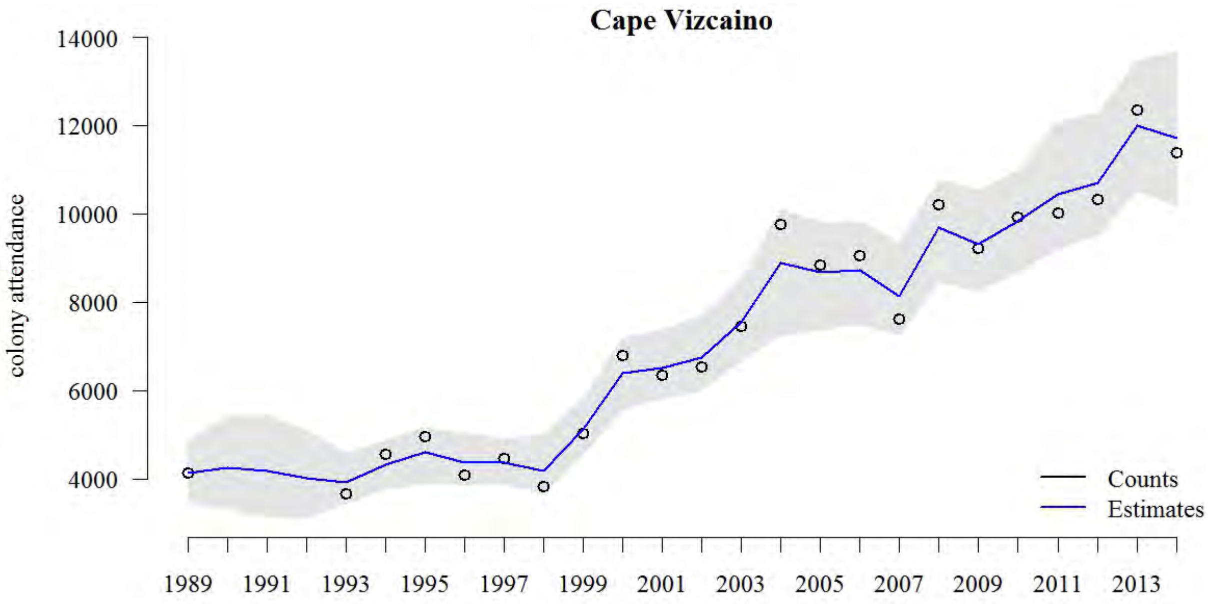

Figure 23. Cape Vizcaino Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

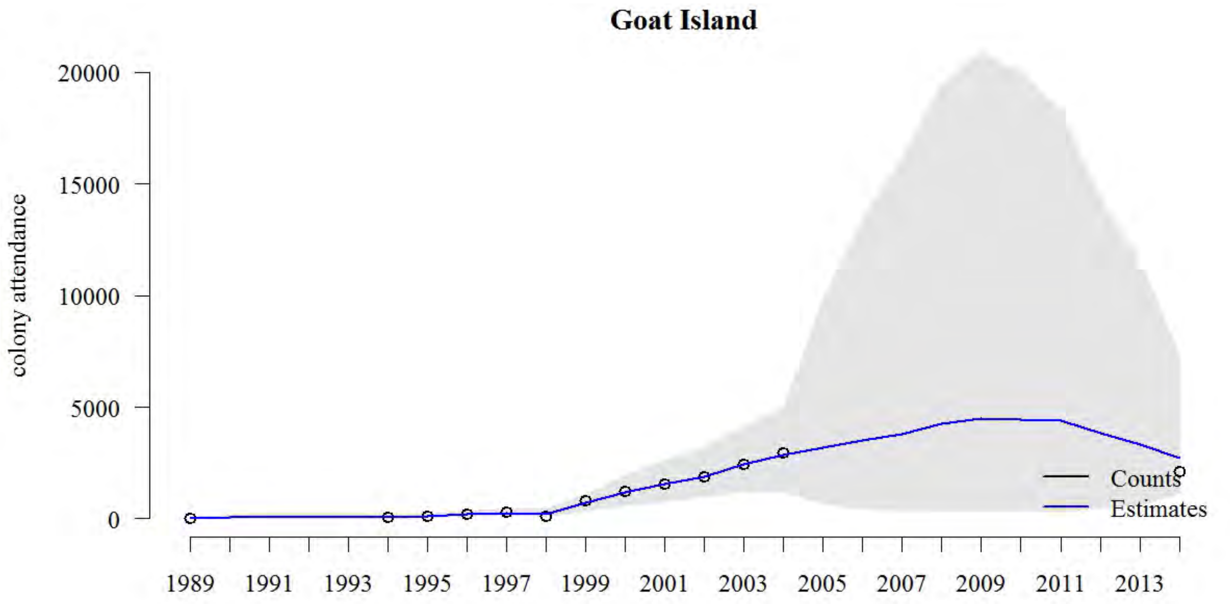

Figure 24. Goat Island Common Murre trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

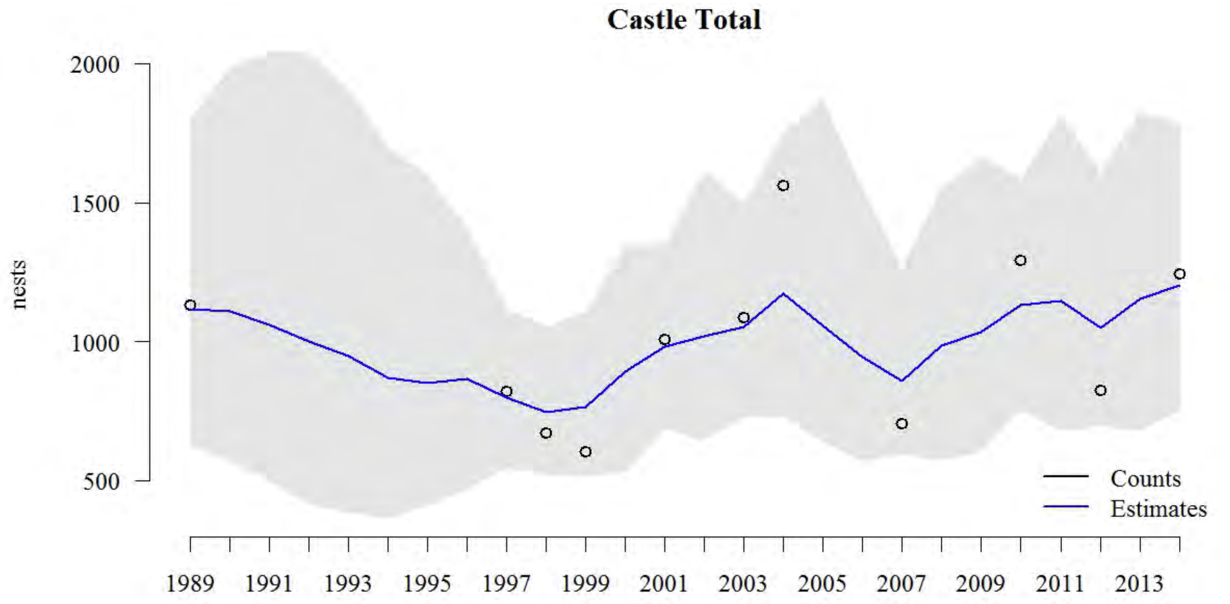

Figure 25. Castle Rock NWR Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

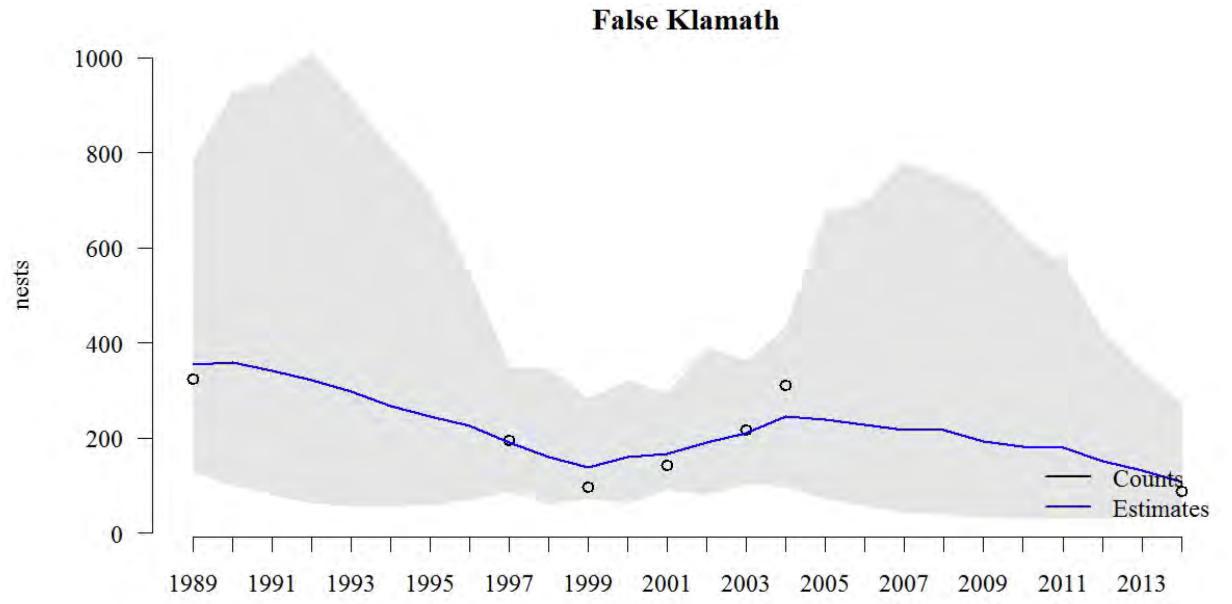

Figure 26. False Klamath Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

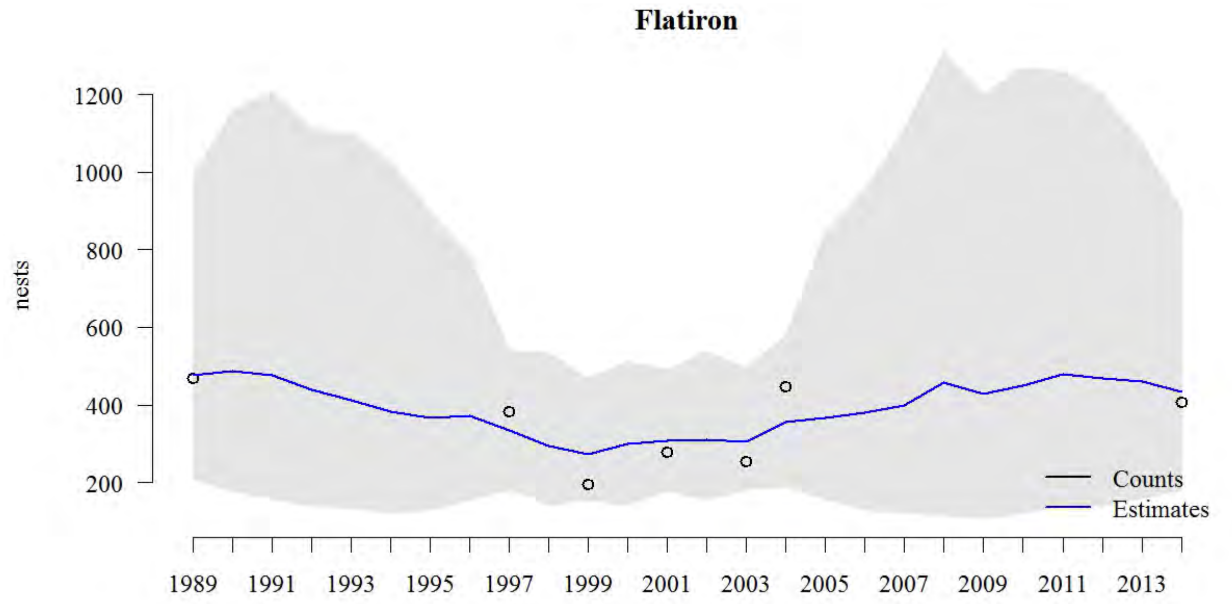

Figure 27. Flatiron Rock Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

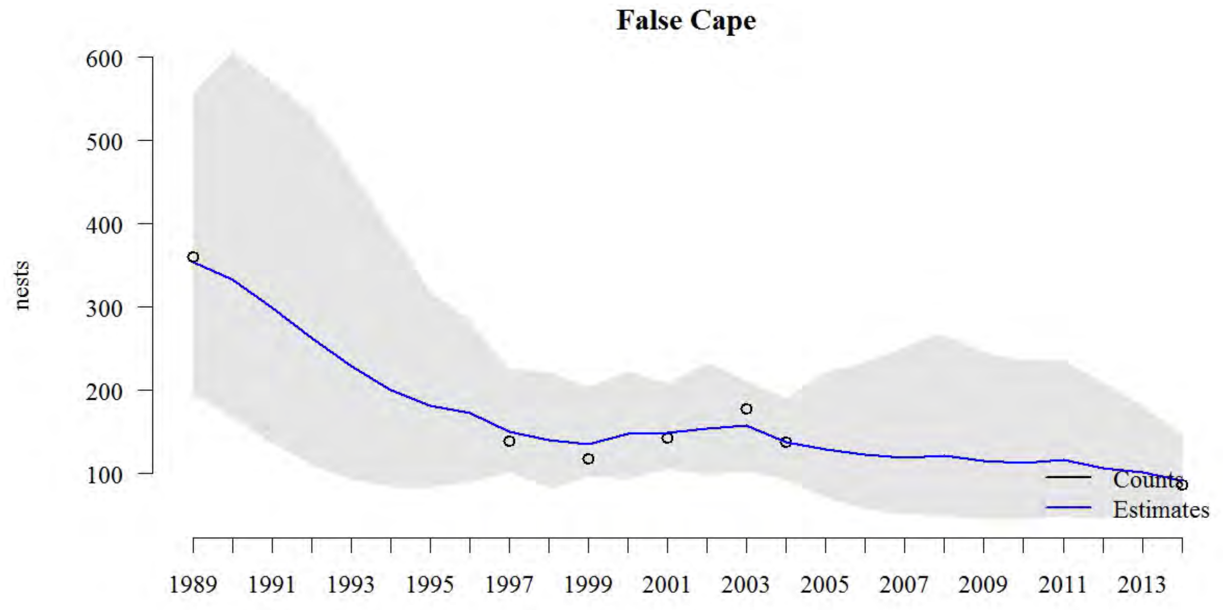

Figure 28. False Cape Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

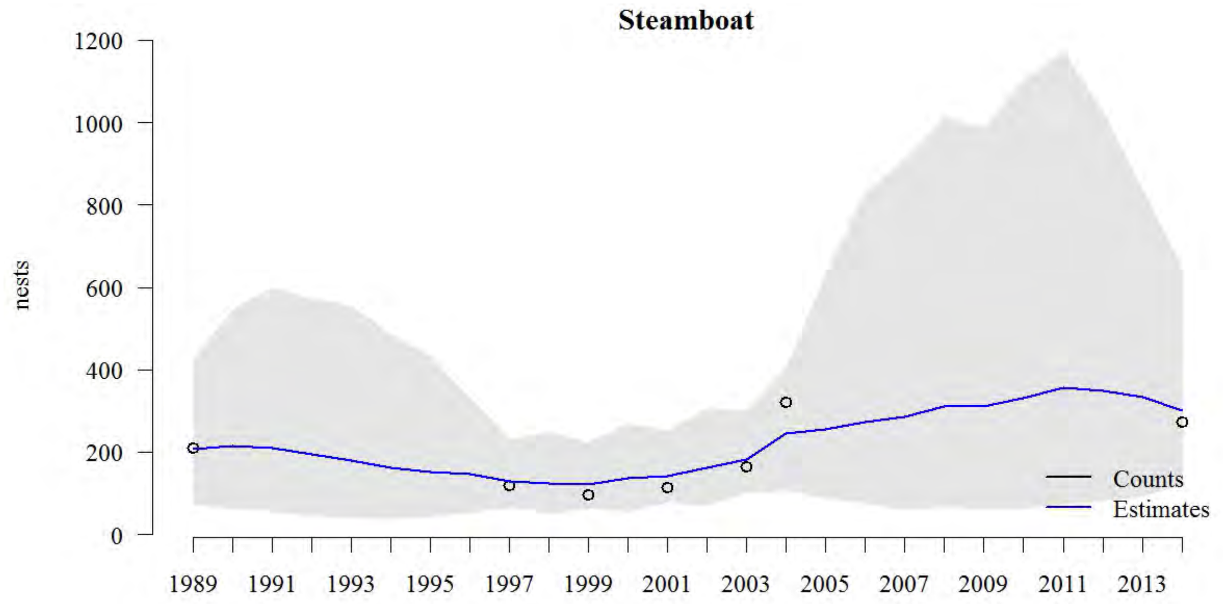

Figure 29. Steamboat Rock Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

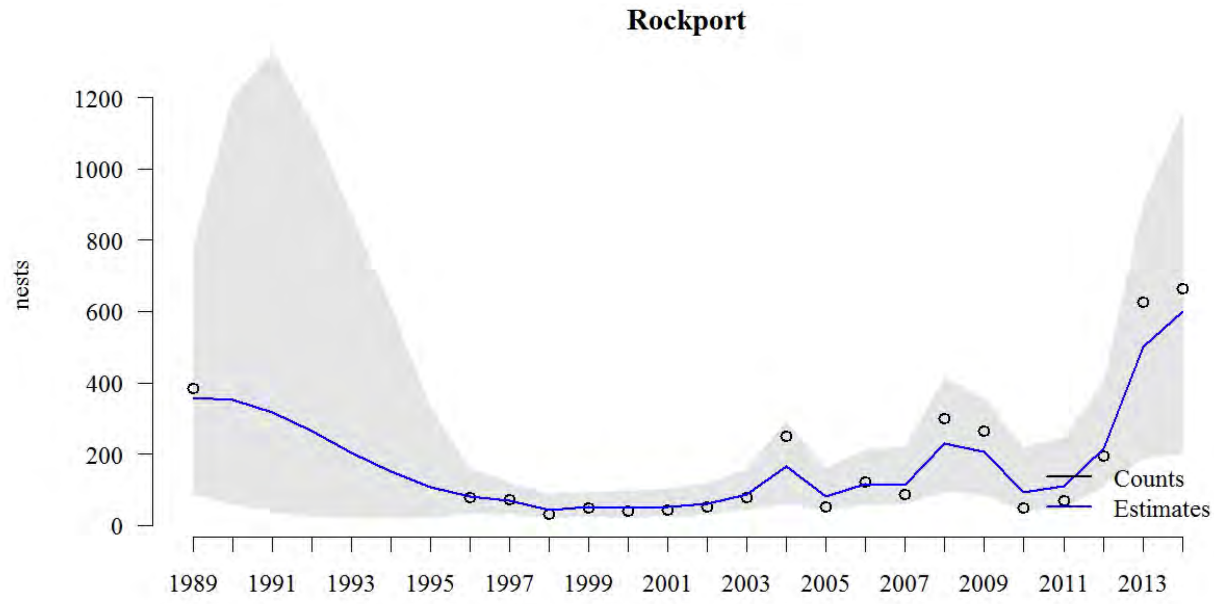

Figure 30. Rockport Rock Brandt’s Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

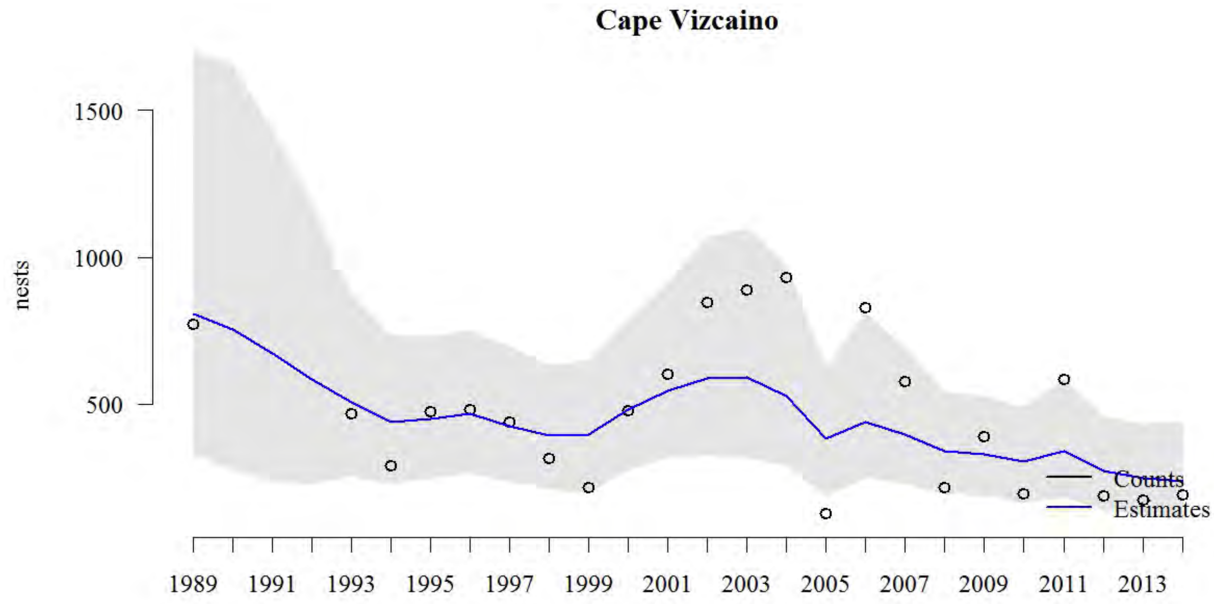

Figure 31. Cape Vizcaino Brandt’s Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

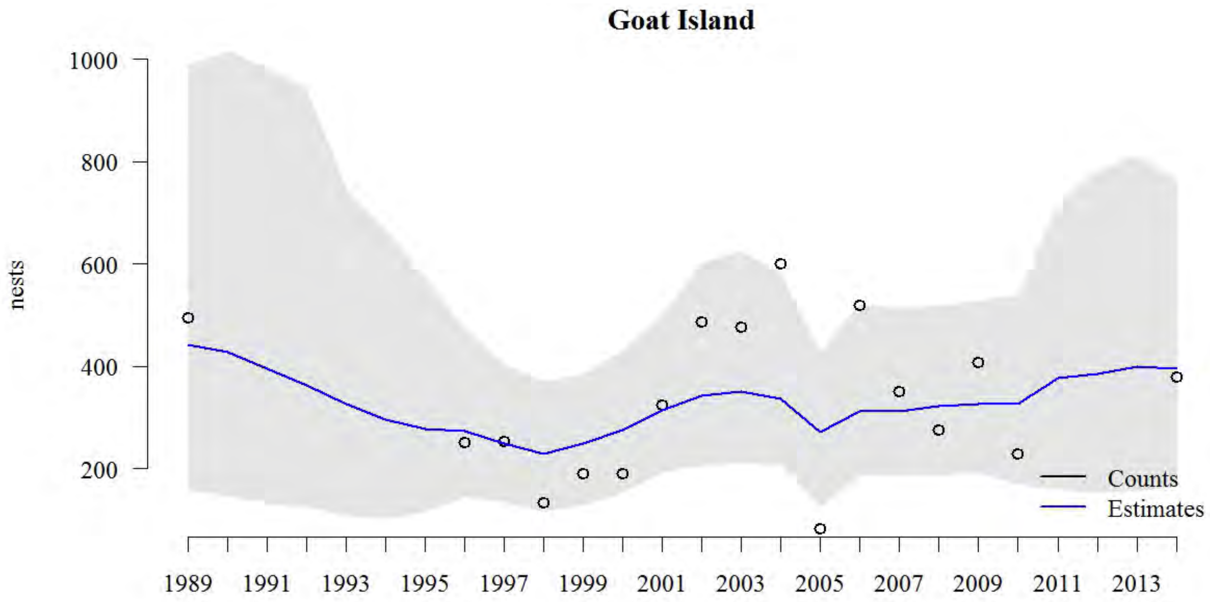

Figure 32. Goat Island Brandt's Cormorant trend estimate and observed counts, 1989-2014. Open black circles are aerial counts of attendance. Solid blue line and gray ribbon represent colony population size estimates and 95% credible interval from an autoregressive state-space model of population growth.

Table 6. Estimated annual population growth rates (mean of posterior probability distribution) by colony and year for monitored NCSR Common Murre colonies. Annual means and site means are reported and shown in bold.

year	**Castle East**	**Castle West**	**False Klamath**	**Redding**	**White**	**Green**	**Flatiron**	**Blank**	**Pilot**	**False Cape**	**Steam-boat**	**Rockport**	**Cape Vizcaino**	**Goat Island**	**mean**
1989	-0.004	-0.010	0.019	-0.309	-0.018	0.002	0.044	0.042	-0.027	0.040	0.007	0.053	0.021	0.143	**0.000**
1990	-0.045	-0.046	-0.020	-0.590	-0.057	-0.033	0.006	0.007	-0.062	0.011	-0.026	0.019	-0.015	0.094	**-0.054**
1991	-0.064	-0.069	-0.038	-0.591	-0.076	-0.054	-0.010	-0.015	-0.081	-0.016	-0.050	-0.003	-0.036	0.081	**-0.073**
1992	-0.044	-0.053	-0.025	-0.577	-0.062	-0.038	0.002	0.007	-0.073	0.001	-0.024	0.005	-0.022	0.110	**-0.057**
1993	-0.065	-0.073	-0.038	0.817	-0.072	-0.049	-0.013	-0.012	-0.088	-0.014	-0.037	-0.009	0.102	0.089	**0.038**
1994	0.003	-0.008	0.020	0.059	-0.011	0.012	0.049	0.048	-0.018	0.050	0.016	0.062	0.059	0.581	**0.066**
1995	0.057	0.053	0.020	-0.189	0.041	0.065	0.008	0.107	0.029	0.012	0.076	0.114	-0.055	0.617	**0.068**
1996	-0.054	-0.056	-0.087	-0.375	-0.064	-0.047	-0.106	0.000	-0.074	-0.093	-0.038	0.010	0.003	0.251	**-0.052**
1997	-0.013	-0.061	0.011	-0.518	0.009	0.025	0.026	0.064	0.068	0.054	0.055	-0.009	-0.046	-0.308	**-0.046**
1998	0.335	0.210	0.078	0.693	0.078	0.102	0.095	0.141	0.142	0.128	0.129	0.134	0.201	1.275	**0.267**
1999	0.156	0.113	0.044	0.061	0.064	0.083	0.074	0.080	0.048	0.083	0.082	0.130	0.225	0.519	**0.126**
2000	0.132	0.063	-0.006	-0.240	0.012	0.028	0.024	0.032	-0.004	0.032	0.030	0.030	0.018	0.281	**0.031**
2001	0.026	0.012	-0.034	-0.330	0.007	-0.003	0.005	0.021	0.016	0.021	0.049	0.029	0.036	0.206	**0.004**
2002	0.032	0.023	-0.021	-0.065	-0.029	0.009	0.022	0.036	0.031	0.037	0.061	0.064	0.114	0.242	**0.040**
2003	0.104	0.082	0.025	0.132	0.034	0.068	0.096	-0.033	0.106	0.062	0.112	0.179	0.163	0.142	**0.091**
2004	-0.057	0.035	-0.012	0.316	-0.011	0.005	0.017	0.027	0.023	0.007	0.014	0.057	-0.026	-0.059	**0.024**
2005	0.232	0.045	0.005	0.731	0.007	0.028	0.038	0.047	0.032	0.027	0.025	0.073	0.007	-0.039	**0.090**
2006	0.108	0.067	0.017	0.262	0.019	0.042	0.057	0.061	0.059	0.041	0.052	0.026	-0.072	-0.018	**0.051**
2007	0.094	0.107	0.098	0.276	0.098	0.117	0.127	0.137	0.126	0.116	0.124	0.348	0.178	0.028	**0.141**
2008	0.015	0.023	0.017	-0.246	0.015	0.036	0.047	0.052	0.046	0.033	0.041	0.001	-0.041	-0.047	**-0.001**
2009	0.055	0.085	0.075	-0.196	0.079	0.095	0.113	0.114	0.105	0.099	0.103	0.012	0.055	0.039	**0.059**
2010	0.128	0.124	0.115	-0.136	0.117	0.135	0.150	0.156	0.151	0.138	0.141	0.255	0.056	0.077	**0.115**
2011	0.029	0.021	0.011	-0.230	0.011	0.032	0.047	0.051	0.046	0.034	0.038	0.131	0.025	-0.027	**0.016**
2012	0.151	0.069	0.037	-0.202	0.039	0.050	0.071	0.074	0.074	0.057	0.062	0.257	0.116	-0.016	**0.060**
2013	0.009	0.022	-0.008	-0.274	-0.005	0.012	0.024	0.032	0.017	0.007	0.019	0.157	-0.027	-0.047	**-0.004**
mean	**0.053**	**0.031**	**0.012**	**-0.069**	**0.009**	**0.029**	**0.040**	**0.051**	**0.028**	**0.039**	**0.042**	**0.085**	**0.042**	**0.169**	**0.040**

Table 7. Common Murre state-space model variance parameters by site, including the mean, lower and upper 95% credible interval (CI) bounds, and coefficient of variation (CV) of the observation variance component σ_i^o and the process variance component σ_i^p. Greater values of the observation variance component σ_i^o indicate greater uncertainty in population size estimates, and greater values of the process variance component σ_i^p indicate greater variance in the annual population growth rate. The large CV of the process variance relative to observation variance suggest that population growth in Common Murre is highly variable relative to observation error.

		observation variance				**process variance**			
site	$\bar{r}$	σ_i^o	**lower CI**	**upper CI**	**CV**	σ_i^p	**lower CI**	**upper CI**	**CV**
Castle East	**0.053**	0.123	0.014	0.234	0.015	0.141	0.026	0.273	2.66
Castle West	**0.031**	0.223	0.058	0.431	0.027	0.136	0.006	0.384	4.38
False Klamath	**0.012**	0.130	0.012	0.343	0.016	0.091	0.003	0.279	7.58
Redding	**-0.069**	0.690	0.159	0.984	0.084	0.664	0.141	0.985	9.62
White	**0.009**	0.125	0.022	0.310	0.015	0.082	0.002	0.290	9.11
Green	**0.029**	0.132	0.015	0.384	0.016	0.081	0.003	0.283	2.79
Flatiron	**0.04**	0.145	0.010	0.362	0.018	0.118	0.006	0.314	2.95
Blank	**0.051**	0.136	0.010	0.404	0.017	0.100	0.005	0.315	1.96
Pilot	**0.028**	0.221	0.011	0.673	0.027	0.192	0.012	0.577	6.85
False Cape	**0.039**	0.150	0.011	0.355	0.019	0.108	0.002	0.317	2.76
Steamboat	**0.042**	0.182	0.016	0.512	0.023	0.138	0.008	0.452	3.28
Rockport	**0.085**	0.254	0.045	0.408	0.031	0.163	0.012	0.425	1.91
Cape Vizcaino	**0.042**	0.081	0.008	0.158	0.010	0.116	0.044	0.203	2.76
Goat Island	**0.169**	0.348	0.024	0.824	0.041	0.605	0.289	0.941	3.57

Table 8. Estimated annual population growth rates (mean of posterior probability distribution) by colony and year for North Coast Study Region Brandt's Cormorant colonies. Annual means and site means are reported and shown in bold.

year	**Castle**	**False Klamath**	**Flatiron**	**False Cape**	**Steam-boat**	**Rockport**	**Cape Vizcaino**	**Goat Island**	**mean**
1989	-0.021	-0.058	-0.019	-0.089	-0.043	-0.176	-0.076	-0.044	**-0.066**
1990	-0.053	-0.099	-0.055	-0.118	-0.073	-0.213	-0.109	-0.075	**-0.099**
1991	-0.067	-0.091	-0.065	-0.139	-0.073	-0.236	-0.118	-0.098	**-0.111**
1992	-0.062	-0.104	-0.070	-0.130	-0.077	-0.227	-0.120	-0.072	**-0.108**
1993	-0.075	-0.109	-0.068	-0.142	-0.068	-0.245	-0.133	-0.089	**-0.116**
1994	-0.012	-0.052	-0.026	-0.084	-0.031	-0.179	0.040	-0.033	**-0.047**
1995	0.037	0.003	0.024	-0.038	0.012	-0.125	0.032	0.010	**-0.006**
1996	-0.056	-0.082	-0.052	-0.127	-0.083	-0.171	-0.097	-0.085	**-0.094**
1997	-0.074	-0.191	-0.125	-0.077	-0.063	-0.483	-0.081	-0.091	**-0.148**
1998	0.015	-0.140	-0.073	-0.016	-0.001	0.190	0.004	0.089	**0.008**
1999	0.146	0.125	0.091	0.070	0.094	-0.099	0.200	0.108	**0.092**
2000	0.105	0.086	0.043	0.030	0.053	0.064	0.115	0.130	**0.078**
2001	0.028	0.104	-0.009	0.018	0.118	0.196	0.080	0.091	**0.078**
2002	0.045	0.108	0.009	0.033	0.134	0.342	-0.011	0.019	**0.085**
2003	0.091	0.124	0.135	-0.129	0.268	0.628	-0.112	-0.044	**0.120**
2004	-0.115	-0.124	-0.030	-0.082	-0.037	-0.714	-0.324	-0.231	**-0.207**
2005	-0.104	-0.119	-0.016	-0.065	-0.001	0.385	0.151	0.164	**0.049**
2006	-0.083	-0.096	0.006	-0.057	0.013	-0.039	-0.090	-0.011	**-0.045**
2007	0.119	-0.033	0.067	0.015	0.061	0.702	-0.158	0.039	**0.102**
2008	0.054	-0.109	-0.010	-0.055	-0.004	-0.104	-0.041	0.019	**-0.031**
2009	0.102	-0.048	0.059	-0.004	0.059	-0.877	-0.090	-0.018	**-0.102**
2010	0.006	-0.011	0.090	0.025	0.095	0.211	0.103	0.112	**0.079**
2011	-0.089	-0.077	-0.009	-0.059	0.004	0.712	-0.225	0.017	**0.034**
2012	0.089	-0.089	0.021	-0.041	0.016	0.816	-0.093	0.033	**0.094**
2013	0.048	-0.120	-0.017	-0.084	-0.026	0.167	-0.059	-0.002	**-0.012**
mean	**0.003**	**-0.048**	**-0.004**	**-0.054**	**0.014**	**0.021**	**-0.048**	**-0.002**	**-0.015**

Table 9. Brandt's Cormorant state-space model variance parameters by site, including the mean, lower and upper 95% credible interval (CI) bounds, and coefficient of variation (CV) of the observation variance component σ_i^o and the process variance component σ_i^p. Greater values of the observation variance component σ_i^o indicate greater uncertainty in population size estimates, and greater values of the process variance component σ_i^p indicate greater variance in the annual population growth rate. The large CV of the process variance relative to observation variance suggest that population growth in Brandt's Cormorant is highly variable relative to observation error.

	observation variance				**process variance**			
site	**σ**	**lower CI**	**upper CI**	**CV**	**σ**	**lower CI**	**upper CI**	**CV**
Castle	0.223	0.058	0.431	0.049	0.136	0.006	0.384	43.95
False Klamath	0.130	0.012	0.343	0.074	0.091	0.003	0.279	1.89
Flatiron	0.145	0.010	0.362	0.070	0.118	0.006	0.314	29.79
False Cape	0.150	0.011	0.355	0.039	0.108	0.002	0.317	2.01
Steamboat	0.182	0.016	0.512	0.069	0.138	0.008	0.452	9.98
Rockport	0.254	0.045	0.408	0.081	0.163	0.012	0.425	7.74
Cape Vizcaino	0.081	0.008	0.158	0.098	0.116	0.044	0.203	2.40
Goat Island	0.348	0.024	0.824	0.100	0.605	0.289	0.941	247.86

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CHAPTER 3: REPRODUCTION, FORAGING EFFORT, AND DIET OF COMMON MURRES NESTING AT CASTLE ROCK NATIONAL WILDLIFE REFUGE

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INTRODUCTION

Marine Protected Areas (MPAs) promote conservation of marine environments by protecting natural resources and restricting consumptive (e.g., fishing) and non-consumptive (e.g., recreation) uses of specific areas that contain those significant natural or cultural resources. Because MPAs limit disturbance to and resource extraction from marine communities, the establishment of a network of MPAs could alter species composition and abundance in protected communities and enhance the availability of primary and secondary consumers to apex predators, including seabirds (Nur et al. 2011, Croxall et al. 2012). This impact of MPAs may even extend beyond their boundaries. MPAs were implemented in the North Coast Study Region (NCSR) in 2012, and alterations in marine communities following protection may occur at various temporal (short- and long-term) and spatial (local, regional) scales (Agardy 2000). A baseline characterization of marine environments in and around these newly protected areas was needed to inventory current resources so that future changes in the marine environment can be detected and, when possible, attributed to MPAs. Importantly, baseline characterizations must provide strong measures of natural variability so that it is possible to distinguish between changes occurring due to natural variability from anthropogenic actions.

Seabirds have become widely used as indicators of marine conditions (Diamond & Devlin 2003, Piatt et al. 2007). Measures of seabird abundance can provide information on long-term changes in the regional marine ecosystem, but changes in population size typically lag behind changes in the marine ecosystem because many seabirds are long-lived, reproduce at a slow rate even in the best conditions (many lay only one egg per year), and can travel long distances (Schreiber & Burger 2001). Thus, changes in seabird abundance effectively indicate long-term (decadal) and large-scale (regional) changes in marine productivity. However, there are various circumstances that cause seabird populations to change at specific locations, which makes it difficult to directly link MPAs to enhanced productivity without understanding complimentary parameters that (1) accurately reflect current ocean conditions and (2) provide insight into mechanisms that alter seabird abundance.

Establishment of a baseline characterization that is representative of current conditions requires parameters that respond rapidly and predictably to changes in the marine environment. First, parameters that rapidly reflect marine conditions are necessary for the detection of short-term changes. Second, because these time-sensitive parameters are the first indicators of change, they are essential to determine if management actions are effective in a timely manner and to identify new management needs resulting from unforeseen environmental or anthropogenic perturbations. Third and finally, when parameters that reflect current conditions are monitored through time and space, these collectively facilitate understanding of marine conditions at broader temporal and spatial scales.

Using seabirds as accurate and rapid indicators of change requires monitoring aspects of their biology that immediately track alterations in marine productivity and prey communities. While nesting, seabirds can indicate rapid (weekly/annual) changes in relevant marine conditions at local and regional scales. During this time, large portions of the population congregate at near-shore rocks and islands where it is relatively easy to observe many individuals simultaneously. Because individuals must regularly return to incubate eggs and feed chicks, seabirds necessarily remain within flight distance of breeding colonies (Orians & Pearson 1979). For many seabirds nesting in the NCSR, adaptations enabling them to pursue prey underwater has reduced the efficiency of flight (Pennycuick 1987). Because reproduction is one of the most energetically demanding phases of the lifecycle (Schreiber & Burger 2001, Watanuki et al. 2009), each individual depends on an abundance of energy-rich prey to successfully fledge young (Cury et al. 2011). Thus, reproductive performance, chick diet, and behaviors related to foraging effort depend on and immediately reflect the prey abundance and composition in localized areas of the marine environment.

Measures of reproductive success are also essential to relate increases in seabird abundance to increased productivity in the marine system; the production of new individuals is a precursor for population growth and, if the annual production of new individuals can be quantified, inferences about the contribution of reproduction relative to other factors potentially altering population size (i.e., immigration, emigration, and adult survival) can be made. Seabird diet, including prey fed to young, is determined by the composition of the prey in waters within flight distance of a breeding colony. Finally, the abundance of prey influences the foraging behaviors of chick-rearing seabirds, including the time they spend away from the colony to obtain prey and the number of times they feed chicks each day.

Colony-based surveys are an effective tool to quantify reproductive success, chick diet, and behaviors indicative of foraging effort for nesting seabirds. However, colony-based surveys are labor intensive. There are various circumstances for certain seabirds that cause substantial challenges for determination of one or more of these parameters using non-invasive colony-based observations. Specifically, many seabird species nest below ground, are active only at night, or feed chicks by regurgitation rather than providing identifiable prey. Of the 11 seabird species nesting in the NCSR, Common Murres (*Uria aalge*) are the only species that nest on the surface, are diurnally active, and deliver whole prey to their chicks. Furthermore, Common Murre are the most abundant seabird nesting in the NCSR and are 88 times more abundant that other surface-nesting, diurnally-active seabirds nesting at the largest seabird breeding colony in the NCSR (Castle Rock National Wildlife Refuge). Thus, Common Murre (hereafter murre) were selected as the best candidate for intensive colony-based monitoring to establish a baseline characterization of this region. Like other seabirds, murre reproductive performance, chick diet, and behaviors related to foraging vary as a function of marine productivity and changes in their prey community (Burger & Piatt 1990, Monaghan et al. 1994, Uttley et al. 1994, Harding et al. 2007, Kadin et al. 2016).

Castle Rock is the most populous single-island breeding colony of murre south of Alaska. Castle Rock has been protected by a year-round Special Closure to minimize disturbance to breeding seabirds and marine mammals. A substantial proportion of the seabirds nesting in the NCSR are at Castle Rock (see Element 1, Chapter 2), making this a good location to establish a baseline characterization for this region by using murre reproduction, and behaviors indicative of foraging effort, and diet. Because all seabirds nesting at Castle Rock rely on waters within flight distance of this location to capture prey for themselves and their young, it is logical to conclude that murres would serve as an effective indicator of marine conditions experienced by many of the seabirds nesting in the NCSR. Furthermore, various MPAs are within flight distance of seabirds nesting at Castle Rock including the Pyramid Point SMCA, Point St. George Reef Offshore SMCA, Redding Rock SMCA, and Redding Rock SMR (Figure 33).

This study provides insights into the marine conditions in the NCSR, as experienced by murres and other seabirds nesting in this region, prior to and immediately following the establishment of MPAs across this region. We characterized the current state of the NCSR in terms of seabird productivity and prey availability, including a quantitative assessment of murre diet. We report colony-based surveys of murre reproduction, behaviors related to foraging effort, and diet by conducting surveys in 2014 (supported by the MLPA baseline characterization) and combining these surveys with the previous 7 years (2007-2013). Collectively, this long-term characterization provides a baseline of variance to assess future changes in marine condition and strengthen our ability to distinguish MPA-induced changes from natural environmental variability.

METHODS

Study Area

Castle Rock is a 0.068 km^2 island (Del Norte County Assessor's Office 1954, 1967) located in the in the northern portion of the NCSR approximately 0.8 km offshore from Crescent City, CA (41^o 43´37´´N, 124 ^{o}15´00´´W; Figure 33). To minimize disturbance to Castle Rock, public access to this island has been prohibited since its designation as a National Wildlife Refuge in 1979 (USFWS 2009). Seabird-monitoring efforts at this colony have not been possible from the mainland due to distance and fog. Additionally, boat-based observations of nesting seabirds in northern California are challenging due to typically rough seas that can compromise safety and create logistical difficulties. To facilitate seabird monitoring of reproductive parameters and diet, a video-based monitoring system (SeeMore Wildlife Systems Inc., Homer, Alaska) was installed at the top of a rocky slope on the north side of the island. The U.S. Fish & Wildlife Service has allowed very limited access outside of the breeding season to facilitate maintenance and repair of this system. The location was randomly selected and does not contain unique habitat features that would enhance or diminish reproductive efforts of seabirds. As such, observations of seabirds within the vicinity of the monitoring system have provided a representative sample of the entire colony. All research at Castle Rock associated with this video-based monitoring system has been approved by Humboldt State University's IACUC (Protocol #'s: 05/06.W.70.A, 08/09.W.54.A, 11/12.W.88-E, 15/16.W.01-E).

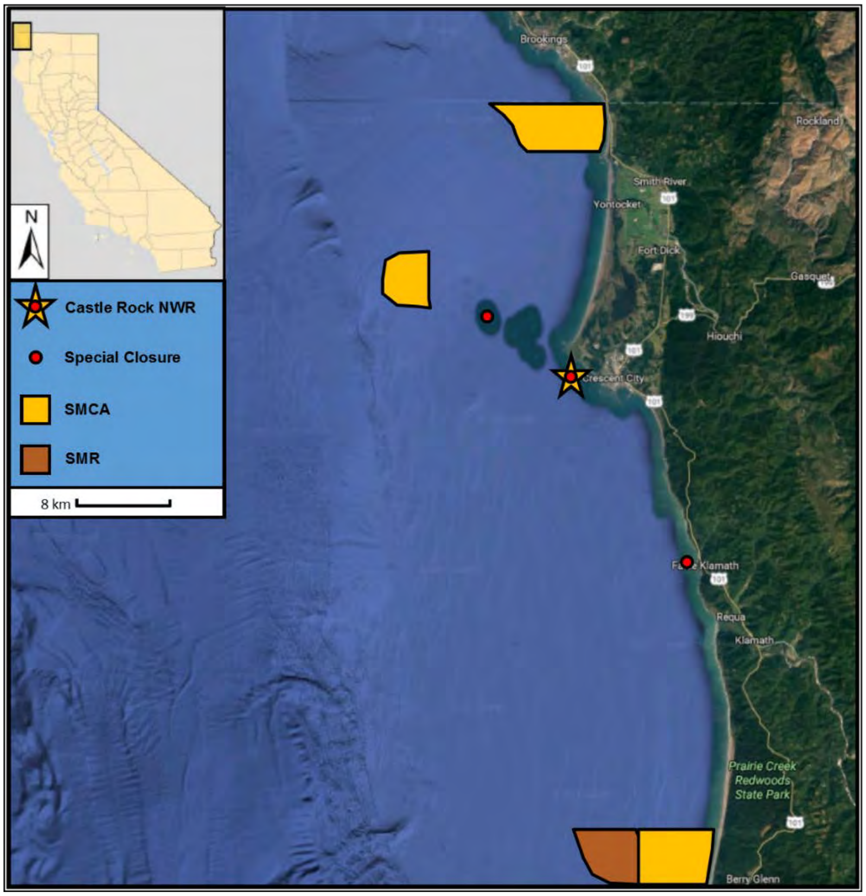


Figure 33. Map of California [inset] with a detailed view of the coastline and continental shelf surrounding Castle Rock National Wildlife Refuge (NWR), Del Norte County, CA. The star indicates the approximate location of Castle Rock. State Marine Protected Areas near Castle Rock including Special Closures (n=3, including Castle Rock), State Marine Conservation Areas (SMCA, n=3), and State Marine Reserves (SMRs, n=1) within foraging range of seabirds nesting at Castle Rock. See Figure 1 for a full depiction of seabird colonies and MPAs within the NCSR.

Colony-based Observations

To observe Common Murre and other diurnally active seabirds we used two weatherproof, remotely controlled video cameras capable of real-time panoramic scanning (360°), tilting (120°), zooming, and auto-focusing. This system was initially installed on the island in 2006. Video was transmitted via microwave from the island to a receiving station on the mainland where the signal was recorded as high-resolution (29 frames per second at a resolution of 720x480 pixels) digital video onto hard drives. Using this system, the island could be viewed by both researchers and the public in real-time (via the internet) or at a subsequent point in time (via these hard drives). Video recordings were essential for repeated viewing and frame-by-frame inspection.

Surveys were conducted to assess: reproductive performance (nest surveys), behaviors of chick-rearing murres indicative of foraging effort (time allocation surveys), and prey diversity and composition (diet surveys) during each breeding season between 2007 and 2014 (see also Golightly and Schneider 2016). To ensure validity of interannual and cross-colony comparisons, all surveys followed specific protocols that approximated methods used at other breeding colonies in the CCS (Boekelheide et al. 1990, Suryan et al. 2014, Fuller et al. 2015).

Nest surveys

Nest surveys were used to assess the overall success of nests, which was a function of the number of nests initiated, eggs hatched, and young fledged. Nests included in these surveys were near the monitoring system because it was necessary to make detailed observations of breeding pairs, eggs, and chicks. In 2007 and 2008, all nests within a $25m^2$ area were observed and, in 2009, this area was doubled to include more nests; typically, these surveys included between 60 and 100 nests. To ensure accurate identification of each nest, still-images of the survey area were generated and each site was labeled with a unique number (Figure 34). These surveys required an observer to remotely move the cameras to view each nest and observe if an egg or chick was present. Observations began prior to nest initiation and continued every other day until all nesting attempts were completed.

Figure 34. Photo illustrating how breeding pairs were numbered so they could be reliably identified for the duration of the breeding season.

Murres only lay one egg and the overall success of nests is determined by the number of nests initiated that successfully fledged a chick. We calculated the average nest success and 95% confidence intervals for each year using the Mayfield Method (a maximum-likelihood estimator of daily survival rate contingent upon accounts for the number of days nests were exposed) to determine nest survival probabilities (Mayfield 1961, 1975). The daily survival probability can be extrapolated across the duration of nesting to determine overall nest success. There are two components that contribute to overall nesting success: hatching success, a measure of the proportion of eggs laid that hatched, and fledging success, the proportion of chicks fledged relative to the number of hatched eggs. Distinction between these two periods is important because of differences in energy demands and sources of mortality. Chicks were considered fledged if they were documented alive at the colony for a minimum of 15 days and were not known to perish afterwards. This 15-day period was based on the duration of chick-rearing reported from various murre colonies in the CCS and elsewhere (Boekelheide et al. 1990) and has been commonly used elsewhere in the CCS to quantify murre nest success (Eigner 2009, Eigner et al. 2012, Fuller et al. 2015). Both hatching and fledging success were estimated using the Mayfield Method. Calculations of nesting, hatching, and fledging success excluded replacement nests which, for Common Murres, are uncommon and typically fail (Golightly & Schneider 2016).

To describe the timing of nesting, dates for nest-initiation (egg-lay) were determined for each nest. The average date of nest initiation was calculated for each year for nest-sites with egg lay dates accurate to ±3 days. Egg lay dates for replacement clutches were not used in calculations of nest initiation dates. An ANOVA was used to determine if there was interannual variation in the timing of nest initiation and, if so, homogenous subsets were identified using Tukey's HSD (Tukey 1949).

Time allocation surveys

Directly measuring the availability of prey for seabirds is challenging and the assumptions needed to define availability may lead to inaccurate assessments (Grémillet et al. 2004). For Common Murre, prey availability (or more accurately, energy availability) has been inferred by quantifying colony-attendance patterns of chick-rearing pairs (Smout et al. 2013). In general, when food is abundant chick rearing pairs can spend up to half of their day at the colony together (known as co-attendance; Zador & Piatt 1999). As food becomes difficult to obtain, the duration of co-attendance is reduced so that chicks continue to receive enough energy to survive (Burger & Piatt 1990, Zador & Piatt 1999, Harding et al. 2007, Smout et al. 2013). In extreme cases, both birds could leave a chick unattended to acquire food.

Time allocation surveys were used to make inferences about prey abundance using a suite of metrics that are related to foraging effort, specifically the duration chick-rearing pairs spent together at the colony and chick provisioning rates. Time allocation surveys required an observer to position the camera to record 12-24 chick-rearing pairs simultaneously. Once positioned, the camera remained stationary for an entire day (from dawn until dusk). This method resulted in all focal sites being clustered in the same area of the colony, however it is unlikely this biased observations because these parameters are influenced by prey availability rather than site quality (Burger & Piatt 1990, Zador & Piatt 1999, Harding et al. 2007, Smout et al. 2013). Time allocation surveys began when 66% of the chicks had hatched and continued until all chicks at focal sites left the colony (approximately 3-4 weeks). Each year, these time allocation surveys occurred six to eight days apart since they required a full day where no other surveys occurred.

Video recordings of each time allocation survey were subsequently reviewed to quantify daily time allocation and provisioning rates for each site using methods described by Parker (2005) and Eigner (2009). Data for time allocation surveys were collected from the video recordings, rather than in real-time, because recordings could be paused or re-watched to ensure that all arrivals, departures, and chick-provisioning events at each nest-site were observed. During video review, the exact time of all arrivals, departures, and chick-provisioning events at each nest-site was noted. A visual scan of each chick-rearing pair occurred every 15 minutes, to confirm that arrivals and departures were not missed. If an arrival or departure was missed, the observer re-reviewed the previous 15 minutes of video to determine the exact time of the relevant change. When an individual briefly left its site (e.g., to kleptoparasitize a neighboring murre) but remained at the island, this event was not classified as a departure event. Additionally, when chicks were left unattended at the colony, an individual from a neighboring site would sometimes brood the unattended chick; although the chick was under the supervision of an adult, for purposes of quantifying time allocations of chick-rearing pairs, we still classified these chicks as unattended until at one of their parents returned.

Once time allocation surveys were concluded, the average amount of time where both, one, and neither member of chick-rearing pairs were present at the colony was quantified. The proportion of time that chick-rearing pairs spent in co-attendance was calculated by dividing the duration co-attendance by the total observation time for each pair and then calculating an average across nests. The same calculation was used to quantify the proportion of time that chick-rearing pairs left their chick unattended at the colony. Annual variation in colony attendance patterns of chick-rearing pairs was assessed using ANOVAs and homogenous subsets were identified using Tukey's HSD procedure (Tukey 1949).

Chick-provisioning events occurred whenever a parent attempted to provide a chick with food and were considered successful only when the chick consumed at least part of the prey provided by one of its parents. Typically, chicks were provisioned with prey captured at sea. Less commonly, adults would provision their chick by stealing prey from conspecifics without leaving the colony. Although murres that stole prey to provision chicks expended less energy relative to murres that obtained prey at-sea, both methods of provisioning represented energy gained by the chick. Therefore, both methods of prey acquisition were used to quantify annual chick-provisioning rates. The rate of provisioning was calculated as the number of prey fed to chicks per hour. Annual variation in chick-provisioning rates was assessed using an ANOVA and homogenous subsets were identified using Tukey's HSD procedure (Tukey 1949).

Diet surveys

Diet surveys were used to quantify the composition of prey brought to the colony by murres during chick-rearing by obtaining high-resolution recordings of prey (Figure 35). To ensure that details sufficient for prey identification were recorded, these surveys focused on alilmurres within ~100 m of the camera and required an observer to actively scan through the colony in real-time and locate adult murres possessing prey. Once located, the observer re-focused the camera onto prey to maximize the recording of potential morphological characteristics needed for identification. Diet surveys began when 10% of eggs hatched and continued until 90% of chicks had fledged, as determined by nest surveys (described above).

In 2007 through 2009, the schedule of diet surveys followed a specific protocol developed by the Common Murre Restoration Project in central California (Eigner 2009, Fuller et al. 2015). This called for two types of surveys: entire-day surveys (06:00-20:00 PDT) that occurred three times during the chick-rearing period at weekly intervals, and two-hour surveys that occurred daily (except one day per week when time allocation surveys occurred). Daylight hours were broken into seven 2-hour intervals (beginning at 06:00 and ending at 20:00) and surveyed on a rotating schedule such that each interval was sampled approximately every seven days. In 2010, this protocol was modified to maximize the total number of prey identified and, since prey deliveries were most frequent between 06:00 and 08:00, diet surveys in 2010 through 2014 occurred six days per week during this time interval.

Video of each diet survey was subsequently reviewed and identified to the most specific level possible using characters such as fin placement, distance between fins, tail shape and body shape based on descriptions provided in Eschmeyer and Herald (1983), Papish (1996), Appendix B of Eigner (2009), and Orben (2009). Video facilitated accurate identification of prey as they were generally visible from various angles, video could be reviewed frame by frame, and experts could be consulted to confirm ID of each prey type. During the identification process, coordinates (x,y) indicating the position of each prey were noted (these coordinates are exact and built into the monitoring system). If more than one prey is observed at the same location during a survey, these prey were directly compared to guarantee that each prey observation was unique.

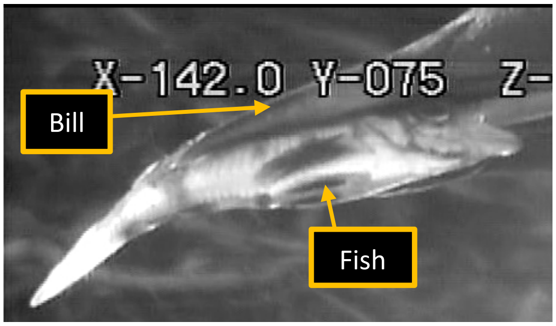


Figure 35. Photo of a murre holding a juvenile rockfish in its bill to illustrate the level of detail recorded by the cameras during diet surveys so that prey could be accurately identified.

The composition of prey observed across all years (2007-2014) was summarized. After establishing the overall diversity of prey delivered to the colony by Common Murre, interannual changes in prey composition were examined. A chi-squared test was used to determine if the prey composition observed during individual years matched the composition expected if prey communities were identical for all years. To meet the assumptions of the chi-squared test, prey types that were observed fewer than five times were excluded rather than lumped into a category called “other” as this would cause prey types rarely fed to chicks to be overrepresented in the chi-squared analyses. Unidentified prey were also excluded from this analysis since the ability to identify prey delivered to the colony was a function of video quality rather than prey identity.

RESULTS AND DISCUSSION

Surveys quantifying Common Murre reproductive performance, behaviors related to foraging effort, and chick diet at Castle Rock were successfully executed in 2014 and these data were added to a 7-year (2007-2013) time series. In 2012, the video monitoring system failed early in the nesting season so only the initiation of nesting by some murres was observed. As such, only the date of the first nest initiated could be determined in this one year.

This long-term analysis will aid in the development of a baseline characterization of marine conditions in the NCSR. By understanding current variability in the reproduction, foraging effort, and diet of seabirds nesting in the NCSR, this study will allow for: (1) assessment of the efficacy of each parameter in serving as an indicator for future changes in marine conditions within the NCSR and (2) strengthen our ability to distinguish MPA-induced changes from natural environmental variability.

Reproductive Performance

Between 2007 and 2014, 619 breeding pairs were monitored (155±12 pairs per year) to quantify the reproductive performance of murres, including the timing of nest initiation and the success of these nests. Nest initiation varied between years (ANOVA: $F_{6,596}$=209.3, P<0.001, Figure 36) with nest initiation occurring relatively early in 2008 and 2009 and relatively late in 2010 and 2011, with nesting in other years initiating at intermediate dates. Notably, the first nest in 2012 was initiated on 15 May, which was the latest date for the earliest nest initiation observed in any year of the study; although the average date of nest initiation in this year could not be quantified due to early failure of the camera system, the average date of nest initiation would have been later in 2012 relative to all other years.

The initiation of nesting by many seabirds, including murres, is thought to be determined by seasonal increases in productivity of the marine system (Reed et al. 2006, Grémillet & Boulinier 2009, Watanuki et al. 2009). In the CCS, seasonal increases in productivity are caused by changes in prevailing wind patterns that transition from downwelling-favorable to upwelling-favorable, which causes deep nutrient-rich water to reach the surface. The timing of the transition to upwelling-favorable conditions in the CCS (commonly referred to as the “Spring Transition”, date determined using the CBR Mean Method; Columbia Basin Research 2016) is variable but generally occurs in late winter or early spring; at Castle Rock between 2007 and 2014, the Spring Transition was earliest in 2008, occurring on 5 February, and latest in 2012, occurring on 24 April (range: 78 days; Figure 37a). The initiation of nesting by murres at Castle Rock over these 8 years was less variable than the timing of the Spring Transition, with the date of the earliest nest varying by 23 days. However, there was evidence that the initiation of nesting was positively related to the timing of the Spring Transition (based on the earliest nest: r^2=0.428, P=0.079; Figure 37b). Based on other studies, the timing of nest initiation by murres is a function of female condition because egg production is energetically costly and only females with adequate energy stores can do so. There is a lag between the onset of upwelling, the conversion of nutrients into biomass by primary producers, and the transfer of this energy to higher trophic levels, which may explain the delay between the onset of upwelling in the NCSR and the initiation of nesting by 50% of murres.

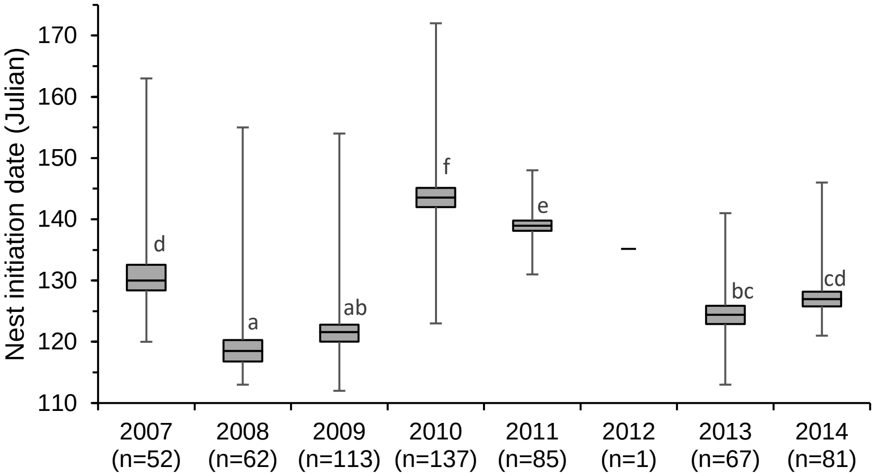


Figure 36. Timing of nest initiation (Julian date) by Common Murre nesting at Castle Rock between 2007 and 2014. For box plots, the center line in each box represents the mean, the extent of each box indicates the 95% confidence interval for the mean, and the whiskers indicate the range of the data. Letters just above each box indicate homogenous subsets (as determined by post-hoc tests from ANOVA). These calculations only included first-clutches (not replacement nests) and the date of initiation was accurate to ±3 days; sample sizes are indicated in parenthesis below each year. In 2012, only the timing of the earliest nest was displayed (—) because the video system failed before all nests were initiated; thus, the full distribution of nest initiation could not be determined.

The ability of murres nesting at Castle Rock to fledge chicks was relatively successful between 2007 and 2014. Even in the year with the poorest reproductive performance (2007), over half of breeding pairs fledged young (Figure 39). Between 2007 and 2014, prey abundance appeared to be the primary factor modifying reproductive success at Castle Rock; other factors known to diminish nest success in murres, specifically predation and disturbance (anthropogenic or natural), were not observed at this colony during the period of study (recordings occurred 24 hours per day for the duration of each breeding season except 2012). In the absence of predation or disturbance, murres are generally successful at raising chicks to fledging age, regardless of variability in prey resources; at many colonies throughout the CCS, long-term averages of reproductive success exceed 70% (Boekelheide et al. 1990, McClatchie et al. 2008, Fuller et al. 2015).

It is important to understand reproductive success as it informs long-term changes in breeding populations. Reproductive success is determined by the ability of individuals to hatch eggs as well as adequately feed and protect chicks until they fledge. Incubation and chick rearing are different; eggs must be kept warm and regularly rotated to hatch whereas chicks must be regularly fed, protected from attack by other murres, and protected from predation by other species. As such, the success of hatching and fledging is influenced by different environmental conditions. To properly interpret variability in overall reproductive success requires insight into factors that influence the success of eggs and chicks.

Hatching success is challenging to interpret relative to fledging success because, based on observations at Castle Rock, it is influenced by both prey availability and inclement weather. In 2007, both hatching and fledging success were low; this low success was based on murre behaviors in this year (discussed below) and prey were inadequate (Figure 39). The scarcity of prey in this year caused some individuals to leave eggs unattended so that they could feed and maintain their own survival (Eigner 2009). However, in other years there were significant rain events that appeared to diminish hatching success, specifically in 2010 and 2013 (Rian 2012, Golightly and Schneider 2016). Because both prey availability and weather influence hatching success at Castle Rock, attributing the cause of failure requires careful attention to weather and the prevalence of egg neglect. At Castle Rock, hatching success seems to be influenced more frequently by weather than by changes in ocean productivity, making this component of reproductive success less informative when trying to make inferences about the productivity of waters surrounding Castle Rock, which includes MPAs within foraging range of murres nesting at this location.

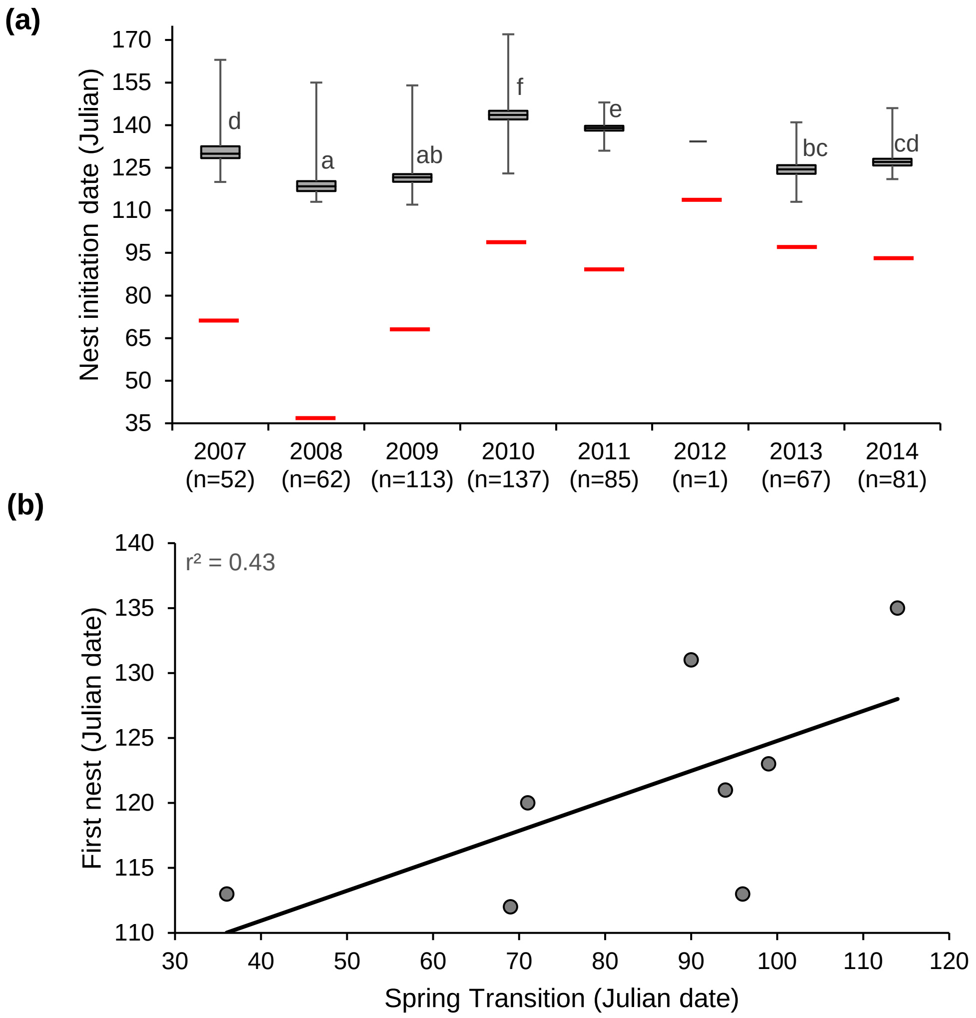


Figure 37. The (a) timing of Common Murre nest initiation (Julian date) at Castle Rock between 2007 and 2014 (symbols as in Figure 4) plotted relative to the Spring Transition date (—) and (b) the functional relationship between Spring Transition and the initiation of nesting. The date of Spring Transition was calculated using the CBR Mean Method for latitudes of 42ºN to 48ºN (Columbia Basin Research 2016). In 2012, the timing of nest initiation was only determined for the earliest nest (**—**), as the camera system failed before nesting was initiated by most individuals. These calculations only included first-clutches (not replacement nests) and the date of initiation was accurate to ±3 days; sample sizes are indicated in parenthesis below each year.

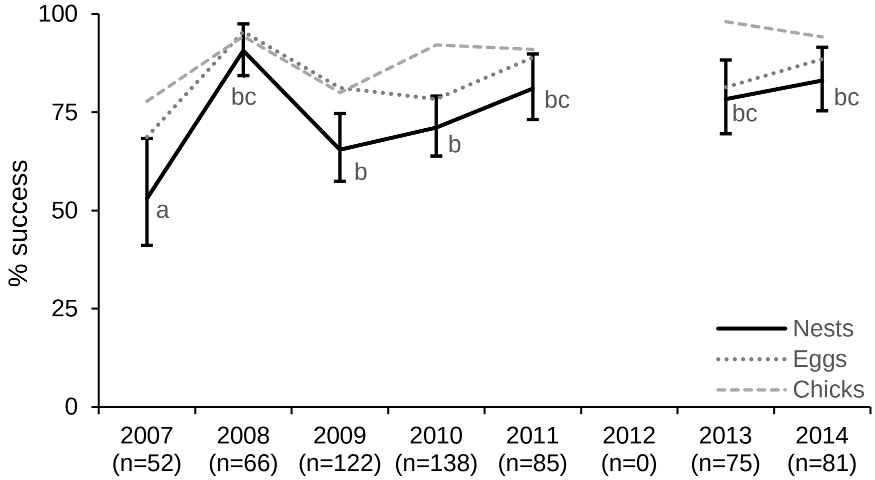


Figure 38. Reproductive performance of Common Murre nesting at Castle Rock between 2007 and 2014. The proportion of nests that were successful (nests), as estimated using the Mayfield Method, are depicted for each year. Error bars associated with this estimate represent 95% Confidence Intervals (CIs). Estimates of hatching (eggs) and fledging (chicks), the two components of nest success, are also depicted but 95% CIs were excluded for clarity. Replacement nests were not included in these calculations. Letters below each error bar represent similarities and differences between years. The number of nests (n) monitored to assess reproductive performance are provided in parenthesis below each year. In 2012, the video system failed early in the nesting season and reproductive performance could not be determined.

The energy demands of reproduction markedly increase during the chick-rearing period, as chicks must be fed regularly (Harding et al. 2007), and prey abundance sets the upper limit of fledging success as it determines the difficulty for a breeding pair to adequately feed their chick. However, fledging success can be further diminished by predation and disturbance events, but these events do not necessarily occur at all breeding colonies (Horton 2014, Suryan et al. 2014, Fuller et al. 2015). At Castle Rock, there was no evidence of large-scale disturbance and predation events during the study period and the variability observed in fledging success was thus likely determined by prey availability within flight distance of the colony. At Castle Rock, fledging success was relatively low in 2007 and 2009 (Figure 38); in these years, there were multiple observations of chicks starving and the behavior of chick-rearing individuals was indicative of prey scarcity (discussed in detail below). In these years, this inadequacy of prey was not limited to areas near Castle Rock. In both years, there was strong evidence that El Niño conditions resulted in a scarcity of prey in many areas of the CCS, which resulted in the fledging success of murres to be abnormally low at many colonies both north and south of Castle Rock (2007: Suryan et al. 2007, McChesney et al. 2008, McClatchie et al. 2008; 2009: Warzybok and Bradley 2009, Eigner et al. 2010). At Castle Rock, and other colonies where other sources of chick mortality are negligible, fledging success clearly indicated whether prey resources were adequate or inadequate to facilitate successful reproduction by seabirds (and other top predators) during each breeding season at a specific location.

Importantly, although successful reproduction indicates that prey were adequate to support the breeding population and their offspring, this parameter does not provide information about magnitudes of change in prey availability above this threshold. While fledging success highlights years when prey were extremely scarce and chick starvation became frequent, this parameter was not sensitive to fluctuations in prey that were less extreme or occurred over hours, days, and weeks. Murres can maintain fledging success across a wide gradient of prey abundance by modifying the amount of time spent acquiring prey (Davoren & Montevecchi 2003, Harding et al. 2007, Smout et al. 2013). When prey availability drops below a threshold where breeding pairs reach their behavioral limit to further compensate, then fledging success drops (Harding et al. 2007, Ashbrook et al. 2008, Schrimpf et al. 2012)

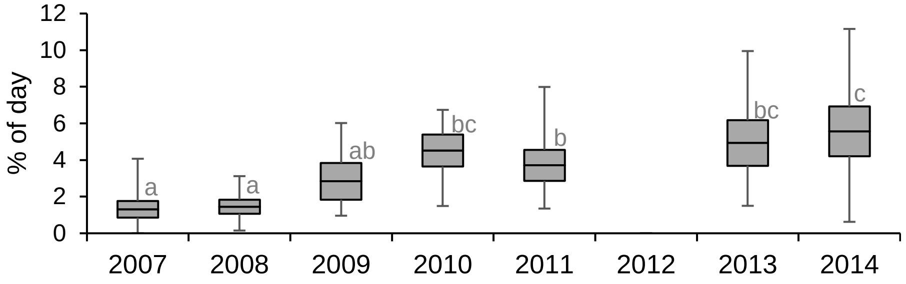
Foraging Effort

To understand the magnitude by which prey resources change requires parameters that respond to minor fluctuations of prey, as opposed to ones that only signal a transition between “good” to “bad”. Murres rely on behavioral flexibility to maintain reproductive success despite variability in prey. Specifically, chick-rearing murres compensate for fluctuations in prey by modifying the amount of time they dedicate to prey acquisition (Davoren and Montevecchi 2003, Harding et al. 2007, Smout et al. 2013). When prey are abundant, chick rearing pairs can spend up to half of their day at the colony together (Zador and Piatt 1999). As prey become scarce, the duration of co-attendance is reduced to maintain feeding rates required by young (Burger and Piatt 1990, Zador and Piatt 1999, Harding et al. 2007, Smout et al. 2013). In extreme cases, both birds leave their chick unattended to maintain provisioning but, when unattended, chicks are at increased risk of mortality from attacks by neighboring murres and opportunistic predators (e.g., gulls).

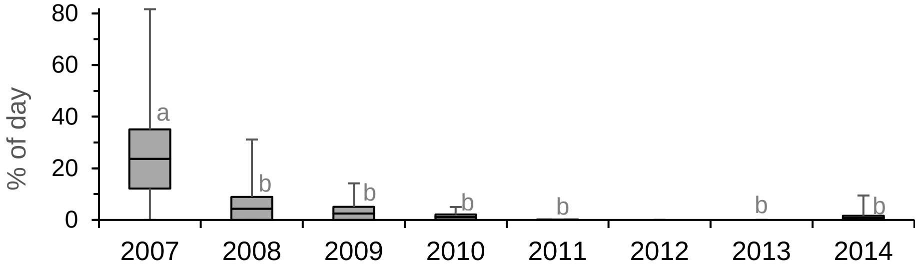
The amount of time that chick-rearing murres spend at the colony is sensitive to, and rapidly changes in response to, fluctuations in prey available near the breeding colony. To quantify behaviors indicative of foraging effort at Castle Rock, 125 chick-rearing pairs were observed between 2007 and 2014. Over this 8-year period, 349 hours of video from 24 observation days were reviewed for these purposes.

At Castle Rock, it was typical for a single member of each pair to remain at the nest with their chick; attendance by a single parent encompassed 91±1.4% of daylight hours across the 8-year study period. Co-attendance was regular, occurring 3.8±0.2 times per day, but was limited in duration to 3.4±0.2% of daylight hours across the 8-year study period (29.5±1.9 minutes per site per day). The duration of time that chick-rearing pairs were together at the colony was lowest in 2007 and 2008 and greatest in 2014 (ANOVA: $F_{6,124}=15.4$, $P<0.001$, Figure 39a). Non-attendance, when chicks were left unattended, also occurred and averaged 5.9±1.4% of daylight hours across the 8-year study period (51.0±12.2 minutes per site per day). In 2007, chicks were left alone longer than other years (ANOVA: $F_{6,124}=9.92$, $P<0.001$, Figure 39b); the duration of non-attendance averaged 23±5.5% of the day in 2007, which was 5.4 times greater than in 2008, the year that ranked second in terms of the duration of non-attendance. The relatively low co-attendance and the prevalence of non-attendance observed in 2007 was indicative of insufficient prey availability near Castle Rock during this year. Also in 2007, the rate at which chicks were fed was lowest relative to other years (ANOVA: $F_{6,124}=6.06$, $P<0.001$, Figure 39c) and starvation became relatively frequent.

(a) Co-attendance



(b) Non-attendance



(c) Provisioning rate

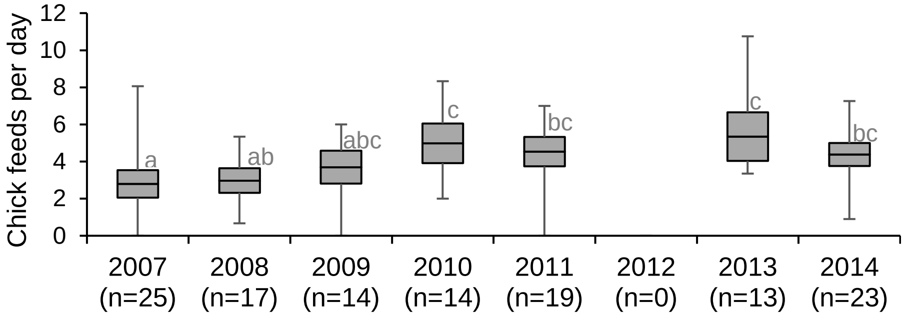


Figure 39. The amount of time that chick-rearing murres spent in co-attendance (a), the amount of time that chicks were left alone (b), and the rate of chick provisioning at Castle Rock in 2007 through 2014 (symbol as in Figure 36). The number of chick-rearing pairs (n) monitored to assess time allocated to foraging are provided in parenthesis below each year. In 2012, the video system failed early in the nesting season and these parameters could not be determined.

Generalizing across the 8-year study period, co-attendance and chick provisioning trended upward while non-attendance trended downward (Figure 39). This indicates that prey availability near Castle Rock generally increased between 2007 and 2014. Although there was an upward trend in prey availability, average time spent in co-attendance never exceeded 6% and, in all years except 2013, chick-rearing pairs had to leave chicks unattended in an attempt to locate adequate prey. At other colonies, co-attendance typically remains above 5-10% unless the colony is experiencing food shortages (Uttley et al. 1994, Ainley et al. 2002, Ashbrook et al. 2008, Golightly & Schneider 2016). Also, chick-rearing pairs often left chicks unattended at Castle Rock while they both searched for prey. Based on long-term studies from other breeding colonies, murres only leave chicks unattended in circumstances of extreme prey scarcity, when the risk of chick mortality from starvation exceeds the risk of chick mortality due to predation and conspecific attack (Ashbrook et al. 2008, 2011).

Prey availability near Castle Rock, as indicated by murre foraging effort, was lower than is typical for other breeding colonies monitored in the CCS; elsewhere in the CCS, murres generally have co-attendance greater than 10% and chicks are rarely left alone (see Appendix G of Golightly and Schneider 2016). There are many explanations for this observation and none are mutually exclusive. It is possible that: (1) there were actual differences in the absolute abundance of prey near Castle during 2007 and 2014; (2) there were differences in the distribution and density of prey near Castle Rock caused by movement of the upwelling jet offshore in this area due to interactions between the current and bathymetry of Cape Blanco (120 km north of Castle Rock; (Barth et al. 2000, 2005, Huyer et al. 2005); or (3) the depletion of prey by murres occurs much quicker near Castle Rock due to the extreme abundance of murres nesting at this particular location (Birt et al. 1987, Gaston et al. 2007, Elliott et al. 2009). To resolve uncertainty about the exact circumstances causing prey to be limited near Castle Rock and determine the actual extent this prey scarcity (i.e., just near Castle Rock or across the entire NCSR) would require additional studies of prey in the NCSR.

The baseline characterization of murre behavior at Castle Rock provided evidence that the energy required to support nesting almost matches the energy available to them within flight distance of this breeding colony. Any event that shifts the balance availability and need could result in large-scale reproductive failure of seabirds nesting at Castle Rock. Although prey availability in the waters surrounding Castle Rock was characterized using murres behavior, this observation is relevant to other top predators that live and reproduce in this area of the NCSR. It is likely that there is great overlap in the prey base and foraging environment of murres and other seabirds, especially pursuit-diving piscivores such as other Alcids (guillemots, puffins, and some auklets) and cormorants. Sympatrically nesting seabirds experience similar foraging conditions because the distance they can travel from the colony to forage is limited by their need to incubate eggs and feed chicks (Orians & Pearson 1979, Elliott et al. 2009, Fauchald 2009). Although the maximum foraging range is species-specific, physical adaptations of diving seabirds such as Alcids and cormorants add constrains on their ability to fly (Pennycuick 1987). Like murres, many piscivorious species nesting in the CCS opportunistically capture prey, which has the consequence of dietary overlap as prey composition is determined by availability (Ainley et al. 1990, Forero et al. 2004, Gladics et al. 2014, Webb & Harvey 2014). Furthermore, prey can concentrate in specific areas where upwelling and ocean productivity is greatest. This aggregation of prey resources causes seabirds to forage in multi-species feeding flocks, further facilitating overlap of prey base (Diamond 1983) despite some specialization (Ainley et al. 1990). Based on long-term observations at the Farallon Islands, the diets of piscivorious species can overlap extensively, especially in years when upwelling is strong and prey are abundant (Ainley et al. 1990, 1996, 2005).

Chick Diet

Between 2007 and 2014, 508 hours of diet surveys occurred and 4302 prey deliveries were observed. In total, 74.8% of prey were classified to family or better and 20 distinct prey types were identified (range: 12-15 per year). Although 20 distinct prey types were identified, 90.6% of all prey identified were one of four prey types: smelt (Osmeridae), rockfish (*Sebastes sp.*), salmon (Salmonidae), and sand lance (*Ammodytes hexapterus*; Figure 40). Smelt (60.7% of prey) and rockfish (22.7% of prey) were the two most common prey types observed in each year (Figure 40). Based on the four most common prey, the observed composition differed from expected in all years except 2011 (χ^2_3=8.5, *P*=0.07; Figure 40). In 2009, prey deviated most from the average condition (χ^2_3=86.5, *P*<0.001, Figure 40). In 2009, differences were primarily due to rockfish being 78% less common than expected and smelt being 26% more common than expected. In 2010, differences were primarily caused by more salmon.

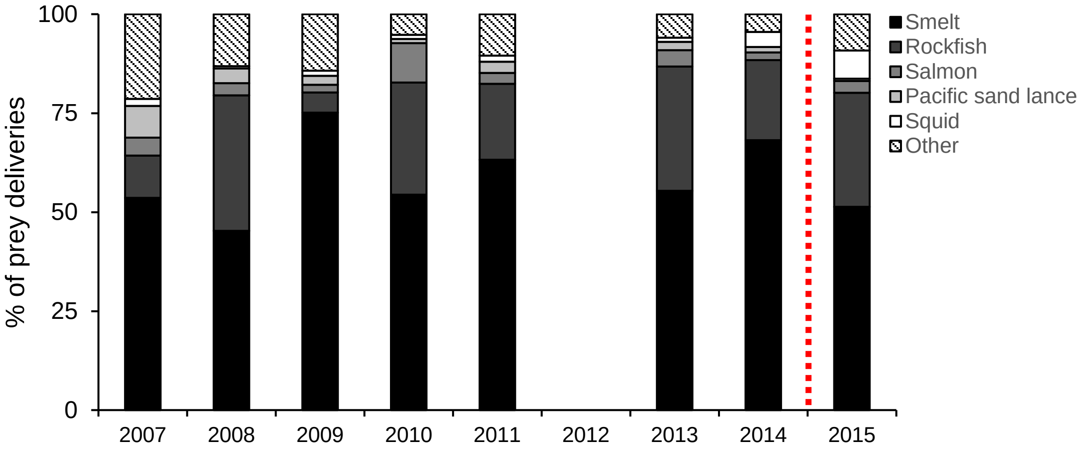


Figure 40. Annual and overall ("ALL") composition of prey delivered to chicks at Castle Rock by Common Murre between 2007 and 2014. The dotted line was used to visually separate prey composition for individual years from the overall composition. Sample sizes (n) indicate the number of identified prey each year because prey that could not be identified were excluded from this analysis. Prey composition could not be determined in 2012 due to premature failure of the video monitoring system.

Long-term identification of prey delivered to Castle Rock during the chick-rearing period revealed a relatively stable composition of prey available to top-predators in the nearby waters. While there is a diversity of prey, the same four prey types dominated the diet every year between 2007 and 2014. This 8-year dataset provided a robust understanding of the current prey composition in the northern sections of the NCSR and any future changes in prey composition can be readily detected by diet studies.

When compared with long-term studies of murre diet at other locations in the CCS, prey composition near Castle Rock is quite different from other areas, particularly to the south. While smelt dominated the diet at Castle Rock, clupeids, rockfish, and hake dominated the diets of murres nesting in central California. Murres in central California typically consume Northern Anchovy, other clupieds, and rockfish (Ainley et al. 1990, Roth et al. 2008, Eigner 2009, Gladics et al. 2014, Suryan et al. 2014). In fact, the prey community at Castle Rock was more like areas north of Point Blanco, based on reports from at Yaquina Head during the same period (Gladics et al. 2014, 2015). This was especially true in 2010, when smelt and rockfish accounted for about 80% of all prey observed at both Yaquina Head and Castle Rock. Despite similarities between the prey communities near Yaquina Head and Castle Rock, there were also noticeable differences; rockfish abundance was generally much lower at Yaquina Head and prey types uncommon at Castle Rock, including herring, sardine, flatfish, and Pacific sand lance were prevalent in the diet of murres at Yaquina Head (Gladics et al. 2015). The observations of chick diet at Castle Rock confirm that prey communities are not homogenous between northern and central regions of California and diet studies at Castle Rock enhance understanding of temporal variability in prey composition across the CCS, including the NCSR.

Synthesis

This long-term study of Common Murre reproductive performance, foraging effort, and diet provides robust baseline understanding of current conditions of the NCSR. This effort will allow for future alterations in marine condition to be detected by using murres nesting at Castle Rock as a responsive and reliable indicator. Importantly, this characterization incorporates 8 years of variability into the characterization of murre reproduction, foraging effort, and diet. Importantly, this characterization extends beyond the 2014-2015 study period typical of many other NCSR baseline characterizations, for which the observations are confounded by anomalous ocean conditions occurring during the 2014-2015 period (Leising et al. 2015, McClatchie et al. 2016). This 8-year time series will provide a more accurate representation of baseline condition in the NCSR prior to the establishment of MPAs and will strengthen our ability to distinguish MPA-induced changes from natural environmental variability. If the impact of MPAs on the marine system expands beyond their geographic boundary, seabirds nesting in the NCSR will likely utilize and benefit from these changing resources.

Each parameter characterized by seabird-monitoring efforts at Castle Rock were effective and interpretable indicators of current condition, which would allow for detection of both short- and long-term trends (Table 10). Furthermore, murre reproductive performance, foraging effort, and chick diet collectively provided valuable information about the health of seabirds nesting in the NCSR and a better understanding of the abundance and composition of prey that murres and other top predators within the NCSR rely on. This research relied on video-based method which allowed us to permanently record detailed observations without disturbing breeding seabirds or nesting habitat, allowing for accurate and repeatable data collection and also provide a repository of video that can be revisited to answer unanticipated questions that develop in the future.

Table 10. Biological parameters measured by studies of Common Murre nesting at Castle Rock and their relationship to environmental influences.

Parameter	Environmental Drivers	Response Time	Character of Response
Reproductive performance			
Nest initiation	upwelling, prey abundance	annual	sensitive; continuous
Hatching success	weather, prey abundance	annual	threshold; good/bad
Fledging success	prey abundance	annual	threshold; good/bad
Foraging effort			
Time allocation	prey abundance	day	sensitive; continuous
Chick provisioning	prey abundance/quality	day	sensitive; continuous
Chick diet			
Prey composition	prey available near colony	day	sensitive; continuous

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CHAPTER 4: BASELINE MONITORING OF COASTALLY BREEDING SEABIRDS WITHIN THE NORTH COAST STUDY REGION OF THE CALIFORNIA MARINE LIFE PROTECTION ACT INITIATIVE

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INTRODUCTION

Seabird Life History and Potential MPA Benefits

Marine protected areas (MPAs) can have both direct and indirect benefits to seabird populations. Direct benefits include 1) reduced disturbance to breeding and roosting sites and 2) decreased human interaction (e.g., bycatch, light attraction, gear entanglement) at foraging sites. Indirect benefits include 1) reduced competition with humans for food resources and 2) greater prey supplies resulting from increased prey production (Tasker et al. 2000).

As upper-level predators, seabird populations are regulated primarily from the bottom up (see Ainley et al. 1995) and show quick responses to changes in prey availability. Prey availability has been shown to affect coloniality (whether birds form large or small colonies), the timing of reproduction, clutch sizes, chick growth, non-predator related chick mortality, and reproductive success (Anderson and Gress 1984, Safina and Burger 1988, Pierotti and Annetti 1990, Massey et al. 1992, Ainley et al. 1995, Monagham 1996, Golet et al. 2000). Though top-down regulation does occur, it is often exacerbated by human activities that disturb breeding and resting sites. The impacts of human disturbance tend to be most pronounced when humans enter the immediate area (see Carney and Sydeman 1999). Intrusions often result in most, if not all, birds fleeing from the immediate area, leaving eggs and chicks vulnerable to predators such as gulls and ravens. While some birds return to nests once an intruder has gone, others will abandon nesting efforts. For example, Brandt's Cormorants have been observed to abandon nests en masse from even single events of human intrusion to the colony (McChesney 1997). Although often not as easily identified, activities such as close approaches (e.g., by boats, surfers, etc.) to colonies and roosts can evoke responses similar to direct human intrusions (Jacques et al. 1996, Carney and Sydeman 1999, Jaques and Strong 2002). Several studies have shown reductions in breeding success or population sizes as a result of close approaches (e.g., Wallace and Wallace 1998, Carney and Sydeman 1999, Thayer et al. 1999, Beale and Monaghan 2004, Bouton et al. 2005, Rojek et al. 2007).

Not all seabird species are equal in their potential to benefit from MPA establishment. Thus, the Science Advisory Team for the North Coast Study Region (NCSR) developed a list of species that would likely benefit from MPA establishment. We selected species from this list to serve as the focus of our baseline monitoring (see Focal Species below). Additionally, we focused on species with a high susceptibility to human disturbance and dependence on locally available prey. For example, Pelagic Cormorants can forage up to 15 km away from the breeding colony, but typically stay much closer (Hobson 1997). In California, their diet is dominated by mid-sized rockfish, sculpins, and other rocky-bottom demersal fishes (Ainley et al. 1981). Pigeon Guillemots typically forage within six kilometers of the breeding colony in depths of 6-45 m (Clowater and Burger 1994, Litzow et al. 2000). In California, guillemot diet is dominated by young rockfish and sculpins (Farallon Islands; Ainley and Boekelheide 1990) and young sanddabs (Point Arguello; Robinette et al. 2007). Furthermore, Litzow et al. (2000) found that changes in guillemot diet were sensitive to local prey abundance rather than regional prey abundance. Black Oystercatchers maintain breeding and foraging territories along rocky shores and, in California, feed primarily on intertidal mussels and limpets (Andres and Falxa 1995).

Before-After-Control-Impact (BACI) Monitoring Approach

The ultimate goal of an adaptive management program is determining whether management actions result in their intended consequences. With regard to MPA management, biologists and resource managers must determine whether or not changes observed within a given MPA are due to the establishment of that MPA versus factors that are simultaneously acting on communities both inside and outside of MPAs (Rice 2000, Gerber et al. 2005). There are several ways to accomplish this. Some programs may take a 'before-after' approach by comparing performance indicators measured before MPA establishment to those measured afterward. If baseline or 'before' data do not exist, a program may take a 'control-impact' approach by comparing performance indicators inside an MPA (the 'impact' area) to those at a control site outside the MPA. The more robust approach to establishing causation is to combine these into a ‘before-after-control-impact’ (BACI) monitoring program (McDonald et al. 2000). Such a program involves measuring indicators at impact and control sites before and after MPA establishment. There are two general approaches to BACI monitoring. If a long period of baseline data exists, then the investigator can take a time series approach, monitoring a single pair of impact and control plots. However, if a baseline time series does not exist, then multiple sites must be used (McDonald et al. 2000).

We chose six MPAs from three areas (see Figure 41) for our BACI monitoring design: Pyramid Point SMCA (Del Norte), South Cape Mendocino SMR (Humboldt), and Ten Mile SMR, MacKerricher SMCA, Point Cabrillo SMR, and Russian Gulch SMCA (Mendocino). We are using the BACI monitoring design to assess MPA-related changes in 1) breeding population size, 2) roost utilization, 3) rates of human-caused disturbance, and 4) foraging rates. It is important to document the proportion of breeding populations protected by MPAs in order to track changes in population size attributable to MPA protection. The establishment of MPAs should result in decreased disturbance rates due to reduced boat traffic; much of the nearshore boat traffic in the NCSR is due to fishing activity and should be reduced in areas where fishing is prohibited. If MPAs are effective in reducing boat traffic, then there will be a decrease in both the number of boat approaches and disturbance events at colonies within these areas compared to unprotected areas. Because most species can forage up to several kilometers from the nest site, a seabird colony does not have to reside within an MPA to benefit from MPA establishment. As long as an MPA is within foraging range for a given species, then that species can potentially benefit from the increased prey availability created by the MPA. Thus, we are using the BACI design to look at foraging rates inside and outside of MPAs. We are not, however, using the BACI design to assess MPA-related changes in breeding productivity. Breeding productivity will be influenced by factors acting adjacent to the colony as well as those away from the colony (e.g., foraging areas). Thus, the benefits of MPA establishment to breeding productivity are likely to be experienced over a broader spatial scale. Our monitoring design therefore focuses on tracking changes in productivity at each of the three focal areas (Del Norte, Humboldt, and Mendocino) over time and performing before-after types of comparisons to measure MPA-related changes within these areas.

Baseline Monitoring Objectives

This report represents a baseline characterization of coastal seabird ecology within the NCSR and the ‘before’ component of our BACI monitoring program. The objectives of our baseline monitoring efforts were five-fold:

1. Assess baseline seabird breeding population size at sites inside and outside of MPAs.
2. Assess seabird roost utilization at sites inside and outside of MPAs.
3. Assess baseline levels of human-caused disturbance at breeding colonies and roosting sites inside and outside of MPAs.
4. Assess baseline breeding productivity at each of the three focal areas.
5. Assess baseline seabird foraging rates at sites inside and outside of MPAs.

Baseline monitoring was conducted in 2014 and 2015. In order to fully implement our BACI monitoring program, it will be important to revisit these monitoring sites with a minimum of five-year intervals. Additionally, it will be necessary to monitor for multiple years within each interval to account for the effects of oceanographic and prey variability on seabird metrics. The NCSR is greatly influenced by the California Current, an eastern boundary current that creates some of the most oceanographically variable conditions in the world (Ainley et al. 1995). Interannual variability in the California Current, in addition to variability in larger scale processes such as the El Niño Southern Oscillation and Pacific Decadal Oscillation, creates high interannual fluctuation in biological productivity and food web structure within the NCSR. Continued long-term monitoring, coupled with available oceanographic data, will allow us to use statistical models to determine the degree to which MPAs and oceanographic processes are affecting seabird metrics.

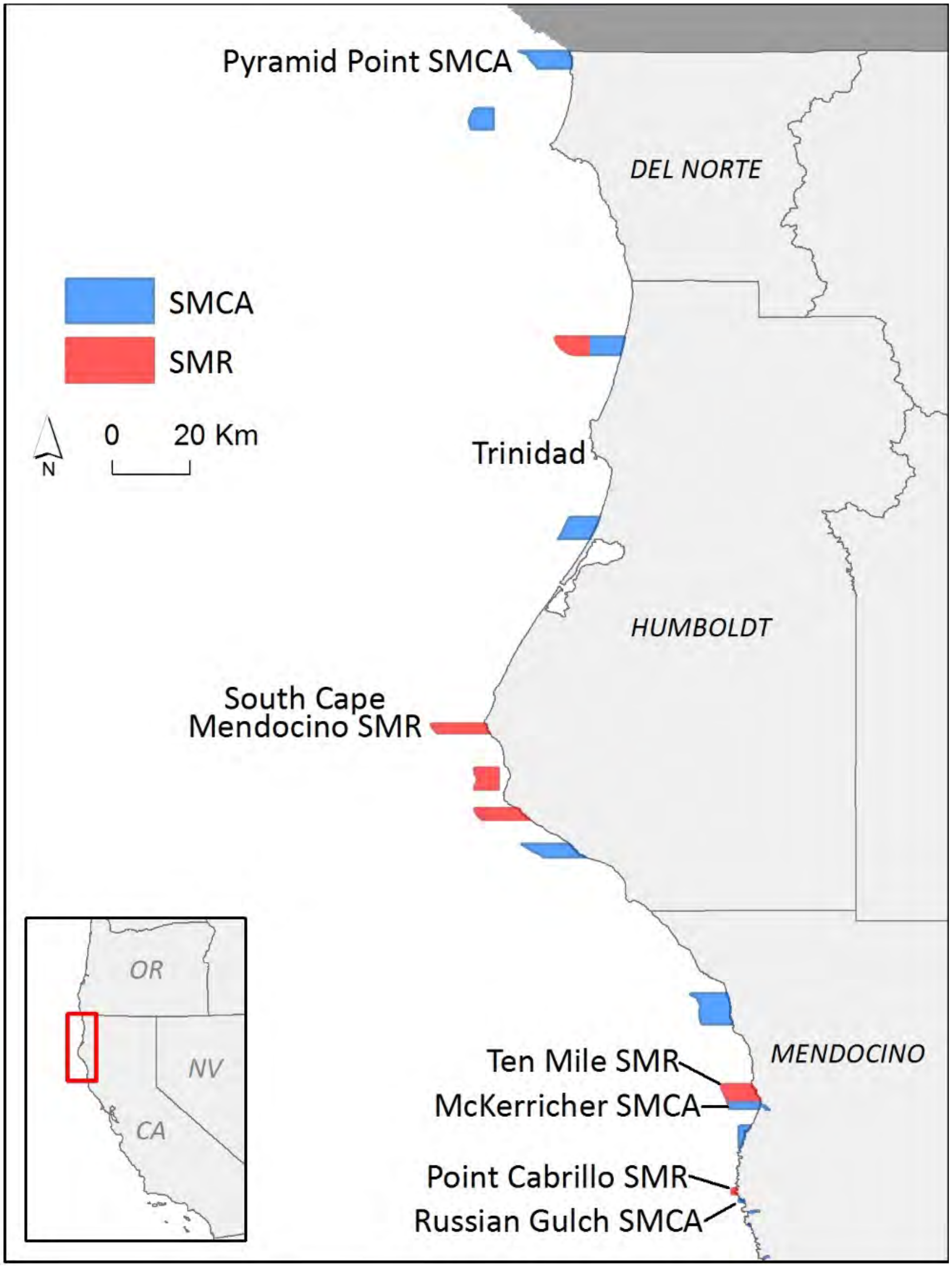

Figure 41. Map of the North Coast Study Region showing the six marine protected areas surveyed for the coastal seabird baseline.

METHODS

Study Area

We selected multiple monitoring sites distributed across three counties (Del Norte, Humboldt, and Mendocino) in order to maximize our coverage of the NCSR coastline. Figure 42 shows the stretches of coastline monitored within each county. When selecting sites, the distribution of coastal breeding habitat and coastal access were major challenges. There is little suitable breeding habitat along the Del Norte coast. The largest seabird colony in Del Norte is at Castle Rock which is too far from the coast to be monitored during coastal surveys. Baseline monitoring at Castle Rock is summarized in Chapter 2 of this report. Foraging surveys (see methods below) were conducted within the Pyramid Point SMCA and at the Crescent City control site. There is no suitable breeding or roosting habitat within the Pyramid Point SMCA. We therefore only conducted transect surveys (for monitoring breeding population and roost utilization, see methods below) at the Crescent City control site. The majority of breeding habitat within Humboldt is located along the Trinidad and Cape Mendocino coasts. There are no MPAs at Trinidad. Therefore, the Trinidad transect and foraging sites served as controls. There are few roads to access the Cape Mendocino coast. We conducted foraging and transect surveys within the South Cape Mendocino SMR, foraging and transect surveys at the Devil’s Gate control site, and transect surveys at the Cape Mendocino control site. There are several roads with coastal access within the Mendocino area, but multiple stretches of inaccessible coastline as well. We conducted foraging surveys within the Ten Mile SMR and the Cabrillo Point SMR and at the Kibesillah and Mendocino Headlands South control sites. We conducted transect surveys within the Ten Mile SMR, MacKerricher SMCA, Point Cabrillo SMR, and Russian Gulch SMCA and at the Kibesillah, Mendocino Headlands North, and Mendocino Headlands South control sites.

Focal Species

The Science Advisory Team (SAT) for the NCSR identified locally breeding and migratory species that will likely benefit from MPA establishment based on their susceptibility to human disturbance and dependence on locally available prey. We monitored the breeding population size of five of these species: Black Oystercatcher, Double-crested Cormorant, Brandt’s Cormorant, Pelagic Cormorant, and Pigeon Guillemot (see Chapter 1 for scientific names and summaries of breeding biology), the latter three species were further elevated by the SAT to a list of species “most likely” to benefit because of their dependence on the types of nearshore prey that will likely benefit from MPA establishment, namely nearshore demersal fishes. Additionally, we monitored the breeding population size of Western Gulls as they are an endemic species that can be impacted both positively and negatively by human activities. We were able to follow the breeding productivity of Black Oystercatchers, Double-crested Cormorants, Brandt’s Cormorants, Pelagic Cormorants, and Western Gulls. Common Murres were also listed by the SAT as a species “most likely” to benefit from MPAs. This species breeds on offshore rocks that are difficult to view from the coast. Long-term data on breeding dynamics and diet for Common Murres breeding on Castle Rock were analyzed as part of the baseline NCSR monitoring and are reported in Chapter 3. We monitored roost utilization for Double-crested Cormorants, Brandt’s Cormorants, Pelagic Cormorants, and California Brown Pelicans. Brown Pelicans breed in southern California and Baja California, Mexico. They migrate north after the breeding season and begin arriving in the NCSR during the summer months. We monitored nearshore foraging rates of Double-crested Cormorants, Brandt’s Cormorants, Pelagic Cormorants, Pigeon Guillemots, Common Murres, and Marbled Murrelets. Marbled Murrelets breed in coastal old growth forests and breeding population size and productivity was not monitored as part of the NCSR baseline project. Finally, though we recorded human-caused disturbances to all species, we used data on Brown Pelicans, Brandt’s Cormorants, and Western Gulls to index disturbance rates because these are the most conspicuous for breeding and roosting and their reactions to human activities are easily recognized.

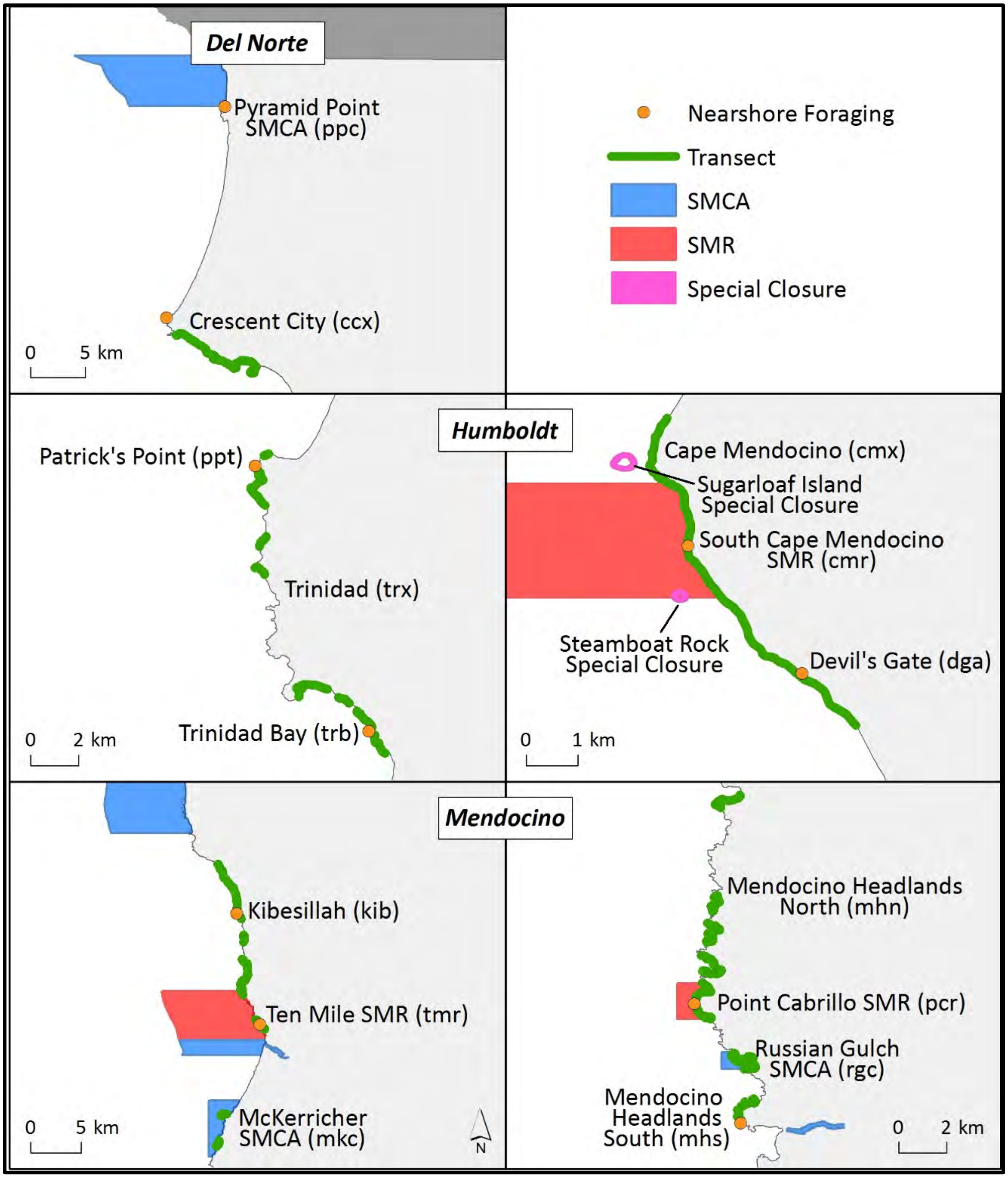

Figure 42. Map showing coastal seabird monitoring sites within the Del Norte (top), Humboldt (middle), and Mendocino (bottom) areas of the North Coast Study Region (NCSR). See Figure 41 for location of areas within the entire NCSR.

Monitoring Methods

Beginning April 15 of each year (when seabird nest initiation is typically well under way), we conducted four types of surveys: transect monitoring, nest monitoring, foraging surveys, and disturbance monitoring.

Each survey type was conducted weekly through the end of July. Sample sizes for each survey type are shown in Table 11.

Transects

Goals. The goals of transect monitoring are three-fold: 1) to document the size of annual breeding populations for each focal species inside and outside of MPAs, 2) to document roost utilization for each focal species inside and outside of MPAs, and 3) to identify nests that can be followed for estimating annual productivity.

Methods. We conducted area count surveys along the coastal sections highlighted in Figure 42. We divided each transect into counting blocks viewable from predetermined observation points. Beginning the week of April 15, we conducted one transect survey per week along each coastal section. We conducted surveys between the hours of 0600 and 1000 as this is the peak time for Pigeon Guillemot rafting activity and roosting activity by non-breeding birds. Nests counts were not possible for Pigeon Guillemots as this species nests in mostly inaccessible rock crevices. However, guillemots often raft on the water or roost on rocky shorelines adjacent to nesting areas. Peak numbers usually occur in early morning and in the pre-breeding season (Point Blue, unpubl. data). For each survey, we began at one end of a given transect and visited each observation point. We alternated starting points between the north and south ends of the transect on a weekly basis to minimize time bias on guillemot raft counts. From each observation point, we scanned the adjacent counting blocks using binoculars and a 20-60x spotting scope. We recorded the number of nesting, roosting, and rafting (for guillemots only) birds observed within each counting block. We recorded data on each of the focal species identified above. We report breeding population size as the peak number of nesting birds (i.e., peak number of nests multiplied by two) observed during area count surveys for all species but Pigeon Guillemots. For guillemots, we report the peak number of rafting birds for each site. We report roost utilization as the mean ± SE number of birds roosting per week at each site.

Nest Monitoring

Goals. The overarching goal of nest monitoring is to record annual nesting phenology and estimate annual breeding productivity. Both phenology and productivity are good indicators of the underlying oceanographic conditions affecting annual population size. Recording phenology requires weekly checks on individual nests within a given colony. Productivity can be calculated as either 1) the number of fledglings produced per adult breeding pair or 2) the percentage of total eggs laid that hatched and successfully grew into fledglings. The first calculation requires only knowledge of the number of fledglings produced within a given nest. The second requires more detailed knowledge of how many eggs were laid, how many of those eggs hatched, and how many of those chicks fledged. In this report, we use the first method to calculate productivity as we were able to collect this data at all areas. However, in some areas, we were able to obtain views of nests to collect data on number of eggs laid. These data can be analyzed at a later date if a more detailed analysis of productivity is warranted.

Methods. We identified monitorable nests during our transect surveys. A monitorable nest is one for which eggs, chicks, and fledglings can be clearly viewed and enumerated without disturbing the nesting adults; though in some cases we were only able to view chicks and fledglings. Once nests were identified, we monitored them every 7 days. During each monitoring visit, we recorded 1) nest condition, 2) number of adults attending the nest and whether one is in incubating posture, 3) number of eggs, 4) number of chicks, 5) the feather condition of chicks, 6) number of fledglings and 7) if nest fails, the reason for nest failure to the extent possible (i.e., Were abandoned eggs left in the nest? Were dead chicks observed in the nest? Was there evidence of predation?). We report breeding productivity as number of fledglings produced per breeding pair for each site. Table 11 shows the number of nests followed for each species at each location while Figures 49 through 52 show breeding productivity for Brandt's Cormorants, Double-crested Cormorants, Pelagic Cormorants, Western Gulls, and Black Oystercatchers at each of the three areas surveyed (Del Norte, Humboldt, and Mendocino).

Table 11. Sample sizes for each of the four survey types conducted in 2014 and 2015 summarized by area and site. See Figure 42 for full site names.

	Del Norte		Humboldt		Mendocino	
	2014	2015	2014	2015	2014	2015
Transect Monitoring – Sample Size = Number of transects completed						
Inside MPA	No Transect	No Transect	cmr - 15	cmr - 14	tmr - 15 mkc - 15 pcr - 15 rgc - 15	tmr - 15 mkc - 15 pcr - 15 rgc - 15
Control	ccx - 8	ccx – 13	trx - 15 cmx - 14 dga - 15	trx - 16 cmx - 14 dga - 15	kib - 14 mhn -15 mhs - 14	kib - 15 mhn -15 mhs - 15
Breeding Productivity – Sample Size = Number of nests monitored						
Double-crested Cormorant	0	0	28	30	0	0
Brandt’s Cormorant	0	0	4	0	31	10
Pelagic Cormorant	36	13	44	21	52	24
Black Oystercatcher	2	0	1	1	7	7
Western Gull	3	3	2	2	7	10
Nearshore Foraging Rates – Sample Size = Number of 1-hr periods						
Inside MPA	ppc - 22	ppc - 17	cmr - 45	cmr - 41	tmr - 41 pcr - 32	tmr - 41 pcr - 42
Control	ccx - 25	ccx - 32	ppt - 44 trb - 44 dga - 44	ppt - 42 trb - 45 dga - 40	kib - 42 mhs - 40	kib - 37 mhs - 41
Disturbance Rates – Sample Size = Number of hours of observation						
All sites combined	126	126	253	217	259	219

Monitoring Nearshore Foraging

Goals. The goals of nearshore foraging surveys are to document 1) the number of seabirds foraging individually inside and outside of MPAs and 2) the number of seabirds foraging in flocks inside and outside of MPAs. We distinguish between the foraging behaviors for two reasons. First, the numbers of birds participating in foraging flocks can be orders of magnitude higher than the numbers of birds foraging individually and must be analyzed separately. Second, the behaviors represent foraging on different types of prey. Foraging flocks are formed by birds foraging on pelagic schooling prey such as anchovies while individual foraging birds typically prey on mid-water and demersal prey such as juvenile rockfish and flatfish.

Methods. Beginning about April 15 of each survey year, we conducted seabird foraging surveys at each of the survey sites shown in Figure 42. We surveyed each site once a week. We conducted surveys during one of the following time periods: 0600-0900, 0900-1200, 1200-1500, or 1500-1800, rotating sites among the four time periods per week to develop a complete 12-hour assessment of foraging activity. We conducted weekly surveys through the last week of July. We made observations from a single observation point, using binoculars and a 20-60x spotting scope. We divided each three-hour period into 15-minute blocks. During each 15-minute block, we scanned all water within a one-kilometer radius of our observation point and recorded the numbers of actively foraging individuals for all species. We also recorded the presence of all foraging flocks and collected data on the numbers of each species participating in the flock. We defined a foraging flock as five or more birds foraging on an aggregation of prey (e.g., an aggregation of anchovies). For individual foragers, we averaged all 15-minute blocks over a given hour of observation. If 100% of the study area was not visible (e.g., due to fog, sun glare, etc.) during two or more 15-minute blocks for a given hour, that hour was not included in the analysis. Here, we report the mean ± SE number of foraging individuals per hour of observation. We report results for abundance (total birds for all species combined), species richness, and species diversity using the Shannon diversity index (Brower et al. 1998). Additionally, we present results for the following species: Pigeon Guillemot, Double-crested Cormorant, Brandt's Cormorant, Pelagic Cormorant, Common Murre, and Marbled Murrelet. For foraging flocks, we report the number of flocks observed at each site and the mean number of individuals participating in flocks.

Disturbance Monitoring

Goals. The goals of disturbance monitoring are 1) to identify human activities that cause disturbance, 2) to identify human activities that do not cause disturbance, and 3) to estimate rates of human-caused disturbance at individual colonies. Disturbance is defined as any event that results in one or more of the following:

1) Birds flushing (birds flying off the rock)
2) Birds displacing (moving from their nest or resting site)
3) Eggs or chicks being:
    a. exposed (adult moves away from the egg or chick),
    b. displaced (egg or chick moves from nest site), or
    c. taken (egg/chick is depredated).
4) Birds becoming visibly agitated

Methods. We recorded all disturbances observed during any of the surveys mentioned above. At the beginning of each survey, we recorded the number of breeding and roosting birds present for each species. We recorded all land-based human activity and boat traffic within 1,500 feet, and aircraft flying at altitude of $\leq$1000 feet and within 1,500 horizontal feet of breeding/roosting seabirds, regardless of whether disturbance occurred or not. When a disturbance occurred, we recorded the following information:

1. Number of birds disturbed and reaction type for each species
2. Number of nests with eggs and chicks exposed for each species
3. Source of disturbance
4. Source altitude and distance from nesting area affected
5. Activity of disturbance source
6. Identification information (e.g., type of vessel or aircraft and any identifying information like license number)
7. Direction of travel
8. Duration of disturbance event

We calculated the monitoring effort (total hours of observation) for each area (Table 11). Here, we present the number of human-caused disturbances per hour of observation.

RESULTS AND DISCUSSION

Seabird Breeding Populations

Breeding Population Size

Figures 43 through 48 show the breeding population size and distribution of six focal species within the areas we surveyed: Brandt's Cormorant, Pelagic Cormorant, Double-crested Cormorant, Pigeon Guillemot, Western Gull, and Black Oystercatcher. We documented all species but Brandt's Cormorants and Double-crested Cormorants breeding within all three focal areas. We documented Brandt's Cormorants breeding in Mendocino and Humboldt and Double-crested Cormorants in Humboldt only.

Brandt's Cormorants (Figure 43). We documented a total of 325 breeding pairs in 2014 and 123 breeding pairs in 2015. Of these, 16.6% were found breeding inside SMRs in 2014 and 21.1% inside SMRs in 2015. All of these were breeding within the South Cape Mendocino SMR. The remaining population (83.4% in 2014 and 78.9% in 2015) were within the Mendocino Headlands South control site. Populations were largest at both sites in 2014 compared to 2015. This was especially true at Mendocino Headlands South where the 2014 population was more than twice the size of the 2015 population.

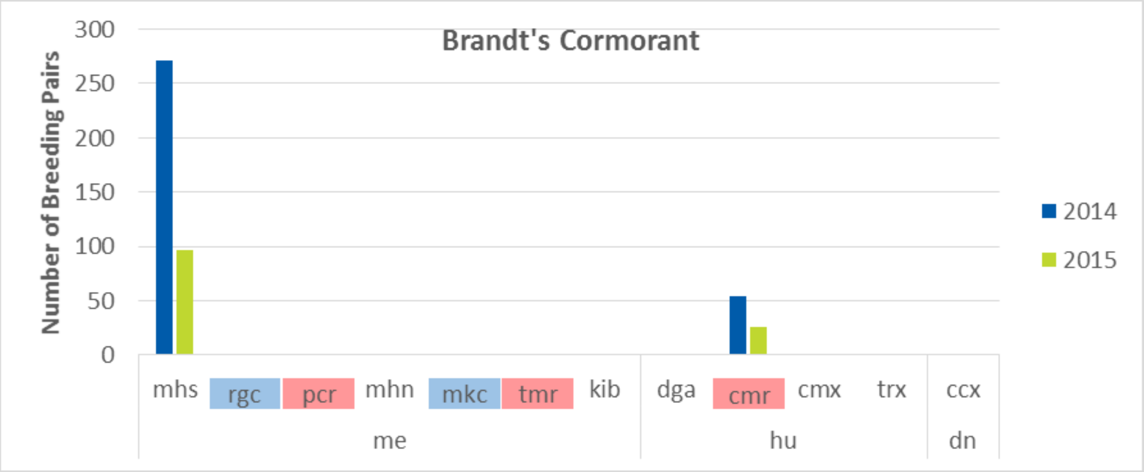

Figure 43. Population size (number of breeding pairs) for Brandt's Cormorants at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs. For area labels, me = Mendocino County, hu = Humboldt County, dn = Del Norte County. For site names, mhs = Mendocino Headlands South, rgc = Russian Gulch SMCA, pcr = Point Cabrillo SMR, mhn = Mendocino Headlands North, mkc = MacKerricher SMCA, tmr = Ten Mile SMR, kib = Kibesillah, dga = Devil's Gate, cmr = South Cape Mendocino SMR, cmx = Cape Mendocino, trx = Trinidad, ccx = Crescent City.

Pelagic Cormorants (Figure 44). We documented a total of 211 breeding pairs in 2014 and 95 breeding pairs in 2015. Of these, we found 1.4% breeding inside SMRs in 2014 and 7.4% breeding inside SMRs in 2015. Pelagic Cormorants bred within the Point Cabrillo SMR in both 2014 and 2015 and within the Ten Mile and South Cape Mendocino SMRs in 2015 only. We found 5.2% of the population breeding within the Russian Gulch SMCA in 2014. Pelagic Cormorants did not breed within the SMCAs we monitored in 2015. The remaining population (93.4% in 2014 and 92.6% in 2015) bred within control sites, with the largest populations at the Cape Mendocino control site in 2014. Breeding pairs appeared evenly distributed among control sites in 2015. As with Brandt's Cormorants, the 2014 population was more than twice the size of the 2015 population. Most of this difference was driven by large decreases at the Cape Mendocino and Trinidad control sites in 2015. There were small between-year differences at all other sites.

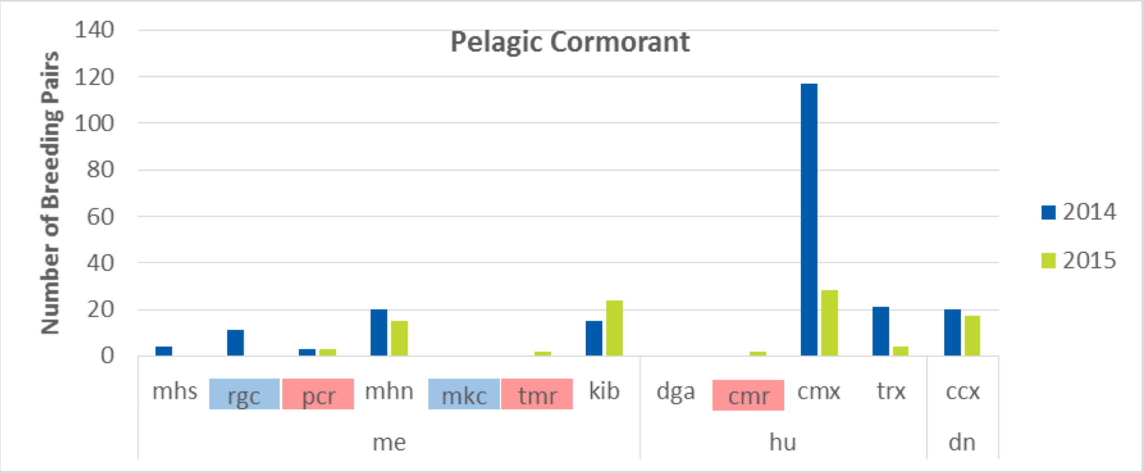

Figure 44. Population size (number of breeding pairs) for Pelagic Cormorants at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

Double-crested Cormorant (Figure 45). We documented a total of 111 breeding pairs in 2014 and 70 breeding pairs in 2015. All breeding pairs were found at two control sites. We found 41.4% of the population at Cape Mendocino and 58.6% at Trinidad in 2014 and 50% at each of the two sites in 2015. The 2014 population was almost twice the size of the 2015 population, with decreases occurring at both control sites in 2015.

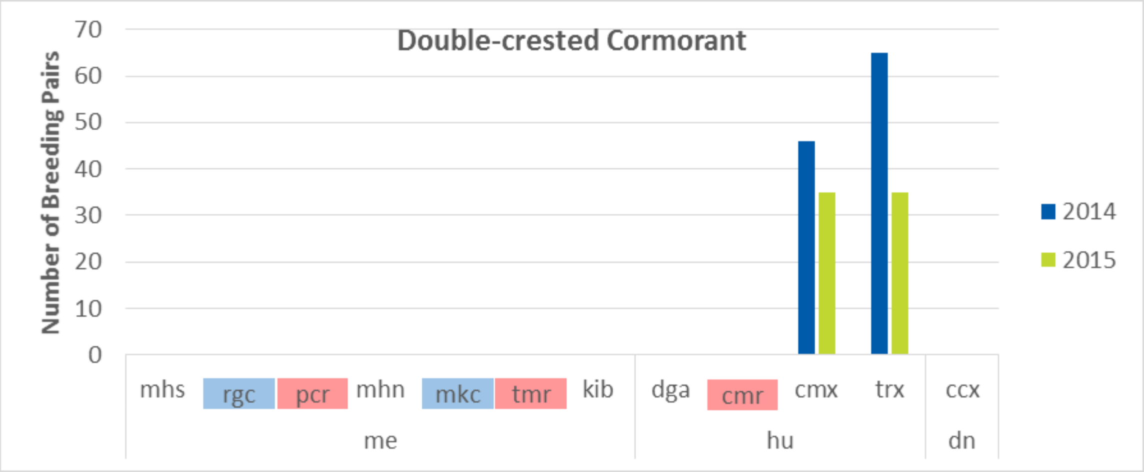

Figure 45. Population size (number of breeding pairs) for Double-crested Cormorants at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

Pigeon Guillemot (Figure 46). Because Pigeon Guillemots breed in inaccessible rock crevices, we were unable to document the number of active nests for this species. Rather, we used raft counts to estimate breeding population size for this species. This is the only species that we observed at all SMR, SMCA, and control sites. We recorded a maximum of 366 guillemots in 2014 and 384 guillemots in 2015. Of these, 15.0% were documented inside SMRs in 2014 and 19.3% inside SMRs in 2015. We documented guillemots inside all three SMRs monitored, with the largest numbers at the South Cape Mendocino SMR. We observed 6.8% of the population inside SMCAs in 2014 and 3.1% inside SMCAs in 2015.

Guillemots appeared evenly distributed among the MacKerricher and Russian Gulch SMCAs, with a slightly larger population at Russian Gulch SMCA in 2014. We observed 78.1% of the population at control sites in 2014 and 77.6% at control sites in 2015. The largest populations were documented at the Cape Mendocino, Trinidad, and Crescent City control sites. There was little difference in total guillemot population size between 2014 and 2015. However, we observed large increases at the South Cape Mendocino SMR and Crescent City control site in 2015 and a large decrease at the Cape Mendocino control site in 2015.

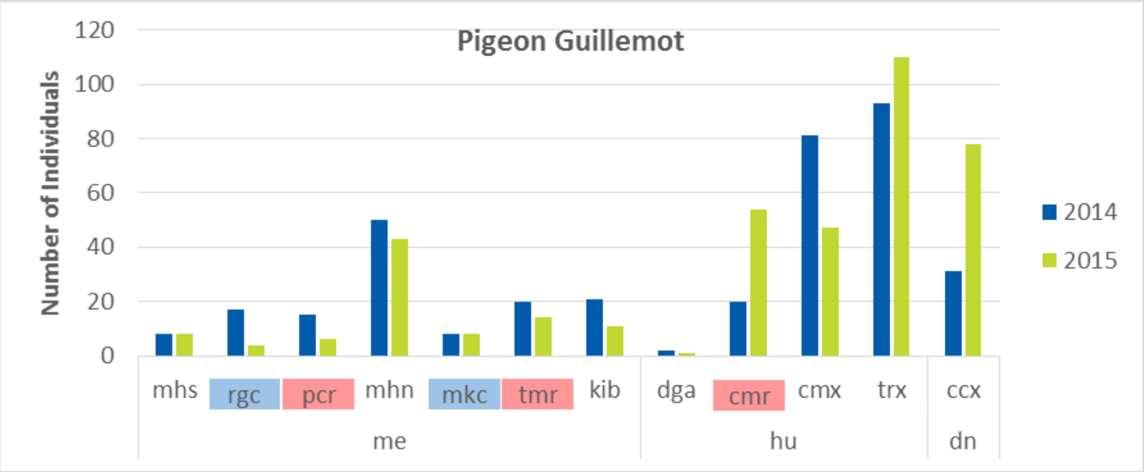

Figure 46. Population size (number of individual breeding birds) for Pigeon Guillemots at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

<u>Western Gulls (</u>Figure 47). We documented a total of 247 breeding pairs in 2014 and 259 breeding pairs in 2015. Of these, 9.3% were found breeding inside SMRs in 2014 and 5.0% were found breeding inside SMRs in 2015. We documented Western Gulls breeding inside the Ten Mile SMR in 2014 and 2015 and inside the Point Cabrillo and South Cape Mendocino SMRs in 2015 only. The remaining population (90.7% in 2014 and 95.0% in 2015) were breeding at control sites, with the majority (54.7% in 2014 and 61.0% in 2015) breeding at the Mendocino Headlands South control site. There were no large differences in breeding population size at any site between 2014 and 2015.

<u>Black Oystercatchers (</u>Figure 48). Due to the cryptic nature of Black Oystercatcher nests, it is difficult to confirm breeding activity in this species, especially if the research location is new to the researchers. We were able to confirm nesting at seven sites. However, we documented oystercatchers consistently using all monitoring sites. Thus, we present both the number of confirmed breeding pairs and the number of birds assumed to be holding territories in Figure 48. We documented a total of 10 breeding pairs in 2014 and 7 breeding pairs in 2015. Of these, 30.0% were found breeding inside SMRs in 2014 and 42.9% were found breeding inside SMRs in 2015. We documented two breeding pairs inside the Point Cabrillo SMR and one breeding pair inside the Ten Mile SMR in both years. We documented one breeding pair in each the Mendocino Headlands South, Mendocino Headlands North, Trinidad, and Crescent City control sites in 2014 and three pairs in 2014 and four pairs in 2015 at the Kibesillah control site.

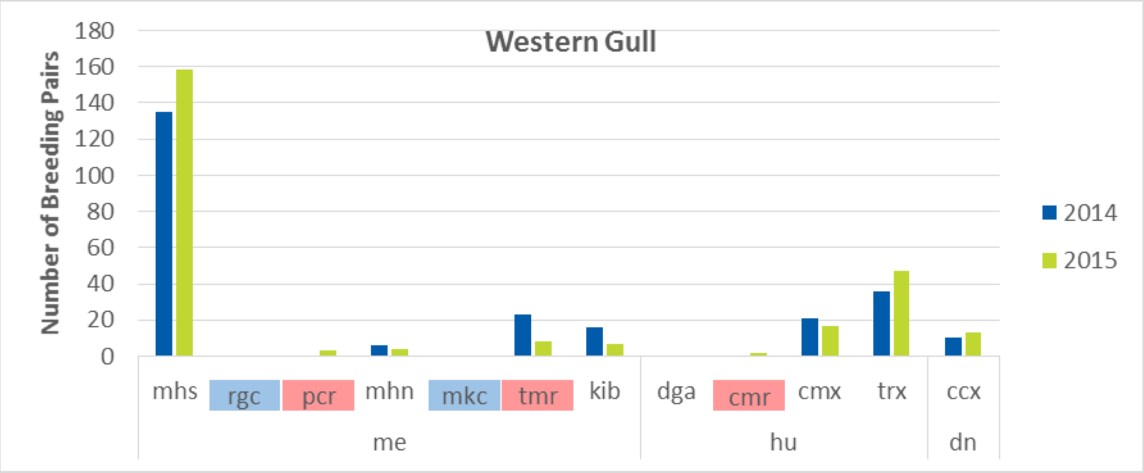

Figure 47. Population size (number of individual breeding birds) for Western Gulls at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

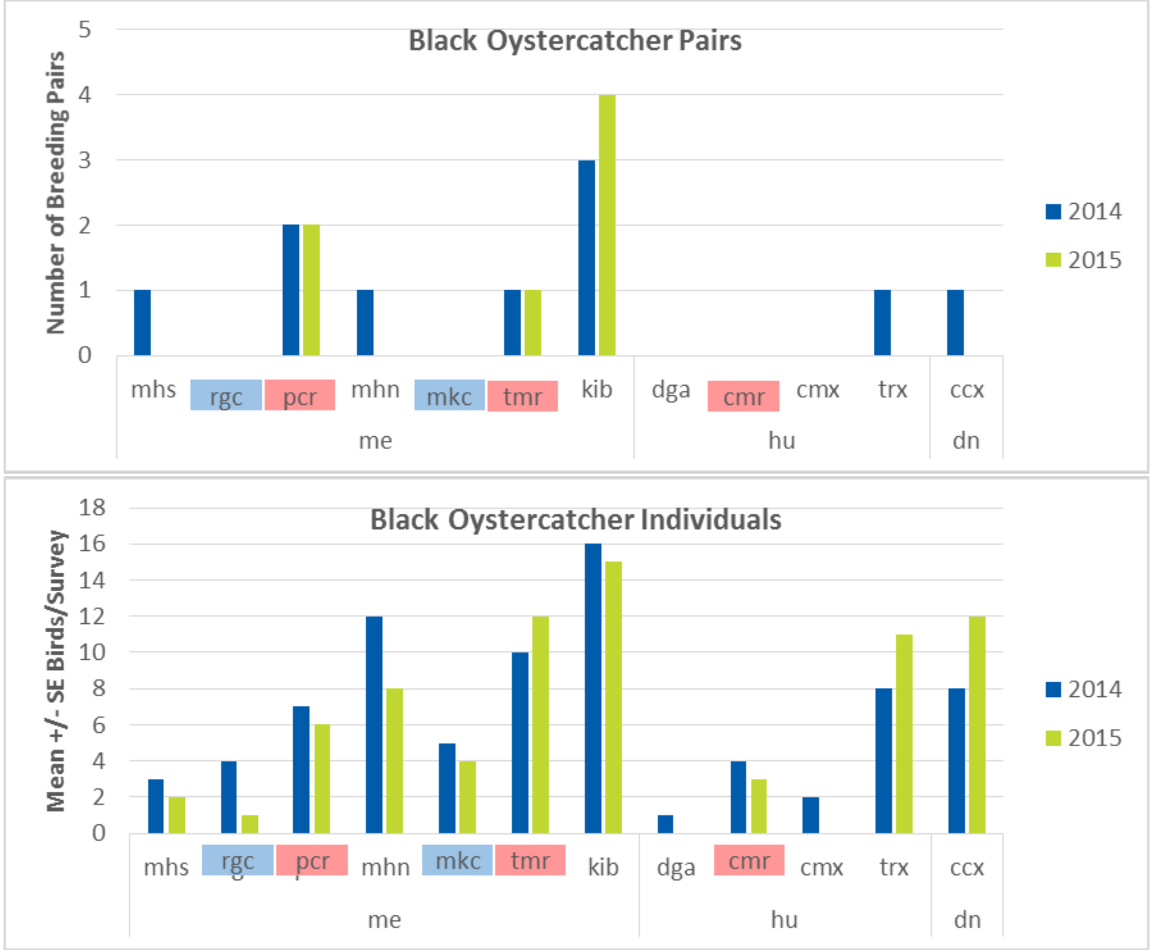

Figure 48. Population size (number of individual breeding pairs; top) and mean +/- SE number of birds observed at potential breeding territories (bottom) for Black Oystercatchers at sites monitored within each of the three NCSR areas for 2014 and 2015.Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

We documented a minimum of 80 oystercatchers on territories in 2014 and 74 oystercatchers on territories in 2015. Of these, 25.3% were observed inside SMRs in 2014 and 28.4% inside SMRs in 2015. The largest numbers of oystercatchers were observed inside the Ten Mile SMR, followed by the Point Cabrillo SMR, then the South Cape Mendocino SMR. We observed 11.3% of oystercatchers inside SMCAs in 2014 and 6.8% inside SMCAs in 2015. We observed similar numbers at the MacKerricher and Russian Gulch SMCAs. The remaining oystercatchers (62.5% in 2014 and 64.9% in 2015) were observed at control sites. The largest numbers of oystercatchers were observed at the Kibesillah, Montecino Headlands North, Trinidad, and Crescent City control sites. Total numbers of oystercatchers were similar between years, though there were large decreases in numbers at Russian Gulch SMCA and the Mendocino Headlands North control in 2015 and large increases at the Trinidad and Crescent City control sites in 2015.

Seabird Breeding Productivity

Figures 49 through 52 show breeding productivity for Brandt's Cormorants, Double-crested Cormorants, Pelagic Cormorants, Western Gulls, and Black Oystercatchers at each of the three areas surveyed (Del Norte, Humboldt, and Mendocino). We were able to follow nests for Pelagic Cormorants, Western Gulls, and Black Oystercatchers at all three areas. Brandt's Cormorants only bred at Humboldt and Mendocino while Double-crested Cormorants only bred at Humboldt.

<u>Brandt's Cormorant and Double-crested Cormorants (</u>Figure 49)

Brandt's cormorants mostly bred on offshore rocks in the NCSR, making nests difficult to follow. We were able to follow four nests at Humboldt in 2014, but were unable to locate any nests that could be monitored at Humboldt in 2015. Fledging success was highest at Mendocino in 2014 with 2.0 fledglings per pair. Fledging success was 1.25 fledglings per pair at Mendocino in 2015. Fledging success at Humboldt in 2014 was 1.25 fledglings per pair.

We followed 28 Double-crested cormorant nests at Humboldt in 2014 and 30 nests at Humboldt in 2015. Fledging success was higher in 2014 (1.43 fledglings per pair) than 2015 (1.07) fledglings per pair.

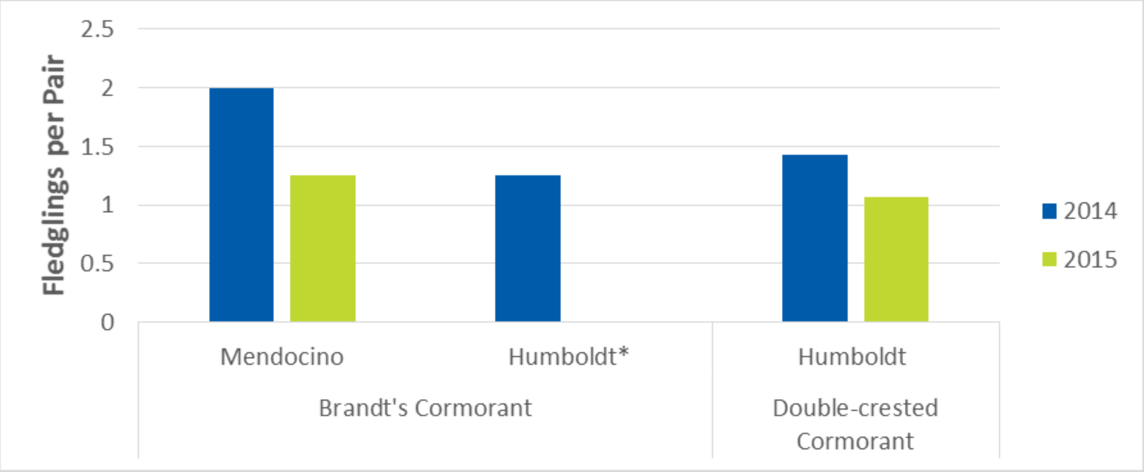

Figure 49. Breeding productivity (fledglings produced per breeding pair) for Brandt's Cormorants breeding at Mendocino and Humboldt and Double-crested Cormorants breeding at Humboldt. *Though Brandt's Cormorants bred at Humboldt in 2015, we were not able to locate any monitorable nests.

Pelagic Cormorant (Figure 50)
We were able to monitor at least 10 Pelagic Cormorant nests at each area and more than 20 nests in most cases. Fledging success was highest in 2014 at all three areas, ranging from 1.49 fledglings per pair at Mendocino to 1.64 fledglings per pair at Del Norte. Fledging success in 2015 ranged from 0.53 fledglings per pair at Humboldt to 1.23 fledglings per pair at Del Norte. Fledging success was highest in both years at Del Norte.

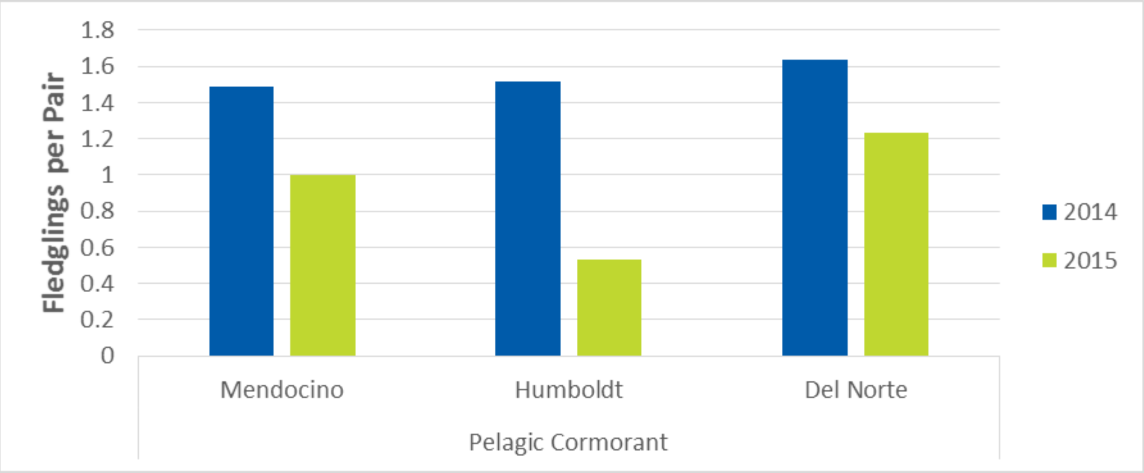

Figure 50. Breeding productivity (fledglings produced per breeding pair) for Pelagic Cormorants breeding at Mendocino, Humboldt, and Del Norte in 2014 and 2015.

Western Gull (Figure 51)
Fledging success was more varied among sites and years than that observed in the cormorants, though samples sizes were also much lower than those for the cormorants. We observed the highest fledging rates at Del Norte in 2014, but did not observe any fledglings at Del Norte in 2015. We did not observe fledglings in Humboldt in 2014 and a fledging rate of 0.5 fledglings per pair in 2015. Mendocino had the largest sample sizes in both year and we calculated a fledging rate of 1.0 fledglings per pair in 2014 and 1.22 fledglings per pair in 2015.

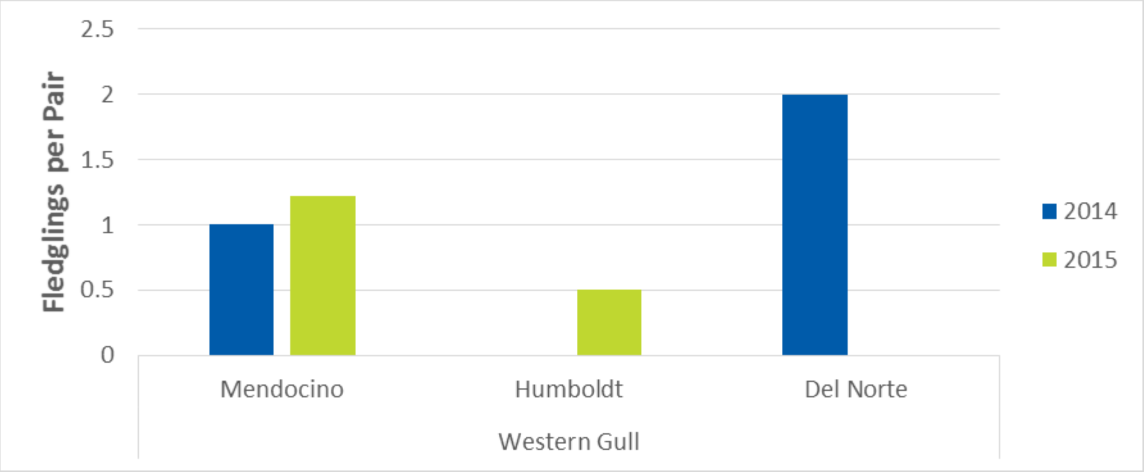

Figure 51. Breeding productivity (fledglings produced per breeding pair) for Western Gulls breeding at Mendocino, Humboldt, and Del Norte in 2014 and 2015.

Black Oystercatcher (Figure 52)
Sample sizes for Black Oystercatchers were very low, though we were able to follow seven nests at Mendocino in each of 2014 and 2015. We followed two nests at Del Norte in 2014 and calculated a fledging rate of 0.5 fledglings per pair. We did not locate any nests to follow at Del Norte in 2015. We followed one nest at Humboldt in each of 2014 and 2015 and did not observe any fledglings. Fledging rates at Mendocino were lower in 2014 (0.14 fledglings per pair) than in 2015 (0.33 fledglings per pair).

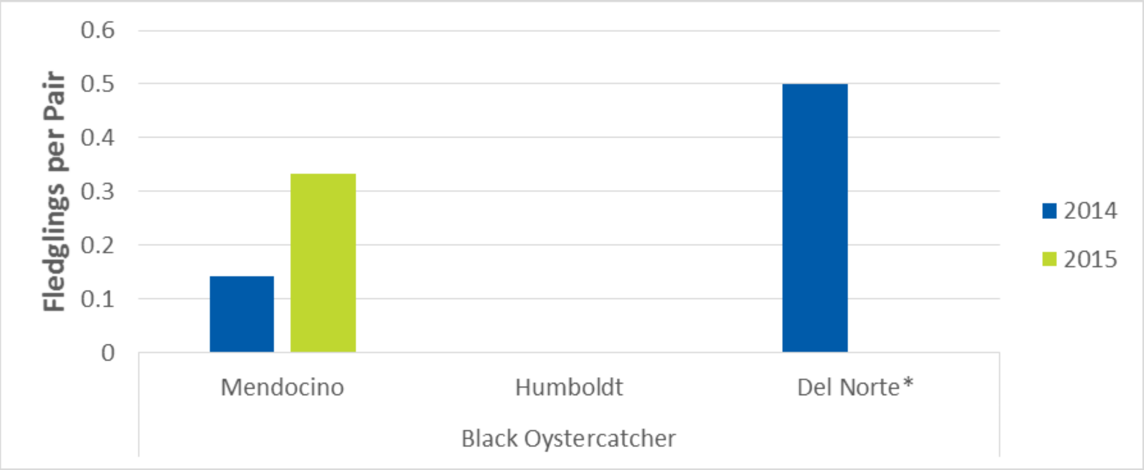

Figure 52. Breeding productivity (fledglings produced per breeding pair) for Black Oystercatchers breeding at Mendocino, Humboldt, and Del Norte in 2014 and 2015. *Black Oystercatchers were not observed breeding at Del Norte in 2015.

Seabird Roost Utilization

We investigated patterns of roost utilization for four species: Brandt's Cormorant, Pelagic Cormorant, Double-crested Cormorant, and California Brown Pelican. Analysis of Variance results for all four species are shown in Table 12. We observed significant differences in numbers of roosting Brandt's Cormorants between years and among sites, but not inside versus outside of MPAs. There was also a significant interaction between year and site. The largest numbers of roosting birds were observed at Mendocino Headlands South in both years (Figure 53). The next largest numbers for both years were observed inside the South Cape Mendocino SMR. There were larger numbers of birds roosting in 2014 for both sites. There were no between year differences for the remaining sites. Patterns for roost utilization were similar for Pelagic and Double-crested Cormorants. There were significant differences between years, among sites, and inside versus outside MPAs. There were significant year x site interactions, but no year x MPA interactions. Mean number of roosting birds was higher at control sites than MPAs for both species (Figure 53). Pelagic Cormorant roosting was more spread out among sites than Double-crested Cormorant roosting. We observed the largest numbers of Pelagic Cormorants roosting at Mendocino Headlands North, Kibesillah, South Cape Mendocino, Trinidad, and Crescent City. Double-crested Cormorant roosting was more concentrated in the north and we observed the largest numbers at South Cape Mendocino, Trinidad, and Crescent City. There were higher numbers of Pelagic and Double-crested cormorants roosting at some sites in 2014 and no difference between years at other sites. There were significant differences among sites and inside versus outside MPAs for Brown Pelicans as well as a significant interaction between year and site. As with Double-crested Cormorants, we observed larger numbers of pelicans roosting outside of MPAs at the northern sites (Figure 53). Roosting numbers were higher in 2014 at Crescent City and higher in 2015 at Trinidad.

Table 12. Results for Analysis of Variance comparing mean number of birds roosting 1) between 2014 and 2015 (Year), 2) inside and outside of MPAs (MPA), and 3) among individual sites (Site), as well as interactions between year and MPA (Year x MPA) and year and site (Year x Site) effects. Degrees of Freedom for each comparison are shown in the column headers.

	Year (df = 1, 325)	MPA (df = 1, 325)	Site (df = 10, 325)	Year x MPA (df = 1, 325)	Year x Site (df = 10, 325)
Brandt's Cormorant	F = 104.44 p <0.001	F = 2.57 p = 0.110	F= 344.03 p <0.001	F =0.01 p = 0.908	F = 58.05 p <0.001
Pelagic Cormorant	F = 11.56 p <0.001	F = 73.64 p <0.001	F = 37.77 p <0.001	F = 0.02 p = 0.885	F = 8.64 p <0.001
Double-crested Cormorant	F = 6.80 p = 0.010	F = 78.94 p <0.001	F = 109.40 p <0.001	F = 2.70 p = 0.101	F = 9.52 p <0.001
Brown Pelican	F = 0.30 p = 0.582	F = 54.53 p <0.001	F = 7.90 p <0.001	F = 1.28 p = 0.240	F = 4.88 p = 0.038

Rates of Human-caused Disturbance

The disturbance rates observed within the NCSR were the lowest we have recorded for all four MLPA study regions (Figure 54). We have observed the lowest disturbance rates in areas where human presence is low (e.g., Santa Cruz Island in the SCSR and Point Reyes in the NCCSR) and the highest disturbance rates where human presence is high (e.g., San Diego in the SCSR and Shell Beach in the CCSR). The disturbance rates observed in the NCSR were similar to those observed at Santa Cruz Island and Point Reyes.

Figure 55 shows disturbance rates for each individual monitoring site in 2014 and 2015. There are no obvious patterns of disturbance within the NCSR. We documented disturbances at six sites, but disturbances were not documented for both study years at any site. We documented disturbances at Point Cabrillo SMR, MacKerricher SMCA, and the Kibesillah control site in 2014 and at Ten Mile SMR and the Trinidad and Mendocino Headlands South control sites in 2015. The highest disturbance rates were recorded at the Trinidad control site. We did not observe any disturbances in the Del Norte area in either year, though the total hours of observation were much lower for this area (Figure 54).

Figure 56 and Figure 57 show the sources of potential disturbance (activities that occurred close enough to breeding and roosting sites but did not cause a reaction) and the sources of actual disturbance. In the Del Norte area, we observed one recreational fishing boat get close enough to a seabird colony to potentially cause a disturbance in 2014, but no disturbance was observed. We did not observe any close approaches at Del Norte in 2015. At Humboldt, a low flying airplane flew close to a colony, but no disturbance was observed. A recreational fishing boat, helicopter, and human with dog were potential sources of disturbance in 2015, with only the recreational fishing boat causing a disturbance. Mendocino had a higher diversity of potential disturbances than the other areas, especially in 2014. Recreational fishing boats came close to seabird colonies in both years, but did not cause disturbances. Humans and dogs caused disturbances in both 2014 and 2015 and a human powered boat (i.e., kayak) caused an additional disturbance in 2015.

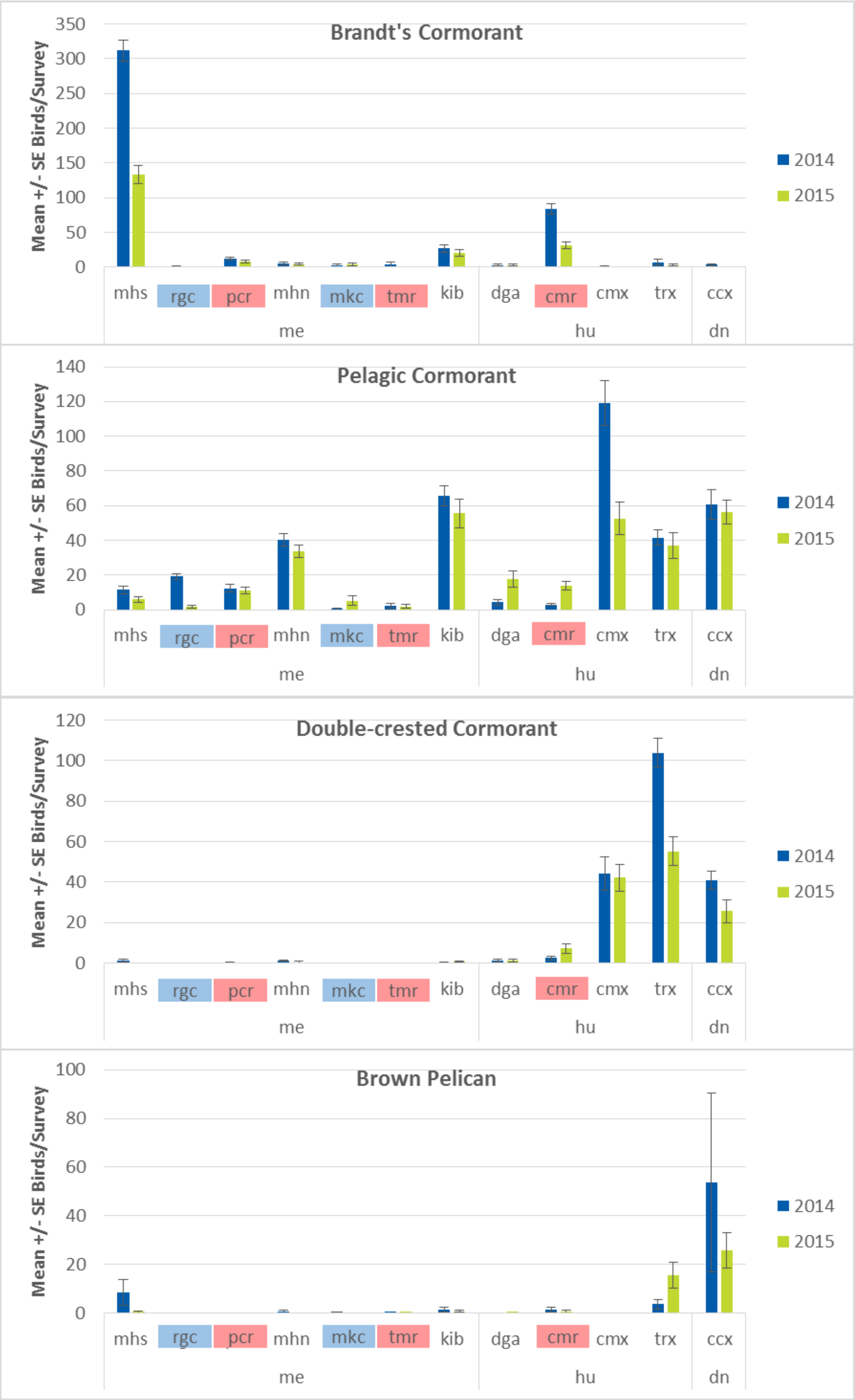

Figure 53. Mean +/- SE number of Brandt’s Cormorants, Pelagic Cormorants, Double-crested Cormorants and Brown Pelicans roosting at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

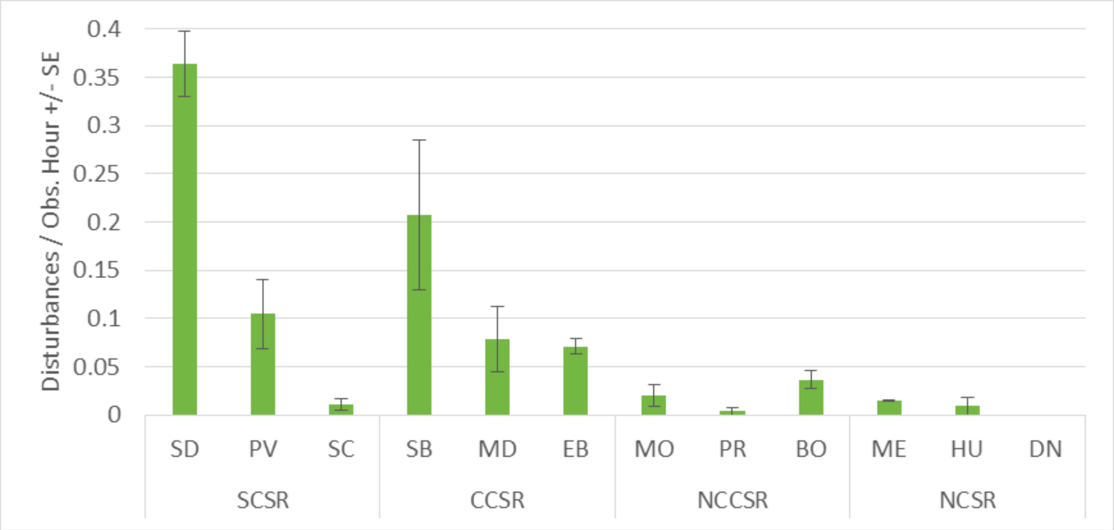

Figure 54. Mean +/- SE number of disturbances observed per hour of observation for areas within each of the four MLPA study regions. SCSR = South Coast Study Region, CCSR = Central Coast Study Region, NCCSR = North Central Coast Study Region, NCSR = North Coast Study Region, SD = San Diego, PV = Palos Verdes Peninsula, SB = Shell Beach, MD = Montaña de Oro, EB = Estero Bluffs, MO = Montara, PR = Point Reyes, BO = Bodega, ME = Mendocino, HU = Humboldt, and DN = Del Norte.

Figure 55. Number of human-caused disturbances observed per hour of observation at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

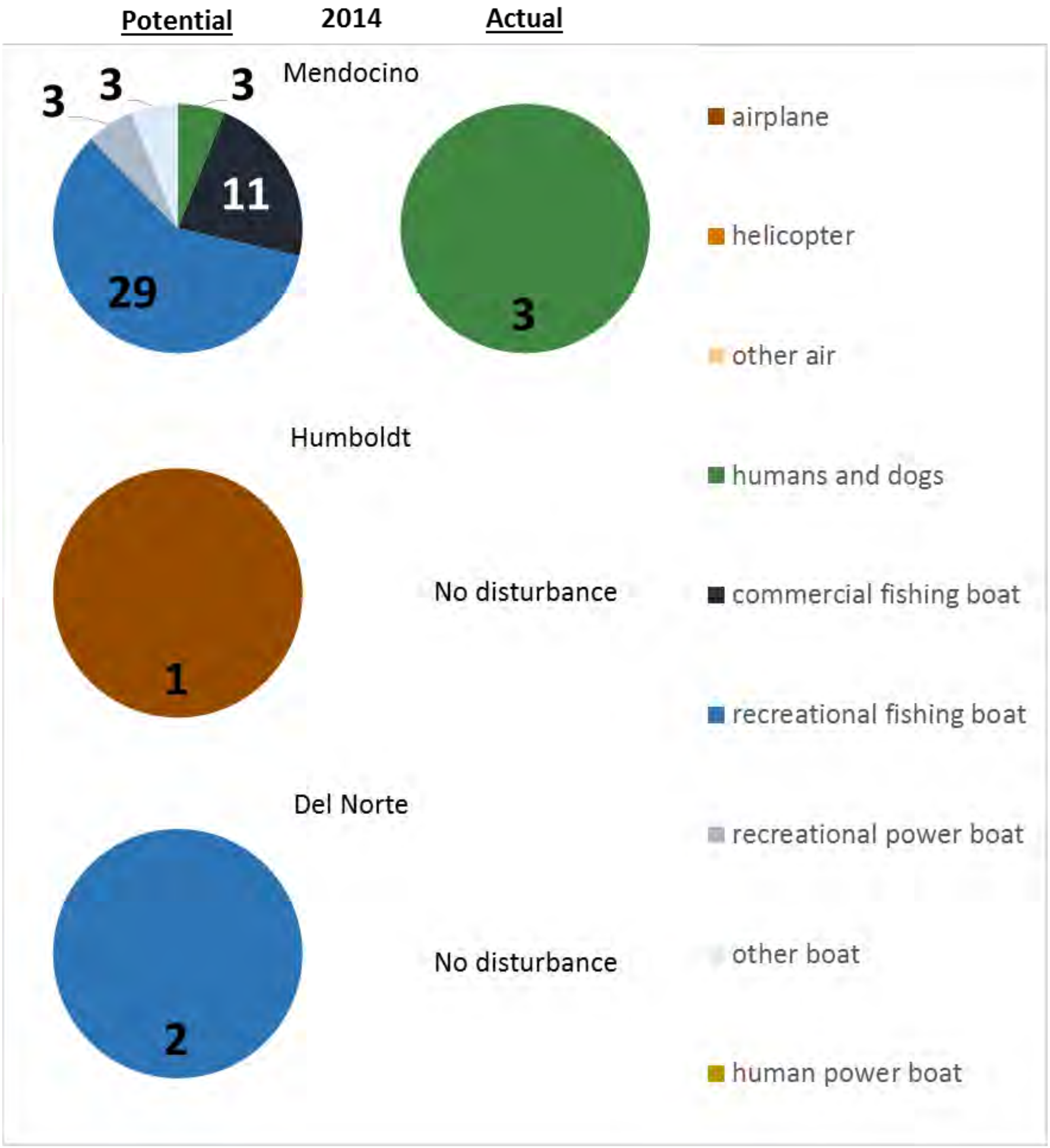

Figure 56. Sources of potential and actual human-caused disturbance at each of the three NCSR areas in 2014.

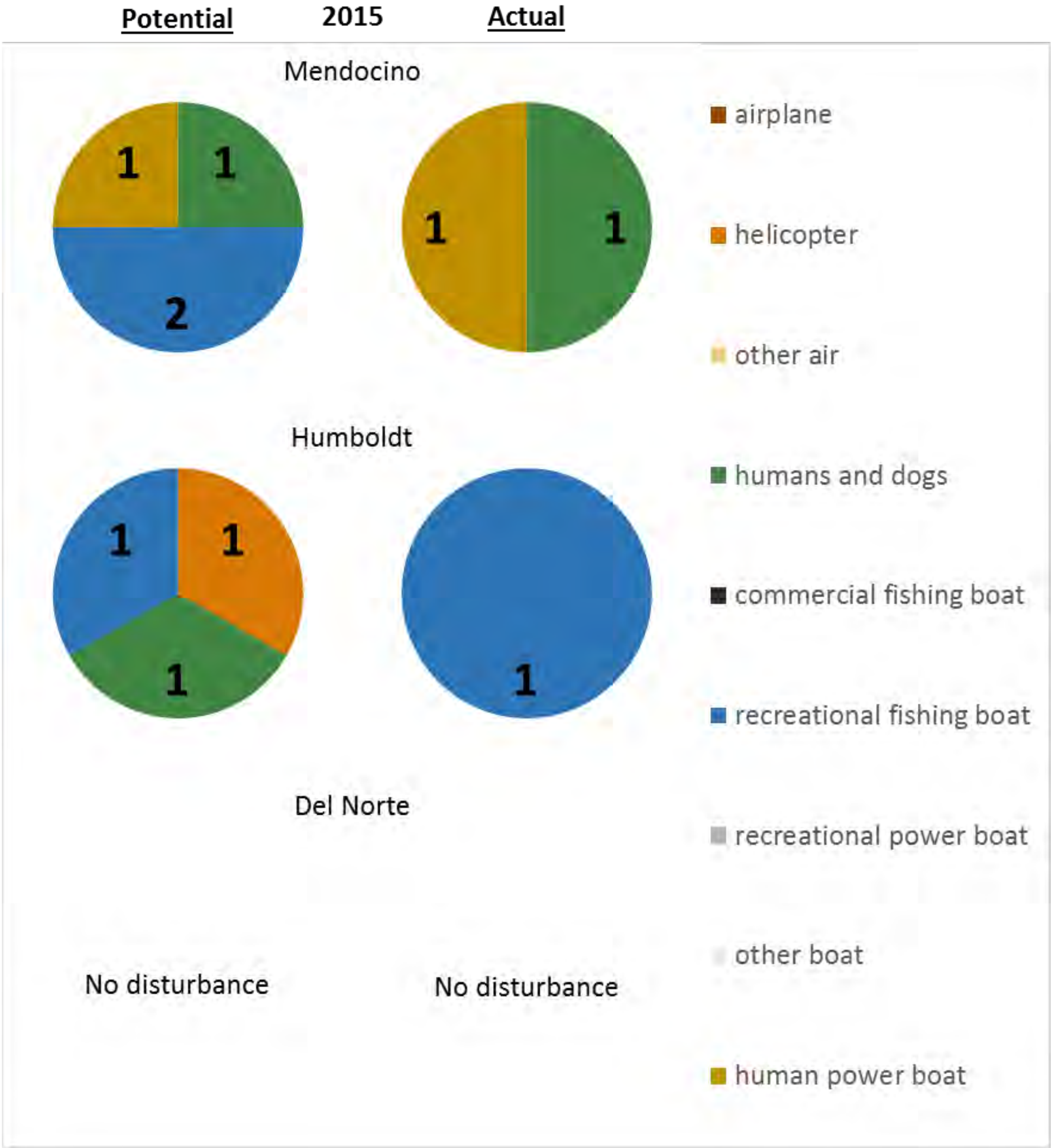

Figure 57. Sources of potential and actual human-caused disturbance at each of the three NCSR areas in 2015.

Seabird Foraging Rates

Individual Seabird Foraging

Table 13 shows Analysis of Variance results for total seabird abundance, species richness, and species diversity (Shannon diversity index (H′)) as well as the six focal species for our nearshore foraging analysis: Brandt's Cormorants, Double-crested Cormorants, Pelagic Cormorants, Pigeon Guillemots, Common Murres, and Marbled Murrelets. Figure 58 shows the mean ± SE abundance, species richness, and species diversity per hour of observation for all foraging survey sites. There were significant differences between years and among sites and significant year x site interactions for all three variables. Abundance, richness, and diversity were highest in 2014 for all sites except Pyramid Point SMCA where values for all three metrics were highest in 2015. Values for all three metrics were highest at the northern sites (Trinidad Bay to Pyramid Point SMCA). There was no significant difference in abundance inside versus outside of MPAs, but significant differences in richness and diversity. There were no significant year x MPA interactions, indicating that differences observed inside and outside of MPAs were consistent across years. Species richness and diversity was higher at control sites than inside MPAs in both years. Much of these differences are being driven by high foraging rates at the Trinidad Bay, Patrick's Point, and Crescent City control sites.

Figure 59 shows mean ± SE number of birds foraging per hour for the three cormorant species. There were significant differences in Brandt's Cormorant foraging rates between years, among sites, and inside versus outside MPAs. There were higher foraging rates at control sites than MPAs and no interaction between MPA and year affects, indicating these results were consistent between years. There were higher foraging rates in 2014 than 2015, especially at the Crescent City control site. Overall, Crescent City showed the highest foraging rates for both years. The Mendocino sites showed the next highest foraging rates while the Humboldt sites showed the lowest foraging rates. There were no significant year or MPA effects for Pelagic Cormorants, but there were significant differences among sites and a significant year x site interaction. Foraging rates were higher in 2014 for all but the Del Norte sites. The Del Norte sites showed higher foraging rates in 2015. Overall, foraging rates were highest at Crescent City and Patrick's Point and lowest at the remaining Humboldt sites. There were no significant year or MPA effects for Double-crested Cormorants, but there were significant differences among sites and a significant year x site interaction. Foraging rates were higher in 2014 for all sites but Pyramid Point SMCA where foraging rates were higher in 2015. Foraging rates were highest at Trinidad Bay and lowest at all sites to the south of Trinidad Bay.

Figure 60 shows mean ± SE number of birds foraging per hour for the three alcid species (Pigeon Guillemots, Common Murre, and Marbled Murrelet). There were significant differences in Pigeon Guillemot foraging rates between years, among sites, and inside versus outside MPAs. There were higher foraging rates outside of MPAs, especially at Trinidad Bay and Patrick's Point; though foraging rates at Ten Mile SMR and Pyramid Point SMCA were the second highest among sites. The Devil's Gate control and South Cape Mendocino SMR sites showed the lowest foraging rates. There was no significant year x MPA interaction, but a significant year x site interaction. Foraging rates were higher in 2014 for Trinidad Bay and Patrick's Point and higher in 2015 for Mendocino Headlands South and Crescent City. There were no differences between years for the remaining sites. There were significant between-year and among-site differences in Common Murre foraging rates, but no difference inside versus outside MPAs. Foraging rates were highest at Crescent City. Foraging rates were also high at Patrick's Point, South Cape Mendocino SMR and Point Cabrillo SMR in 2014. There was a significant year x site interaction with foraging rates higher in 2014 for Mendocino Headlands South, Point Cabrillo SMR, Ten Mile SMR, South Cape Mendocino SMR, and Patrick's Point. Foraging rates were higher in 2015 at Pyramid Point SMCA and there were no differences at the remaining sites. We observed Marbled Murrelets foraging at multiple sites in 2014, but did not observe them foraging at any site in 2015. We therefore did not test for between-year differences. There was no significant difference in foraging rates inside versus outside of MPAs, but there were significant among-site differences. Foraging rates were highest at the northernmost sites (Patrick's Point, Crescent City, and Pyramid Point SMCA), followed by Trinidad Bay. We observed Marbled Murrelets foraging at only two sites south of Trinidad Bay (Kibesillah and Devil's Gate).

Table 13. Results for Analysis of Variance comparing mean number of birds foraging 1) between 2014 and 2015 (Year), 2) inside and outside of MPAs (MPA), and 3) among individual sites (Site), as well as interactions between year and MPA (Year x MPA) and year and site (Year x Site) effects. Degrees of Freedom for each comparison are shown in the column headers. Data from 2014 only were used for Marbled Murrelets. Degrees of Freedom are shown separately for this species only.

	Year (df = 1, 737)	MPA (df = 1, 737)	Site (df = 8, 737)	Year x MPA (df = 1, 737)	Year x Site (df = 8, 737)
Abundance	F = 44.26 p <0.001	F = 2.31 p = 0.129	F = 27.92 p <0.001	F = 0.31 p = 0.580	F_8 = 4.73 p <0.001
Richness	F = 62.47 p <0.001	F = 13.48 p <0.001	F = 23.13 p <0.001	F = 0.48 p = 0.489	F = 7.95 p <0.001
Diversity (H′)	F = 24.78 p <0.001	F = 12.76 p <0.001	F = 15.10 p <0.001	F = 0.29 p = 0.588	F = 5.98 p <0.001
Brandt’s Cormorant	F = 18.82 p <0.001	F = 4.46 p = 0.035	F = 24.67 p <0.001	F = 1.09 p = 0.297	F = 6.12 p <0.001
Pelagic Cormorant	F = 2.30 p = 0.130	F = 1.13 p = 0.288	F = 32.74 p <0.001	F = 3.21 p = 0.074	F = 8.81 p <0.001
Double-crested Cormorant	F = 1.41 p = 0.236	F = 0.02 p = 0.900	F = 48.08 p <0.001	F = 0.03 p = 0.866	F = 3.27 p = 0.001
Pigeon Guillemot	F = 10.02 p = 0.002	F = 7.01 p = 0.008	F = 18.47 p <0.001	F = 1.77 p = 0.184	F = 12.36 p <0.001
Common Murre	F = 7.64 p = 0.006	F = 0.02 p = 0.886	F = 7.80 p <0.001	F = 0.66 p = 0.416	F = 1.94 p = 0.051
Marbled Murrelet	Not Tested	F = 0.00 p = 1.000 (df = 1, 379)	F = 2.55 p = 0.010 (df = 8, 379)	Not Tested	Not Tested

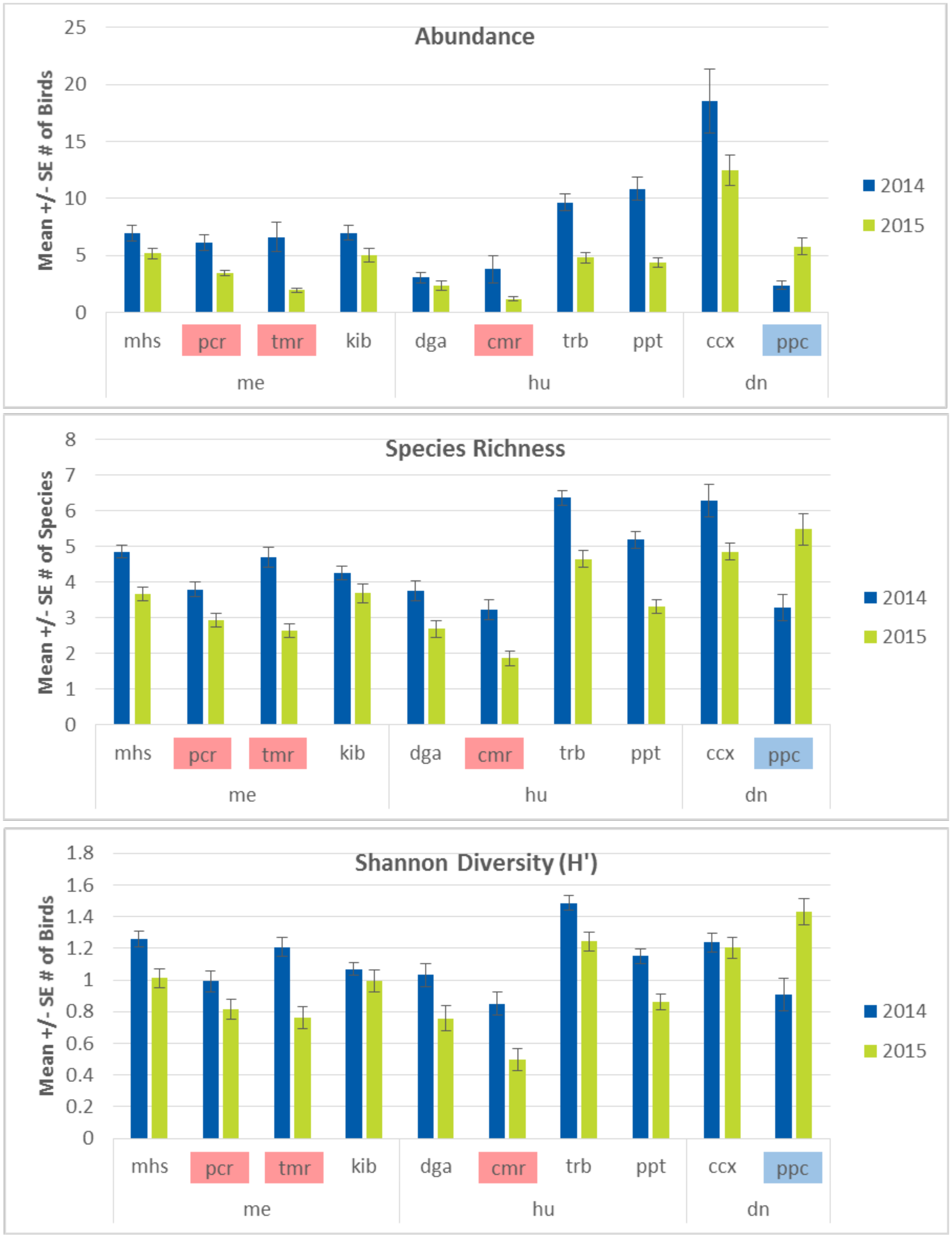

Figure 58. Mean +/- SE abundance (top), species richness (middle), and species diversity (bottom) for nearshore foraging seabirds at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

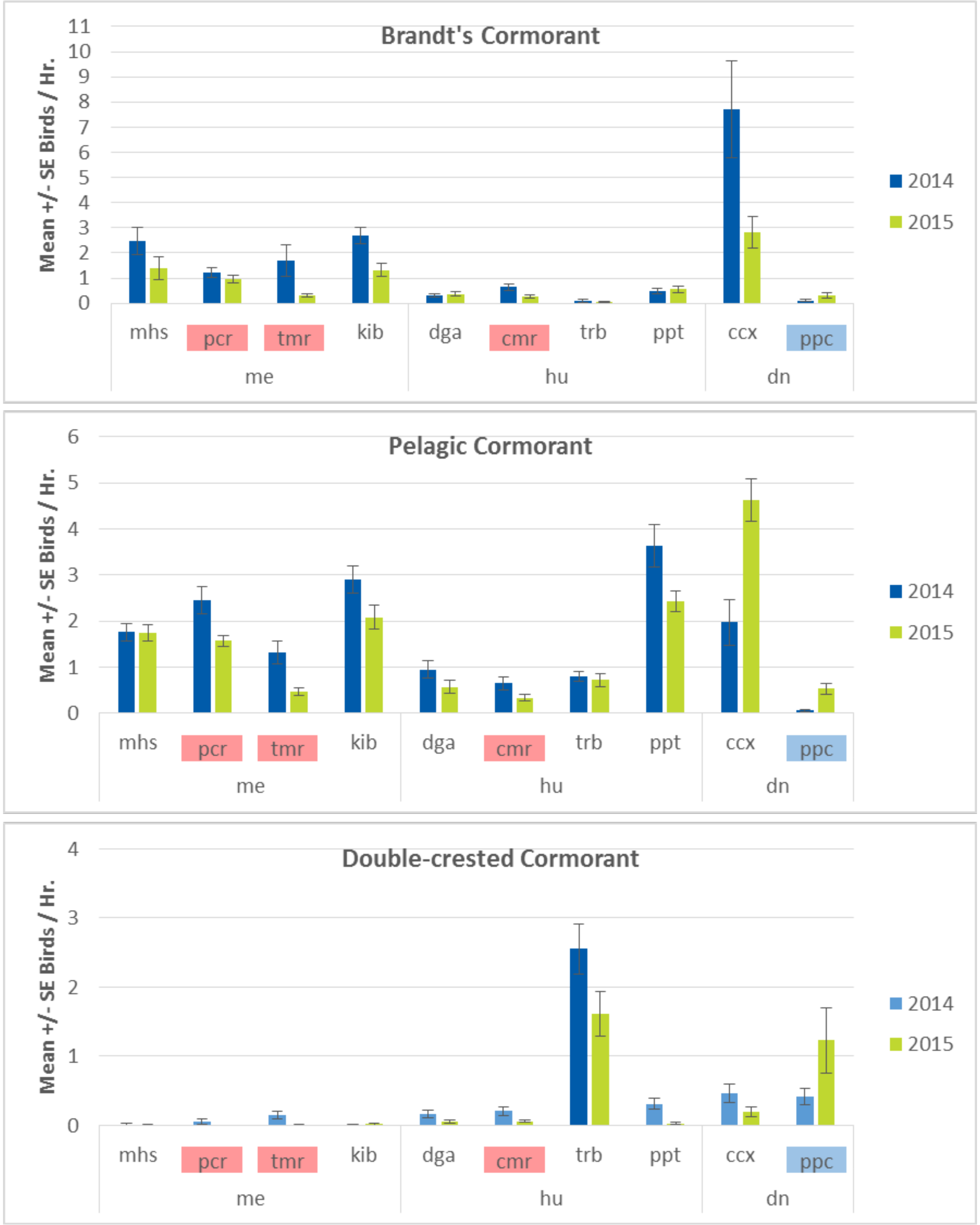

Figure 59. Mean +/- SE number of foraging Brandt's Cormorants (top), Pelagic Cormorants (middle), and Double-crested Cormorants (bottom)per hour of observation at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

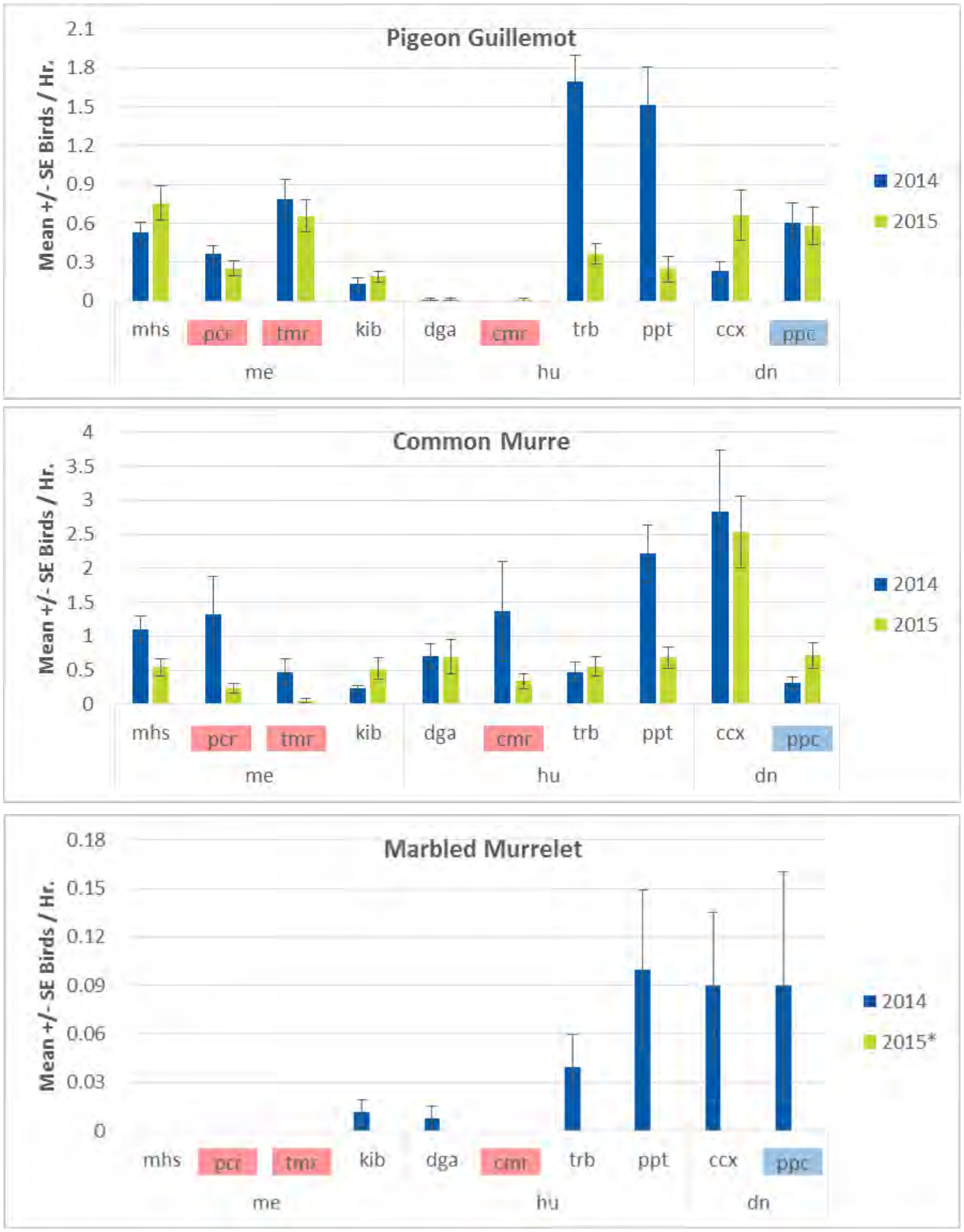

Figure 60. Mean +/- SE number of foraging Pigeon Guillemots (top), Common Murres (middle), and Marbled Murrelets (bottom) per hour of observation at sites monitored within each of the three NCSR areas for 2014 and 2015.Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

Foraging Flocks

We observed very few foraging flocks within the NCSR (Figure 61). We observed a total of three flocks in 2014, one each at Mendocino Headlands South, Devil's Gate, and Crescent City. In contrast, we observed 14 foraging flocks in 2015. All but one were observed at the four northernmost sites, and the majority of these were inside Pyramid Point SMCA. Figure 62 shows the mean number of birds foraging per flock. The largest flock was observed at Crescent City in 2014 and was dominated by Common Murres and Western Gulls. In 2015, Common Murres dominated the largest flocks which were observed at Patrick's Point and Crescent City. Though the Pyramid Point SMCA had the largest number of flocks in 2015, mean flock size was small. The flocks at Pyramid Point SMCA were dominated by Double-crested Cormorants and Western Gulls.

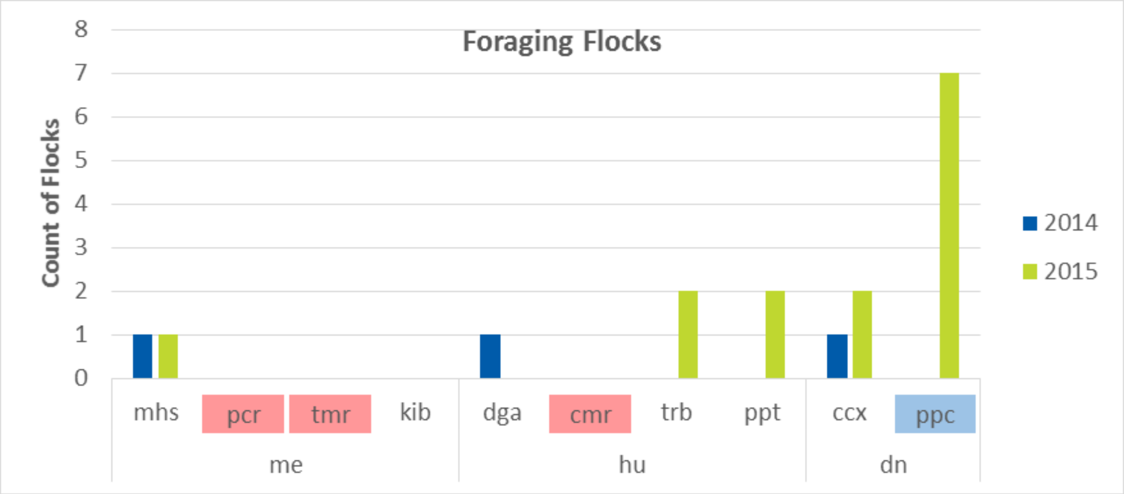

Figure 61. Number of foraging flocks observed at sites monitored within each of the three NCSR areas for 2014 and 2015. Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs.

Baseline Characterization of Seabird Ecology

Our NCSR baseline monitoring built on previously established seabird monitoring programs in the SCSR, CCSR, and NCCSR (McChesney and Robinette 2013, Robinette et al. 2013, Robinette et al. 2014) to provide baseline data on several important parameters for tracking the influence of MPAs in the NCSR over time. With the completion of NCSR baseline monitoring, we now have baselines for all four study regions of the MLPA Initiative. Within the NCSR areas that we investigated, the majority of breeding, roosting, and foraging activity occurred outside of MPAs. However, all of the MPAs that we surveyed are protecting marine birds to some degree. All MPAs but Pyramid Point SMCA protect at least one Black Oystercatcher territory, with some MPAs protecting multiple territories. The Pyramid Point SMCA had moderate foraging rates for Double-crested Cormorants, Pigeon Guillemots, and Marbled Murrelets and had the greatest number of foraging flocks. The South Cape Mendocino SMR protects important Pigeon Guillemot breeding habitat and had moderate foraging rates for Common Murres in 2014. The Ten Mile SMR protects important breeding habitat for Black Oystercatchers and small breeding populations of Pigeon Guillemots and Western Gulls. It also had moderate Pigeon Guillemot foraging rates. The MacKerricher SMCA protects important Black Oystercatcher breeding habitat and a small Pigeon Guillemot breeding population. The Point Cabrillo SMR protects important breeding habitat for Black Oystercatchers and small Pelagic Cormorant and Pigeon Guillemot populations and showed moderate foraging rates for Pelagic Cormorants and Common Murres. Finally, the Russian Gulch SMCA protects small populations of Pigeon Guillemots and Pelagic Cormorants.

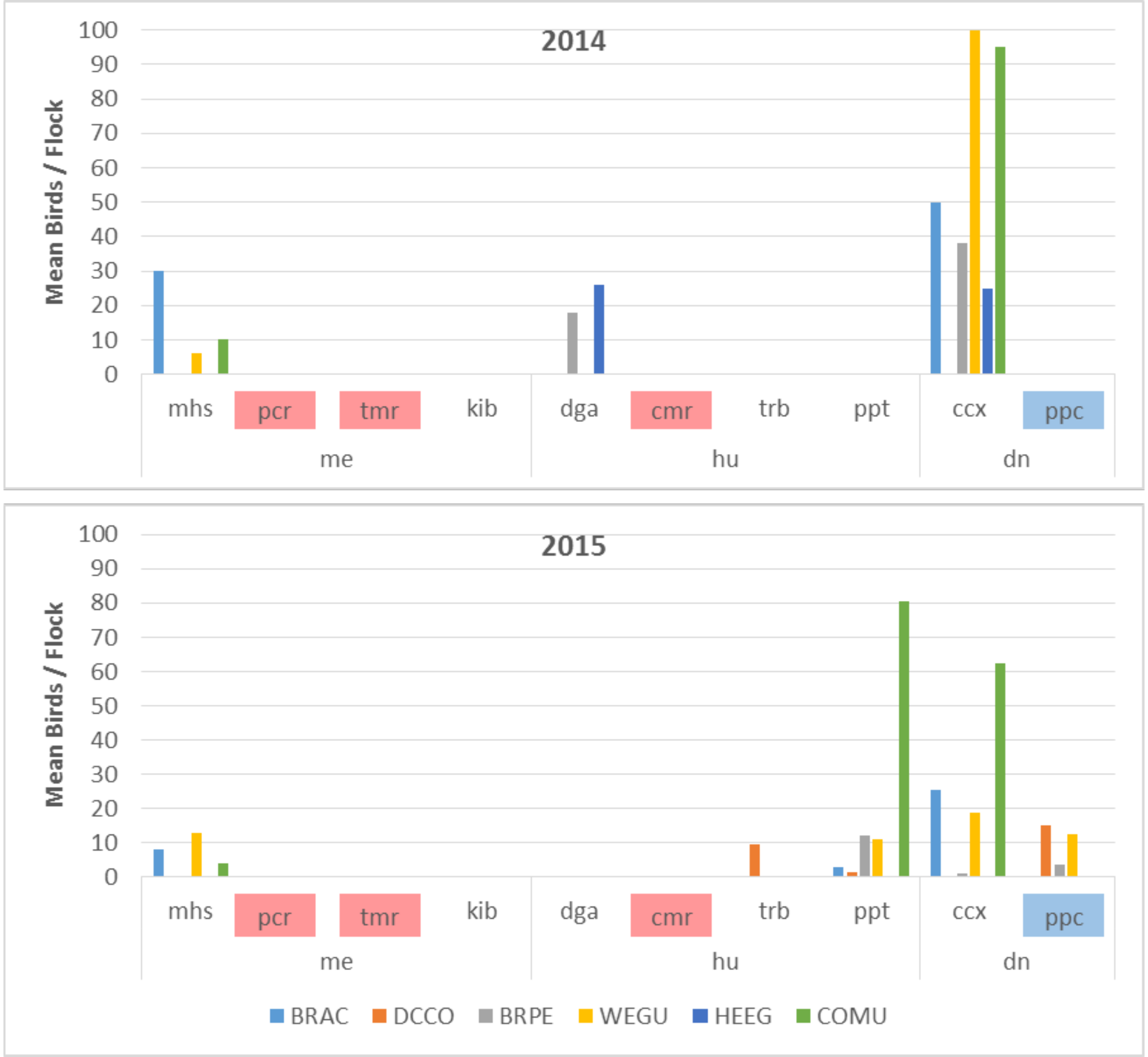

Figure 62. Mean number of birds per foraging flock for the six most abundant species observed at sites monitored within each of the three NCSR areas for 2014 (top) and 2015 (bottom).Red sites are SMRs, blue sites are SMCAs, and white sites are outside MPAs. BRAC = Brandt's Cormorant, DCCO = Double-crested Cormorant, BRPE = Brown Pelican, WEGU = Western Gull, HEEG = Heerman's Gull, and COMU = Common Murre.

While MPAs will have the greatest direct impacts on species targeted by fisheries, seabirds will receive indirect benefits as the abundance of their prey species increases; and this should hold true regardless of whether a given prey species is targeted by fisheries. There is broad consensus among marine scientists that MPAs have community-wide impacts inside and adjacent to their boundaries (Lubchenco et al. 2003). In fact, many studies have shown reserves to increase mean body size, density and biomass of both targeted and non-targeted species (Guénette and Pitcher 1999). We expect MPA-related changes in prey abundance to translate into measurable responses initially in seabird metrics like foraging rates and diet (not investigated as part of the coastal monitoring, but see Chapter 2) and eventually in metrics like breeding productivity and population size. In general, breeding productivity and population size will respond to long-term changes occurring within the NCSR, while variability in diet and foraging rates will respond to short-term, localized changes in prey abundance and distribution. Robinette et al. (2012) showed persistent use of an upwelling retention area by nearshore foraging seabirds over a six-year period, though foraging rates for each species varied among years. A preliminary analysis of more recent data from the same area shows that foraging rates are highest at the retention area during years of persistent upwelling and that seabirds spread out to more sites during years of variable upwelling (Robinette unpublished data). Furthermore, Robinette et al. (In Review) compared the distributions of juvenile kelp forest fishes to seabird foraging rates and found correlations at local (15-30 km) and regional spatial scales. Like our NCSR focal species, the seabirds in the Robinette et al. studies preyed heavily on the juveniles of subtidal fish species and foraging rates were likely responding to variability in the recruitment rates of these fishes. These results showed that seabirds will not only respond to variability in prey abundance and distribution, but will benefit from the types of localized increases in prey abundance predicted to occur within MPAs.

The direct benefits that seabirds will receive from the NCSR MPAs will likely be in the form of decreased human-caused disturbance to breeding colonies. While seabird populations are primarily regulated from the bottom up (see Ainley et al. 1995), top-down regulation does occur and is often exacerbated by human activities that disturb breeding and resting sites. Compared to the other MLPA study regions, rates of human-caused disturbance in the NCSR were low. Furthermore, we did not observe any persistent patterns of disturbance between years that would indicate disturbance is a chronic issue for the NCSR. If disturbance rates remain low, we should expect to see greater and clearer responses of seabird metrics to the benefits of increased prey abundance. The NCSR MPAs and special closures should help maintain low levels of human-caused disturbance. Additionally, a new chapter of the Seabird Protection Network (SPN) is currently being established in the Trinidad area. The SPN uses monitoring data to guide outreach efforts to educate coastal ocean user groups about the impacts of human caused disturbance (see http://www.seabirdprotectionnetwork.org). Targeted outreach by other SPN chapters has proven effective in reducing rates of human-caused disturbance (see Robinette et al. 2013).

Baseline breeding productivity for our nest monitoring focal species (Brandt’s Cormorants, Pelagic Cormorants, Western Gulls, and Black Oystercatchers) varied between years and among species. When compared to long-term means reported in Robinette and Howar (2016) and Warzybok et al. (2015), productivity for Brandt’s and Pelagic Cormorants was average to above average during 2014 and below average during 2015. Conversely, productivity for Western Gulls and Black Oystercatchers was below average in 2014 and average to above average in 2015. Furthermore, trends for all species were similar to those reported in Robinette and Howar (2016) and Warzybok et al. (2015), with productivity decreasing from 2014 to 2015 for the two cormorant species and increasing from 2014 to 2015 for Western Gulls and Black Oystercatchers. The NCSR baseline period was somewhat unique oceanographically. Beginning in late 2013, the California Current System appeared to be transitioning to a warmer, less productive state (Leising et al. 2015). Additionally, a marine heat wave (also referred to as the ‘Warm Water Blob’) developed in the Gulf of Alaska in late 2013 and persisted into 2015 when a strong El Niño began to develop (McClatchie et al. 2016). Despite these developing conditions, the NCSR was overall productive in 2014. Upwelling intensity was high during the 2014 spring and summer, resulted in high juvenile rockfish abundance (a preferred prey for cormorants). However, juvenile rockfish abundance was low by the 2015 spring and summer and likely contributed to the decreased breeding productivity observed in the cormorants. Conversely, Black Oystercatchers forage solely within intertidal habitats and diet results from Robinette and Howar (2013) show that coastally breeding Western Gulls forage heavily within this habitat as well. Thus, Black Oystercatchers and Western Gulls are likely responding to different oceanographic signals than the cormorants. If the transition of the California Current System to a warm state persists, then the initial years of MPA establishment within the NCSR will experience low oceanographic productivity and biological responses to MPA establishment will likely be slow.

Initial Changes Within the NCSR

We did not to expect to observe changes in the parameters we measured within our two-year baseline study period. While there is short-term variability in all the parameters we measured, changes due to MPA implementation will happen over longer periods of time. For example, changes in breeding productivity will respond to variability in ocean productivity over the short term and to MPA establishment over the long term as adult fish stocks, and thus spawning biomass, are built up within protected areas. Similarly, breeding populations may initially respond to reduced disturbance rates to breeding colonies, but more sustained population growth will happen as fish stocks are replenished. However, it is possible for rates of human-caused disturbance to show short-term responses to MPA implementation, especially if targeted outreach efforts are a component of MPA implementation. In the NCCSR, there was a measurable decrease in boat disturbances inside the Egg (Devil's Slide) Rock to Devil's Slide Special Closure (McChesney and Robinette 2013). Baseline disturbance rates within the NCSR were low and results from the NCCSR provide encouraging evidence that rates of human-caused disturbance will remain low with the added MPA and special closure protections.

Behavioral parameters for seabirds like foraging rates and distribution may also show short-term responses to MPA implementation. In South Africa, Pichegru et al. (2012) illustrated how a fishing closure can have immediate impacts on African Penguin (*Spheniscus demersus*) foraging behavior. They found more penguins foraging inside the closed area and an overall decrease in foraging effort by breeding penguins. Similarly, Bertrand et al. (2012) showed seabird behavioral responses to intense localized fishing effort in the Humboldt Current System off Peru. Intense fishing created regional depletion, taking 100 times more than the requirement of breeding seabirds over the same period. With the onset of fishing, breeding seabirds increased their range of daily trips and depths of dives. The more the fishery depleted local prey abundance, the farther the breeding seabirds needed to forage from the colony to obtain food. One main difference between our study and those of Pichegru et al. and Bertrand et al. is that birds in those studies were competing directly with fisheries for prey, whereas the birds we are monitoring are consuming the juvenile age classes of fished species. While we expect our focal species to benefit from decreased fishing inside MPAs, the response will take longer as fishing within the NCSR does not directly target seabird prey.

Seabirds as Indicators of Ecosystem Condition

Seabirds have proven to be reliable indicators of change in the marine environment. Seabirds are highly visible and easily enumerated and dietary information can be obtained for many species when conditions allow. Several studies conducted over the past 30 years have shown that seabirds respond predictably to changes in prey abundance and can thus be used as reliable indicators of change in prey populations (see Cairns 1992, Hatch and Sanger 1992). Changes in a variety of seabird demographics and foraging parameters have been successfully used to, among other things, detect changes in prey abundance on several temporal and spatial scales (e.g., Montevecchi and Myers 1995), changes in prey age-class structure (e.g., Sunada et al. 1981, Davoren and Montevecchi 2003), responses of prey populations to climate change (e.g., Miller and Sydeman 2004), and changes in local food-web structure (e.g., Montevecchi and Myers 1996). Thus, studies of seabird ecology can provide timely and important information on local oceanography and marine ecosystem structure that would otherwise be difficult and expensive to obtain. Monitoring seabird ecology can contribute to MPA management in two ways: 1) tracking variability in regional oceanographic conditions and 2) indexing temporal and spatial variability of fish recruitment to nearshore habitats.

The recovery rate of populations released from fishing pressure will be largely determined by the degree to which new individuals recruit to MPAs (Warner and Cowen 2002). Juvenile recruitment in marine organisms is largely dependent on both biophysical processes such as upwelling and the life history strategies of the organisms being considered (Caley et al. 1996). For species with pelagic larval stages, recruitment will be largely dependent on 1) the number of larvae produced in a given year, 2) the survival of those larvae to settlement age, and 3) delivery of those larvae to adult habitat (Jenkins and Black 1994, Levin 1996, Wing et al. 1995a). The first two conditions are greatly affected by regional oceanographic conditions while the third condition is greatly affected by nearshore ocean currents. Robinette et al. (2007) investigated sanddab (Citharichthys spp.) recruitment around the Vandenberg SMR and illustrated how seabird diet can be integrated with estimates of regional larval abundance and upwelling to investigate spatial and temporal variability in recruitment. They found that regional larval sanddab abundance was highest when upwelling was persistent. They also showed that recruitment of sanndabs differed on opposing sides of a coastal promontory, with leeward recruitment strongest during persistent seasonal upwelling and windward recruitment strongest during variable upwelling. Dispersal patterns of planktonic larvae are often influenced by the phasing and amplitude of coastal upwelling, showing offshore transport during periods of persistent upwelling and onshore transport during periods of relaxation (Sakuma and Larson 1995, Sakuma and Ralston 1995, Wing et al. 1995a). However, many studies have provided evidence that localized retention areas prevent the offshore transport off planktonic larvae (Wing et al. 1995b, 1998, Graham and Largier 1997, Mace and Morgan 2006a,b). These studies have found persistent, predictable retention areas in the lee of coastal promontories central California. Robinette et al. (2012) investigated the foraging distribution of multiple seabird species around the Vandenberg SMR and showed that foraging distributions were consistent over a six-year period. Seabirds that feed on juvenile fishes foraged mostly in the lee of the coastal promontory where Robinette et al. (2007) showed fish recruitment should be highest. Together, these studies suggest that the geographic location of an MPA will influence the rate of juvenile recruitment and thus the rate of population and community-level change within MPA boundaries. Furthermore, seabirds can play an important role in identifying areas of high juvenile fish recruitment and tracking variability in recruitment through time.

The success of MPA management will be determined by managers’ ability to 1) understand MPA effectiveness and 2) adapt to shortfalls in MPA performance. Both of these will require an understanding of the mechanisms causing change within MPAs. We propose that the best way to understand these mechanisms is to take a two-pronged approach, looking at 1) broad-scale oceanographic conditions to understand variability in regional primary and secondary productivity and 2) fine scale tracking of how regional primary and secondary productivity is delivered to MPAs and areas outside MPA boundaries. Seabirds can provide information for both of these approaches. Monitoring seabird breeding population sizes and reproductive success can complement indices of ocean climate to track interannual variability in ocean productivity while monitoring seabird diet and foraging can provide information on temporal and spatial variability in fish recruitment. Understanding and tracking both of these mechanisms will allow managers to set realistic expectations for how quickly change should occur within individual MPAs and the SCSR as a whole.

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CHAPTER 5: CITIZEN SCIENCE CROSS-VALIDATION

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INTRODUCTION

Implementation of citizen science in natural resource monitoring programs provides several opportunities while simultaneously posing several challenges. Natural resource monitoring programs, especially those involved in management, can benefit from increased public engagement and investment in monitoring and management, as well as increased spatial and temporal extent of sampling (Haywood et al. 2010). Challenges posed include sustaining involvement in citizen science programs over time, development of protocols that can be implemented by available participants ("citizen scientists"), and data quality (Cohn 2008). Cross-validation of citizen science monitoring data is an important step towards implementing citizen science in natural resource monitoring (Dickinson et al. 2010).

California's network of marine protected areas (MPAs) is spatially extensive, constituting 139 individual reserves and covering approximately 1300 km of the state's coastline (Botsford et al. 2014). Many of California's marine protected areas are logistically difficult to access, particularly in the North Coast Study Region (NCSR), which has relatively little road development resulting in poor access via shore, and generally difficult conditions for small boats (i.e. inclement weather including rough seas, wind, and poor visibility). Questions regarding how to sustain monitoring the management of California's Marine Protected Areas over the long term, as well as how to increase public investment in the success of California's MPAs given the difficulty of enforcement of such a wide-ranging network, have naturally led towards proposal or implementation of citizen science for a diversity of marine natural resources (Sayce et al. 2013). Seabirds are a natural target of a potential citizen science monitoring program given their observability relative to other potential targets of monitoring (Ward et al. 2015), and their status as apex predators (Einoder 2009). Seabird abundance and distribution are thought to reflect changing abundance in prey (Einoder 2009), and seabird nesting and roosting use of nearshore rocky habitats is often responsive to anthropogenic disturbance regimen (Durant et al. 2009). However, cross-validation of seabird monitoring data collected by citizen scientists with data collected by professional scientists has rarely been a part of protocol development or implementation.

We cross-validated citizen science data on seabird abundance and distribution in the Trinidad Head and Patrick's Point, Humboldt County, California (Figure 63). We collected citizen science shore-based monitoring data in collaboration with local volunteer groups during the 2016 field season. We developed a simple survey-based citizen science program with a flexible and easy-to-replicate sampling protocol, and conducted outreach to three local interest groups (Redwood Region Audubon Society, California North Coast Chapter of The Wildlife Society, and the Trinidad Gateway to the California Coastal National Monument). We simultaneously collected data with a trained professional biologist at the same study site to allow comparison and cross-validation of the results, and report on that cross-validation here. These protocols are currently being adopted by (and potentially refined by, on the basis of this analysis) the new Trinidad Seabird Protection Network Chapter coordinated by the Bureau of Land Management, beginning in 2017.

METHODS

Biologist Data

Transect monitoring data were collected by a single trained observer following the transect protocol described in Element 3 (Shore-based Monitoring) during 13 April – 30 Aug 2016, at the sites abbreviated in that element as "trx" for "Trinidad Transect."

Citizen Scientist Data

We developed a simple survey protocol for citizen scientist participants based on a photograph of the sampling area, onto which participants would record the numbers of birds and nests present on particular rocks (Figure 64). Nine sampling locations were designated throughout the Trinidad Head and Patrick's Point region (Figure 63). Observers also recorded disturbance or potential disturbance events observed.

No temporal sampling structure was imposed on the protocol (other than conducting surveys during daylight hours) but spatial sampling structure was strong given the nature of the data recording method and protocol (Figure 64). Outreach presentations by DCB in 2015 and 2016 through Redwood Region Audubon Society, California North Coast Chapter of the Wildlife Society, Humboldt State University, and the Trinidad Gateway to the California Coastal National Monument resulted in initial contact with a maximum of 110 individuals (there may have been some overlap in initial contacts). Two training sessions conducted by DCB and SEM on-site at Trinidad Bay in April and May 2016 resulted in contact with 17 potential observers. 39 surveys with complete data were ultimately collected by 7 participants in the field (range per observer was 1 to 18). An additional set of citizen scientist surveys (n=4) was not analyzed here because the observations of seabirds were not identified to species (although these may still be useful data).

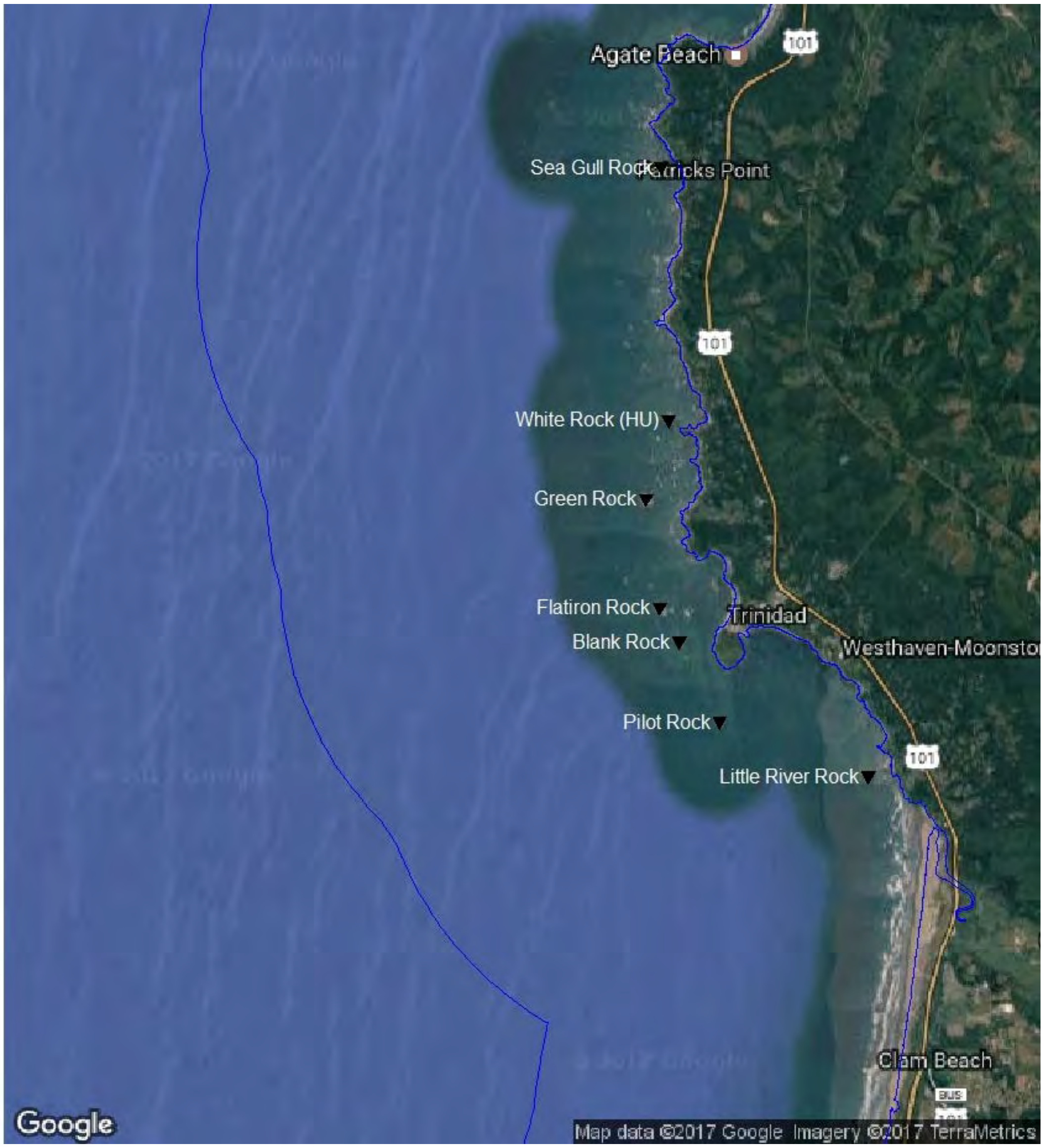
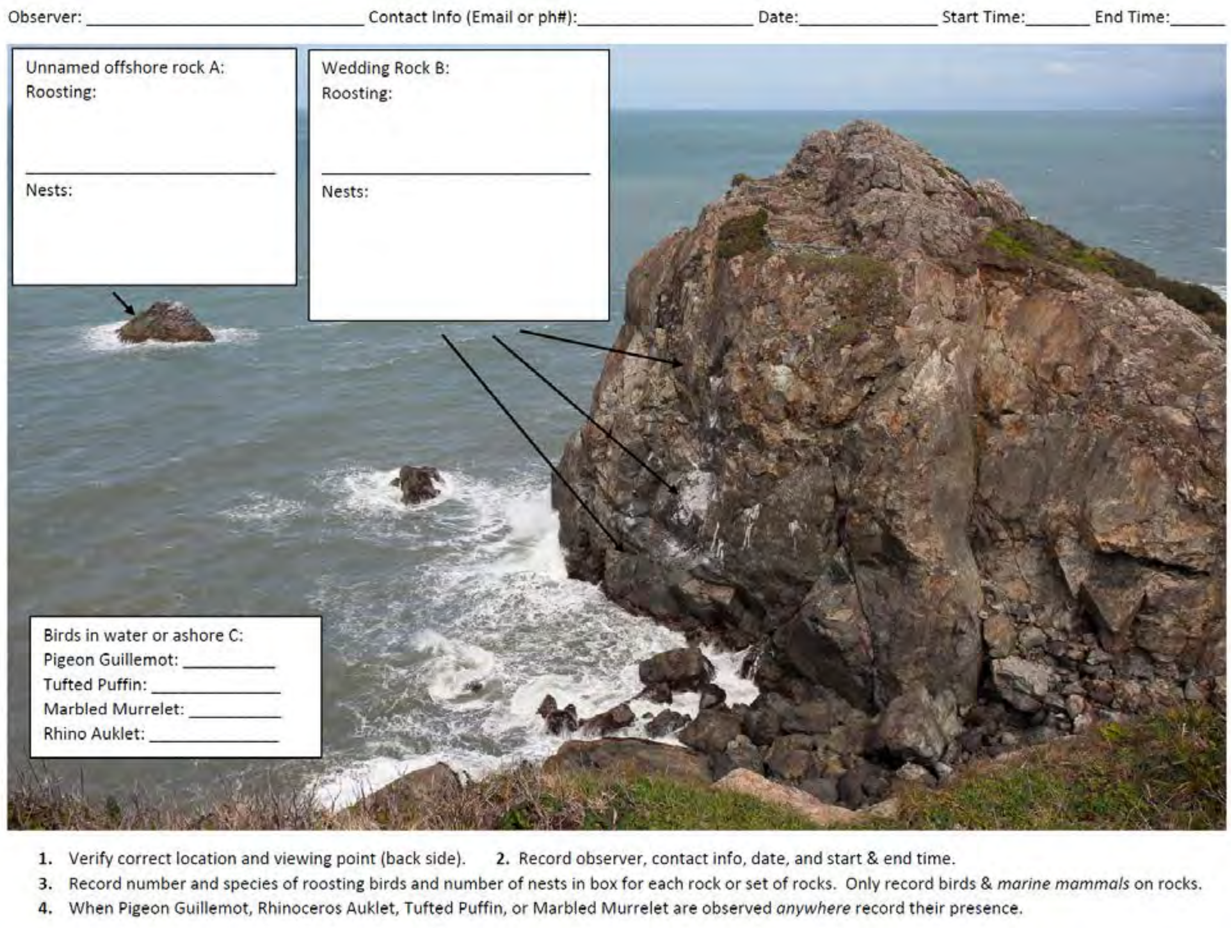


Figure 63. Trinidad Head / Patrick's Point study area. Large colony monitoring sites (inverted black triangles) from the trend analysis element are shown. The 9 citizen science survey locations ranged from Little River Rock to Agate Beach, shown as red circles.



Observer: ____________________ Contact Info (Email or ph#):____________ Date:__________ Start Time:_____ End Time:_____

Unnamed offshore rock A:
Roosting:

Nests:

Wedding Rock B:
Roosting:

Nests:

Birds in water or ashore C:
Pigeon Guillemot: ________
Tufted Puffin: __________
Marbled Murrelet: ________
Rhino Auklet: __________

1. Verify correct location and viewing point (back side). 2. Record observer, contact info, date, and start & end time.
3. Record number and species of roosting birds and number of nests in box for each rock or set of rocks. Only record birds & *marine mammals* on rocks.
4. When Pigeon Guillemot, Rhinoceros Auklet, Tufted Puffin, or Marbled Murrelet are observed *anywhere* record their presence.

Figure 64. Example citizen science data form. Data forms for each site are unique, but provide easy-to-reference cues that match the appearance of the field site from the observation point. Observers recorded birds observed by species at each recognizable feature.

RESULTS AND DISCUSSION

We analyzed data for the three most abundant taxa detected on biologist and citizen science surveys. We were forced to pool cormorant data across species, because many citizen scientist observations included uncertainty regarding cormorant identification. We excluded surveys conducted in inclement weather with poor observation conditions, yielding 39 citizen scientist observations and 135 biologist observations. We collected data at 11 sites between the two methods, with repeated observations using both methods at 8 sites (Table 14). All data, including the non-overlapping sites, is shown in Table 14 for comparison.

Citizen scientist data appeared to undercount relative to biologist data (Figure 65), but the two appeared to index each other well across the three species (and likely, to index true spatial variation in abundance within this study area) although we are not certain that in all cases the boundaries of count areas were perfectly aligned between the two methods. The coefficients of variation were similar between citizen scientist and biologist data (Table 14). The biologist reported 20 potential disturbance events and 1 response to disturbance event resulting in birds flushing from a roost site (following definitions in

Element 3, Chapter 5) while the citizen scientists reported 1 potential disturbance event and no responses to disturbance. The temporal range of dates sampled by the biologist (April-August) was different than the range of dates sample by citizen scientists (May-June).

Citizen science studies are widely used in natural resource monitoring programs and basic research () yet cross-validation studies are relatively infrequent (Cohn 2008). Seabirds have been or are currently monitored by various citizen science schemes, including coastline survey programs (Parrish et al. 2007, Ward et al. 2015) and occurrence-based programs such as eBird (Sullivan et al. 2009). Our results show that an index of rocky shoreline habitat use collected by citizen scientists, at least at a relatively fine spatial scale in one region, correlated strongly with data collected by professional scientists and thus is likely to be of use in detecting differences across space or time in this metric. Comparison with other citizen science programs conducted in the same region, such as beached-bird surveys conducted by COASST (Parrish et al. 2007) may further complement the approach we cross-validated, and may provide further insights

Table 14. Comparison of biologist and citizen scientist mean counts and coefficient of variation (CV) for three seabird taxa at 10 sites surveyed using both methods. Only 8 sites were surveyed via both methods.

		Biologist							Citizen Scientist					
		Cormorants		Western Gull		Pigeon Guillemot			Cormorants		Western Gull		Pigeon Guillemot	
Site	n	mean	CV	mean	CV	mean	CV	n	mean	CV	mean	CV	mean	CV
Camel Rock Overlook	12	16.8	0.4	19.3	0.5	21.9	0.7	4	6.5	1.0	11.4	0.4	6.2	0.4
Tepona Point	13	21.9	0.8	27.3	0.3	14.9	1.0	4	18.0	0.3	13.9	0.3	2.9	0.7
Trinidad Bay	14	3.2	1.1	0.9	1.5	14.1	0.9	5	14.4	0.7	0.0		4.7	0.6
Trinidad Bay #2	14	0.3	1.6	8.2	0.5	1.6	1.2							
Lighthouse	13	1.9	0.8	48.8	0.3	22.4	0.3	5	1.1	1.7	31.6	0.5	5.4	1.0
Wharf	13	0.2	2.6	2.1	0.5	2.0	1.2	4	1.6	0.9	4.7	0.7	1.5	1.2
Patrick's Point Drive	14	1.3	1.3	2.0	0.8	2.1	1.6	4	4.5	0.8	1.9	0.8	0.0	
Scotty Point	13	22.3	0.8	6.5	0.5	0.5	2.6	4	6.8	0.5	5.5	0.4	0.0	
Mussel Point	16	4.4	1.2	0.9	1.5	17.1	0.9							
Palmer's Point	13	16.8	0.6	38.8	0.4	1.2	1.5	4	7.7	0.3	24.3	0.4	0.0	
Wedding Rock Overlook		0.0		0.0		0.0		5	14.0	0.3	1.1	0.5	4.0	0.8

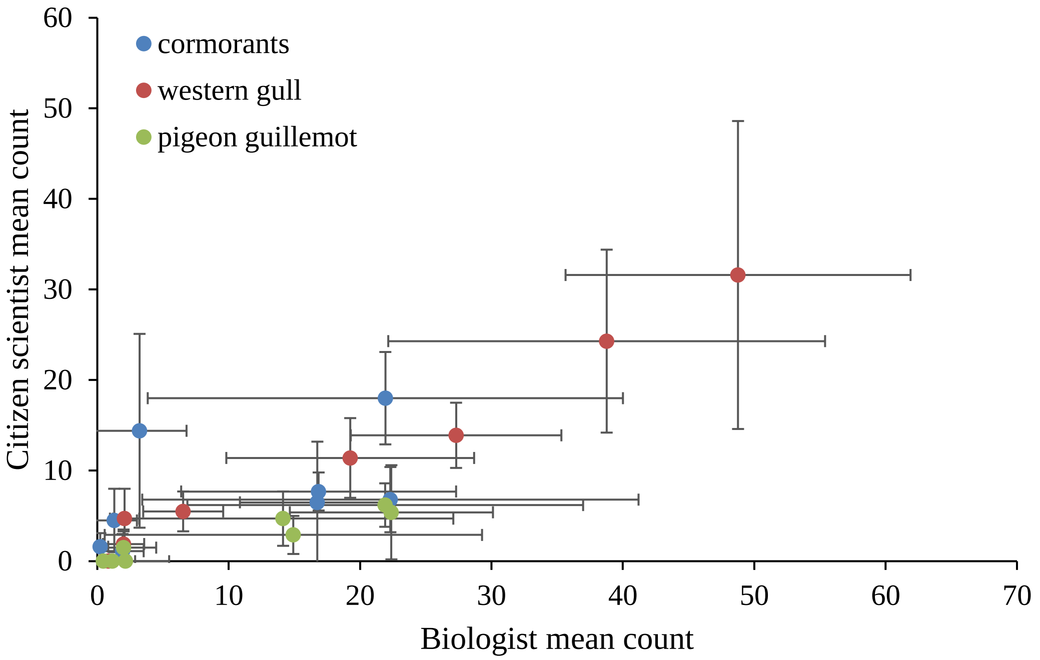


Figure 65. Pairwise comparisons of biologist and citizen scientist mean counts of three seabird taxa at 8 sites in the Trinidad Head / Patrick's Point region surveyed using both methods. The error bars are SD.

Long-term Monitoring Recommendations

Citizen scientist data may be useful in long-term monitoring programs, given the results of the simple cross-validation we performed. However, several important considerations should be made: first, that even in the most important unprotected seabird breeding area (the Trinidad Head and Patrick's Point area) on the California coast, near the North Coast Study Region's largest population center and only university, attracting and (perhaps) retaining citizen scientists to this program during the summer months appeared relatively difficult. Six outreach and training presentations yielded a small number of observers (7) that varied widely in the numbers of samples they collected. Second, that we were not able to cross-validate this method across sites with and without MPAs (the Trinidad Head area has none) because of budget limitations. Third, most of the NCSR's MPAs are located in remote, difficult-to-access (or for many potential citizen scientists, impossible to access) areas and thus problems in attracting and retaining citizen scientists to this cross-validation study would likely be multiplied if this method were applied more broadly to remote MPAs. Thus, expansion to the entire region or to monitoring changes in MPAs over time may prove logistically difficult given the inaccessibility of many of the region's MPAs, with the exception of several easily accessible MPA sites along the Mendocino County coastline, several of which are partly monitored by a pre-existing citizen science protocol dating to 2009 (Weigand 2014).

However, despite these caveats, the protocol and partnerships developed via this project element will continue and be refined, given the arrival of the newly-established Trinidad Chapter of the Seabird Protection Network. It is likely some of the shortcomings of initial implementation (especially in terms of number of participants and outreach coordination among different interest groups) will be reduced by this collaborative effort started by California Coastal National Monument (BLM), Humboldt State University, Point Blue Conservation Science, and several other stakeholder groups. In particular, dedication of a full-time biologist to specific development of this monitoring program, and work with a dedicated BLM education and outreach specialist, will likely increase participation, retention, and quality of this approach.

ACKNOWLEDGMENTS

The efforts of the 7 citizen scientist observers that participated by collecting data and the additional 10 potential observers that participated in training are obviously instrumental. Leisyka Parrott (Bureau of Land Management – Arcata Field Office) and Ben Morehead (Trinidad Coastal Land Trust) were invaluable partners.

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CHAPTER 6: BASELINE DISCUSSION AND RECOMENDATIONS

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The three elements of this baseline characterization each provided insights into the present state of seabirds in the North Coast Study Region (NCSR). They each established a benchmark for the parameters they measured, each has potential to detect change at different spatial and temporal scales, each describes somewhat different ecosystem attributes, and has potential to detect wide-ranging changes that might result from the establishment of Marine Protected Areas (MPAs). Seabirds use wide areas to find prey and are good regional indicators of change in the system. However, the impacts of specific MPAs are more difficult to assign to a wide-ranging animal like a seabird. An exception to this are the Special Closures within the system of MPAs which provide specific protections at some of the seabird colonies and thus fine scale effects. With adequate data histories, these fine scale targeted closures could be compared to colonies without such protections to assess their effectiveness. Additionally, seabird foraging rates and distribution are likely to respond to short-term changes in juvenile fish availability and can provide an index of spatio-temporal variability in nearshore fish recruitment. Conversely, because MPAs may affect forage fish more broadly than within the boundaries of a single MPA, seabirds in the NCSR will be likely indicators of change in productivity across the NCSR.

Each of the three elements monitored different population parameters and scales (Table 15). All of the monitoring by each element produced adequate samples to detect change (at least for common species like the Common Murre, but not for rarer species like Marbled Murrelets; see Appendix A). Burrow nesters (Cassin’s Auklets, Rhinoceros Auklets, Tufted Puffins) or storm-petrels were not monitored directly but, in some cases, change attributes can be inferred from data on Common Murre, but in other cases different techniques would be needed (e.g.: thermal imaging cameras, sound recordings) that would rely on additional technologies.

The aerial photographic element (Element 1, Chapter 2) successfully identified population change over multiple years. It would also be able to detect the processes of distributional change. This approach allowed calculation of minimum breeding population size across the region (being robust to movements between colonies) while at the same time identifying changes in abundance at individual colonies. Aerial photographic counting is limited to surface nesting seabirds, and was useful for counting the most common seabird in the NCSR, the Common Murre. This technique may also be used to count numbers of Brandt’s Cormorant, Double-crested Cormorant, and Western Gull. Further, large numbers of Common Murre reflect long-term changes in prey availability, and thus reflect forage fish changes that may impact other seabird species. Aerial counting cannot directly identify sources of disturbance at colonies, or the mechanisms of population decrease or increase (although with long-term data we did identify an association between population growth and oceanographic condition as measured by the Southern Oscillation Index; see Element 1, Chapter 2). Although aerial counts generally occur annually, it might take multiple years to detect the effect of a small but directed change in the population. Unless combined with additional data from observational surveys or remote video of diet and reproduction, the mechanisms of change may be unknown. This approach using aerial photos to monitor populations could also be used to monitor colonies inside and outside of special closures and marine reserves, and thus presents the opportunity for before-after-control-impacts (BACI) designs.

The Castle Rock National Wildlife Refuge reproductive and foraging seabird element (Element 2, Chapter 3) provided temporally quick responses to changing conditions (in some cases weekly) that could ultimately affect population size. The timing of daily reproductive and forage information can allow concurrent inference from reproductive success or failure to assist and inform managers in a time frame consistent with the change, and thus an opportunity to alter management. Changes in diet or effort in delivery of prey can also be indicative of ocean changes. Importantly, deleterious disturbance events will be reflected early by changes in reproductive parameters and may be the first indicators of population-level effects from disturbance. Similar to aerial counting, this technique does not identify the source of the disturbance but can quickly indicate the occurrence of a disturbance event. Trend in reproductive output can be measured if conducted annually, and can be compared to elsewhere in the California Current. With the exception of single-site disturbances, the reproductive and forage measures can be extrapolated to nearby colonies or to the entire NCSR. This technique is dependent on the use of specific technologies (e.g., video) which initially requires a developed infrastructure. However, this infrastructure also allows viewing of the science in progress by the public via the live-streaming video over the web and the large screen real time display in Redwoods National and State Parks visitor center in Crescent City. The technique was useful for monitoring Common Murre, but can also be adapted for Brandt's Cormorants.

The coastal habitat monitoring element (Element 3, Chapter 4) focused on coastally breeding species (Brandt's Cormorant, Double-crested Cormorant, Pelagic Cormorant, Pigeon Guillemot, Western Gull and Black Oystercatcher), but included some coastally roosting and nearshore foraging species as well (California Brown Pelican, Common Murre, and Marbled Murrelet). Despite the inclusion of the Common Murre, there was no overlap with the first two elements as they investigated Common Murre population size and breeding success while Element 3 focused on estimating Common Murre foraging rates. Although the measures in this element are conducted at several sites in the NCSR, in combination they can be indicative of the region. Both foraging rates and productivity of the focal species were measured and characteristic of areas surrounding each sampled site, and this method also provides the basis for future experimental BACI designs for several monitoring metrics. The technique uses human observers in real time who can also monitor disturbance rates (though only detecting disturbance when the observer is present and the adequacy of this measure is related to effort). Systematic application of the technique allows for trend analysis through years. Assessment of distribution of seabirds using this technique is dependent on the placement of sites in the NCSR. Marbled Murrelets were detected, but adequate samples for estimation of population trend or response were not obtained (see Appendix A).

None of the three elements could detect the establishment of MPAs, despite two of the elements using sensitive long-term pre-existing data sets. This is not surprising because seabirds are long-lived and some species such as the Common Murre produce only one egg each year. Thus immediate changes resulting from MPAs may not be quickly reflected in population abundance, and responses might have significant time lags. Further, any small change will be masked by natural variation and it would be expected that multiple years of monitoring, and, ideally, use of a BACI design may be necessary.

There were pros and cons to each approach. The aerial photographic technique monitors the important logistical measure of population abundance, that is often needed by managers in planning and assessment of the ocean resources. With annually repeated surveys, it is possible to detect trends and even correlate those trends with large scale oceanographic change. However, without knowledge of reproduction and foraging effort, it lacks the causal mechanisms of any detected change. This technique also requires multiple years and longer time series.

For quicker responses, the observations resulting from the remote video and from the shore based coastal surveys can provide timely insights. These are also more likely to suggest the sources of population change. Common Murre are good indicators of the state of the NCSR for seabirds, but are generally occur at locations that are difficult for human observers to access visually from shore, even with high powered scopes. Alternatively, observational surveys from shore may be better suited to site specific comparisons using appropriate species to make conclusions about localized changes in conditions. The observational surveys from shore also have the possibility for the utilization of citizen science participants.

Citizen science has potential to be effective in the shore-based monitoring (Chapter 5), though the data from the cross-validation study herein had higher sampling variance for the citizen science data (lower potential to detect change). The Castle Rock reproductive element (Element 2) also has potential to involve the public, but may be better suited as an outreach effort due to the complexity of the camera system, and citizen science should not be relied on as a cost-effective data collection program for this technique.

Recommendations for Continued Seabird Monitoring

Successful adaptive management of the NCSR network will depend on continued long-term monitoring to inform managers of the network's ongoing status. Long-term monitoring is important due to the highly variable nature of the California Current System. There are two compelling reasons to include seabirds in continued MPA monitoring. First, seabirds are an integral component of nearshore ecosystems and will benefit from MPA protection. However, the benefits of MPAs on seabirds have not been well studied. California's network of MPAs offers a unique opportunity to document these benefits. Second, seabirds are reliable indicators of change within marine ecosystems and can help track the underlying mechanisms governing change within MPA boundaries as well as across the region. Below, we outline four recommendations for continued seabird monitoring within the NCSR.

1) The NCSR Monitoring Plan should be updated so that individual marine bird species are represented within the appropriate ecosystem feature. Multiple marine bird species can be used to contribute to the monitoring of ecosystem features (Table 16). Specifically, most coastally-breeding seabirds are dependent on prey from multiple ecosystem features. By monitoring both the breeding and foraging ecology of these species, it is possible to gain information on multiple ecosystem features without additional surveys. Other marine birds (i.e., shorebirds, waterfowl and other piscivorous birds) could be used to monitor the intertidal and estuary/wetland ecosystem features.

2) The sources and rates of human-caused disturbance should continue to be documented inside and outside of MPAs as well as monitoring the special closures. MPAs can provide direct benefits to seabird populations, but outreach and enforcement will be a necessary component of MPA success. Quantification of human-caused disturbance can be used to guide the efforts of MPA Watch and similar groups. Further, while monitoring human-caused disturbance

observers can also document illegal fishing. Such efforts will reinforce the protection provided by MPAs.

3) Measures of seabird breeding productivity should be integrated with indices of ocean climate and direct measures of ocean productivity. It is important to recognize that much of the change occurring within MPA boundaries will be driven by regional oceanographic conditions governing primary and secondary productivity. Integrating seabird metrics with direct measures of ocean productivity will create a more holistic index of annual oceanographic conditions. Combining this regional approach with the fine scale approach of monitoring inside and outside of individual MPAs will help scientists and resource managers track the mechanisms leading to change within the NCSR network and better interpret the changes observed within individual MPAs. Finer-scale data on local and mesoscale oceanographic conditions are being generated by another NCSR baseline project, and when available will provide an opportunity for such synthesis.

4) Seabird foraging rates should continue to be monitored in order to: 1) better interpret annual variability in breeding population size and breeding productivity by documenting annual variability in prey distribution and 2) track where fish recruitment is likely occurring within nearshore habitats. Data on foraging rates can be integrated with indices of ocean climate, estimates of regional larval abundance, and fine-scale maps of near-surface currents to investigate both temporal and spatial variability in the ocean conditions affecting fish recruitment. Understanding annual variability in fish recruitment for individual MPAs will help managers interpret the changes observed within these MPAs and establish realistic expectations for their performance. Furthermore, it will help managers determine if MPA boundaries need to be moved to increase the effectiveness of a given MPA.

Specific recommendations for long-term monitoring depend on the specific concerns and objectives of the effort. For example, if disturbance rates were a major concern for seabird monitoring, then a technique that includes human observers could have the best probability of encountering and recording those events. For efficient long-term indicators of change, the aerial photographic surveys can provide sensitive indication of changes in the populations attending the colonies, and would be an effective indicator of change in the region due to changes in forage in the region (and the MPAs). However, it is important to note that aerial surveys alone would not provide managers with timely insight in order to manage short term influences (e.g.: human disturbance). Understanding the mechanisms of change would require detailed understanding of reproduction and diet. The combination of diet and reproduction with a longer term data set can assist in this effort. If a goal of the monitoring includes outreach to the public, the video from Element 2 could provide an additional opportunity in that effort. The costs of all three techniques varied with the sampling effort. New techniques for counting birds from aerial photographs are being developed that may make aerial counting extremely cost effective in the future.

Table 15. Outcome characteristics and benefits of parameters used for monitoring seabirds derived from the baseline characterization evaluating effects of marine protected areas on seabirds in the North Coast Study Region.

	Outcome characteristic				Benefits			
Monitoring Parameters	Reflective of multiple species	Adequate samples	Measurable variance	Responsive to change	Trend	Population Size	Disturbance observations	Distribution
Element 1 - Aerial Colony Counts								
Counts of Aerial photos	yes	yes	yes	year(s)	yes	yes	no	yes
Element 2 - Castle Rock NWR								
Reproductive success	yes	yes	yes	in season	yes	no	possible	no
Foraging effort	yes	yes	yes**	season/year	yes	no	no	no
Diet	yes	yes	yes**	weekly/annual	yes	no	no	no
Element 3 -- Coastal Habitat								
Foraging rates	yes	yes	yes	annual	yes	no	no	yes
Roost Utilization	yes	yes	yes	in season	yes	yes	yes	yes
Breeding population size	yes	yes*	yes**	annual	yes	yes	yes	yes
Breeding productivity	yes	yes*	yes**	annual	yes	yes	yes	no

*for some species, namely cormorants

**variance measurable among years

Table 16. Recommended inclusion of marine birds as indicators/focal species for future monitoring efforts within the NCSR.

Ecosystem Feature	Key Attribute	Indicator/Focal Species
Kelp and Shallow (0-30m) Rock	Primary: Seabird Breeding & Foraging Ecology	Pelagic Cormorant Breeding Population Size & Fledging Rate
		Brandt's Cormorant Breeding Population Size & Fledging Rate
		Pigeon Guillemot Breeding Population Size
		Pelagic Cormorant Foraging Rates
		Brandt's Cormorant Foraging Rates
		Pigeon Guillemot Foraging Rates
	Optional: Seabird Diet	Pigeon Guillemot Diet
		Brandt's Cormorant Diet
Mid-Depth (30-100m) Rock	Primary: Seabird Breeding & Foraging Ecology	Brandt's Cormorant Breeding Population Size & Fledging Rate
		Common Murre Breeding Population Size & Fledging Rate
		Pigeon Guillemot Breeding Population Size
		Brandt's Cormorant Foraging Rates
		Common Murre Foraging Rates
		Pigeon Guillemot Foraging Rates
	Optional: Seabird Diet	Brandt's Cormorant Diet
		Common Murre Diet
		Pigeon Guillemot Diet
Rocky Intertidal	Primary: Black Oystercatcher Breeding & Foraging Ecology	Black Oystercatcher Breeding Population Size & Fledging Rate
		Black Oystercatcher Foraging Rates
		Black Oystercatcher Diet
	Optional: Predatory Marine Birds	Abundance of Shorebirds & Piscivorous Birds
		Diversity of Shorebirds & Piscivorous Birds
		Abundance of Black Oystercatchers
Soft-Bottom Subtidal (0-100m)	Primary: Seabird Breeding & Foraging Ecology	Brandt's Cormorant Breeding Population Size & Fledging Rate
		Pigeon Guillemot Breeding Population Size
		Brandt's Cormorant Foraging Rates
		Pigeon Guillemot Foraging Rates

Table 16 (cont'd.)

Ecosystem Feature	**Key Attribute**	**Indicator/Focal Species**
Soft-Bottom Subtidal (0-100m)	Optional: Seabird Diet	Brandt's Cormorant Diet
		Pigeon Guillemot Diet
Estuary & Wetland	Primary: Waterbird Habitat Use	Abundance of Shorebirds, Waterfowl, and Piscivorous Birds
		Diversity of Shorebirds, Waterfowl, and Piscivorous Birds
		Least Tern Diet
	Optional: Predatory Marine Bird Foraging	Foraging Rates of Piscivorous Birds
Soft-Bottom Intertidal & Beach	Primary: Predatory Marine Birds	Abundance of Shorebirds
		Diversity of Shorebirds
	Optional: Western Snowy Plover Breeding	Western Snowy Plover Breeding Population Size & Fledging Rate
Nearshore Pelagic	Primary: Seabird Breeding Ecology	Brandt's Cormorant Breeding Population Size & Fledging Rate
		Common Murre Breeding Population Size & Fledging Rate
	Optional: Seabird Diet	Brandt's Cormorant Diet
		Common Murre Diet

APPENDIX A:
MARBLED MURRELETS

The Marbled Murrelet (*Brachyramphus marmoratus*) is a small seabird in the family Alcidae and is native to the northern coast of California. Marbled Murrelets (hereafter murrelets) live and forage at sea but are unique among California seabirds because they nest high in old-growth coniferous trees (Hébert and Golightly 2006, Golightly et al. 2009, Baker et al. 2006), primarily redwoods (*Sequoia sempervirens*). During nesting the adults must return to sea to feed and must deliver food caught at sea to the nest to feed the maturing chick over a 28 day period before the chick fledges to fly to sea. Murrelets in California are federally listed as “threatened” as well as listed under the California Endangered Species Act (CESA) as “endangered”. Their low abundance has been primarily due to the loss of nesting habitat (USFWS 1997) and nest predation (Hamer and Nelson 1995, Hébert and Golightly 2006, Peery and Henry 2010, Hébert and Golightly 2007); most murrelets in California now nest in trees protected in parks and preserves. However, their population size may also have been affected by ocean condition and forage availability (see Becker and Beissenger 2006, Becker et al. 2007). Little is known about the specifics of their diet, though they were reported to take krill in spring and small schooling fishes like northern anchovy (*Engraulis mordax*), Pacific herring (*Clupea harengus*), and surf smelt (*Hypomesus* sp.) in the summer when rearing chicks (Nelson 1997). Studies have been initiated using DNA sequencing to establish more robust diet characterization (Peery and Golightly in progress).

Murrelets are near-shore foragers, and average distance that individuals are found from shore is 1.4 km (Hébert and Golightly 2008). Thus murrelets can be present in almost all northcoast Marine Protected Areas (MPA), though some MPA’s may be used more than others due to variation in the distribution of their prey and the proximity to adjacent nesting habitat. Those MPA units nearest coastal stands of old-growth forest are probably more likely to be used by or have potential impact on murrelets. Geographically, murrelets are most abundant in the North Coast Study Region (NCSR) north of the Eel River mouth (this is based on existing populations and may change slowly as terrestrial nesting conditions change or recover in the southern part of the NCSR). In the shore-based surveys of our project (Element 3), murrelets were detected foraging nearshore in 2014, but not in 2015. Foraging rates in 2014 were highest at the northern most study sites: Pyramid Point SMCA and the Crescent City and Patrick’s Point control sites (see Element 3 for map of study sites). Foraging rates were close to the region-wide average at the Trinidad Bay control site and low foraging rates were detected at the Devil’s Gate and Kibesillah control sites. Murrelets may have been responding to a greater availability of nearshore prey in 2014 as conditions within the NCSR were more oceanographically productive in 2014, with higher abundances of juvenile rockfish being reported (Leising et al. 2015).

Although murrelets are ecologically important and of significant management concern, they were not deemed appropriate for a study indicator or baseline survey in the NCSR. There were three reasons we did not include them in our field surveys. First is that their abundance is low; small sample sizes in surveys would render any analysis difficult and would lack statistical power. Secondly, within a day they fly long distances and do not consistently remain at the same location, which is problematic for a limited sampling effort. They also move long distances across time (Hébert and Golightly 2008, Lorenz et al. 2016), making it difficult to associate any particular observation with specific areas of the ocean. Third, there is already a rigorous effort to monitor trend and population sizes conducted as part of the Marbled Murrelet Effectiveness Monitoring for the Northwest Forest Plan (NFP; see Falxa and Raphael 2016). The NFP monitoring is better supported than could be accomplished in our baseline studies and has now been conducted over 16 years. Therefore we chose to summarize the reports from the NFP efforts and used the summary report from 2015 (Lynch et al. 2016).

In California in 2015, the NFP monitoring effort conducted at-sea surveys in the Marbled Murrelet Conservation Zone 4 (an area along the coast from southern Oregon to southern end of Humboldt County in California) which overlaps the NCSR. Lynch et al. (2016) reported a slightly positive trend (3+/- 2.6 %) in the zone for the period from 2000 to 2015. They reported a density estimate of 7.54 birds/km^2 and an overall population estimate of 8743 (95% CL was 7409-13125 birds) for Zone 4. Because of individual state concerns, they also calculated an estimate for California only (north of the Golden Gate) which produced a density of 3.62 birds/km^2 and a population size of 5666 (95% CL was 3970-7362). These California-only numbers include low density areas south of the NCSR and exclude numbers from southern Oregon. The NCSR north of the Eel River mouth is probably better characterized by the numbers for Zone 4 in general.

The slightly positive trend in Conservation Zone 4 should be interpreted cautiously. First the lower end of the 95% confidence interval is only slightly positive (0.4%, or very little population change). New forest nesting habitat requires long time frames and habitat gains will not be quickly reflected in 15 years of monitoring. However, acres of higher suitability nesting habitat in California has declined only slightly during this time (Raphael et al. 2016) in Zone 4, due in part to most of the nesting habitat being protected in parks and preserves. Efforts have been made to improve or protect existing terrestrial habitat in the past 10 years which has potential to increase productivity by murrelets in the existing habitat (e.g. see CASP 2013, Gabriel and Golightly 2014, Goldenberg et al. 2016, West et al. 2016). Anecdotally, the period sampled by the NFP monitoring in Zone 4 has also been free from oil spills (the last big oil spills that killed murrelets occurred in or near Humboldt Bay in the 1997 Kurre oil spill and in 1999 Stuyvesant oil spill; CDFG et al. 2007, CDFG and USFWS 2008) which may have reduced anthropogenically sourced mortality.

Although the NFP monitoring has provided region-wide population estimates and trend detection, it would be difficult to identify smaller spatial-scale estimates or subunits that could be tested objectively. Numbers surrounding single MPA units would lack meaningful power to detect change. Thus the MPA program will need to evaluate murrelets at the regional scale. Continued monitoring by the NFP initiatives in Zone 4 is essential to that evaluation. Beginning in 2015, NFP monitoring has changed to every-other-year surveys (Lynch et al. 2016).

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APPENDIX B:
STUDENT INVOLVMENT, PRESENTATIONS AND OUTREACH

STUDENTS

Shannon Murphy – Humboldt State University MS Thesis, Fall 2016
Stephanie Schneider—Moss Landing Marine Lab MS Thesis, anticipated Fall 2017
Jasmine Mejia– Undergraduate Intern at Moss Landing Marine Lab, Summer 2016

THESES

Murphy, S. E. 2016. Evaluating Brandt's Cormorant (Phalacrocorax penicillatus) reproductive success: effects of parental care behaviors and estimating individual chick survival. M.S. Thesis, Humboldt State University, Arcata, CA.

Schneider, S. R. In progress. Environmental and ecological insights from colony-based monitoring of an apex predator, the Common Murre. M.S. Thesis, San Jose State University, San Jose, CA. [Anticipated September 2017].

PUBLICATIONS TO DATE

Leising et al. 2015. State of the California Current 2014-15: Impacts of the warm-water " Blob " CalCOFI Report, 56, 31–68.

McClatchie et al. 2016. State of the California Current 2015-16: Comparisons with the 1997-98 El Niño. CalCOFI Rep. 57:5-61.

OUTREACH AND PRESENTATIONS

Daniel Barton and Shannon Murphy. 2015 (April). Seabird monitoring for Marine Protected Areas: Cross-validation of a new citizen science protocol. Scientific & Outreach Presentation to California North Coast Chapter of The Wildlife Society.

Daniel Barton. 2015 (Sept). Seabird monitoring in north coast Marine Protected Areas: Are seabirds indicators? What do they indicate?, Scientific & Outreach Presentation to Redwood Region Audubon Society.

Daniel Barton, Dan Robinette, and Richard Golightly. 2015 (Nov). MLPA North Coast Study Region Baseline Monitoring Program Baseline Characterization of Seabirds. Presentation to the North Coast Collaborative Forum.

Daniel Barton. 2016 (April) Seabird monitoring in north coast Marine Protected Areas: Are seabirds indicators? What do they indicate? Scientific & Outreach Presentation to Conservation Unlimited.

Daniel Barton. 2016 (May). Seabird monitoring in north coast Marine Protected Areas: Are seabirds indicators? What do they indicate? Scientific & Outreach Presentation to Trinidad Gateway Group.

Daniel Barton. 2016 (June). Seabird monitoring in north coast Marine Protected Areas: Are seabirds indicators? What do they indicate? Scientific & Outreach Presentation to Trinidad Gateway Group.

Daniel Barton. 2016 (Dec). Seabird conservation and monitoring on the North Coast. Outreach Presentation to Trinidad Chapter of the Seabird Protection Network.

Shannon Murphy, Richard Golightly, Stephanie Schneider, and Daniel Barton. 2016 (Feb). Parental care behaviors in Brandt's Cormorant (*Phalacrocorax penicillatus*): Effects on reproductive success and use as indicators of the marine environment. Presentation to the Western Section of the Wildlife Society (*winner of best student presentation award*).

Stephanie Schneider and Richard Golightly. 2017 (Feb). Living at the edge: A 10-year timeseries of Common Murres nesting at a large colony in the California Current. Presentation to the Pacific Seabird Group's International Symposium.

Stephanie Schneider. 2017 (Feb). Nesting seabirds: A brief history of monitoring in the California Current. Guest Lecture for San Jose State University, Advanced Topics in Marine Vertebrates [MS 204].

Schneider, S.R. and R.T. Golightly. 2015 (April). Castle Rock: A Seabird Sanctuary. Invited speaker for Friday Night at the Refuge hosted by Humboldt Bay National Wildlife Refuge.

APPENDIX C:
BUDGET SUMMARIES

HUMBOLDT STATE UNIVERSITY, BUDGET SUMMARY

The Humboldt State University (HSU) Sponsored Programs Foundation budget for California Sea Grant project R/MPA-35A were spent on all three elements within the Seabird Baseline Characterization. Element 1 of the characterization shared expenditures with UC Santa Cruz. Element 2 was exclusive to HSU. Element 3 shared expenditures with Point Blue Conservation Science. All expenditures were within 10% of proposed. For Sea Grant funds, the salaries and benefits totaled $93,691, travel totaled $10,961, supplies totaled $996. Matching funds were also derived as proposed, with the exception of using only $17,574 of a trust fund for Castle Rock (instead of $20,000) and substituting an additional uncompensated salary by one PI in the amount of $2434 for Element 2. Match, including deferred indirect, totaled $126,902.

UNIVERSITY OF CALIFORNIA, SANTA CRUZ, BUDGET SUMMARY

Expenditures by University of California, Santa Cruz (UCSC), for California Sea Grant Project R/MPA-35C were as planned: Salary and Benefits $27,998.28 ($28,042 Budgeted), Supplies, i.e. CDFW Aircraft Costs, $7,000 ($7,000 Budgeted), Domestic Travel $299.16 ($300 Budgeted). Funds budgeted for costs of the California Department of Fish and Wildlife were included as Domestic Travel in our award budget but as Supplies by the University. UCSC deferred indirect costs on Salary and Benefits was $8399.48 (45% vs 15%).

POINT BLUE CONSERVATION SCIENCE BUDGET SUMMARY

As of March 1, 2017, Point Blue Conservation Science has billed $53,048.01 to grant RMPA-35B. An additional $344.99 will be billed in a final invoice. Thus, the final cost of this project will be $53,393.00 and will not exceed our proposed budget (see Table C-1). We were $1,914.36 (7.13%) over budget in the Salaries and category, $890.50 (7.13%) over budget in the Benefits category, and $2,803.50 (82.46%) under budget in the Travel category. After our initial visit to Humboldt State University, during which we established monitoring sites, we determined that all other communication could be conducted remotely. We therefore used the additional $2,803.50 budgeted for travel to increase the amount of staff time for data management, data analysis, and report writing.

TABLE C-1. Final summary of funds spent by Point Blue Conservation Science for California Sea Grant Project R/MPA-35B.

Budget Category	Year 1	Year 2	Year 3	Total	Budgeted	Difference
Salaries	$7,432.18	$7,921.98	$13,395.20	$28,749.36	$26,835.00	-$1,914.36
Benefits	$3,455.94	$3,683.74	$6,228.82	$13,368.50	$12,478.00	-$890.50
Travel	$596.50	$0	$0	$596.50	$3,400.00	$2,803.50
Indirect Costs	$2,871.15	$2,901.47	$4,906.02	$10,678.64	$10,680.00	$1.36
Total Costs	$14,355.77	$14,507.19	$24,530.04	$53,393.00	$53,393.00	$0