Chapter 2

Population Trends of the Common Murre *(Uria aalge californica)*

by

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Abstract: Population trends for the common murre (*Uria aalge californica*) were determined from available whole-colony counts of murres in California, Oregon, Washington, and British Columbia from 1800 to 1995. From 1800 to 1978, historical counts were sporadic and not standardized. From 1979 to 1995, standardized whole-colony counts from aerial photographs were conducted in many years in California, Oregon, and Washington. In contrast, no aerial photographs of murre colonies in British Columbia have been taken and only a few other whole-colony counts have been conducted. Direct comparisons and statistical treatment of

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whole-colony counts were conducted using 1979–95 data. Complete data for all colonies were available only in 1988–89 when the breeding murre population was estimated to be 1.1 million, about 5–8% of the world population and 13–28% of the Pacific Ocean population. A summary of various natural and anthropogenic factors affecting murre populations in western North America since 1800, and particularly in 1979–95, also is provided.

A relatively good history exists for murre colonies in central California. The well-known colony at the South Farallon Islands may have numbered 1–3 million birds in the early 1800s. Egging and human disturbance throughout most of the nineteenth century, plus mortality from oil pollution in the early twentieth century, caused the near extirpation of this colony by the 1930s. Since the 1950s, this colony has grown and, by the early 1980s, again was the largest colony in central California. Two other large colonies also are present in central California at the North Farallon Islands and Point Reyes. In the early twentieth century, Prince Island in southern California was the southernmost breeding colony of *U. a. californica*, but the colony was extirpated in about 1912. Hurricane Point Rocks in central California is now the southernmost colony. In 1980–82, the central California breeding population was estimated at 194,000–224,000 breeding birds at nine active colonies. From 1979 to 1989, this population declined 9.9% per annum ($P = 0.002$) because of mortality from gill nets and oil spills, in concert with detrimental effects from the severe 1982–83 El Niño. All colonies declined significantly and the Devil's Slide Rock colony was extirpated. In 1989, the population was estimated at 90,200 breeding birds at 8 active colonies (i.e., 8% of the *U. a. californica* population). From 1985 to 1995, the population increased 5.9% per annum ($P = 0.002$), mostly since 1989–90, but had only partly recovered to 1979–82 levels by 1995. Increase since the late 1980s has occurred despite continuing anthropogenic impacts and low reproduction during the severe 1992–93 El Niño. The Devil's Slide Rock colony did not recover between 1986 and 1995, but breeding has been restored in 1996–2000, using social attraction techniques.

In northern California, limited historical data indicated that murre colonies were heavily affected by early settlers in the late nineteenth century, as well as oil pollution in the early twentieth century. Only two colonies (i.e., Castle and Green Rocks) were specifically known prior to the late 1940s. Detrimental effects apparently lessened in the mid-twentieth century, allowing substantial population growth over several decades since the 1930s, including many recolonization events prior to the 1970s. Little change in available population numbers occurred from 1979 to 1989, which suggests a possible leveling of population numbers and little or no long-term detrimental effects from the 1982–83 El Niño. Lower numbers at Castle Rock in 1986 and 1989 appear related to differences in survey techniques. In 1989, the breeding population was estimated at about 261,400 breeding birds at 11 active colonies (i.e., 24% of the *U. a. californica* population). The largest colonies were at Castle Rock, False Klamath Rock, Green Rock, Flatiron Rock, and False Cape Rocks. Colonies had lower numbers in 1993 indicating short-term abandonment during the severe 1992–93 El Niño, but with few long-term detrimental effects. Recolonization and population increase have continued since the 1970s at the southern end of this population.

The vast majority of murres in western North America, south of Alaska, now breed in Oregon. Numbers of murres in the nineteenth and early twentieth centuries were much lower owing to extensive use of coastal rocks and islands by native peoples, followed by egging and human disturbance by early settlers. However, the population increased for several decades in mid-twentieth century. By 1988, about 711,900 breeding birds were estimated at 66 active colonies (i.e., 66% of the *U. a. californica* population). The largest colonies were at Shag Rock, Finley Rock, Middle Rock, Gull Rock (Cape Blanco), 270-110, Cat and Kittens, and 219-018. A sample of 15 colony sites indicated that murre numbers changed little from 1988 to 1995, except for short-term abandonment during the severe 1992–93 El Niño. Long-term detrimental effects from severe El Niños in 1982–83 or 1992–93 have not been detected.

Historical accounts indicate that murre populations in Washington increased from 1907 to 1979. In 1979, about 53,000 breeding birds were estimated at 18 active colonies. The largest colonies were at Split Rock, Willoughby Rock, Grenville Arch, and Rounded Island. Between 1979 and 1986, a 43.7% per annum ($P = 0.006$) decline occurred in the number of murres attending breeding colonies in southern Washington. Overall numbers of murres in Washington declined 13.3% per annum ($P = 0.003$) from 1979 to 1995. By 1988, about 7,000 breeding birds (i.e., less than 1% of the *U. a. californica* population) remained. Declines apparently were related to the 1981 warm water event, the 1982–83 El Niño, and anthropogenic factors (i.e., human disturbance at colonies and gillnet and oil-spill related deaths). No recovery occurred in southern Washington from 1984 to 1995 but limited increase in the number of murres attending some colonies in northern Washington was documented. Increase at Tatoosh Island from 1984 to 1995 involved intercolony movements and intrinsic growth. The Washington murre population size has recovered little since its decline in the early 1980s, and remained low through 1995.

Small numbers of murres breed in British Columbia and there is no evidence to suggest they are more numerous than in 1900. About 8,300 breeding birds (i.e., less than 1% of the *U. a. californica* population) were estimated at two active colonies in 1989, although five other small colonies had been active in the 1970s. The northernmost colony of *U. a. californica* is at the Kerouard Islands at the southern tip of the Queen Charlotte Islands. In British Columbia, the vast majority of murres breed at the large colony at Triangle Island off the northern tip of Vancouver Island. Population trends at Triangle Island have not been well assessed.

Key words: Alcidae, breeding colony, breeding distribution, British Columbia, California, colony disturbance, colony extirpation, colony formation, common murre, egging, El Niño, gill net, habitat change, oil spill, Oregon, population size, population trends, predators, seabird, *Uria aalge*, Washington

Information on populations of the common murre (*Uria aalge californica*) in California, Oregon, Washington, and British Columbia are of two types, whole-colony counts of birds, which can be adjusted to derive estimates of the number of breeding adults at each colony, and transect counts of birds at sea, which describe at-sea densities. These two types of population data serve as primary baseline information for monitoring and assessing trends in populations of murres in various geographic areas in western North America (Sowls et al. 1980; Briggs et al. 1987, 1992; Speich and Wahl 1989; Takekawa et al. 1990; Rodway 1991; Wilson 1991; Carter et al. 1992, 1995; Byrd et al. 1993; Tyler et al. 1993).

Standardized whole-colony counts include a large proportion of breeding birds (i.e., each egg or chick had one or two attending adults) and some nonbreeding individuals attending the colony. Thus, whole-colony counts of all colonies in a geographic area constitute a primary population index wherein most of the population is counted directly rather than sampled. This kind of population index increases our ability to measure trends by greatly reducing potential variation or bias from sampling. At each colony, this index is related directly to the number of breeding adults or the total number of murres (breeding and nonbreeding) attending the colony, but the exact relation has not been determined. Estimates of the number of breeding birds at a colony can be derived from whole-colony counts with the use of a correction factor k (see Appendix A). Similarly, estimates of the numbers of nonbreeding birds can be derived through population modeling. However, k correction factors and demographic variables used in population models have been determined in only a few studies at certain locations and may not apply widely.

We considered trends in sums of standardized whole-colony counts from aerial photographs for all or many colonies in a geographic area to best reflect trends in murre populations over time. Within the range of *U. a. californica*, whole-colony counts can be conducted at all colonies, which reduces the potential for sampling error (e.g., if one colony were selected for monitoring in an area). Source colonies of birds also are known for colony counts but must be interpreted using various sources of information for at-sea counts. In general, standardized whole-colony counts are less variable, more repeatable, and subject to fewer biases than at-sea counts. However, numbers of murres attending colonies during the breeding season are subject to variation because of several factors, especially time of season, time of day, and colony disruption by human disturbance or interactions with other seabirds or marine mammals.

Transect counts of birds at sea also can be extrapolated over large areas to derive estimates of total population (i.e., adults and subadults) in a defined geographic area, but must account for murres attending colonies. At-sea counts and total-population estimates provide important data on the density, distribution, abundance, and movements of murres at sea, which are important in connection with various conservation issues. Significantly more baseline population information useful for monitoring purposes is available for colonies than for at-sea murre distribution and abundance.

Since 1979, monitoring of murres in California, Oregon, and Washington has focused primarily on standardized whole-colony counts from aerial photographs of birds attending colonies during the breeding season. The monitoring is so focused because (1) most colonies are comparatively small (fewer than 20,000 breeding birds) and are present on small islands with open habitats that can be aerially photographed on a regular basis, (2) intensive monitoring of plots within most colonies is impractical because most colonies are inaccessible or cannot be accessed without extensive disturbance to breeding birds, (3) potential biases are associated with monitoring plots from the ground (i.e., plot selections, number of plots, variation between plots, and counting error [Harris et al. 1986; Mudge 1988; Harris 1989]), and (4) monitoring plots can involve high cost and effort. The U.S. Fish and

Wildlife Service (USFWS) manages and surveys most murre colonies in Washington and Oregon, and important colonies in California, within the National Wildlife Refuge System. In California, however, most colonies are managed by the California Department of Fish and Game and the National Park Service, and surveys have been conducted by a combination of personnel from the USFWS, Humboldt State University, U.S. Geological Survey and University of California. Aerial photographic surveys can be conducted by refuge staff or other researchers at a reasonable cost in a short period of time during the breeding season, although the subsequent counting of photographs requires substantial effort. In certain other parts of the breeding range of the common murre, sample plots have been established as the primary method for monitoring where deriving accurate whole-colony counts from aerial photographs of entire colonies is either too difficult, too costly, or impossible. Overall, researchers have used a combination of survey and census techniques to monitor murre populations around the world, with techniques varying between colonies and geographic areas (Birkhead and Nettleship 1980; Gaston and Nettleship 1981). However, standardized or nonstandardized whole-colony counts at one or more colonies over several years have been used by many researchers to describe common murre population trends in various parts of the Atlantic and Pacific Oceans (Hudson 1985; Nettleship and Evans 1985; Vader et al. 1990; Byrd et al. 1993). Aerial photographic surveys of murre colonies at Funk Island, Newfoundland, and several colonies in eastern Canada have been employed since 1972 (D. N. Nettleship, personal communication).

Whole-colony counts of murres provide the best available baseline information for analysis of trends in the number of murres attending colonies in California, Oregon, and Washington. Available data sets are hampered, however, by four main factors: (1) incomplete or irregular survey coverage (i.e., surveys not conducted in some years, certain colonies omitted in certain years), (2) incomplete colony coverage (i.e., poor quality or incomplete sets of photographs at certain colonies in certain years), (3) incomplete counting of available aerial photographs in northern California and Oregon, and (4) single counts in most years (i.e., variation in whole-colony counts has not been fully assessed). For central California, Oregon, and Washington, such problems were limited, have been reduced over the past decade, and did not greatly affect the use of whole-colony counts for assessing murre population trends. However, we have identified and accounted for serious problems in certain cases. In northern California and British Columbia, available information was much more limited for assessing recent trends than in other areas. Under unusual circumstances, whole-colony counts may not accurately reflect the actual colony size; for instance, during severe El Niño-induced weather conditions, large numbers of murres may not attend colonies during annual surveys. Such circumstances must be identified and accounted for in assessments of population trends, using whole-colony count data.

In Alaska, common murres often breed sympatrically with thick-billed murres (*U. lomvia*) and it is often difficult to determine the proportions of each species (Sowls et al. 1978). This problem does not exist throughout most of the geographic area of western North America covered in this chapter. The current southern limit of breeding thick-billed murres is at Triangle Island, British Columbia, where up to 70 thick-billed murres have been recorded attending the colony (Vallee and Cannings 1983; Rodway 1991).

In this chapter, we have examined population trends of common murres using available information from whole-colony counts, primarily from aerial photographs, in California, Oregon, Washington, and British Columbia through 1995. In addition, we have reported estimates of the size of breeding populations of *U. a. californica* in different geographic areas. We have not attempted to collate information on at-sea densities or total-population estimates, but aspects of at-sea distribution, abundance, and movements are summarized in Manuwal and Carter (2001).

Methods

We used a broad framework for assessing murre population trends within six geographic areas along the west coast of North America: central California, northern California, Oregon, southern Washington, northern Washington, and British Columbia. Information provided for each area includes summaries of (1) qualitative and nonstandardized quantitative historical data from 1800 to 1978 of numbers of murres attending colonies and known or suspected human activities at colonies, (2) current breeding population size and distribution of colonies, and (3) major population changes identified between 1979 and 1995 (using standardized whole-colony count methods) and factors known or suspected to be associated with changes observed. Significant events documented after 1995 were noted where appropriate but data used for population trend analysis were restricted to the 1979–95 period.

Historical information on murres at colonies helped to derive a general concept of long-term colony and

regional population trends prior to 1979. To summarize historical information, and without the aid of computerized databases or search processes, we consulted all available published and unpublished sources known to us from prior research as follows: California (H. R. Carter, see Appendix B), Oregon (R. W. Lowe), Washington (U. W. Wilson and H. R. Carter), and British Columbia (M. S. Rodway and H. R. Carter). To augment historical information, H. R. Carter also examined egg records in California, Oregon, Washington, and British Columbia in major museum collections, including Western Foundation of Vertebrate Zoology, Camarillo, California (WFVZ); Humboldt State University, Department of Wildlife Museum, Arcata, California (HSUWM); Santa Barbara Natural History Museum, Santa Barbara, California (SBNHM); San Diego Natural History Museum, San Diego, California (SDNHM); University of California Berkeley, Museum of Vertebrate Zoology, Berkeley, California (BMVZ); National Museum of Natural History, Smithsonian Institution, Washington, D.C. (USNM); Museum of Comparative Zoology, Harvard University, Cambridge, Massachusetts (MCZ); Royal British Columbia Museum, Victoria, British Columbia (RBCM); and University of British Columbia Zoology Museum, Vancouver, British Columbia (UBCZM). Substantial historical information was rediscovered in these museum collections.

Annual survey data for each colony in 1979–95 were collated as follows: California (H. R. Carter and J. E. Takekawa; see Appendices C, D), Oregon (R. W. Lowe; see Appendix E), Washington (U. W. Wilson; see Appendixes F and G), and British Columbia (M. S. Rodway). Regional estimates of the total number of murres attending colonies during surveys were determined by summing single representative whole-colony counts from aerial photographic surveys for each colony within a particular year in central and northern California and southern and northern Washington (see Appendix A for survey methods and summary). For these areas, we summed counts only when all or most colonies were judged to have been surveyed in a generally standardized and compatible fashion in the same year. For Oregon, we summed counts for 15 sample colonies that were surveyed and counted annually between 1988 and 1995. This sample of colonies is spread along the entire Oregon coast, but several large colonies are not included because of extensive counting time required. Annual sums of whole-colony counts in each geographic area are presented in Appendix H. In keeping with seabird colony catalogs (Sowls et al. 1980; Speich and Wahl 1989; Rodway 1991; Carter et al. 1992), we referred to specific colonies or subcolonies as they have been previously defined, which allowed easy cross referencing between sources.

We also summed whole-colony counts within "colony complexes" in California and Washington. We considered a colony complex to be a geographic subunit, composed of several colonies close together. Such subunits reflected major geographic assemblages of breeding murres that resulted from the distribution of suitable breeding habitat, accounted for the greater potential for interaction between nearby colonies, and accounted for inconsistent definitions of what constituted a colony in seabird colony catalogs (Sowls et al. 1980; Speich and Wahl 1989; Carter et al. 1992). We lumped adjacent rocks, islands or mainland cliffs with groups of breeding murres into colony complexes when they were within about 5 km of each other. Colony complex totals are presented in Appendix H.

Regression analysis has been used extensively to assess avian population trends and is a widely accepted method of demonstrating and measuring the rate of population change over a period of time (Sauer and Droege 1990). We used regression analysis to calculate rates of population change (percent per annum change with 95% confidence intervals) for each area within certain periods and to determine statistical significance for population trends identified in these periods (see Appendix H). We conducted linear regression analyses on single, annual sums of whole-colony counts in geographic areas over a period of years, only including standardized and compatible data. Because of availability of data and previously identified population changes, we conducted regressions over all years of available data between 1979 and 1995, as well as on series of years where our direct inspection of data indicated a distinct trend (i.e., increasing, decreasing, or no change). This approach led to the following regression periods: central California (1979–89, 1985–95, 1979–95), northern California (1979–89), Oregon (1988–95), and Washington (1979–86, 1984–95, and 1979–95). Significant regressions ($P < 0.050$) are reported in the text and presented in figures. Where significant trends were not detected, changes in whole-colony counts are discussed in the text and presented in figures.

For certain objectives, we estimated the number of breeding adults at a colony and summed colonies to determine the size of a breeding population. Whole-colony counts of murres can be adjusted with a k correction factor to convert whole-colony counts to either "number of breeding pairs" or "number of breeding individuals" (Nettleship 1976; Birkhead and Nettleship 1980; Sowls et al. 1980; Takekawa et al.

1990; Carter et al. 1992; Sydeman et al. 1997). We applied a *k* correction factor to estimate the number of breeding individuals. Sydeman et al. (1997) calculated *k* from data collected between 1985 and 1995 at the South Farallon Islands in central California. An average *k* of 1.671 (SE = 0.026; *n* = 11) was obtained with relatively little variation among years (see Appendix A). A *k* was not determined for any colony in northern California, Oregon, or Washington. A very different *k* was found for Triangle Island in British Columbia (see Appendix A). For all estimates of the number of breeding murres in California, Oregon, and Washington, we used a constant *k* correction factor of 1.67. This approach allowed rough estimation of population sizes for general comparisons. However, given concerns about potential variation in *k* correction factors between different parts of the breeding range, season, times of day, and years, we did not apply a *k* correction factor to whole-colony count data before examining population trends.

Summary of Population Data

California

Historical Background on Breeding Colonies in California, 1800–1978

The history of the common murre on the west coast of North America before 1900 is best documented in California. Settlement of southern and central California by the Spanish began in the eighteenth century, much earlier than European colonization farther north along the west coast of North America south of Alaska. Frequent activity by early settlers probably occurred at many colonies, mainly from the mid-nineteenth through the early twentieth centuries when rapid immigration occurred after California was ceded to the United States by Mexico in 1848. Except at the South Farallon Islands, little documentation is available. In northern California, native people may have occasionally visited certain murre colonies by canoe to obtain eggs or birds until the late nineteenth century when populations of native people were reduced to very low levels. Whereas diets of native people in northern California did not focus on seabirds, they did feed extensively on marine foods, which probably included seabirds on occasion (Heizer and Elsasser 1980). Such food gathering and hunting activities were limited to accessible offshore rocky stacks and islets. In central California, visitation of murre colonies by native people probably was infrequent because they did not use large ocean-going canoes in this coastal area. Native people were not known to visit the South Farallon Islands, which are located far from shore. Below, we present a brief synopsis of the known history of murre colonies in California, and we refer the reader to Appendix B for a detailed account with citation to historical literature. Given extensive historical changes in California murre populations, current population status and trends of murres must be viewed with these earlier events in mind.

At the South Farallon Islands, the harvest of murres and their eggs and the human occupation of the islands for nearly two centuries have greatly impacted the murre population. In 1818, the Russian sealing station on the South Farallon Islands (operated from 1812 to 1838) reported killing birds (probably murres) for meat and feathers. Egging was first reported in 1827. Commercial egging began in 1849, was made illegal in 1881, but continued until at least 1904. From 1850 to 1892, between 180,000 and 600,000 eggs were harvested annually, before falling to about 90,000 in 1896. Ainley and Lewis (1974) estimated that 400,000 birds may have bred at the South Farallon Islands, based on their review of egging records. However, our reinterpretation of historical records suggests numbers of murres were probably much higher, possibly between 1 and 3 million breeding birds (Appendix B). Hunting, egging, human occupation, and disturbance of these small islands, as well as heavy oil pollution, led to a dramatic decrease in the size of the murre colony at the South Farallon Islands. In 1909, the North Farallon Islands were included in the Farallon Reservation for Protection of Native Birds (later the Farallon National Wildlife Refuge). By 1911, there were fewer than 20,000 murres and very small numbers were reported in 1923, 1930, and 1933. Several thousands of murres died in the 1937 *Frank Buck* oil spill at the Golden Gate (Aldrich 1938; Moffit and Orr 1938). In the 1950s and 1960s, murre numbers at the South Farallon Islands grew and 6,718 were counted in 1959. In 1969, the South Farallon Islands were added to the Farallon National Wildlife Refuge. Additional protection from human disturbance was provided when the California Department of Fish and Game prohibited low overflights (although some still occurred) over the Farallon Islands Game Refuge in 1971. A detailed ground survey in 1972 revealed about 20,000–45,000 birds and the colony continued to increase to about 30,000–60,000 from 1975 to 1979. Estimates of population size varied widely owing to differences in census techniques, the degree of completeness of surveys, and irregular use of *k* correction factors. The increase between 1950 and 1982 reflects high levels of breeding success, reduction in human disturbance at the islands (especially since the early 1970s), and low levels of anthropogenic-related deaths at sea except for the 1971 San Francisco oil spill when many thousands died (Smail et al. 1972; Carter 1986; Boekelheide et al. 1990).

Few other islands in central and northern California were large enough for occupation by settlers (see Appendix I), but many colonies were accessible to people with small boats. Several colonies may have been extirpated during this period by egging and other activities. However, only the loss of colonies at Prince Island (c. 1912) and San Pedro Rock (c. 1908) are well documented. Egging was documented at other colonies, including the North Farallon Islands in the 1880s and 1890s, Point Reyes in 1897, and possibly Mendocino County in 1900. Egging probably occurred at colonies near settlements at Trinidad and Crescent City. In addition to egging, extensive disturbance and human access resulting from construction and operation of the Ocean Shore Railroad may have contributed to the loss of the San Pedro Rock colony. Similarly, egg gathering for private collections may have contributed to the loss of the Prince Island colony, the only location in southern California where murre eggs were known to be collected between 1885 and 1912. Extensive oil pollution in the early twentieth century probably affected all colonies in central California. Colonies in northern California also may have been affected by oil pollution, judging by observations of oiled murres on beaches in 1909–10 (C. I. Clay, unpublished field notes). Other murre colonies may have been extirpated by eggers or others before documentation in the Channel Islands and throughout the coasts of San Luis Obispo, Monterey, Sonoma and Mendocino Counties where appropriate breeding habitats exist. Murres were rarely seen in southern California before the 1960s until populations in central California began to increase, with some murres moving south after breeding (Pyle 1953; Unitt 1984; Lehmann 1994; Manuwal and Carter 2001).

The murre population in northern California seems to have increased markedly from the 1940s to the late 1970s following earlier decreases that resulted from activities of early settlers and use of certain islands by native people. A small colony was reported at Castle Rock between 1917 and 1935, but the population increased to 5,000–10,000 breeding pairs in 1956–61 and to 20,000–40,000 breeding pairs in 1970. In 1980, Castle Rock was included in the National Wildlife Refuge System. Increases at Castle Rock in the mid-twentieth century appear to reflect growth and recovery following use by native peoples, egging, and the use of the island for grazing by domestic animals. At Whaler Island, near Crescent City, breeding was documented in 1928. Since 1939, the island was partly quarried and a breakwater has connected it to the mainland allowing easy access by rats and humans, which has prevented breeding by most seabirds. Murres did not breed at Flatiron Rock from 1910 to 1934, but 1,000 breeding pairs were noted in 1969 and many thousands currently breed there. This colony is close to the long-settled port of Trinidad and would have been very accessible to commercial eggers by boat and native people by canoe (see Appendix I: Figure I-15). In fact, large numbers of eggs were collected in 1897–1901 from several unidentified islands, apparently in the Trinidad area, indicating that higher population levels may have existed at that time. Nearby Green Rock seemed to be the only murre colony that existed in the Trinidad area from 1917 to 1941, with about 2,000 birds noted in 1941. By 1969–70, murres were found at most colonies where they have been recorded regularly since 1979 (except for Mendocino County), indicating population increase between the 1940s and late 1960s.

Prior to the 1980s, certain California murre colonies outside of the Farallon and Castle Rock National Wildlife Refuges were protected within the Point Reyes National Seashore (i.e., Point Reyes, Point Resistance, Millers Point Rocks, and Double Point Rocks), Redwood National Park (i.e., False Klamath Rock and Sister Rocks), and Channel Islands National Park (i.e., Prince Island) in 1968, 1972, and 1980, respectively. Earlier, Prince Island had received partial protection when reserved for lighthouse purposes in 1917 and transferred to the U.S. Navy in 1934. Additional protection for murre colonies after 1980 in California are mentioned later in this chapter.

Current Population Size and Distribution of Breeding Colonies in California

By 1995, 26 murre colonies had been described in California, including 22 colonies used between 1979 and 1995 and 4 colonies extirpated earlier in the twentieth century (Appendixes B–D). The colonies separate into two groups: the northern California group consisting of 15 colonies in Del Norte, Humboldt, and northern Mendocino Counties (Figure 2.1); and the central California group consisting of 10 colonies in Marin, San Francisco, San Mateo, and Monterey Counties (Figure 2.2). One colony was previously reported in southern California (Santa Barbara County) at Prince Island, a record that represented the southern breeding limit known for the species in California and the world (Figure 2.2). Breeding has been confirmed with observations of eggs or chicks at all colonies in central California, the extirpated Prince Island colony in southern California, and most colonies in northern California (see Appendix B; Sowls et al. 1980, unpublished data archive; Boekelheide et al. 1990; Carter et al. 1992, unpublished data archive; McChesney et al. 1994; H. R. Carter, unpublished data; M. W. Parker, unpublished data; G. J. McChesney,

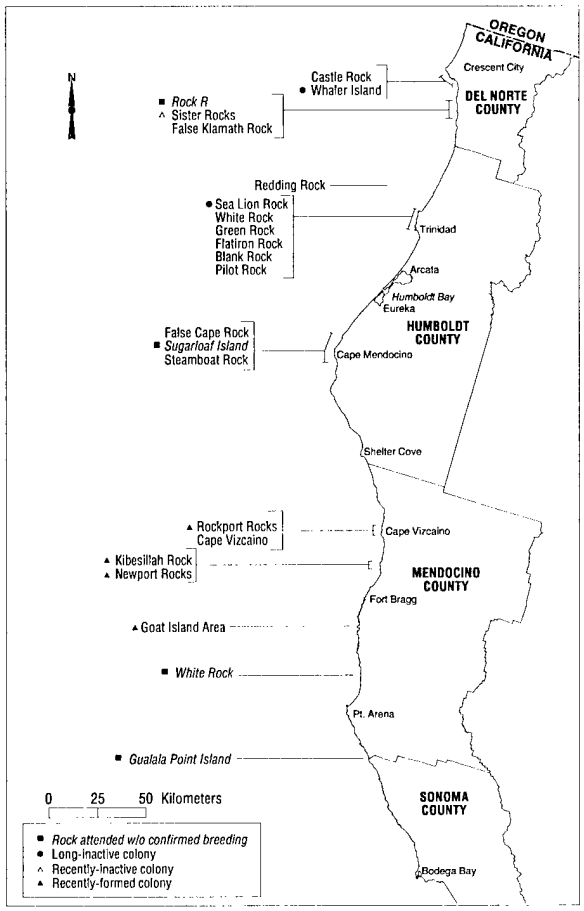

Figure 2.1. Distribution of common murre colonies in northern California (Del Norte to Sonoma Counties).

personal observation). Murres breed regularly at the South Farallon Islands, Point Reyes, and Castle Rocks and Mainland colonies where long-term studies of murre biology and reproductive success are under way (see Chapter 1; Boekelheide et al. 1990; Parker et al. 1997, 1998, 1999; McChesney et al. 1998, 1999).

The entire breeding population of murres in California was estimated in 1980, 1982, and 1989 (Sowls et al. 1980; Briggs et al. 1983; Carter et al. 1992). On the basis of summed, whole-colony counts for all colonies, with a *k* correction factor, we calculated total populations of 467,100, 514,900, and 351,600 breeding murres in 1980, 1982, and 1989, respectively. The central California population held 42, 43, and 26% of the total in each of the 3 years surveyed, respectively. The lower percentage in 1989 reflects extensive decline in central California from 1982 to 1989, and little change in northern California, except for lower revised estimates at Castle Rock (see later).

In central California, colonies can be grouped into six colony complexes—two offshore complexes at the South and North Farallon Islands (about 20–30 km from the mainland) and four nearshore complexes (i.e., coastal rocks within 1 km of the mainland and adjacent mainland cliffs) at Point Reyes, Points Resistance–Double, Devil's Slide, and Castle–Hurricane (Figure 2.2; Appendixes C and D). The largest colony complex was the South Farallon Islands where an estimated 102,700 murres bred in 1982 (Table 2.1). Whole-colony counts at the South Farallon Islands averaged 38,019 birds per survey from 1979 to 1995, which corresponded to an estimated 63,500 breeding birds (Table 2.1). Two other large colony complexes were at the North Farallon Islands and Point Reyes, which averaged 34,600 and 23,000 breeding birds, respectively, during the same time period. Breeding on inaccessible mainland points occurs only at Point Reyes and Castle Rocks and Mainland. Three smaller colonies (Point Resistance, Millers Point Rocks, and Double Point Rocks) exist south of Point Reyes within the Points Resistance–Double complex. Colonies exist south of San Francisco at the Devil's Slide complex (including the Devil's Slide Rock and Mainland colony and the long-inactive colony at San Pedro Rock), as well as at the Castle–Hurricane complex (Figure 2.2). The southernmost colony in California in 1979–95 was Hurricane Point Rocks.

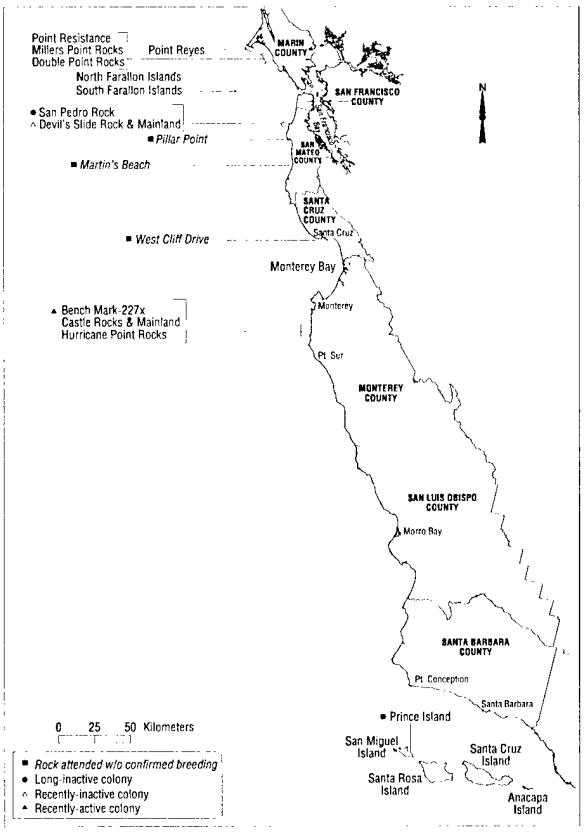

Figure 2.2. Distribution of common murre colonies in central California (Marin to Monterey Counties) and southern California (Santa Barbara County).

Table 2.1. Average and maximum sizes for nine breeding colonies of common murres in central California, 1979–1995 (see Appendixes C and D).

Rank[a]	Colony name	Mean count[b]	Mean number of breeding adults[c]	Years of data	Maximum count	Maximum number of breeding adults[c]	Year of maximum
1	South Farallon Islands	38,019	63,500	12	61,510	102,700	1982
2	North Farallon Islands	20,717	34,600	12	31,428	52,500	1980
3	Point Reyes	13,755	23,000	13	26,337	44,000	1982
4	Double Point	4,116	6,900	13	8,850	14,800	1980
5	Point Resistance	3,046	5,100	13	4,440	7,400	1980
6	Castle Rocks & Mainland	1,140	1,900	9	2,275	3,800	1980
7	Hurricane Point Rocks	692	1,200	10	1,500	2,500	1981
8	Devil's Slide Rock & Mainland[d]	446	700	11	1,750	2,900	1980
9	Miller's Point Rocks	256	400	9	713	1,200	1995

[a] Ranked in order of mean colony size.

[b] Only sites with at least five years of data considered suitable for trend analysis were included. Lower quality data for certain years and colonies were not included.

[c] Number of breeding adults was obtained by multiplying mean or maximum count by a *k* correction factor of 1.67, and rounding to the nearest hundred.

[d] No breeding occurred from 1986 to 1995 (see text).

A new colony, "Bench Mark-227x," was temporarily established within the Castle–Hurricane complex in 1996–98 but subsequent breeding did not occur in 1999–2001 (Parker et al. 1998, 1999; McChesney et al. 1999; M. W. Parker, unpublished data). In March–June 1999, 3–9 murres attended but did not breed at Prince Island, in association with nesting Brandt's cormorants (*Phalacrocorax penicillatus*; H. R. Carter, unpublished data). Use of the Bench Mark-227x colony area had not been noted previously, but murres had been recorded in the vicinity of Prince Island since 1991 (Carter et al. 1992; McChesney et al. 1995).

In northern California, colonies are most often on offshore rocks within 1 km of the mainland, except for the small isolated colony at Redding Rock — 7 km offshore. The largest colony complex in northern California (and the state) in recent decades was at Castle Rock, where 142,400 breeding birds were estimated in 1982 (Briggs et al. 1983; Appendix C). However, 1979–82 counts may have overestimated the size of this dense colony, which was estimated to be about 100,000 and 107,700 breeding birds in 1986 and 1989, respectively (Takekawa et al. 1990; Carter et al. 1992). Four other large colony complexes are found at False Klamath, Trinidad (including colonies at White Rock, Green Rock, Flatiron Rock, Blank Rock, and Pilot Rock), Cape Mendocino (including False Cape Rocks and Steamboat Rock colonies), and Vizcaino (including Cape Vizcaino and Rockport Rocks colonies; Table 2.2). Mainland breeding occurs only at one subcolony on an inaccessible point at Rockport Rocks. Smaller colonies are present at Sister Rocks (within the False Klamath complex) and Redding Rock. By 1995, the southernmost colony (where breeding was certain) in northern California was Cape Vizcaino. However, in 1997, breeding was confirmed at three small colonies (Newport Rocks, Kibesillah Rock, and Goat Island Area) south of Cape Vizcaino in northern Mendocino County where attendance had been noted in recent years (Carter et al. 1992, 1996; see below).

Between 1979 and 1995, murres attended several rocks in California where breeding was not confirmed (Sowls et al. 1980, unpublished data; Briggs et al. 1983;

Table 2.2. Average and maximum sizes for eight breeding colonies of common murres in northern California, 1979–1995 (see Appendixes C and D)[a].

Rank	Colony name	Mean count	Mean number of breeding adults	Years of data	Maximum count	Maximum number of breeding adults	Year of maximum
1	False Klamath Rock	26,650	44,500	6	31,801	53,100	1982
2	Green Rock	24,327	40,600	6	32,934	55,000	1980
3	Flatiron Rock	16,799	28,100	8	25,494	42,600	1995
4	False Cape Rocks	8,847	14,800	7	12,426	20,800	1995
5	Cape Vizcaino	4,194	7,000	6	4,950	8,300	1995
6	Steamboat Rock	4,089	6,800	5	5,454	9,100	1989
7	White Rock	2,614	4,400	5	3,277	5,500	1981
8	Redding Rock	923	1,500	6	1,632	2,700	1989

[a] The largest colony at Castle Rock was excluded (see text). Symbols and format as in Table 2.1.

Carter et al. 1992, 1996; Appendixes C and D). Near colonies in northern and central California, such attendance has been noted at Rock R, Sugarloaf Island, and Martin's Beach (Figures 2.1 and 2.2; Appendixes C and D). In addition, such attendance was recorded south of known breeding areas in Mendocino and Sonoma Counties at Newport Rocks, Kibesillah Rock, Goat Island Area, White Rock, and Gualala Point Island (Figure 2.1; Appendixes C and D). Briggs et al. (1983) also noted murres on Bruhel Point Rocks (herein referred to as Newport Rocks). H. L. Cogswell (unpublished field notes) also noted at least 30 murres "resting on coastside rock in ocean below sea cliff" at Pillar Point or Moss Beach on 27 November 1952 and at least 6 murres on "a small rocky islet offshore" of West Cliff Drive at Santa Cruz on 23 July 1967 (Figure 2.2).

Carter et al. (1992) classified attendance at Rock R and Goat Island Area in 1989 as newly-formed colonies without determining whether eggs were laid. Here, we reclassified these observations as "attendance without confirmed breeding." Breeding was ultimately verified at Kibesillah Rock, Newport Rocks, and Goat Island Area on 12-13 July 1997, when about 5–10 chicks and clumps of other birds in incubation or brooding postures were observed by telescope from the mainland (G. J. McChesney, personal observation). In retrospect, these colonies appearred to be forming during the 1989–95 period. Numbers of murres at Newport Rocks and adjacent Kibesillah Rock increased from 7 birds in 1993 to 542 birds in 1995 (Carter et al. 1996). Similarly, small numbers of birds were noted at Goat Island Area in 1989, 1994, and 1995 and at Rock R in 1980, 1989, and 1994 (Sowls et al. 1980; Carter et al. 1992, 1996). From 1989 to 1995, birds were present in clumps or rows, with some individuals in incubation postures (as seen in aerial photographs), which suggests possible breeding. In addition, one murre was observed carrying a fish (possibly to feed a chick or for courtship) in flight to the Goat Island Area in June 1989 (Carter et al. 1992; unpublished survey data). However, breeding probably was not occurring at Newport Rocks (1993–95) and Sister Rocks (1989–95), where all birds were standing and scattered during aerial photographic surveys.

Population Trends in Central California, 1979–1995

From 1979 to 1982, overall numbers of murres attending colonies in central California increased (Figures 2.3 and 2.4; Appendix C; Sowls et al. 1980; Briggs et al. 1983). This increase was well documented, mainly at the largest colony at the South Farallon Islands where boat and ground surveys also documented the increase (Boekelheide et al. 1990; Takekawa et al. 1990; Sydeman et al. 1997). At the South Farallon Islands, the increase reflected part of a long-term increase that began in the 1950s (Appendix B; Carter 1986; Boekelheide et al. 1990; Sydeman et al. 1997). No increase occurred at the nearby North Farallon Islands during that time, possibly because of total occupation of more limited available breeding habitat (see Appendix I: Figure I-9) and lower levels of past human disturbance compared to the South Farallon Islands. Counts at Point Reyes and the Points Resistance–Double complex varied, but also seemed to increase from 1979 to 1982. The effects of low overflights by aircraft (and possibly close approach by boats) may have contributed to this variation in numbers. However, the Gulf of the Farallones National Marine Sanctuary was created in 1981, which prohibited low overflights (below 1,000 feet or 305 m) over the colonies. McChesney et al. (1998) also clarified that counts in 1979–81 at Point Reyes underestimated numbers of murres present because of incomplete and low-quality photographs. Taking this into account, little change was evident at Point Reyes between 1979 and 1982. Between 1980 and 1982, murre numbers were reduced at the Devil's Slide complex and decline was evident at the Castle–Hurricane complex.

Between 1979 and 1989, all colony complexes in central California underwent large declines of 8.7 to 28.5% per annum, ($0.001 < P < 0.020$; Figure 2.3; Appendix H). The overall population declined 9.9% per annum ($P = 0.002$; Figure 2.4; Appendix H). Most decline occurred between 1982 and 1985, as further verified with ground-based observations at the South Farallon Islands (Boekelheide et al. 1990; Takekawa et al. 1990). Plot observations and ground and boat surveys at the South Farallon Islands showed low attendance and low breeding success during the severe El Niño-related breeding conditions in 1983–84 (Boekelheide et al. 1990; Takekawa et al. 1990; Sydeman et al. 1997). Although no aerial photographs were taken in 1983–84, numbers at colonies in 1985 and 1986 after breeding conditions had returned to normal were still much lower than in 1981 and 1982 (Appendix C). By 1986, the Devil's Slide Rock and Mainland colony had essentially disappeared with between 0 and 128 murres in 1986–87 and 0–5 murres in 1988–95.

Between 1987 and 1990, counts at most colonies reached their lowest levels compared to 1981–82 (Figure 2.3; Appendix D). The Castle–Hurricane complex reached the lowest level of all extant colony complexes (1,047 and 1,093 birds counted in 1988 and 1989, respectively) with loss of subcolonies and only small groups of birds on several remaining subcolonies. The small Millers Point Rocks colony (within the Points

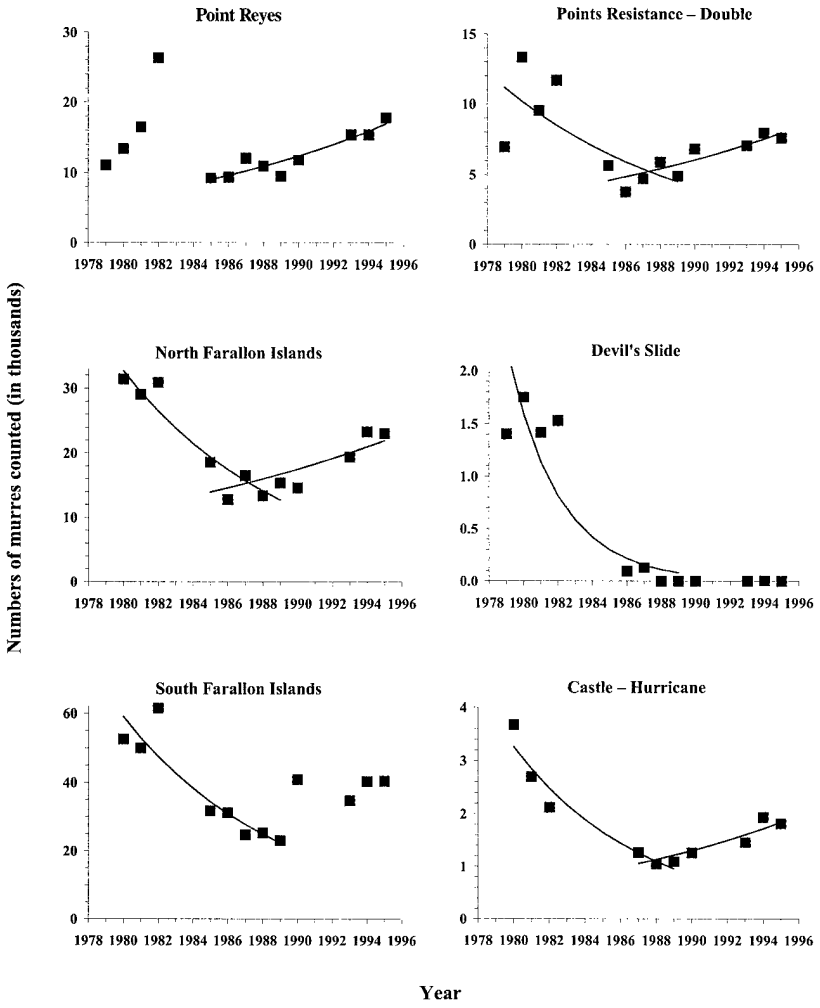

Figure 2.3. Trends in whole-colony counts for six colony complexes of common murres in central California, 1979–1995 (see Appendix H).

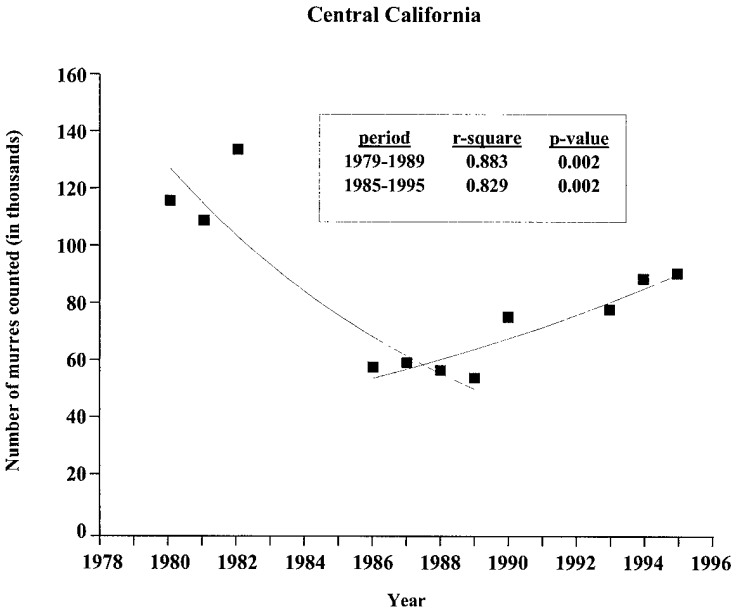

Figure 2.4. Trends in whole-colony counts for common murres in central California, 1979–1995 (see Appendix H).

Resistance–Double complex) may have been nearly extirpated during this decline since only 23 birds were found in 1987. However, this colony grew to 380 birds by 1990, possibly because of intercolony movements from nearby colonies within this complex. However, a remarkable increase occurred at the South Farallon Islands between 1989 and 1990. In retrospect, this upswing signaled the end of the decline and the start of an increase for the central California population. Ground and boat surveys and plot counts from 1989 to 1995 at the South Farallon Islands also confirmed the end of decline by 1990 (Sydeman et al. 1997, 1998).

The decline of the central California population between 1979 and 1989 and the loss of the Devil's Slide Rock and Mainland colony have been attributed mainly to extensive gill-net and oil-spill deaths, and reduced productivity related to the severe 1982–83 El Niño (Carter 1986; Carter and Ainley 1987; Salzman 1989; Takekawa et al. 1990; Wild 1990; Piatt et al. 1991; Swartzman and Carter 1991; Carter et al. 1992, 1995; Ainley et al. 1994; Sydeman et al. 1997; McChesney et al. 1998, 1999). More than 75,000 murres died in 1979–87 in central California as a result of gill-net fisheries (Takekawa et al. 1990). High mortality was attributed to the consistent spatial and temporal overlap of large numbers of feeding murres and high gill-net fishing effort in nearshore waters of Monterey Bay, Gulf of the Farallones, and Bodega Bay area from 1980–86. Most severe declines occurred at colonies located nearest areas of highest gill-net mortality. Two major oil spills occurred during this period and killed more than 8,000 murres. In November 1984, the *Puerto Rican* oil spill occurred off the Golden Gate, killing 1,500–2,000 murres (PRBO 1985; Ford et al. 1987). Mortality probably was focused on large colonies at the Farallon Islands and Point Reyes. In January–February 1986, the *Apex Houston* oil spill occurred between San Francisco and Monterey Bay, killing 6,300–7,500 murres (Page and Carter 1986; Ford et al. 1987; Page et al. 1990; Siskin et al. 1993). Mortality probably was spread more widely over all colonies with greatest impacts at Devil's Slide Rock and Mainland, Castle Rocks and Mainland, and Hurricane Point Rocks. The loss of the Devil's Slide Rock and Mainland colony (first noted in June 1986) was associated with this mortality, although earlier gill-net mortality had reduced the colony beforehand (Takekawa et al. 1990; Piatt et al. 1991; Swartzman and Carter 1991). Many smaller spills also killed thousands of murres between the late 1970s and 1989 (Stanzel et al. 1988; Carter 1997; Nur et al. 1997).

Low productivity in the 1982–83 El Niño undoubtedly affected the ability of the central California population to recover in the late 1980s. However, it was not possible to detect whether or not increased deaths of adult or subadult murres resulted during severe El Niño-induced winter weather conditions in 1982–83. At this time, high numbers of murres killed in gill nets were washing up on beaches but the cause of death for many nonoiled beached birds could not be determined (Stenzel et al. 1988). A small part of the reported decline at certain colonies may have been related to (1) methodological differences between surveys in 1979–82 and 1985–89, (2) undocumented human disturbances from low overflights and boats, or (3) depredation at colonies by peregrine falcons (*Falco peregrinus*) and common ravens (*Corvus corax*; Sydeman 1993; McChesney et al. 1998, 1999; M. W. Parker, unpublished data).

The marked decline in the central California murre population between 1979 and 1989 far outweighed the relatively small increase by 1995 after this decline. Between 1985 and 1995, the total population increased 5.9% per annum ($P = 0.002$), whereas colony complexes increased between 4.6 and 7.2% per annum ($0.001 < P < 0.020$), excluding the extirpated Devil's Slide complex (Figures 2.3 and 2.4; Appendixes C, D, and H). In 1992, severe El Niño breeding conditions occurred and murre attendance at the South Farallon Islands was low (Sydeman et al. 1997). However, whole-colony counts at the South Farallon Islands and Points Resistance–Double complexes were still higher in 1993 than in 1987–89. The North Farallon, Point Reyes, and Points Resistance–Double complexes increased after 1990. At the Castle–Hurricane complex, increase was not noted between 1987 and 1993, but higher numbers did occur in 1994–95. Overall, increases that began at the South Farallon Islands in 1990, and later at most other colonies, were sustained despite interruption by the severe 1992–93 El Niño. Highest colony complex counts in the 1993–95 period were still lower than peak counts in the 1979–95 period (Figure 2.3; Appendixes C and D). Increases in 1993–95 at the largest complexes (i.e., South Farallon Islands, North Farallon Islands, Point Reyes, and Points Resistance–Double) are encouraging, but natural recovery of the central California population to 1979–82 levels may require at least another decade without additional major detrimental effects. Human disturbance at colonies has been reduced greatly through additional regulations and enforcement. However, sporadic disturbance events continue. For example, extensive disturbance to breeding murres resulted from low overflights by a U.S. Coast Guard helicopter responding to the grounding of the M/V *Wayfarer* at Point Reyes in 1995 (McChesney et al. 1998; Thayer et al. 1998, 1999).

Funds from the settlement of the *Apex Houston* oil spill litigation were used for a restoration project at the Devil's Slide Rock and Mainland colony (Graham 1996; Parker et al. 1997, 1998, 1999; Helmuth 1999; Parker 1999). Breeding did not occur at this colony between 1986 and 1995. Using social attraction techniques, the USFWS, Humboldt State University, and the National Audubon Society restored breeding by small numbers of murres at this colony in 1996–2000 (i.e., increasing to 98 breeding pairs by 2000)

The southernmost colony at Hurricane Point Rocks also did not increase significantly between 1987 and 1995. The increase in the Castle–Hurricane complex in 1993–95 has occurred mainly at the Castle Rocks and Mainland colony. Both of these colonies are geographically isolated from other colonies in central California, were affected greatly during the decline, and remain susceptible to extirpation. Slow recovery at these colonies by 1995 may reflect poor breeding success, immigration, and continued anthropogenic effects (especially gill-net deaths and human disturbance; Julian and Beeson 1995; Carter et al. 1998; McChesney et al. 1999; M. W. Parker, unpublished data). Establishment of the California Islands Wildlife Sanctuary in 1983, which prohibited disturbance of seabirds and marine mammals, may have reduced human disturbance at the Castle–Hurricane and Devil's Slide colony complexes. In 1992, the Castle–Hurricane and Devil's Slide complexes were provided more protection from human disturbance through the creation of the Monterey Bay National Marine Sanctuary that prohibited most overflights below 305 m (1,000 feet). However, disturbances from low-flying aircraft still continue. Depredation by peregrine falcons does not seem to be seriously affecting these colonies (M. W. Parker, unpublished data).

In central California, slow population recovery since 1990, and no recovery at certain colonies, probably resulted from long-term and extensive anthropogenic effects, especially mortality from gill nets and oil spills and human disturbance from 1979 to 1987. Natural factors (i.e., reduced breeding effort and success during the severe 1982–83 El Niño) contributed to the decline and also increased recovery time. Between 1988 and 1995, the effects of deaths from gill-nets and oil-spills continued, but at reduced levels compared to 1982–88 (Julian and Beeson 1995; Sydeman et al. 1997; McChesney et al. 1998, 1999). High breeding success at the South Farallon Islands has occurred throughout 1979–95, except during severe El Niños in 1983–84 and 1992–93 (Boekelheide et al. 1990; Sydeman et al. 1997). Thus, the increase seems mainly a result of reduced anthropogenic factors. However, continuing low-level anthropogenic effects from oil pollution, gill-net fishing, and human disturbance may limit recovery. Reduced breeding effort and success during recent El Niños (i.e, 1992–93 and 1997–98) also may slow the rate of recovery. If long-term climate change has caused a significant reduction of prey resources, this factor also may have influenced changes observed and slowed recovery during the 1990–95 period.

Population Trends in Northern California, 1979–1995

From 1979 to 1982, numbers of murres attending many colonies in northern California increased (Figures 2.5 and 2.6; Appendix C; Sowls et al. 1980; Briggs et al. 1983). Increases were noted at all colony complexes, except Trinidad (including Flatiron and Green rocks) which remained relatively stable despite much variation at individual colonies. Some methodological differences between researchers in 1979–80 (Sowls et al. 1980) and 1980–82 (Briggs et al. 1983) may have slightly affected survey results reported for these two periods. In addition, 1981 data quality may have been lower at several colonies (K. T. Briggs, personal communication).

Few anthropogenic or natural factors were documented to affect colonies in northern California at this time. In 1980, Castle Rock received protection through designation as a National Wildlife Refuge, although occasional low overflights may have continued. Both False Klamath Rock and Sister Rocks are located within Redwood National Park, which may have contributed to some disturbance from low overflights related to park viewing. Variations in numbers of murres counted at Redding Rock in 1979–82 (Appendix C) probably reflected disturbance from U.S. Coast Guard crews servicing an automated light on this site during the breeding season. This source of disturbance was first noted in 1979 (Sowls et al. 1980; unpublished survey data), but probably occurred earlier. In addition, California sea lions (*Zalophus californianus*) "haul out" high up on this rock and may adversely affect breeding success of murres in some years (see Appendix I: Figure I-18; H. R. Carter and M. W. Parker, personal observations). Variable patterns within the Trinidad complex appear to represent intercolony movements between five nearby colonies. In fact, corresponding changes in murre numbers at Flatiron Rock and Green Rock (i.e., two large and adjacent colonies in the Trinidad complex) were recorded between May and July surveys in 1980–82 (Figure 2.5; Appendix C). Reasons for intercolony movements were not determined, but hundreds of small dead murre chicks were found on Flatiron Rock on

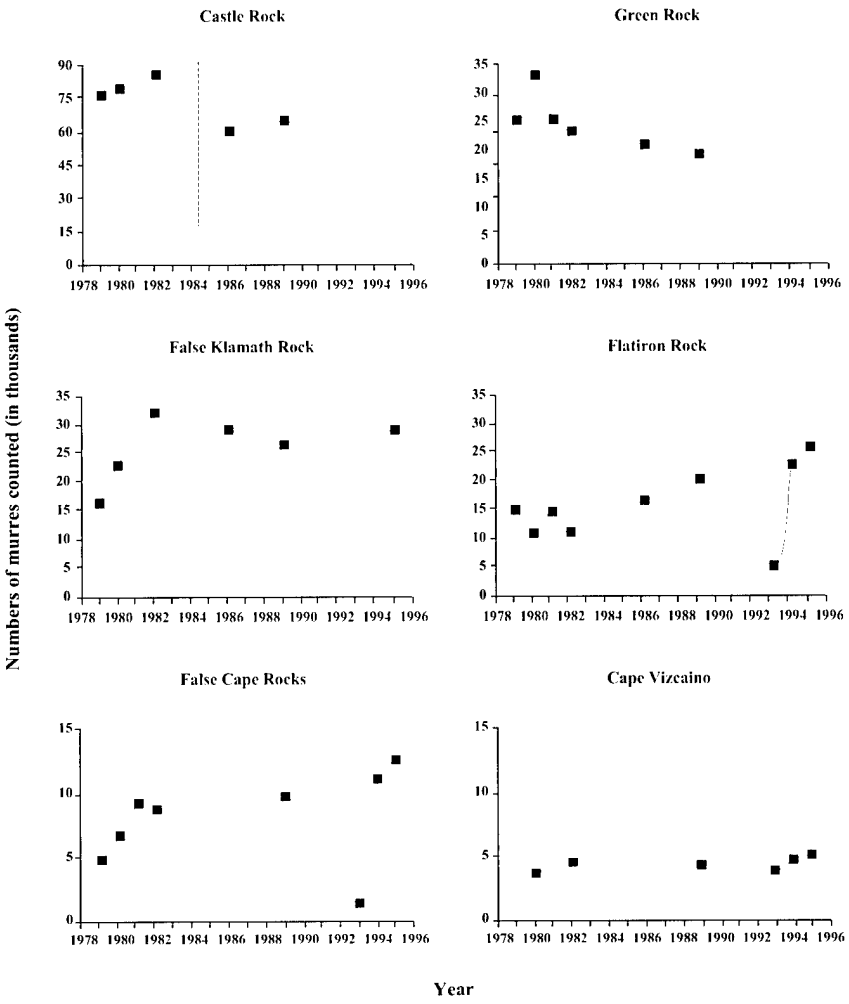

Figure 2.5. Changes in whole-colony counts of common murres at selected colonies in northern California, 1979–1995 (see Appendixes C and D). At Castle Rock, 1979–1982 counts were not considered to be comparable to 1986–1989 counts and are separated by a dashed line. A decline between these periods has not been inferred (see text).

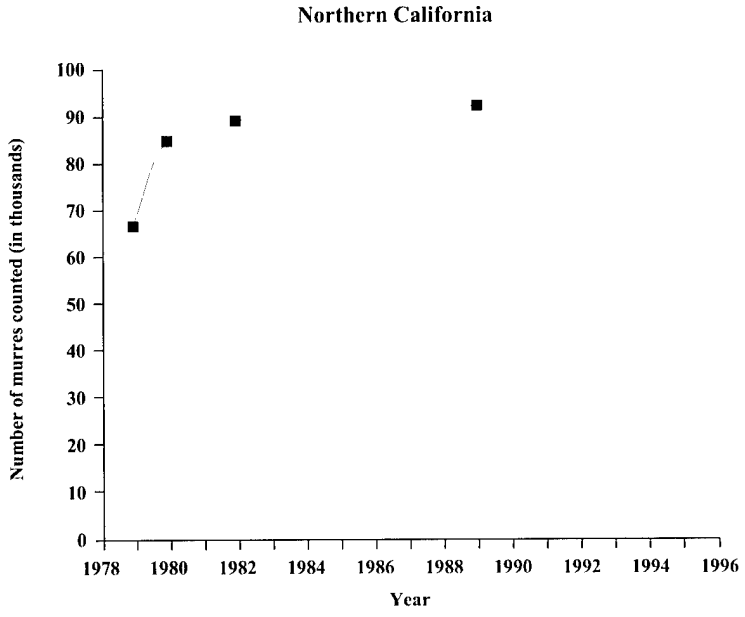

Figure 2.6. Changes in whole-colony counts of common murres in northern California, 1979–1989, excluding the Castle Rock colony (see Appendix H).

21 August 1980 (Sowls et al. 1980, unpublished survey data).

Sowls et al. (1980) reported four suspected new colonies in 1979–80 not thought to have been present in 1969–70 (Osborne and Reynolds 1971; Osborne 1972): Sister Rocks, Blank Rock, Pilot Rock, and Cape Vizcaino. However, we considered that some or all of these colonies may have been overlooked in earlier surveys (i.e., these focused on large colonies) or sporadic attendance may have reflected intercolony movements within colony complexes. For instance, murres were noted by other observers at Blank Rock in March 1965 and at Pilot Rock in 1966–69 (Appendix B). No birds were reported at Blank Rock in 1979, but small numbers were present in 1980–81. On the other hand, Cape Vizcaino was surveyed in August 1969, at a time when breeding may have been finished. We considered small numbers of murres attending Sister Rocks in 1980–82 to represent an active colony because about 30 "brooding" birds were reported on 20 June 1980 and birds were present on 19 May and 25 July 1980 (Sowls et al. 1980, unpublished survey data). However, no birds were reported there on 22 May 1979 and breeding has not been confirmed subsequently.

Overall, the increase in the northern California population (1979–82) seemed to reflect continuation of a long-term increase over several decades, owing to a reduction in levels of human disturbance (Appendix B). For instance, from 1970 to 1979, murre counts increased at Castle Rock (i.e., from about 32,000 to 76,000 birds), Green Rock (i.e., from about 20,000 to 25,000 birds), and Flatiron Rock (i.e., from about 5,000 to 15,000 birds). Although counting techniques were not directly comparable between 1970 and 1979, substantial increase seemed to have occurred during this extended period with continued increases through 1982.

Between 1979 and 1989, little change was noted at many colony complexes in northern California (Figures 2.5 and 2.6). A notable exception was Castle Rock where numbers were much lower in 1986 and 1989 than in 1979–82 (Figure 2.5; Appendixes C and D). However, upon inspection of archived aerial photographs (J. E. Takekawa, H. R. Carter, and K. T. Briggs, personal communication), there was no visible difference in breeding densities or in breeding areas used (Takekawa et al. 1990). Differences seemed to be related primarily to different aerial survey methods used at this large colony (see Appendix I: Figure I-20). In 1979–82, few photographs were taken per survey and numbers were estimated roughly within blocks of high-density murres. In 1986 and 1989, many photographs per survey provided better viewing of all parts of the colony and all murres were counted individually. Survey and counting methods used in 1986 and 1989 were considered to more accurately reflect colony size (Takekawa et al. 1990; Carter et al. 1992). The severe 1982–83 El Niño may have caused lower attendance and breeding success at Castle Rock in 1983, but the lack of a large or sustained decline at most other colonies from 1982 to 1986 supports the view that Castle Rock probably had not declined to a large degree.

In 1983, establishment of the California Islands Wildlife Sanctuary, which prohibited disturbance to seabirds and marine mammals, may have reduced human disturbance at several murre colonies, especially in the Trinidad and Cape Mendocino colony complexes. In 1989, total numbers in the Trinidad complex were similar to 1980–81, (Appendixes C and D). Continued variation in murre numbers occurred at Redding Rock (i.e., low numbers in 1986 and high numbers in 1989), probably reflecting continued disturbance by U.S. Coast Guard personnel. Of interest, Redding Rock was not specifically identified as "withdrawn for lighthouse purposes" when the California Islands Wildlife Sanctuary was created in 1983. The Cape Mendocino and Cape Vizcaino complexes increased from 1982 to 1989. Most growth within the Cape Vizcaino complex occurred at the newly recolonized Rockport Rocks colony where breeding was first noted in 1989.

The northern California population remained relatively stable from 1979 to 1989 (Figure 2.6; Takekawa et al. 1990; Carter et al. 1992; Appendix H). In fact, total whole-colony counts (excluding Castle Rock) were similar in 1982 (88,962) and 1989 (92,080). By not considering early survey problems to be significant at Castle Rock nor examining trends at other colonies, other sources have indicated that the northern California population (or Castle Rock colony) declined greatly between 1979 and 1989 (Ainley et al. 1994; Jaques and Strong 2001). However, as noted above, the large change in numbers at Castle Rock between 1982 and 1986 was not visually evident in aerial photographs (Takekawa et al. 1990). In central California, major declines between 1982 and 1985–86 were obvious in a comparison of aerial photographs (Carter and Ainley 1987; Takekawa et al. 1990; McChesney et al. 1998, 1999). We considered data at Castle Rock to be reasonably comparable within the periods 1979–82 and 1986–89, but not between these periods. Additional efforts are needed to evaluate comparability of data sets and trends at Castle Rock, especially reexamining 1979–82 photographs and counting archived aerial photographs for several years between 1985 and 1995 (Appendixes C and D).

In 1993–95, all colonies were surveyed, but aerial photographs were counted only at False Klamath Rock, Flatiron Rock, False Cape Rocks, and Cape Vizcaino (Carter et al. 1996; Appendix D). Combined murre numbers at these colonies increased from 1989 to 1995, despite the severe 1992–93 El Niño when most murres abandoned colonies, except at Cape Vizcaino. At False Klamath Rock, little change occurred between 1986 and 1995. By 1995, all four colonies had reached their highest recorded levels, exceeding peak counts in 1979–82 (Figure 2.5;Appendixes C and D). At Redding Rock, numbers of murres observed during aerial photographic surveys have declined from 1989 to 1995, although photographs have not been counted. This colony may be extirpated in the near future because of chronic human disturbance by U.S. Coast Guard personnel.

Population stability or limited increase in northern California from 1979 to 1995 may have resulted from three main factors, this region may be nearing the murre carrying capacity of available breeding habitat and prey resources, severe El Niños and other natural events have not had long-term effects, and anthropogenic effects have not been extensive. Murres currently use much of the available and suitable breeding habitat on all large islands in Del Norte and Humboldt counties, although breeding densities could increase further (see Appendix I). The only large islands with substantial breeding habitat that lack murre colonies north of Cape Vizcaino are Hunter Rocks, Prince Island (at the Smith River), and Sugarloaf Island. Although all three islands have colonies of Brandt's cormorants, past and present human disturbance may prevent breeding by murres. In 1912, Prince Island and Hunter Rocks were assigned to the Tolowa tribe, and native people periodically visit these islands. Sugarloaf Island is occasionally visited by climbers and low overflights occur frequently. Human disturbance has occurred regularly at Redding Rock, but has not been well documented (Lowe 1993). During surveys in May (1980 and 1989), murres were observed being flushed from Green Rock and False Cape Rocks by U.S. Coast Guard aircraft flying at or below 152 m (500 feet) elevation (Sowls et al. 1980; Carter et al. 1992, unpublished survey data). Few predators are known to affect murres at northern California colonies. Few peregrine falcons and bald eagles (*Haliaeetus leucocephalus*) are present, although numbers of falcons have been increasing. On several dates in 1980, extensive egg predation by common ravens was noted at False Klamath Rock, causing colony disruptions (Sowls et al. 1980; unpublished survey data). Although few oil spills occurred in northern California by 1995, two recent oil spills near Humboldt Bay (1997 *Kure* and 1999 *Stuyvesant*) killed large numbers of murres (P. R. Kelly, personal communication).

High numbers of breeding birds at colony complexes at Cape Mendocino and Trinidad Area may have contributed to the production of source birds that recolonized the Cape Vizcaino and Rockport Rocks colonies between 1969 and 1989. Between 1877 and 1942, log loading operations at and around Cottaneva Wharf, which extended directly onto Rockport Rocks, probably caused the earlier extirpation of these colonies (Appendix B). Recolonization and subsequent growth at the Cape Vizcaino colony complex may have contributed to the production of source birds for more recent colony formations at Newport Rocks, Kibesillah Rock, and Goat Island in the mid-1990s. Such colony formations in Mendocino County apparently occurred over several decades of favorable conditions.

Oregon

Historical Background on Breeding Colonies in Oregon, Prior to 1980

Before the arrival of settlers from Europe and the United States, native people occupied many locations along the Oregon coast. Shellfish, fish, seabirds, and marine mammals were of great importance in the diet of native people (Berreman 1944; Heflin 1966; Gould 1966, 1976; Zontek 1983; Minor et al. 1987; Lyman 1988, 1989, 1991; Gard 1990, 1992). Large mainland village sites were associated with offshore rocks (Chase 1873; Schumacher 1877a, 1877b; Berreman 1944; Ross 1977). The pursuit and harvest of these food resources by native people undoubtedly had great influence on seabird colonies. Native people may have regularly visited certain accessible murre colonies (especially near village sites) by canoe to obtain eggs or birds. All known colonies are located close to former village sites or seasonal camps (Figure 2.7). Even colonies 4.8–6.4 m (3–4 miles) offshore on Orford Reef (i.e., Redfish Rocks, Colony numbers 270-043 to 270-047; Figure 2.7) could have been reached by local residents during calm ocean periods and were probably exploited on occasion for food.

Some rocks and islands were actually occupied by native people, at least seasonally. At Goat Island, shellfish remains were the most common items found in a large midden (radiocarbon dated to 880 $\pm$ 70 b.p.), along with small numbers of bones of marine mammals, fish, and seabirds (though not murres; Gard 1990, 1992). Murres probably did not breed on Goat Island during coastal occupation by native peoples because the entire island is easily accessible to humans. Seasonal

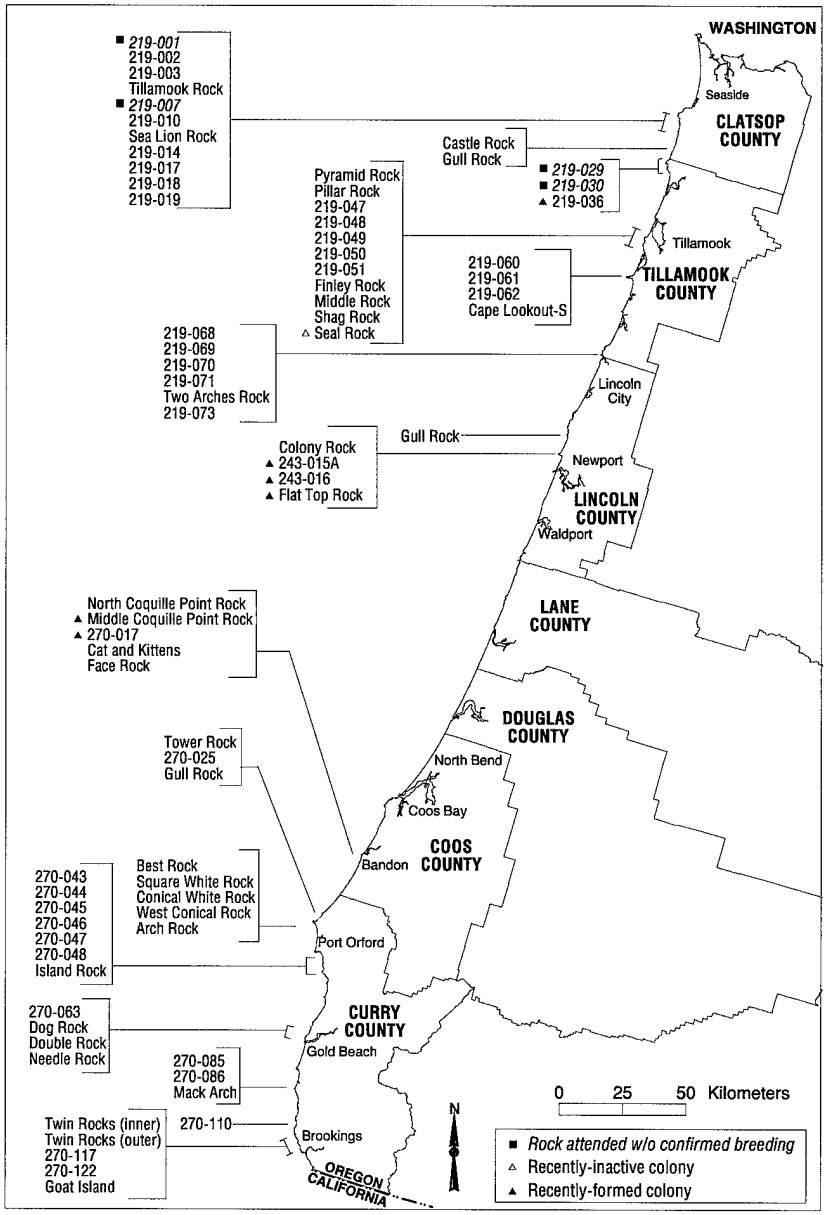

Figure 2.7. Distribution of common murre colonies in Oregon (Clatsop to Curry Counties).

occupation by people also occurred at an unnamed rock near Whaleshead Creek, Curry County (Colony number 270-110), where a large murre colony has occurred at least since the 1950s (see below). At Yaquina Head, near Newport, archaeological investigations of midden sites on the mainland included bones of cormorants, gulls, albatross, and loons, but not murres. Colony Rock, just northwest of Yaquina Head (Colony number 243-015), is connected to the mainland during low tides and would have been accessible to native people. Apparently, murres began nesting at this site in the 1940s or 1950s (see below). Radiocarbon dating of cultural material from various islands and mainland locations indicated that coastal rocks and islands were used for food gathering by native peoples for thousands of years.

Most murre colonies known in 1988 (68%; $n = 66$) are considered accessible by climbing and these support about 90% of the Oregon murre population. Thus, murre numbers probably were much lower during occupation by native people and may have been at lowest levels in recent centuries when settlers arrived.

After Euro–American settlers arrived, native people were decimated by disease, then forcibly relocated to centralized reservations (Gard 1990). The elimination of subsistence harvest and human occupation on rocks and islands probably allowed the Oregon murre population to slowly expand and colonize new locations over time. As early as 1892, murre eggs were harvested along the southern Oregon coast by early settlers. Two

local men began a business to harvest murre eggs from rocks off Humbug Mountain, where "The murre, which a few years ago was not known to exist north of Cape Mendocino are now to be found off Humbug by thousands (*Port Orford Tribune*, 17 May 1892)." Murre colonies on Island Rock (Colony number 270-049) and Redfish Rocks, west of Humbug Mountain, apparently were the targets of this harvest. Harvest of murre eggs on Island Rock and Blanco Reef (probably Redfish Rocks off Cape Blanco) also occurred by 1901 (*Port Orford Tribune*, 11 June 1901), and in 1909 it was reported that "170 dozen eggs [2,040 eggs] were collected in one forenoon's work" (*Port Orford Tribune*, 9 June 1909). In the late nineteenth and early twentieth centuries, an average of 700 dozen murre eggs (8,400 eggs) were gathered each year at Island Rock and Redfish Rocks (Centennial Edition of the *Coos Bay Times*, 3 May 1947). Thus, thousands of murres evidently bred on the rocks and islands near Port Orford at this time, a number sufficient to support commercial harvesting of eggs for at least a decade.

At the start of the twentieth century, W. L. Finley (1902; 1905a, 1905b) documented a large colony of breeding murres at Three Arch Rocks (Colony numbers 219-054 to 219-057), as well as the slaughter of murres by sport shooters for target practice. Through persistent urging by Finley, Three Arch Rocks was declared a Reservation for the Protection of Native Birds (now known as a National Wildlife Refuge) by Executive Order of President Theodore Roosevelt in 1907. Over the years, tremendous numbers of murres have been reported here, described as "countless thousands" (Ferris 1940), or from "hundreds of thousands" to 750,000 murres (Gabrielson and Jewett 1940). S. G. Jewett (Tuck 1961) considered numbers in the late 1930s and early 1940s to have increased since his first visit to the rocks in 1914. From 1930 to 1940, R. Ferris banded many chicks at Three Arch Rocks and Cape Lookout (Colony numbers 219-061 to 219-063; Bayer and Ferris 1987). S. G. Jewett (Tuck 1961) also reported large colonies "on the rocks off Bandon" (Colony numbers 270-015 to 270-020), "off Port Orford" (Redfish Rocks), "off the mouth of the Pistol River" (Colony numbers 270-085 to 270-087), and "other smaller colonies in between". A. Walker (Tuck 1961) reported fewer than 2,500–3,000 on Cape Lookout, large colonies at "Two Arches off Cascade Head, where two of the three rocks are occupied by murres" (Colony numbers 219-069 to 219-073), breeding at "Cape Mears, where some nest on ledges on the cape and others on an offshore rock" (Colony numbers 219-044 to 219-051); and breeding at "another smaller colony on a rock off Falcon Cove" (Colony number 219-030). Museum egg specimens provided additional evidence of breeding and numbers of murres present at several colonies in the first half of the twentieth century (Table 2.3).

Murre numbers increased during the first half of the twentieth century, including colony formations. Murres apparently began breeding on Colony Rock at Yaquina Head in the 1940s or 1950s. Murres were not noted breeding in 1899 (Prill 1901; Bayer 1986a), nor were they recorded during a Portland Audubon Society field trip in May 1940 (Anonymous 1940). However, a Portland Audubon Society field trip on 11 May 1952 did report murres from this location, although it was not clear if they were breeding (Oakes 1952). By 1958, breeding was confirmed by egg collectors (Table 2.3). Visual estimates during aerial surveys by the USFWS from 1966 to 1977 were from 1,800 to 4,800 birds.

The first coastwide survey of murres and other seabirds in Oregon occurred in the 1960s when the Bureau of Sport Fisheries and Wildlife (now U.S. Fish and Wildlife Service) was evaluating the acquisition of the larger rocks and islands for inclusion in the National Wildlife Refuge System. On 9–10 July 1964, biologists conducted an aerial survey along the entire coast and visually estimated 108,700 murres, which did not include 300,000 murres previously reported at Three Arch Rocks (D. B. Marshall, unpublished data). D. B. Marshall (unpublished field notes) noted that visual estimates of the murres were probably inaccurate because "The dense colonies of sea birds, particularly murres, pose a census problem which I feel is not satisfactorily resolved." They did photograph various colonies to provide a comparison with visual estimates, but the photos were never counted (D. B. Marshall, personal communication). The current location of these photos is unknown.

In 1966–67, Browning and English (1967, 1972) surveyed 12 rocks and islands and provided estimates of murres at some colonies, including 225 on North Coquille Point Rock (Colony number 270-015), 450+ on Cat and Kittens (Colony number 270-019), 1,180 on Island Rock, 300 on an unnamed rock NW of Island Rock (Colony number 270-048), and 1,650 on an unnamed rock at Whaleshead Creek. In 1964, D. B. Marshall (unpublished data) recorded 1,500 murres at Island Rock, but there were major differences at other sites. For example, 12,000 and 18,000 murres were reported on Cat and Kittens and the unnamed rock at Whaleshead Creek, respectively. The disparity between these two surveys may have resulted from the early survey dates in 1967 (22–23 April) and rough visual estimates.

Table 2.3. Summary of museum egg specimens[a] of common murres in Oregon.

Area or colony name	Number of Eggs	Dates	Sources	Notations
Yaquina Head	7	05/30/58	U. G. Kubat	Colony Rock
	12	06/08/58	U. G. Kubat	Three Arch Rocks
	15	06/27/40	E. N. Harrison	Noted: "thousands of birds were nesting on top of island as close together as possible"
Cape Lookout	29	06/02/29	A. Walker	
	3	05/29/32	W. E. Griffee	
	2	06/05/35	W. E. Griffee	Noted: "egg on the bare top of a rock about half an acre in extent. The 600 or 700 pairs of murres were closely grouped for protection from the western gulls, which eat murre eggs whenever they are exposed."
	8	06/01/41	L. T. Stevens, B. F. Walker	
	5	06/08/52	U. G. Kubat	
	26	06/14/53	U. G. Kubat	
	13	06/05/54	W. E. Griffee, U. G. Kubat	
	1	06/08/58	U. G. Kubat	
Cascade Head Area	54	06/05/35	L. T. Stevens, W. E. Griffee	
	7	06/15/30	A. and K. M. Walker	Two Arches Rock
	17	06/19/33	A. Walker	Two Arches Rock
	40	06/04/34	A. Walker	Two Arches Rock
Island Rock	23	06/15/30	J. C. Braly	Noted: "between 2,000 and 3,000 murres in the colony"
Port Orford	3	06/15/30	J. C. Braly	Redfish Rocks
Brookings Area	3	07/20 to 22/17	F. J. Smith	Egg Island Colony numbers 270-115 to 270-123
	8	5/18 to 19/30	J. T. Fraser	Noted: "several hundred murres nested here"
	2	06/15/30	J. C. Braly	Noted: "several thousands nesting"
	109	6/6 to 7/49	L. T. Stevens, L. R. Howsley	
	1	07/07/49	L. T. Stevens	

[a] Specimen information was obtained from Western Foundation of Vertebrate Zoology, Camarillo, California; Santa Barbara Natural History Museum, Santa Barbara, California; San Diego Natural History Museum, San Diego, California; and National Museum of Natural History, Washington, D. C.

In 1967, 28 large islands along the Oregon coast were included in the National Wildlife Refuge System. From 1966 to 1977, the Bureau of Sport Fisheries and Wildlife conducted aerial surveys from fixed-wing aircraft and visually estimated numbers of murres attending colonies throughout the Oregon coast. From 1966 to 1974, an average of 122,673 murres was estimated. In 1975 and 1976, 162,350 and 202,960 murres (respectively) were estimated but numbers of birds estimated at each site varied significantly. At Three Arch Rocks, estimates were from 25,000 to 107,000 murres. Even at the small colony at Goat Island, estimates varied significantly (range, 800–3,000) between years. Such large variations have not been noted since aerial photographic survey techniques have been employed, except during severe El Niños.

The first comprehensive survey of breeding seabirds in 1979 employed the first use of aerial photographs at murre colonies in Oregon (Varoujean and Pitman 1980). A total of 259,993 murres were counted at 63 colony sites. Unfortunately, count data from 1979 aerial photographs are considered inaccurate and were not used for trend analyses because (1) U.S. Coast Guard Sikorsky helicopters were operated at great distances from seabird colonies to minimize disturbance, making photographs difficult to count accurately and flushing large numbers of murres at certain colonies; and (2) at 28 (44%) of 63 colonies, counted photographs were taken on 16 July 1979, by which date up to 75% of murres had already departed the colony (based on comparison to other photographs in May or June 1979). Thus, murre numbers at colonies in Oregon were probably underestimated using July 1979 data.

Current Population Size and Distribution of Breeding Colonies in Oregon

By 1995, 75 locations attended by murres had been documented in Oregon (Figure 2.7; Appendix E). We have designated these locations as follows: (1) 49 regularly-attended colonies averaging more than 100 birds per count; (2) 14 regularly-attended colonies

Table 2.4. Numbers of common murres counted at seven colonies formed in Oregon between 1989 and 1995.

Colony number	1988	1989	1990	1991	1992	1993	1994	1995
219-036	0	0	0	0	0	0	Pr[a]	43
219-057	0	157	68	99	0	0	0	0
243-015A	0	0	75	56	33	0	6	73
243-016	0	0	0	0	0	168	119	265
243-017	0	0	8	192	603	783	1,201	1,692
270-016	0	0	0	5	17	0	107	90
270-017	0	0	46	324	795	204	1,180	1,079
Total	0	157	197	676	1,448	1,155	2,613	3,242

[a]Birds present in small numbers.

averaging fewer than 100 birds per count; (3) 7 recently-formed (since 1988) and regularly-attended colonies (Table 2.4), including 6 that are still active and one colony abandoned in 1991 (Colony number 219-057); (4) 2 colonies abandoned after 1979 (Colony numbers 219-007 and 219-029); and (5) two rocks attended by small numbers (fewer than 10 birds) without confirmed breeding. Breeding has been confirmed with observations of eggs or chicks at all regularly-attended colonies in 1988–95. The most complete and accurate survey was conducted in 1988 (R. W. Lowe, unpublished data). Sixty-six attended locations were identified with 59 surveyed using aerial photography and 7 surveyed visually by boat. The largest colony (more than 132,000 breeding birds) was located at Shag Rock within Three Arch Rocks (Colony number 219-056). The Oregon breeding population was estimated at 711,900 breeding birds. Although colonies exist within several colony complexes in Oregon (Figure 2.7), count data were not available for all colonies within complexes. Consequently, murre numbers and trends at the colony complex level in Oregon were not described or assessed.

Population Trends in Oregon, 1988–1995

To measure population trends in Oregon from 1988 to 1995, we analyzed data for 15 sample colonies surveyed and counted annually during this period (Table 2.5; Appendix E). This sample of colonies is spread along the entire Oregon coast. The largest colonies are not included because of extensive counting time required. Most colonies (80% of 75 attended locations in 1988) and most of the Oregon population (87.5% of 426,278 birds counted in 1988) were not counted, except in 1988 when all colonies were counted. In addition, two of the 15 sample locations were not attended until after 1988 (see below). Analyses of population trends were hampered by initiation of standardized aerial photographic surveys of Oregon murre colonies after major declines in central California and Washington in 1979–86. Also, trends from 1988 to 1995 for 15 sample colonies might not be representative of all colonies although obvious differences were not noted (R. W. Lowe, unpublished data).

In addition to USFWS surveys, murre colony counts in Oregon were conducted by a private consulting firm

Table 2.5. Average and maximum sizes for 15 selected breeding colonies of common murres in Oregon, 1988–1995 (see Appendix E). Symbols and format as in Table 2.1.

Rank	Colony number	Mean count	Mean number of breeding adults	Years of data	Maximum count	Maximum number of breeding adults	Year of maximum
1	243-015	15,764	26,300	8	19,147	32,000	1989
2	243-010	12,938	21,600	8	14,377	24,000	1990
3	270-116	6,061	10,100	8	7,588	12,700	1991
4	219-005	5,790	9,700	8	7,199	12,000	1995
5	270-123	2,755	4,600	8	2,968	5,000	1990
6	219-017	2,688	4,500	8	3,145	5,300	1995
7	219-060	2,015	3,400	8	2,506	4,200	1989
8	270-064	2,003	3,300	8	2,389	4,000	1991
9	270-034	1,889	3,200	8	2,317	3,900	1994
10	270-043	1,544	2,600	8	1,888	3,200	1990
11	270-117	1,499	2,500	8	1,918	3,200	1989
12	219-070	790	1,300	8	972	1,600	1988
13	270-122	662	1,100	8	820	1,400	1988
14	270-086	126	200	8	327	500	1988
15	270-085	78	100	8	142	200	1995

(Briggs et al. 1992) in late June 1989. Owing to numerous problems (i.e., different survey techniques and incomplete surveys), data from these surveys are not discussed in this chapter (see Appendix A). In 1995, three replicate aerial surveys were conducted to determine variability in counts over a four week period (Table 2.6). Although a few individual colony counts did show much variation (i.e., Colony numbers 219-003, 219-010, 219-026), overall variation among the 15 colonies was small (Lowe and Pitkin 1996).

The number of murres at sample colonies in Oregon increased from 1988 to 1990, then declined slightly before severe El Niño breeding conditions in 1993 (Figure 2.8; Appendix H). In 1993, warm marine waters persisted along much of the Oregon coast, which resulted in complete murre reproductive failure. Colony abandonment began in late May, prior to the annual aerial photographic survey in early June. Abandonment of this magnitude had not been reported previously in Oregon. Murres returned in large numbers in 1994 and increased further in 1995. The effects of the 1982–83 El Niño were apparently not as severe as in 1993; however, reduced breeding success and, possibly, greater adult and subadult mortality was observed (Hodder and Graybill 1985; Bayer 1986b; Bayer et al. 1991). Effects of the 1992–93 El Niño did not result in large changes in the numbers of breeding murres in Oregon. The breeding population in Oregon has been relatively stable from 1988 to 1995 (Figure 2.8; Appendix H).

Table 2.6. Numbers of common murres counted during replicate aerial surveys at 15 selected colonies in Oregon in 1995.

Colony number	23 May	7 June	21 June	Mean
219-002	80	67	60	69
219-003	106	118	221	148
219-005	7,488	7,199	7,479	7,389
219-010	0	129	244	124
219-013	2,508	2,694	2,623	2,608
219-014	136	130	120	129
219-017	3,047	3,145	2,649	2,947
219-019	6,549	7,143	7,029	6,907
219-026	7,279	6,132	7,679	7,030
219-027	5,312	5,342	4,462	5,039
219-036	38	43	44	42
219-044	4,428	4,926	4,381	4,578
219-045	7,377	7,079	7,192	7,216
219-060	2,350	1,922	718	1,663
219-062	105	132	139	125
Total	46,803	46,201	45,040	46,014
Deviation from 7 June	1.3%	ND	-2.5%	-0.4%
Deviation from mean	1.7%	0.4%	-2.1%	ND

Seven colony formations occurred during 1988–95 (Table 2.4). One colony formed in 1989 on Seal Rock (Colony number 219-057) and persisted until 1991, but was then abandoned, possibly because of disruptions from Steller sea lions *Eumetopias jubatus* that "haul out" on the rock (R. W. Lowe, personal observation). Other colonies formed in 1990, 1991, and 1994 when murre numbers in Oregon reached high levels. One colony formed in 1993, during the 1992–93 El Niño, when the number of murres attending colonies was at a record low. Six of seven new colonies (i.e., except Unnamed Rock, Colony number 219-036) were established at rocks nearby (i.e., within 250 m) large colonies, probably reflecting intercolony movements within colony complexes. For example, colony formation at Seal Rock occurred in close association with the large colony complex at Three Arch Rocks National Wildlife Refuge. Six of the seven colony formations (i.e., except Seal Rock) also occurred in conjunction with newly-formed colonies of Brandt's cormorants. Colony formation at Unnamed Rock (Colony number 219-036) in 1994–95 occurred 5–6 km from the next nearest active murre colony but also was associated with nesting cormorants. Six new colonies still existed in 1995 (i.e., except Seal Rock), despite poor breeding conditions associated with elevated sea-surface temperatures since 1991 (R. W. Lowe, unpublished data).

Stable murre populations from 1988 to 1995 coincided with a period of relatively low anthropogenic effects before and during this period. Between 1982 and 1993, more than 1,200 rocks and islands along the Oregon coast were protected by acquisition or conservation agreements by the USFWS. In some cases, such as Tillamook Rock, human disturbance was

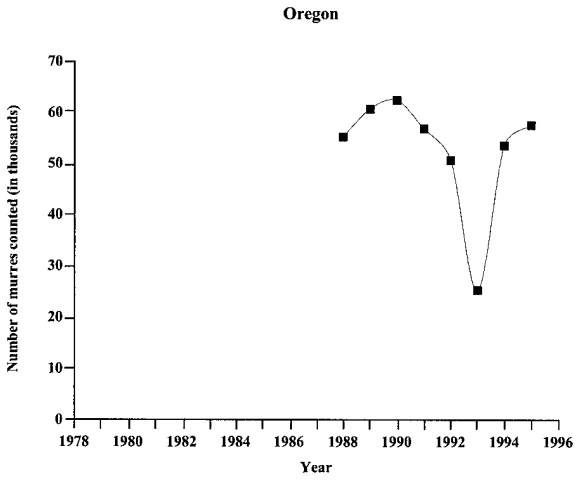

Figure 2.8. Changes in whole-colony counts for 15 sample colonies of common murres in Oregon, 1988–1995 (see Appendix H).

reduced over a period of time and numbers of murres correspondingly increased. In 1957 the Tillamook Rock lighthouse was decommissioned. In 1980 access was limited under private ownership. In 1993 the USFWS obtained a perpetual conservation agreement for this site and human access during the breeding season was prohibited. Numbers of murres attending this site between 1987 and 1995 far exceeded numbers in the 1970s and peaked in 1995 (see Appendix I: Figure I-31). Human disturbance has occurred regularly at many colonies in Oregon but, to date, this has not resulted in colony abandonments. Recently, disturbances from low-flying aircraft and close approach by boats were well documented at Three Arch Rocks (Lowe 1993). Management actions have now been implemented to reduce the problem at this colony complex but human disturbance at other colonies throughout the Oregon Coast National Wildlife Refuge Complex remains a primary management concern.

Gill-net fishing has been prohibited in Oregon since the 1940s and no large oil spills have occurred adjacent to colonies before or during the 1988–95 study period. However, death of murres from gill-net fishing and oil spills in Washington and British Columbia undoubtedly included large numbers of murres from Oregon colonies during or after northward movements that occur after colony departure (Manuwal and Carter 2001). Two major oil spill events in Washington (i.e., 1988 *Nestucca* and 1991 *Tenyo Maru*) killed an estimated 30,000 and a range of 3,740–19,559 murres, respectively (Ford et al. 1991; Tenyo Maru Oil Spill Natural Resource Trustees 2000). Given the relative size of the common murre populations in Washington and British Columbia prior to and after the *Nestucca* spill, it is quite likely that a substantial proportion of the birds killed as a result of this event were from Oregon breeding colonies. An assessment of the origin of murres killed in the *Tenyo Maru* spill indicated that 39–58% of the adult murres killed in the spill were from Washington and the remainder (42–61%) were from Oregon, although a series of assumptions were used to generate this estimate (Warheit 1996). Murre deaths also result from gillnet entanglement in the fall sockeye salmon fishery in Puget Sound, Washington (Pierce et al. 1994). This fishery and associated seabird deaths take place when Oregon birds are typically present (Manuwal and Carter 2001). In 1997, the Washington Department of Fish and Wildlife adopted regulations to reduce seabird deaths in the nontreaty fishery by eliminating early morning (dawn) fishing and requiring net modifications (Melvin et al. 1999). In addition, die-offs of large numbers of chicks after colony departure have been reported for decades in Oregon (Bayer et al. 1991), but the level of deaths may have increased since 1990 (R. W. Lowe, unpublished data).

Along the north and central coasts, predation and disturbance by bald eagles have severely affected breeding murres at some colonies (R. W. Lowe, personal observation). This was first noted in 1994 and continues to increase at colony sites from Tillamook Head to Colony Rock in Newport. The disruption at murre colonies has been concomitant with increased sightings of juvenile bald eagles in this area. Most impacts result from repeated colony disturbance, rather than actual predation. Juvenile eagles often perch within colonies and delay murre egg laying. Disruptions during incubation cause murres to flush, exposing murre eggs to breakage or predation by gulls and corvids. At Bird Rocks at Chapman Point (Colony numbers 219-017 and 219-018), continued harassment by eagles throughout the breeding season has resulted in erratic colony attendance and complete breeding failure. Recent effects from various natural and anthropogenic factors have been localized (e.g., eagle or human disturbance at specific colonies) or dispersed among the numerous colonies and large populations (e.g., oil-spill and gill-net deaths). Efforts to further reduce anthropogenic effects are continuing.

Washington

Historical Background on Breeding Colonies in Washington, 1905–1978

The degree to which native people affected murres in western Washington before the early twentieth century is not clear. Despite large populations of native people and the common use of canoes, the inaccessibility of many rocks and islands on the Washington coast may have limited food gathering activities to certain locations. At some larger islands (i.e., Tatoosh Island), occupation by native people probably prevented breeding by murres. Seagull eggs were harvested in June from colonies at Point Grenville and Cape Elizabeth by Quinault native people (Olson 1936; Speich et al. 1987). However, harvesting of gull eggs apparently did not prevent murre breeding at Willoughby Rock in 1906, although gull egg harvesting by Ozette native people may have prevented breeding by murres at White Rock (Dawson 1907).

In July 1906 and June 1907, most seabird colonies on islands off the outer coast of Washington were surveyed by canoe, and 1,736 murres were counted at seven locations (Dawson 1907, 1908a, 1908b; Dawson and Bowles 1909). Five of these colonies— Erin, Grenville Arch, Grenville Pillar or "Radio Stack,"

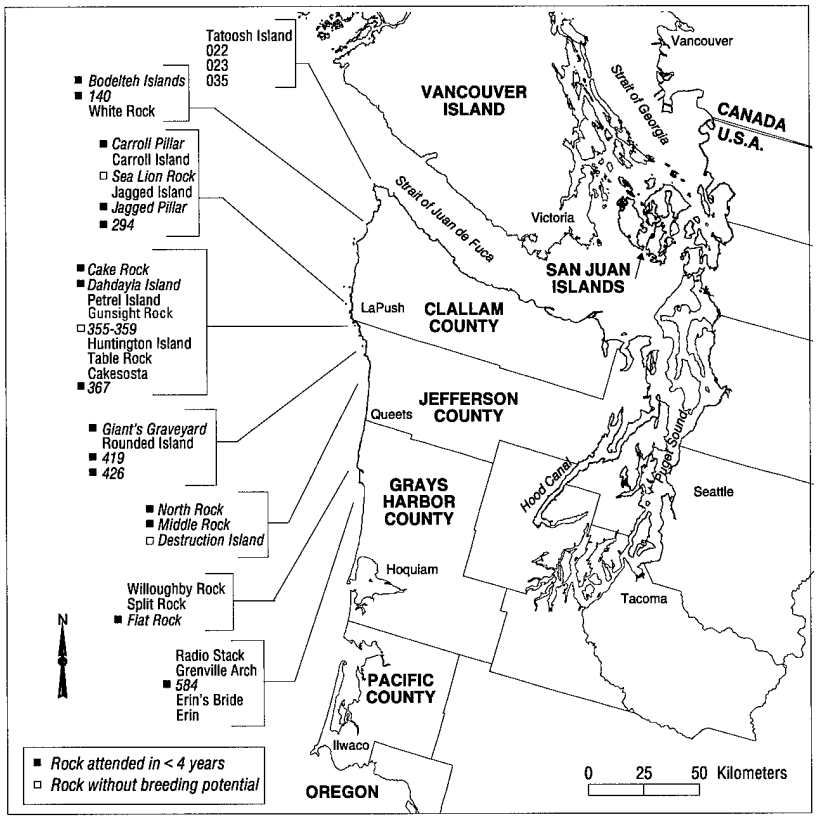

Figure 2.9. Distribution of common murre colonies in Washington (Clallam to Pacific Counties).

Willoughby Rock, and Carroll Island— still exist (Figure 2.9). Eggs or chicks were confirmed only at Willoughby Rock and Carroll Island (Dawson 1907; Jones 1909). One egg was collected by Dawson at Carroll Islet on 21 July 1906 from a colony of about 100 pairs with "most [eggs] hatched" (SBNHM egg records). Three eggs in the WFVZ collection were obtained on 20 June 1907 at "Birdrock," Washington (WFVZ No. 47,472–47,474). Another egg collected on 12 June 1910 at Willoughby Island was from a colony of "500 pairs on south slope" (SBNHM egg records). No murres were observed at several other rocks subsequently attended by murres in historical literature: Erin's Bride, Split Rock, Destruction Island, North Rock, Rounded Islet, Giant's Graveyard, Quillayute Needles, Cake Rock, White Rock, Flattery Rocks (including Old Rock, also known as Bodelteh Islands), Point of the Arches (including Silversides), and Tatoosh Island. At Carroll Pillar (adjacent to Carroll Island, also known as "Paahwoke-it"), 200 murres were recorded, but since then, only small numbers of murres have been observed sporadically in 1917 and 1978 (Speich and Wahl 1989; Appendixes F and G).

On 13–17 July 1959, a combination of aerial and boat surveys of seabird colonies was conducted along the Washington outer coast, which recorded 4,450 murres at seven locations plus 550 at sea off Cape Flattery (Kenyon and Scheffer 1962). The largest colonies were at Carroll Island and Willoughby Rock (2,000 murres each, but they were uncertain of exact locations) with smaller colonies at Tatoosh Island (200) and White Rock (100). Small numbers (100) were noted at Bodelteh Islands on 13 July 1959 but none were recorded on 17 July 1959; fewer than 100 murres were noted in 1978–79 but none between 1980 and 1995 (Speich and Wahl 1989; Appendixes F and G). Murres noted at "Flattery Rocks" in 1914 may have referred to Bodelteh Islands or White Rock (Jewett et al. 1953; Speich and Wahl 1989). At Cake Rock, 50 murres were noted on 13 July 1959; small numbers (25–175) were noted in 1967, 1990, and 1992 (Speich and Wahl 1989; Appendixes F and G). We have considered sporadic observations of murres at Bodelteh Islands and Cake Rock to reflect irregular attendance. Various other observations at several known colonies between 1907 and 1959 indicate long-term use of many colony sites (Jewett et al. 1953; Speich and Wahl 1989).

Manuwal and Campbell (1979) summarized data from USFWS aerial surveys (visual estimates from fixed-wing aircraft) conducted in the early 1970s and tabulated 11,950 murres at 11 locations. The largest estimates of the numbers of murres present were reported at Grenville Arch (3,000), Willoughby Rock (3,000), and Split Rock (2,100). Smaller colonies were found at Point Grenville

(1,100), Quillayute Needles (900), James Island (750), Cake Rock (300), White Rock (250), and Tatoosh Island (100). Murres have not been otherwise observed on James Island, but are known to occupy adjacent rocks now known as Petrel Island ("Kohchaa[uh]") and Gunsight Rock. Quillayute Needles represents a colony complex with several recent colonies: Huntington Island, Cakesosta, and Table Rock. In addition, murres were reported in the early 1970s at Flat Rock south of Split Rock (300) and Giant's Graveyard (150; Speich and Wahl 1989). These observations apparently reflect irregular attendance or colony misidentification. At Point Grenville and Giant's Graveyard, specific islands attended by murres were not identified.

Many of Washington's largest murre colonies are located on top of flat-topped sea stacks or islands (see Appendix I: Figures I-33 to I-37). It is impossible to see all attending birds, and in some cases even to determine if murres are present, when circumnavigating these islands by boat. In historical information, it is also unclear whether all colonies were surveyed, if counts on adjacent colonies were lumped and reported under one colony name, or if colonies were properly identified. In addition, murre attendance at colonies in 1906–07, 1959, and the early 1970s may have been affected by El Niños (Quinn et al. 1987). Colonization of Tatoosh Island by 1956 indicates the population may have expanded in the mid-twentieth century.

Suspected increases may reflect lower levels of activities by native people along the coast because of changes in traditional lifestyles. For example, camps of native people on Tatoosh and Destruction islands were abandoned. However, the decline of native populations and the rate of arrival and number of Euro–American settlers in western Washington was not as pervasive or extensive as in California and Oregon. Native people were confined to reservations along most of the outer coast and early settlers in the 1880s and 1890s tended to move into Puget Sound or the eastern areas on the Olympic Peninsula (Evans 1983). Large coastal areas were included in Mount Olympus National Monument and Olympic National Forest in the early 1900s. Some of these areas became part of Olympic National Park in 1938. Much of the outer Washington coast also remained inaccessible by land until 1931 when the Olympic loop highway (i.e., Highway 101) was completed. Thus, the outer coast of Washington was spared from many effects from early settlers.

Military bombing of Sea Lion Rock (north of Willoughby Rock) in southern Washington occurred from 1944 to 1992 (Speich et al. 1987). Carroll Island, Rounded Island, Sea Lion Rock, and Split Rock also were practice bombing targets during World War II and were bombed extensively with heavy ordinance. Several murre colonies were probably affected by low-flying aircraft en route to and from Sea Lion Rock and other target islands, including Willoughby Rock, Split Rock, and possibly Grenville Arch. Similar problems probably occurred after World War II. Lighthouse keepers and associated activities may have prevented breeding at Tatoosh Island from the late nineteenth century to the 1950s, but rats (*Rattus spp.*) and cats (*Felis catus*) were not introduced (Kenyon and Scheffer 1962). An accidental fire caused by researchers at Carroll Island burned the top of the island in 1969, but it is unclear if murres were affected since they bred on other parts of the island at that time (M. L. Cody, personal communication).

Current Population Size and Distribution of Breeding Colonies in Washington

During 1979–95, murres were recorded at 32 different locations along the outer coast of Washington (Appendixes F and G). Most counts at these locations did not provide information on the breeding status of attending birds. Eighteen locations have been designated as colonies (Figure 2.9) based on historical or recent observations of breeding (i.e., eggs or chicks seen) or regular attendance of rocks with suitable breeding habitats. As noted above, historical breeding (pre-1979) had been confirmed only at Carroll Island and Willoughby Rock (Dawson 1907; Jones 1909; Jewett et al. 1953). In 1980–82, U. W. Wilson (unpublished data) observed chicks during the last week of June and first week of July at Grenville Arch, Split Rock, Willoughby Rock, and Cakesosta. On 27 August 1985, S. M. Speich collected one egg at Grenville Arch and four eggs at Willoughby Rock (WFVZ Nos. 149,537–149,541). On 4 September 1985, S. M. Speich collected an abandoned egg at Split Rock (WFVZ No. 149,536). On 3 June 1987, F. Dobler (unpublished data) collected several murre chicks near colony departure that were accidentally killed on Jagged Island. On 19 June 1995, U. W. Wilson (unpublished data) observed medium-sized chicks on Huntington Island, and noted several large young on 13 July 1995 at this colony. On 18 July 1995, U. W. Wilson (unpublished data) and G. Burrell visited Carroll Island and found one abandoned murre egg. Murres breed regularly at the Tatoosh colony complex where long-term studies of murre behavior and reproductive success are under way (Paine et al. 1990; Parrish 1995). Breeding may have occurred at several of the other 14 sites but adequate documentation (see below) to confirm breeding status is lacking.

To examine population trends, we divided locations attended by murres into two geographic sections, southern Washington—with 6 known colonies in Grays Harbor County—and northern Washington—with 12 known colonies in Clallam and Jefferson Counties (Figure 2.9). These areas had been identified in previous studies as having different murre population trends (Wilson 1991; Parrish 1995). No murre colonies have ever been reported in the inland marine waters of Washington's Juan de Fuca Strait, San Juan Islands and Puget Sound (Speich and Wahl 1989).

Murres were counted annually by the USFWS along the outer coast of Washington from 1979 to 1995, except Tatoosh Island which was surveyed aerially only in 1994–95 (Appendixes F and G). Routine aerial surveys were not conducted at the Tatoosh Island complex from 1979 to 1993 because this colony was not part of the Washington Islands National Wildlife Refuge (consisting of Flattery Rocks, Quillayute Needles, and Copalis National Wildlife Refuges). Few birds (e.g., 200 birds in 1978; Speich and Wahl 1989) attended Tatoosh colonies (Appendixes F and G) in the late 1970s. By adding 200 birds to the USFWS 1979 aerial survey total of 31,520 birds for all other locations, a total count of 31,720 birds for 1979 was derived. Speich and Wahl (1989) derived a similar total (30,780 birds) by combining raw counts from the late 1970s and early 1980s. By applying a *k* correction factor to the 1979 total, we estimated the breeding population for Washington at about 53,000 breeding birds. Southern and northern Washington accounted for 86% and 14%, respectively.

In late June 1989, murre colony counts in Washington were conducted by a private consulting firm (Briggs et al. 1992) but, because of numerous problems (i.e., different survey techniques and incomplete surveys), we relied only on data from standardized USFWS surveys for trend assessments (see Appendix A). However, if we add 830 birds for Tatoosh Island (Briggs et al. 1992) to the USFWS total of 3,925 birds (which excluded the Tatoosh Island complex), a total count of 4,755 murres was derived for 1989, which corresponded to about 7,900 breeding birds. Certainly, numbers of murres in Washington were much lower in 1989 than in 1979 (Figure 2.10), and only 28% occurred in southern Washington.

In 1994 and 1995, breeding population estimates for murres in Washington were 5,900 and 9,600 breeding birds, respectively, based on the results of the 5 July 1994 and 25 June 1995 USFWS aerial photographic surveys (Appendix G), which included the Tatoosh Island complex. The Tenyo Maru Oil Spill National Resource Trustees (2000) estimated the 1995 murre population in Washington at 13,600 birds by adding a median count of 5,230 birds from USFWS 1995 refuge surveys (excluding Tatoosh Island) to a 1995 ground count of 3,720 murres on Tatoosh Island (Parrish 1996) and applying a *k* correction factor of 1.6. The proportion of Washington murres attending southern Washington locations was between 1 and 14% in 1994–95.

In southern Washington, colonies occur in complexes at Point Grenville and Split–Willoughby. Peak numbers were estimated in 1979 at Point Grenville (21,400 breeding birds) and in 1982 at Split–

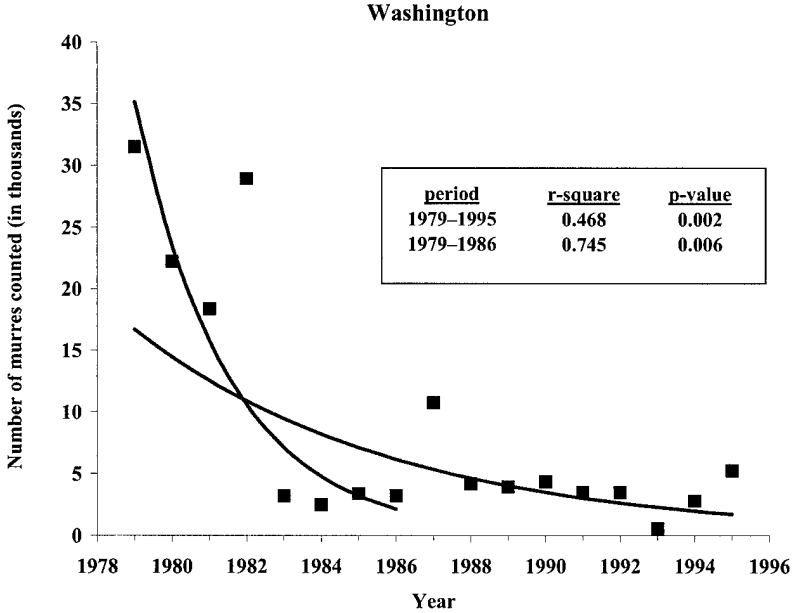

Figure 2.10. Trends in whole-colony counts for common murres in Washington, 1979–1995, excluding Tatoosh colonies (see Appendix H).

Willoughby (26,300 breeding birds). Between 1979 and 1982, breeding murres were centered at these colonies in southern Washington. However, all six colonies in both colony complexes were abandoned or severely reduced by 1994–95 (see below). In northern Washington, colonies exist in relatively small individual colonies and in three colony complexes: Quillayute Needles, Carroll–Jagged, and Tatoosh (Appendixes F, G). In 1979–82, relatively small numbers occurred in northern Washington, but in 1994–95 the Washington population was centered there.

Many instances of irregular murre attendance at rocks, often by small numbers, have been documented in the past (see above), as well as during aerial surveys from 1979 to 1995 (Figure 2.9). Determining breeding activity is difficult in Washington because of colony inaccessibility and poor viewing conditions from adjacent mainland areas or boats. During surveys in 1979–95, murres were reported at eight locations on only one survey in 1 year (i.e., Colony number 584 in 1989, 457 in 1986, 426 in 1986, 419 in 1987, 367 in 1985, 294 in 1985, 140 in 1986, and Jagged Pillar in 1991). We suspect that breeding did not occur at these locations. During this same period, irregular attendance occurred over 2–3 years at Cake Rock (1990–92) and at Carroll Pillar (1993–95). At Jagged Pillar, 17 murres were reported in 1978 and 25 in 1982 (Speich and Wahl 1989). In the past, irregular attendance has been noted at Carroll Pillar, Cake Rock, Bodelteh Island, Flat Rock, and Giant's Graveyard. In addition, irregular attendance has been reported at "Dahdayla" near Cake Rock (2–30 birds in 1967–69), Half Round Rock (250 murres in 1981), and Quillayute Needle (35–276 birds in 1978–80). Otherwise, murres were not reported at these locations (Speich and Wahl 1989; Appendix F). At Middle Rock, large numbers (range, 450–1,800) attended irregularly in 1985–86.

Although attended for a few years, breeding is not suspected at Destruction Island or Colony number 355–359. Murres were not reported attending Destruction Island from 1906 to 1987 (Speich and Wahl 1989; Appendix F). Between 250 and 650 birds were observed loafing around peripheral rocks annually from 1988 to 1992 (Appendixes F and G). In 1995, 215 murres were present, but no eggs or chicks were found during ground visits to the island (U. W. Wilson, unpublished data). Destruction Island lacks suitable murre nesting habitat and murres were present in association with nesting Brandt's cormorants on the island's peripheral rocks. On the Washington coast, small numbers of nonbreeding murres frequently are seen among nesting Brandt's cormorants. Since these cormorants can change their colony locations, irregular murre attendance at certain rocks may be due to attraction of murres to Brandt's cormorant colonies.

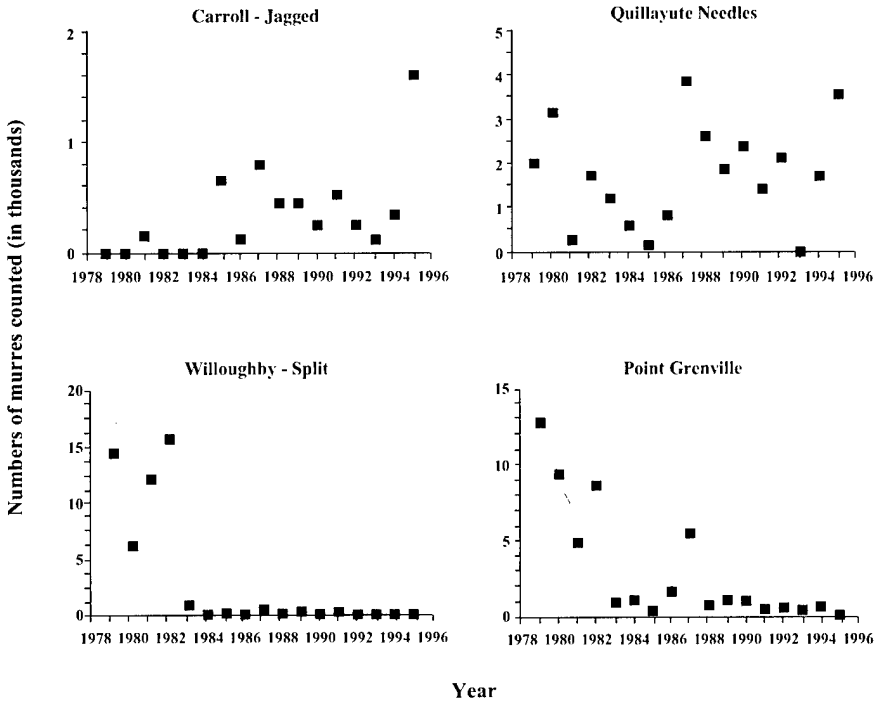

Figure 2.11. Trends in whole-colony counts for four colony complexes of common murres in Washington, 1979–1995 (see Appendix H).

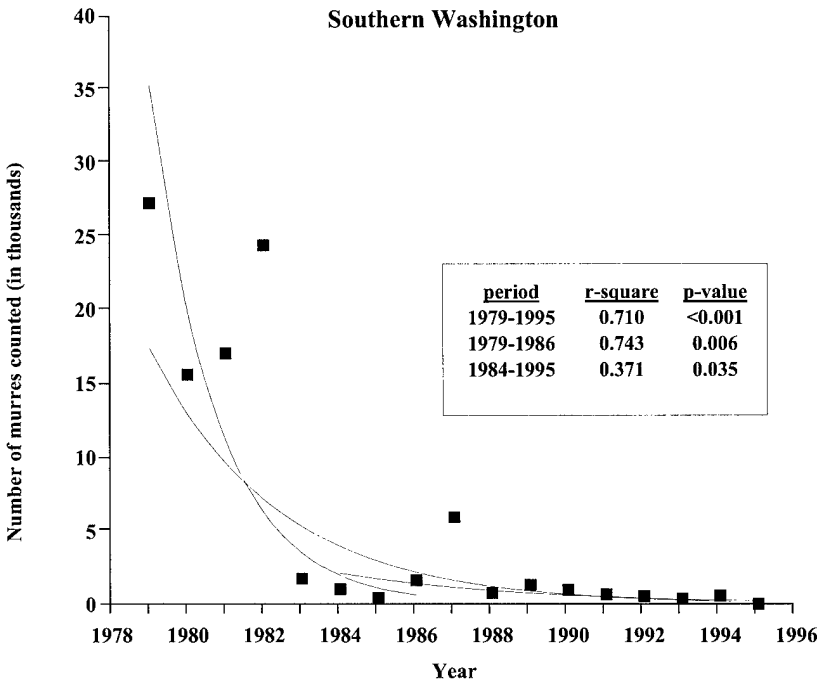

Figure 2.12. Trends in whole-colony counts for common murres in southern Washington, 1979–1995 (see Appendix H).

Population Trends in Southern Washington, 1979–1995

Overall numbers of breeding murres in Washington declined 32.9% per annum between 1979 and 1986 ($P = 0.006$; Figure 2.10). Most decline between 1979 and 1986 occurred in southern Washington where steep downward trends occurred at colony complexes (Figure 2.11), as well as for southern Washington overall (43.7% per annum, $P = 0.006$; Figure 2.12). During the period 1979–82, the southern Washington population was much higher than between 1983 and 1995, when numbers varied extensively at colonies and colony complexes (Figure 2.11; Appendix F). Murre attendance at the Point Grenville complex decreased greatly from 1979 to 1981 before rebounding in 1982. At the Willoughby–Split complex, both colonies decreased greatly in 1980 between similar peak numbers in 1979 and 1982. Large differences in the numbers of murres attending colonies in southern Washington between 1979 and 1982 apparently were related to natural and anthropogenic factors. In 1981, reduced colony attendance probably reflected a response to unusually warm surface waters (similar to moderate El Niños) off the Washington coast between January and April 1981 (Wilson 1991). Numbers of murres attending colonies increased in 1982, apparently reflecting the return of birds that did not breed, remained at sea, or moved temporarily to other colonies in 1980–81. In addition to the warm water episode, various human disturbances such as overflights and military activity occurred along the Washington coast on a regular basis in the late 1970s and early 1980s (Speich and Thompson 1987). These disturbances may have contributed to oscillating colony attendance. Death of common murres from entanglement in Washington gill-net fisheries (i.e., Willapa Bay, Grays Harbor, Puget Sound) in the 1970s and early 1980s probably occurred but was poorly documented.

In 1983, widespread colony abandonment occurred in association with severe El Niño conditions (Wilson 1991). Almost complete abandonment occurred at the Willoughby–Split complex by 1984–86. Impacts from chronic colony disturbances and the 1981 warm water event also may have contributed to this steep decline. The negative effects of private aircraft overflights and military practice bombing of Sea Lion Rock on murre attendance at southern Washington colonies in 1984–85 was documented by Speich et al. (1987). Colony attendance was significantly reduced because of lingering effects of the 1982–83 El Niño during this study, therefore, the full effect of these military disturbances on breeding colonies was unclear. However, such disturbances undoubtedly affected many murre colonies until military bombing and aircraft overflights were greatly reduced in 1992. In 1984–85, disturbance by commercial ground-fishing boats was noted at Point Grenville and Willoughby Rock (S. M. Speich, personal communication). Limited increase between 1984 and 1988 may have reflected the return of some birds and the recruitment of subadults from higher populations in 1981–82.

In 1988, numbers of murres attending colonies again began to dwindle to very low levels during the

1987–88 El Niño (Wilson 1991). From 1988 through 1995, natural and severe anthropogenic factors acted in concert to affect the murre population. In December 1988, an estimated 30,000 murres were killed off the outer Washington coast as a result of the *Nestucca* oil spill (Ford et al. 1991). Because large numbers of murres from breeding colonies in Oregon, and possibly British Columbia, are found along the Washington coast during the fall and winter months (Manuwal and Carter 2001), these deaths probably involved murres originating from colonies in Washington and other areas. The proportion of murres from each area of origin killed by the *Nestucca* spill is unknown. Following the July 1991 *Tenyo Maru* spill, Warheit (1996) estimated that 39–58% of the adult murres killed by this spill originated from Washington. Based on estimates of total mortality (3,740–19,559 murres), the Tenyo Maru Oil Spill Natural Resource Trustees (2000) concluded that a sizable portion of the total Washington state murre population (including nonbreeding adult, subadult, and juvenile birds) may have been killed in the spill.

Between 1991 and 1994, onboard observer programs in various Washington gill-net fisheries documented seabird deaths from entanglements. Further studies in selected Puget Sound fisheries confirmed that common murres represented the majority of the total seabird entanglement (Jefferies and Brown 1993; Erstad et al. 1994; Pierce et al. 1994). In addition to deaths from oil spills and gill nets documented in the early 1990s, low colony attendance and reduced breeding effort occurred in 1993 during the severe 1992–93 El Niño. By 1994–95, small numbers still attended colonies in the Point Grenville complex but almost no birds attended the Willoughby–Split complex. At these low levels, it is doubtful if any murre breeding was still occurring in southern Washington.

In summary, numbers of murres attending colony complexes in southern Washington declined 25.5% per annum between 1979 and 1995 ($P \leq 0.001$; Figure 2.12; Appendix H). Several types of anthropogenic and natural factors apparently acted in concert to greatly affect the population and prevent recovery. These include severe El Niños, chronic human disturbance, and direct deaths from oil spills and gillnet entanglement. These factors presumably resulted in low colony attendance, reduced breeding success and recruitment, increased movements within and outside colony complexes, and deaths at sea. Since the southern Washington population constituted 86% of the entire Washington population in 1979–82, this change represents loss of most of the breeding population of murres within the state of Washington. Thus, overall numbers of breeding murres in Washington also declined 13.3% per annum between 1979 and 1995 ($P = 0.002$; Figure 2.10). The factors affecting the three largest colonies (Grenville Arch, Willoughby Rock, and Split Rock) are largely responsible for the Washington murre decline. Whereas murre colony attendance during severe El Niño years is generally reduced (Wilson 1991) because of changes in the marine food chain (Wooster and Fluharty 1985), the manner in which anthropogenic and natural factors acted to contribute to the decline, and how they may have prevented recovery, are difficult to determine with available evidence.

Population Trends in Northern Washington, 1979-1995

Between 1979 and 1982, numbers of murres attending colonies varied extensively at individual colonies, colony complexes, and overall in northern Washington (Figures 2.11 and 2.13; Appendixes F, G, and H). As in southern Washington, widespread colony abandonment occurred in association with the severe 1982–83 El Niño (Wilson 1991). Colony attendance at the Quillayute Needles complex returned to 1979–82 levels (excluding 1981) between 1987 and 1995 (Appendix H). This increase may have reflected return of some breeding birds which had not attended colonies during surveys in 1983–84 or movements of birds from other colony complexes. At the Carroll–Jagged complex, substantial increase and more regular attendance occurred in 1987–95 than in 1979–86. At this complex, there was an apparent shift of birds from Jagged Island to Carroll Island. U. W. Wilson (unpublished data) considered no birds to be breeding at Carroll Island in 1995, although egg laying and breeding-site failure may have occurred prior to surveys. The lack of recovery at Rounded Island colonies, located closest to southern Washington, may have reflected similar conditions as experienced in southern Washington including a combination of effects from natural and anthropogenic factors. With the exception of Navy practice bombing, the same factors affecting murres in southern Washington also affected the northern colonies (e.g., severe El Niños and gill-net and oil-spill deaths).

At Tatoosh Island and associated rocks (Colony numbers 022, 023, 035), aerial photographic surveys were not conducted until 1994–95 when moderate numbers were recorded. Murres have been reported at this colony since 1956 (Speich and Wahl 1989). Paine et al. (1990) reported fewer than 1,000 birds during 1956–79, with a sharp increase to 2,000 birds during the early 1980s. Briggs et al. (1992) reported 830 birds in 1989. By 1992, islandwide attendance reached 3,871 birds, based on ground counts, ground estimates, and

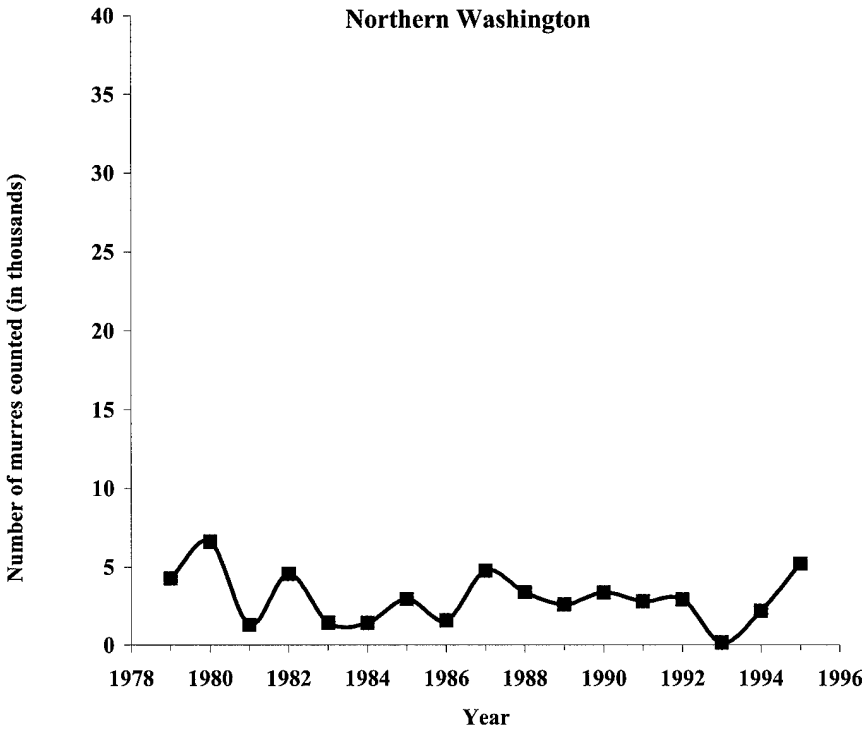

Figure 2.13. Changes in whole-colony counts for common murres in northern Washington, 1979–1995 (see Appendix H).

photographs taken from boats (J. K. Parrish, personal communication). In 1995, a USFWS aerial count of 1,705 birds was obtained on 27 July, corresponding to an estimate of 2,800 breeding birds. This estimate is similar to the 1995 ground-based estimate of 3,270 breeding birds (J. K. Parrish, personal communication). In Washington, Tatoosh Island is currently the only murre colony where murre reproduction has been studied, and is the only colony for which there is evidence of consistent breeding (Parrish 1996). However, murres on this colony have recently experienced adult deaths and reduced reproductive success because of predation and harassment by bald eagles (Parrish 1995; Parrish and Paine 1996). The peregrine falcon population also has increased along the outer coast of Washington (Wilson et al. 2000) and may affect murre colonies (Parrish 1995).

Limited increase in the northern Washington population occurred after the severe 1982–83 El Niño, in contrast to a lack of recovery in southern Washington. Both areas apparently have been subjected to similar problems (i.e., human disturbance, deaths from gill-net fishing and oil spills, reduced colony attendance during El Niños and other warm water events, and possible impacts from climate change). The large and sustained increase in murres attending Tatoosh Island indicated that immigration has contributed to the growth of this northern Washington colony along with the return of first-time breeders natal to Tatoosh colonies (Parrish 1995). This rapid increase of the Tatoosh complex and increase at the Carroll–Jagged complex in the early to mid-1980s occurred at the same general time as the marked declines at both southern and other northern Washington murre colonies.

Overall, the murre population in Washington significantly declined between 1979 and 1995, with the steepest rate of decline occurring between 1979 and 1986. Colony attendance dropped most dramatically during the severe 1982–83 El Niño. Recovery after this event has been poor because of the effects of additional El Niños, continued chronic gill net and oil spill associated deaths, and disturbance from military practice bombing and low aircraft overflights through 1992. Since northern Washington constituted only 14% of breeding murres in Washington prior to 1983, the small relative increases in northern Washington from 1984 to 1995 have not significantly changed the status of the common murre in Washington to date.

British Columbia

Historical Background on Breeding Colonies in British Columbia, 1900–1979

Historical records for murre colonies in British Columbia are scarce and mostly anecdotal. Early in the twentieth century, large colonies were reported on Triangle Island at the northwest tip of Vancouver Island and on the west coast of Graham Island in the Queen Charlotte Islands. Smaller colonies were reported on the west coast of Vancouver Island at Solander Island and near Ucluelet (Figures 2.14 and 2.15; Brooks and Swarth 1925; Taverner 1928). Subsequent breeding was

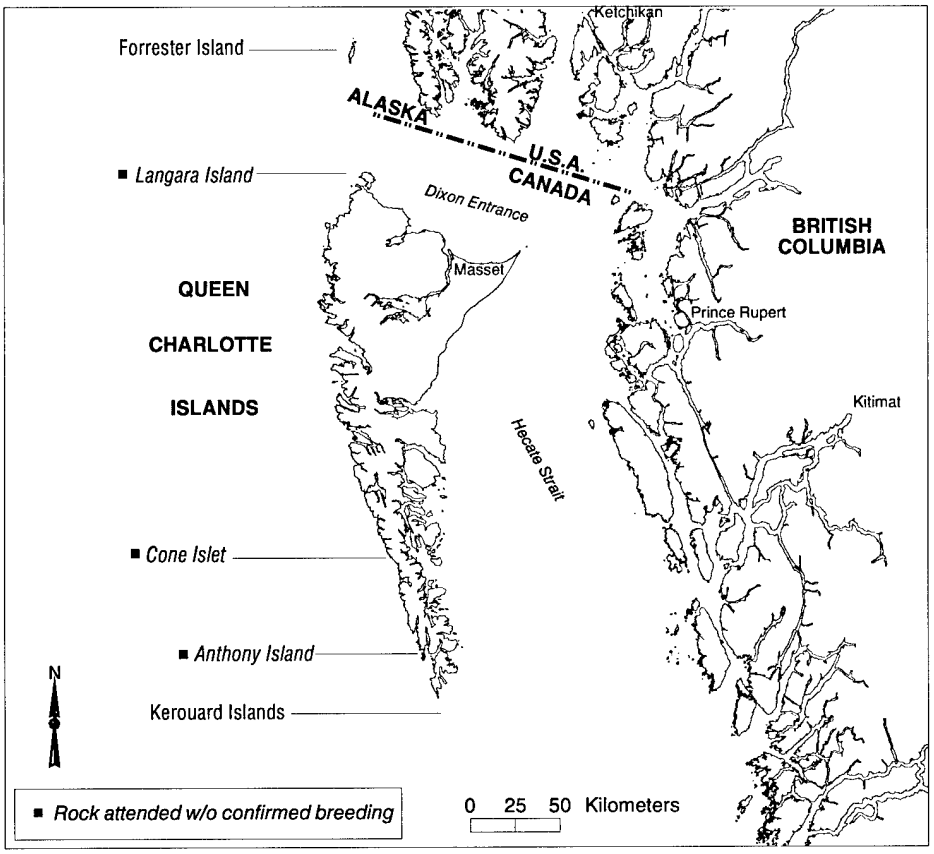

Figure 2.14. Distribution of common murre colonies in northern British Columbia and the southern edge of Alaska.

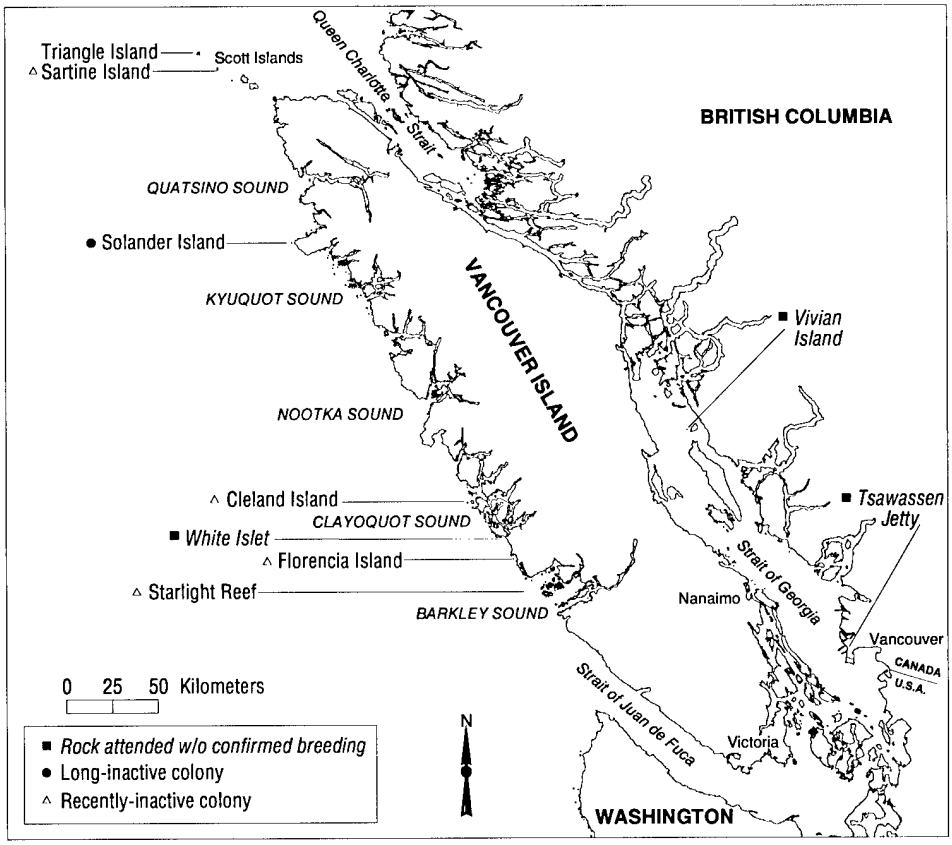

Figure 2.15. Distribution of common murre colonies in southern British Columbia.

documented at Triangle Island and near Ucluelet but not at Solander or Graham Islands.

Triangle Island has been consistently identified as the main murre colony in British Columbia (Brooks and Swarth 1925; Drent and Guiguet 1961; Rodway 1991). Breeding was documented as early as 1900 and has been recorded on all subsequent ornithological expeditions to the island (Kermode 1904; Guiguet 1950; Carl et al. 1951; Drent and Guiguet 1961; Vermeer et al. 1976b; Rodway et al. 1990b; Parrish 1997). A weather station and lighthouse were abandoned in 1921 (Beebe 1960). Between 24 June and 1 July 1949, murre breeding was well documented during a collecting trip by the British Columbia Provincial Museum (now Royal British Columbia Museum [RBCM]; Guiguet 1950; Carl et al. 1951; Drent and Guiguet 1961). Drent and Guiguet (1961) reported 19 eggs collected, although 26 eggs are currently preserved in collections at the RBCM (numbers E1149, E2025–E2038, E228–E229) and the UBCZM (numbers 787–795). One adult male also was "taken off egg" by C. J. Guiguet on 28 June 1949 (RBCM number 9853). In addition, colony size was estimated to be about 1,500 breeding pairs, but egg laying had just commenced (Drent and Guiguet 1961). C. J. Guiguet (unpublished field notes) provides additional details: "Several (7) large roosting colonies observed - each containing several hundreds of birds - nesting sites visited on edge of high cliffs north side - all nests at altitude 525 feet [160 m] - grassy tussocks at edge of sheer drop - eggs only - fresh. Majority of birds apparently haven't laid as yet in areas visited. Total population using island - several thousands."

Several trips were made by the RBCM and British Columbia Ecological Reserves to Triangle Island in the 1960s and early 1970s but few details on murres are available. On 18 and 24 August 1966, small "two-day-old" chicks were collected (RBCM number 11642). On 24 August 1966, a larger chick (RBCM 11645) and an adult (RBCM number 11677) were collected. On 11 August 1974, "many small young" were noted and one egg was collected at Triangle Island (R. W. Campbell, unpublished data; RBCM number E1130). During seabird studies by the Canadian Wildlife Service, Vermeer et al. (1976a) conducted a complete count of 5,934 murres attending Triangle Island (i.e., 5,384 on Puffin Rock and 550 on Castle Rock) on 29 July 1976. In 1977, lower attendance (about 3,000 birds) was noted when almost total breeding failure occurred for murres and tufted puffins *Fratercula cirrhata* (Vermeer et al. 1979).

Breeding was confirmed at Sartine Island, near Triangle Island (Figure 2.15), where 236 and 600 murres were observed on ledges in 1968 and 1975, respectively (Hancock 1971; Vermeer et al. 1976a). Earlier breeding had not been reported in 1950 at Sartine Island or, to the east, at Beresford, Cox, and Lanz Islands (Carl et al. 1951). Mink (*Mustela vision*) and raccoon (*Procyon lotor*) were introduced to Cox and Lanz Islands in 1938–39 and extirpated nesting Cassin's auklets (*Ptychoramphus aleuticus*; Carl et al. 1951; Beebe 1960; Drent and Guiguet 1961; Rodway et al. 1990b). Mammalian predators are not present at Sartine and Beresford Islands, which support large populations of burrow-nesting storm-petrels and alcids, but little breeding habitat for murres exists (see Appendix I: Figure I-43; Rodway et al. 1990b).

Early, unsubstantiated records of breeding murres along the central west coast of Vancouver Island (Kermode 1904; Brooks and Swarth 1925; Taverner 1928) were not accepted in major historical summaries (Munro and Cowan 1947; Drent and Guiguet 1961), but these warrant reconsideration in light of subsequent, confirmed breeding at Cleland Island, Florencia Islet, and Starlight Reef. At Cleland Island, breeding was confirmed between 1969 and 1982 (i.e., 1969–70, 1973–77, 1979, 1982, and 1983), but not in 1967 (Campbell and Stirling 1968; Campbell 1976; Campbell et al. 1975, 1990; British Columbia Nest Records Scheme [BCNRS], see Myers et al. 1957; H. R. Carter and S. G. Sealy, unpublished data; see Appendix I: Figure I-39). Between 2 and 150 murres were reported between 1969 and 1982, but only 1 to 8 breeding pairs laid eggs. When last noted, in 1983, only three murres were seen that may not have been breeding (G. Kaiser, unpublished data).

One and two pairs bred on Starlight Reef in 1975 and 1980, respectively (Hatler et al. 1978; BCNRS). One unsuccessful breeding attempt by a single pair was recorded at Florencia Islet in 1969 (Campbell et al. 1975; Hatler et al. 1978), but no murres bred there in 1970, 1974, or 1979 (Campbell et al. 1975; BCNRS; H. R. Carter and S. G. Sealy, unpublished data). A series of observations at White Islet (between Florencia Islet and Cleland Island) between 1968 and 1970 suggested that 1–2 pairs may have attempted to lay eggs but breeding was not confirmed (R. W. Campbell, unpublished data). On 30 July 1968, 30 murres were noted near shore and two adults were seen in potential nesting habitats, but no eggs or chicks were noted after landing. On 4 August 1969, two adults were again seen in the same location on the rock but no eggs or chicks were found. No murres were noted on 28 June 1970, but one adult was again seen on land in potential breeding habitat on 25 August 1970. Murres were not noted to attend White Islet on

subsequent visits from 1972 to 1979 (R. W. Campbell, unpublished data).

Solander Island is a large seabird colony on the northwest coast of Vancouver Island and contains much suitable breeding habitat (Figure 2.15). However, breeding murres were not observed in 1954, 1975, 1988, or 1989 (Guiguet 1955; Beebe 1960; Drent and Guiguet 1961; Campbell 1976; Rodway and Lemon 1990). We have classified Solander Island as a long-inactive colony, based on available habitat and historic reference as a breeding colony, but without details (Brooks and Swarth 1925). Although 20 murres were noted flying around Solander Island on 27 June 1975 (R. W. Campbell, unpublished data), these birds probably were not attending potential nesting areas.

A large murre colony on the west coast of Graham Island in the northern Queen Charlotte Islands was reported by Haida native people from Masset (Brooks and Swarth 1925). Three large seabird colonies (i.e., Langara, Frederick, and Hippa islands) with some breeding habitat exist along this coast, but breeding murres were not reported in 1927, 1946–47, 1952, 1955–58, 1970–71, 1977, or 1981–88 (Darcus 1930; Beebe 1960; Drent and Guiguet 1961; Campbell and Garrioch 1979; Rodway et al. 1994; C. J. Guiguet, unpublished field notes; S. G. Sealy, personal communication). Large numbers of murres were reported off Langara Island and along the north coast of Graham Island during summer of 1927, 1946–47, and 1952 (Darcus 1930; C. J. Guiguet, unpublished field notes). At Langara Island, Darcus (1930) further noted no murre colonies. On 4 July 1946, C. J. Guiguet (unpublished field notes) noted "six California murres sitting on the rocks below the lighthouse, near sea. This is the first time I've observed these birds on land here". On 18 May 1947, he noted ". . . I have no clues on nesting of these birds in this area". A female with a fully developed egg was collected near Langara on 19 July 1930 (Cumming 1931; Munro and Cowan 1947). We have treated Langara Island as "rocks attended without confirmed breeding." We presume that accounts of large numbers of murres at sea off northern Graham Island during the summer were related to the large colony at nearby Forrester Island in southeastern Alaska (Figure 2.14) that has been documented since 1914 (Willett 1915; Gabrielson and Lincoln 1959; DeGange et al. 1977; Sowls et al. 1978; Slater 1997). In fact, Haida natives may have meant this colony in their original report. Osgood (1901) presumed breeding at the Skedans Islands off the east coast of Moresby Island but breeding was never confirmed subsequently.

Breeding was first confirmed on the Kerouard Islands at the south end of the Queen Charlotte Islands in 1977 (Campbell and Garrioch 1979). This colony represents the northern limit of the known breeding range of the subspecies *U. a. californica*, and is the only confirmed common murre colony in northern British Columbia between Triangle Island and Forrester Island in Alaska. Although breeding was not confirmed, 10 murres were noted on land on 5 August 1977 at the northwest rocks at Anthony Island, north of the Kerouard Islands (R. W. Campbell, unpublished data; H. R. Carter, personal observation). On 3 and 4 June 1982, three birds were flushed from inaccessible cliffs (Rodway et al. 1990a; M. S. Rodway, personal observation). No murres were seen there in 1985 or 1986. In addition, 40 murres were sighted on an unnamed rock ("Cone" Islet) on the west coast of Moresby Island in 1977 (BCNRS; Rodway 1991). Rough weather prevented a close inspection of the rock to confirm breeding in 1977 and the site has not yet been revisited.

Table 2.7. Summary of most-recent surveys of common murres at colonies in British Columbia, 1977–1997.

Colony number	Colony name	Number of murres counted	Number of breeding adults	Survey year
Vancouver Island (Central West Coast)				
WV-550	Starlight Reef	0	0	1982
WV-520	Florencia Islet	0	0	1982
WV-020	Cleland Island	0	0	1997
Vancouver Island (North West Coast)				
WV-080	Solander Island	0	0	1988
SC-020	Sartine Island	113	0	1989
SC-010	Triangle Island	9,943	8,153	1989
Queen Charlotte Islands (South Moresby Island)				
WM-320	Kerouard Islands	200	164	1995
WM-180	"Cone" Islet	40	not confirmed	1977
Total		10,296	8,317	

Current Population Size and Distribution of Breeding Colonies in British Columbia

In British Columbia, murres now breed almost entirely at Triangle Island (Table 2.7), and breeding no longer occurs at four of five other known colonies (Rodway 1991). Murres have not bred recently at Cleland Island, Florencia Island, or Starlight Reef in 1982, 1984, and since 1990 (Rodway and Lemon 1990; BCNRS; A. Dorst, personal communication). At Sartine Island, no murres attended cliffs in 1987 and 1989, but 440 and 113 murres were observed in nearby waters (Rodway et al. 1990b). At the Kerouard Islands, the highest numbers (400 murres) were counted on cliffs in 1987 (Rodway et al. 1990a). Annual records kept by tour-boat operators indicated intermittent attendance of small numbers of birds from 1989 to 1991, and 1994 to 1996 (200 murres in 1995), but none were observed in 1992–93 or 1997 (R. W. Campbell, unpublished data). On occasion, murres also have been noted on land at other locations wherein no breeding occurred—on 2 July 1974, five murres in breeding plumage were observed in intertidal habitats at Vivian Island and on 7 July 1974, two murres were noted on a breakwater off the jetty at the Tsawassen ferry terminal (R. W. Campbell, unpublished data).

Triangle Island was examined extensively from 1980 to 1985 (Vallee and Cannings 1983; Vallee and Carter 1987; Rodway 1990; Rodway et al. 1990b). During this period, murres bred in four main areas: Puffin Rock, Murre Rock, Castle Rock, and Southeast Point (Rodway 1990). On 10 July 1982, a partial count of 4,910 murres was obtained at the main colony on Puffin Rock. In 1984, about 12,000 murres were estimated on the water in early July, but all breeding attempts failed (see below) and only small numbers were present on the breeding slopes. In 1985, murres bred successfully, and 3,956 murres were counted in different breeding areas between 9 and 19 July.

The most complete estimate of colony size for Triangle Island was made in 1989 (Rodway 1990). An average of 5,839 murres (range, 3,335–6,144) was derived from replicated counts of murres from boat photographs between 27 July and 17 August 1989. Numbers from four complete counts (between 1800 and 2000 h [PDT] when daily attendance was highest) ranged from 5,846 to 6,144 birds. To determine a total number of birds attending the colony, the 5,839 mean count was adjusted with a "ground-truthing" or "*g*" correction factor of 1.44. This correction factor was determined by averaging the difference between telescope and photo counts over the 0700–2100 period. Thus, 8,408 birds were estimated to attend photographed areas. An average of 1,535 murres in other areas (i.e., not photographed) were added to obtain a total of 9,943 birds. To derive an estimate of the number of breeding adults at the colony, a *k* correction factor of 0.82 was applied to derive 8,153 breeding birds or 4,077 breeding pairs. This estimate was higher than previous estimates due mainly to more complete coverage of the colony. A repeat of the survey using similar methodology in 1996 probably underestimated total numbers because some chicks and adults had departed from the colony before the count was completed (Parrish 1997).

By adding recent complete counts of murres at Triangle Island and the Kerouard Islands, we obtained a total of 10,296 birds which corresponded to a total breeding population estimate for British Columbia of about 8,300 breeding birds (Table 2.7).

Population Trends in British Columbia

Historical records and recent data were inadequate to determine population trends in British Columbia. Colonization and abandonment of colonies along the west coast of Vancouver Island in the late 1960s and 1970s are difficult to interpret and may indicate an intermittent colonization event, perhaps during a period of colony growth at Triangle Island and colonies in Washington.

Replicated counts at three subcolony sites on Triangle Island in 1982, 1985, and 1989 were highest in 1982 and lowest in 1985 (Table 2.8; Rodway 1990). Counts were conducted at different times in the 3 years, but decrease between 1982 and 1985 and limited increase between 1985 and 1989 were similar to trends at certain northern Washington colony complexes (see Washington section). No murres bred successfully at Triangle Island in 1984 when complete breeding failure of murres and most other surface-breeding species occurred because of severe weather and prey shortage

Table 2.8. Comparison of counts of common murres at subcolony sites on Triangle Island in 3 years from 1982 to 1989 (from Rodway 1990).

Location	Site number	1982	1985	1989
S side W point	13,14	1,140	540	790
W side W point	15	648	400	523
Murre Rock	22, 25, top	1,843	740	1,466

(Rodway et al. 1990b, 1992). A partial failure also occurred in 1989. Large numbers of murre eggs had been eaten by glaucous-winged gulls (*Larus glaucescens*), but it was unknown whether predation contributed to abandonment or occurred afterward (Rodway 1990). Incubating murres that remained at breeding sites sat tight on their eggs when approached by bald eagles or peregrine falcons. Thus, it seemed unlikely that avian predators were the sole cause of failure. Large numbers of murres were killed by the 1988 *Nestucca* oil spill, but a distinct change in the breeding colony at Triangle Island could not be detected (Rodway et al. 1989, 1990b; Burger 1992).

Overall Population Assessment

Current Population Size and Distribution of Breeding Colonies

A complete assessment of the total size and distribution of the overall breeding population of the common murre in California, Oregon, Washington, and British Columbia has been made only once, over a 2-year period from 1988 to 1989. During this period, the overall estimated breeding population was approximately 1.1 million breeding birds (Table 2.9; see Carter et al. 1995). Several previous population estimates of common murres for this portion of western North America were lower and less reliable. Tuck (1961) roughly estimated not more than 1 million murres for California and Oregon without details, and it was not clear if breeding and nonbreeding birds were included in the estimate. Byrd et al. (1993) used a combination of 1979–89 data and reported a total of about 826,000 breeding murres: California (363,000 in 1979–80; Sowls et al. 1980), Oregon (426,000 in 1988; R. W. Lowe, unpublished data), Washington (31,000 in 1978–79; Speich and Wahl 1989), and British Columbia (6,000 in 1988–89; Campbell et al. 1990). However, large population declines occurred in central California and Washington between 1979 and 1989, which makes this combination of data less reliable. Tyler et al. (1993) reported 810,500 breeding murres (minus British Columbia): California (351,000 in 1989; Carter et al. 1992), Oregon (438,100 in 1989; Briggs et al. 1992), and Washington (21,400 in 1989; Briggs et al. 1992). We relied on data largely from the USFWS for murre numbers in Washington and Oregon to maximize compatibility among data sets used to generate population estimates. Rodway (1991) reported 8,640 breeding birds for British Columbia; this estimate was based on the same information as the 8,300 breeding birds estimated in this report.

From 13.0 to 20.7 million breeding individuals, or 6.5 to 10.3 million breeding pairs, of common murre have been estimated in the world, with 54–57% and 43–46% in the Pacific and Atlantic Oceans (including adjacent areas of the Arctic Ocean), respectively (Nettleship and Evans 1985; Byrd et al. 1993; Ainley et al., in preparation). The breeding population in California, Oregon, Washington, and British Columbia (about 1.1 million breeding birds) constitutes 5–8% and 13–28% of the breeding population size of the world and the Pacific Ocean, respectively.

The common murre is the most abundant breeding species of seabird in central California, northern California, and Oregon (Sowls et al. 1980; Varoujean and Pitman 1980; Carter et al. 1992, 1995; Tyler et al. 1993; R. W. Lowe, unpublished data). Suitable habitat (small, bare, nearshore rocks) is abundant and widely distributed along these coasts. Habitat availability and the ability of murres to exploit various abundant prey resources in many different marine habitats near shore and throughout the continental shelf have enabled murres to exist in high abundance within this geographic area. In Washington, Cassin's auklets, rhinoceros auklets (*Cerorhinca monocerata*), and glaucous-winged gulls are more numerous than murres (Speich and Wahl 1989). In south-central California, the Brandt's cormorant becomes the most numerous species of breeding seabird and murres no longer breed south of Monterey County (Hunt et al. 1980; Sowls et al. 1980; Carter et al. 1992). Murres have achieved large breeding populations at most colonies in northern California and Oregon in recent decades, probably in

Table 2.9. Total sum of common murres counted and numbers of breeding adults estimated in California, Oregon, Washington, and British Columbia in 1988–1989.

Geographic area	Year counted	Total sum[a]	Number of breeding adults	Percent of total
Central California	1989	53,985	90,200	8.4
Northern California	1989	156,555	261,400	24.2
Oregon	1988	426,278	711,900	66.0
Washington	1988	4,190	7,000	0.6
British Columbia	1989	10,296	8,300	0.8
Total		651,304	1,078,800	

[a] Sum of whole-colony counts at all colonies in a geographic area.

response to the relatively low levels of colony disturbance and anthropogenic mortality, and excellent prey conditions within the central part of the California Current upwelling system (Briggs et al. 1987; Ainley and Boekelheide 1990; Tyler et al. 1993; Manuwal and Carter 2001).

About 66% of the overall breeding population of common murres (*U. a. californica*) is present in Oregon (Table 2.9). More than 420,000 murres were estimated breeding along the central and north coasts of Oregon with the largest colonies at Bird Rocks, Three Arch Rocks, Two Arches Rock complex, and Gull and Colony Rocks near Newport. The southern Oregon coast contained approximately 290,000 breeding murres with the largest colonies at Cat and Kittens, Gull Rock (Cape Blanco), Orford Reef, Island Rock, Hubbard Mound Reef, Mack Arch, and outer Whaleshead. Significant numbers (24%) also bred in northern California, mainly on several large offshore rocks in Del Norte and Humboldt Counties (especially Castle Rock, False Klamath Rock, Green Rock, Flatiron Rock, and False Cape Rocks). Combined, Oregon and northern California comprise the current population "core" or 90% of the breeding birds which form the geographic center of the entire *U. a. californica* population. This "core" area is located in the central part of the California Current upwelling system, characterized by strong and persistent upwelling during the spring and summer (Briggs et al. 1987; Tyler et al. 1993). Prey resources and breeding habitat in the area appear to have been sufficient to sustain this major part of the population from 1979 to 1995.

Historically, very large numbers of murres were present in central California, which also is located within the central part of the California Current upwelling system (Briggs et al. 1987; Ainley and Boekelheide 1990; Tyler et al. 1993). In the early nineteenth century, central California had a much larger proportion of breeding murres before near extirpation of the immense colony at the South Farallon Islands. This colony may have totaled 1–3 million breeding birds at its peak. By 1989, the number of breeding murres in central California was at the lowest recorded level between 1979 and 1995 and comprised only about 8% of the total population of breeding murres (Table 2.9). In 1979–82, more than twice as many murres bred in central California than in 1989. The largest colonies were present at the South Farallon Islands, North Farallon Islands, and Point Reyes. Presently, the southernmost colony in California is in central California at Hurricane Point Rocks. In the past, murres bred as far south as Prince Island in the Channel Islands off southern California.

The southern limit of the breeding range of the common murre in the eastern Pacific Ocean is roughly aligned with the southern edge of the California Current upwelling system off southern California and western Baja California, where colder subarctic waters are diluted by warmer waters from the central ocean gyre (Tyler et al. 1993). Several other alcids also reach their southern limit in southern or central California (i.e., pigeon guillemot [*Cepphus columba*], marbled murrelet [*Brachyramphus marmoratus*], rhinoceros auklet, and tufted puffin) or their northern limit (i.e., Xantus's murrelet [*Synthliboramphus hypoleucus*]; Hunt et al. 1980; Sowls et al. 1980; Carter et al. 1992; Gaston and Jones 1998). A major change in climate, breeding habitats, prey resources, and natural predators occurs in this area, which affects breeding by several breeding seabird species (Hunt et al. 1980; Briggs et al. 1987; Carter et al. 1992; Tyler et al. 1993). In addition, large populations of native peoples used mainly marine food resources and probably prevented breeding in many parts of the Channel Islands off southern California for thousands of years (e.g., Glassow 1980) until they were extirpated from the area in the mid-nineteenth century. In the nineteenth and twentieth centuries, early European and American settlers also affected seabird populations in southern California and northwestern Baja California, with egg-collecting activities, introduction of predators to islands (McChesney and Tershy 1998), and other activities. Thus, the southern limit of breeding murres *U. a. californica* may have occurred in the southern California or possibly northwestern Baja California for at least tens of thousands of years.

Breeding murres in Washington represented less than 1% of the total breeding population of *U. a. californica* in 1988–89 (Table 2.9) but were several times more numerous between 1979 and 1982. The few colonies in British Columbia also comprised less than 1% of total breeding population (Table 2.9). Most birds in British Columbia bred at one isolated colony at Triangle Island, at the north tip of Vancouver Island. Other colonies are small, widely separated, and irregularly attended. The northernmost colony of *U. a. californica* is located at the Kerouard Islands at the southern end of the Queen Charlotte Islands. There is no evidence that murres ever bred more widely in British Columbia.

A major change in breeding habitat occurs on the west coast of Vancouver Island, British Columbia, where most of the outer coast islands become forested, and those that are not tend to be small, low and rounded, and less suitable for breeding by murres and some other seabirds (Beebe 1960; Campbell et al. 1990). In comparison with Washington, Oregon, and California,

the availability of open breeding habitats on islands in British Columbia are reduced. Washington and southern British Columbia also are located at the northern end of the California Current upwelling system where it meets the Alaska Current during spring and summer (Morgan et al. 1991; Tyler et al. 1993; Wahl et al. 1993). Different prey resources are associated with the estuarine conditions within the extensive fiord system along the coasts of British Columbia and southeastern Alaska. Also, large populations of native people were present for at least thousands of years in British Columbia prior to the mid-nineteenth century (Duff 1997), and probably visited many seabird nesting islands to obtain eggs and birds for food. Certain murre colonies may have been extirpated prior to 1800. However, reduced availability of breeding habitats and change in prey resources may be the primary factors contributing to the current geographic gap between murres breeding in Alaska (*U. a. inornata*) and from southern British Columbia to California (*U. a. californica*).

During glacial periods in the last 1 million years (i.e., the last period ended about 10,000 years ago), continental ice sheets extended to sea level from northern Washington through much of southern Alaska, which coincides to a large degree with the current geographic gap and is related to changes in coastal topography. Glacial history probably is a major factor underlying the current gap in distribution, and also may have strongly influenced the current location of the major portion of the population of *U. a. californica* in Oregon and California. However, the fossil history of *Uria* extends back at least 5 million years in southern California (Barnes et al. 1981; Howard 1949, 1981, 1982; see Bèdard 1985). Various changes in seabird communities, marine environments, and coastal topography have occurred in the North Pacific over millions of years and influenced the distribution and abundance of the common murre.

Recent and Historical Population Trends

The numbers and distribution of common murres in the Oregon and northern California "core" population between 1979 and 1995 seem to represent relatively stable high levels, possibly indicative of near carrying-capacity levels and distribution (Figure 2.16). In this area, murre numbers have stabilized for several decades, apparently in relation to available breeding habitat, prey resources, and relatively low levels of human disturbance at colonies. Most suitable breeding habitat is occupied, although some habitat has been removed historically by either connecting islands to the mainland with breakwaters or modifying islands for lighthouses or other structures. The abundance and availability of prey resources have not been well studied, but evidently have been adequate to maintain populations at current high levels. Few natural factors are known that would disrupt this stability. Lower numbers of breeding birds

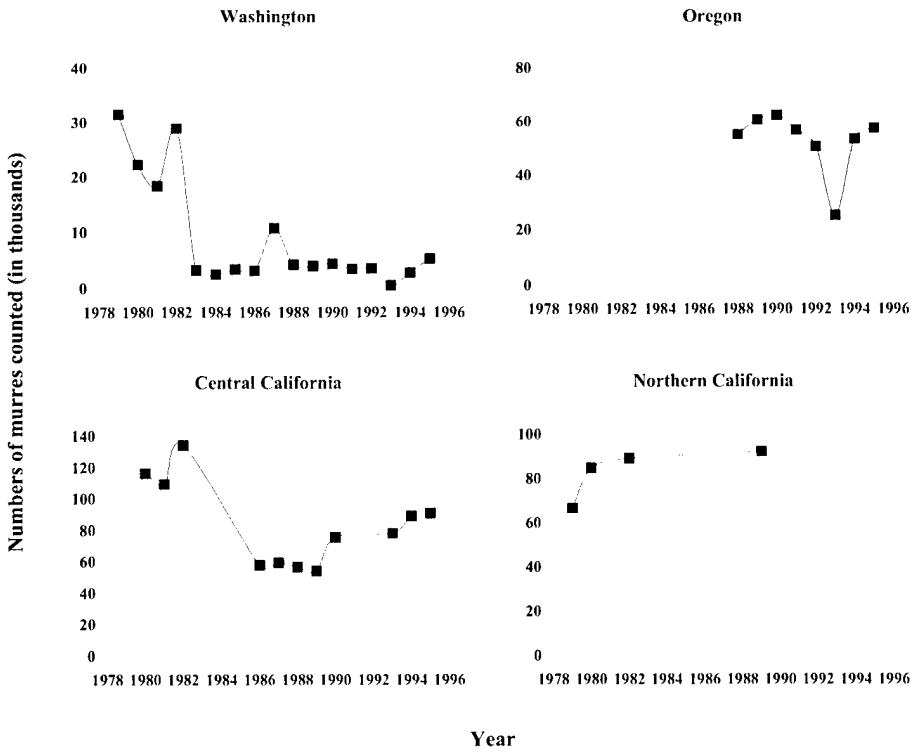

Figure 2.16. Changes in whole-colony counts of common murres at breeding colonies in central California (all colonies), northern California (excluding the Castle Rock colony), Oregon (15 sample colonies), and Washington (excluding Tatoosh colonies), 1979–1995.

attended colonies during severe El Niños in 1983–84 and 1992–93. These events probably caused lower prey availability or accessibility over the short term (e.g., Ainley and Boekelheide 1990). Resultant depressions in reproductive performance apparently have not had long-term effects on the size of the "core" populations in northern California and Oregon. Few anthropogenic factors have affected murres in Oregon and northern California in recent decades except in fall and winter when most Oregon birds move north to Washington and British Columbia. Some Oregon birds undoubtedly have been killed in gill nets and by oil spills in Washington and British Columbia (especially the 1988 *Nestucca* and 1991 *Tenyo Maru* oil spills) between 1979–95. Yet, despite such deaths, the "core" of the overall population did not change to a large degree. The large population size and the wide dispersal of anthropogenic deaths among many colonies may have lessened effects on the Oregon population.

To reach current population levels, murres in northern California and Oregon had to recover over many decades (mainly from the 1940s to the 1970s) from extensive human impacts that occurred in the late nineteenth and early twentieth centuries. As impacts from native peoples and early settlers declined, the numbers and range of breeding murres increased to the current high population levels. During this period, colony formations (including recolonization events) occurred widely. Adequate prey resources, available breeding habitat, and relatively low natural or anthropogenic deaths must have existed for this recovery to occur. In northern California, extensive recovery has occurred in Del Norte and Humboldt Counties but recovery is still occurring in Mendocino and Sonoma Counties where recolonization events and population increase are ongoing.

In central California, historical effects by early settlers reduced this population to low levels. Extirpation of colonies in Sonoma and, possibly, Mendocino counties may have caused the geographic gap between murres breeding in central and northern California. Colony extirpation was recorded at San Pedro Rock and near extirpation at the South Farallon Islands. By the early 1980s, many colonies had increased substantially, but were still well below known historical levels. As in northern California, adequate prey resources, available breeding habitat, and relatively low natural and anthropogenic mortality existed for this limited partial recovery to occur. Breeding habitat at the South Farallon Islands has been reduced from historical conditions, thus, it is unlikely this colony will ever return to levels reached in the early nineteenth century. Declines between 1982 and 1989 occurred at all colonies in central California and one colony (Devil's Slide Rock) was extirpated. Partial recovery in central California between 1989 and 1995 has been slow and limited, possibly reflecting the relative severity of the original decline, as well as continuing effects. Breeding success has remained high at the South Farallon Islands (except during severe El Niños) and is not a factor impeding recovery at most colonies (Hastings et al. 1997; Sydeman et al. 1997; McChesney et al. 1998, 1999; M. W. Parker, unpublished data). However, mortality from recent oil spills (e.g., 1996 *Mohican*, 1997–98 Point Reyes Tarball Incidents, and 1998 *Command*) and the recent resurgence of significant deaths in gill nets in Monterey Bay have increased anthropogenic impacts since 1995 (P. R. Kelly, personal communication).

In Washington, numbers of murres attending colonies in 1979–82 reflected growth since the early twentieth century. Decline and little recovery between 1982 and 1995 in Washington appear to have resulted from severe effects (from natural and anthropogenic factors) on the murre population over the long term (Figure 2.16). Murre attendance at the largest colonies in southern Washington (i.e., Split Rock, Willoughby Island, Grenville Arch, and Rounded Island) plummeted to small numbers of irregularly-occurring birds and evidence of reproductive effort and success has been largely absent since the initial decline. Small colonies in northern Washington also declined, but to a lesser degree and have shown limited growth in recent years, possibly because of intercolony movements from southern Washington colonies. Massive decline and a lack of recovery in southern Washington may be related to the lower initial population size in Washington before the decline (compared with populations in Oregon and California), the high magnitude of natural and anthropogenic impacts over an extended period of time, and intercolony movements of birds to northern Washington colonies. However, small numbers of birds still attend traditional breeding colony locations in southern Washington and some recovery may be possible in the future. The likelihood of rapid natural recovery in Washington is very low because of continued anthropogenic and natural effects and the slow rate of murre recovery documented at severely reduced colonies elsewhere along the Pacific coast.

Overall, murre numbers in central California and Washington have declined substantially since the early 1980s and currently exist well below historical population levels and distribution (Figure 2.16). Major declines occurred rapidly between 1982–86, and low numbers have remained over extended periods of time following these declines. Although limited increase has

occurred in central California in recent years, numbers remain depressed in Washington. Prey resources have been little studied but were apparently adequate to maintain these populations at higher population levels in 1979–82. Large-scale declines between 1979 and 1989 resulted from long-term impacts from anthropogenic factors (i.e., gill-net and oil-spill deaths and human disturbance), coupled with natural factors (i.e., reduced reproductive effort and success associated with severe 1982–83 El Niño, and the 1981 warm water event in Washington). At the same time, climate change has been occurring with a significant warming of coastal waters which also may be affecting murre prey resources (Roemmich and McGowan 1995; Ainley et al. 1996). In central California, climate change has not prevented recent increase in the murre population, but may have reduced the rate of increase (Sydeman et al. 1997). The apparent overall stability of populations in Oregon and northern California between 1979 and 1989 underscored the fact that natural factors alone were not responsible for major declines in central California and Washington (Takekawa et al. 1990; Carter et al. 1995). Washington populations now persist at extremely low levels and are affected by continuing anthropogenic and natural factors, which probably have prevented or slowed recovery.

The status of common murres in British Columbia is poorly known. The isolated colony at Triangle Island has been present since at least the beginning of the twentieth century. At this colony numbers of breeding murres were relatively stable between 1982 and 1989. Small colonies on the west coast of Vancouver Island disappeared in the 1970s and 1980s. In the past, most potential breeding islands for murres in British Columbia probably were visited frequently by native people hunting seabirds. Murres on Triangle Island breed largely on inaccessible cliffs far from the coast of Vancouver Island, enabling this colony to coexist with native peoples over extended periods of time. Few Europeans or Canadians settled the outer west coast of Vancouver Island (except during a brief sardine fishery in the 1930s), which suggests that human effects were probably low during the twentieth century. However, mortality from oil spills or gill nets may have impacted these colonies, either during the breeding season (e.g., Barkley Sound; Carter and Sealy 1984) or in wintering areas in Juan de Fuca Strait, Puget Sound, or the Straits of Georgia.

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