POPULATION TRENDS AND DISTRIBUTION OF COMMON MURRE *URIA AALGE* COLONIES IN WASHINGTON, 1996–2015

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ABSTRACT

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Periodic assessments of population trends and changes in spatial distribution are valuable for managing marine birds and their breeding habitats, particularly when evaluating long-term response to threats such as oil spills, predation pressure, and changing ocean conditions. We evaluated recent trends in abundance and distribution of the Common Murre *Uria aalge* within Copalis, Quillayute Needles, and Flattery Rocks National Wildlife Refuges, which include all murre colonies in Washington except one, off-refuge, on Tatoosh Island. In 1996–2001 and 2010–2015, aerial photographic surveys were conducted during the incubation phase (mid-June through mid-July) each year. Using images from film (1996–2001) and digital (2010–2015) cameras that included all parts of each colony, we manually counted murres. We estimated population trend as annual percent change in whole-colony counts using an overdispersed Poisson regression model. Overall, numbers of murres counted at breeding colonies in Washington increased by 8.8% per year (95% CI 3.0%–14.9%) during 1996–2015. The overall statewide increase was driven by an increase at colonies in northern Washington of approximately 11% per year (95% CI 4.5%–17.8%). Despite an increasing trend, abundance remains lower than levels in the late 1970s, and the spatial distribution has changed. Colonies in southern Washington — where murres were historically the most abundant — are no longer active, or only minimally so, whereas colonies in the north — which were rarely active in the early 1970s — are now the largest. There was high variability in spatial distribution among years, a pattern that indicates a need for coordinated monitoring and movement studies throughout the California Current System to understand dispersal and colonization. Our results indicate that future management of refuge islands could protect both current and historic colony locations, given the patterns of colony dynamics and the uncertainty about long-term effects of a changing ocean ecosystem and predation pressure on the status of murres.

Key words: Common Murre, Washington, trends, distribution, Poisson regression

INTRODUCTION

Common Murres *Uria aalge* are one of the most common surface-nesting seabirds in the California Current System (CCS), from southern British Columbia through California. The most recent estimate for the subpopulation that breeds in the CCS (*U. a. californica;* hereafter murres) was 1.1 million breeding birds in 1988, with approximately 7000 nesting in Washington at that time (Carter *et al.* 2001). Trend analysis conducted by Carter *et al.* (2001) indicated an overall decline of 13.3% per year in Washington during 1979–1995. This trend varied spatially, with declines of 3.7% per year in northern colonies and 25.5% per year in southern colonies (Carter *et al.* 2001). In recent decades, murre population trends appear to have declined in British Columbia (Hipfner 2005), remained stable in Oregon (S. Stephenson, pers. comm.), and increased in California (G. McChesney, pers. comm.).

Murres face numerous anthropogenic threats, such as oil spills and fisheries bycatch, as well as population regulation from both top-down (predation) and bottom-up (environmental or physical) forcing (Wilson 1991, Parrish *et al.* 2001, Parrish & Zador 2003, Ainley & Hyrenbach 2010). The decline of murres in Washington between 1979 and 1995 has been associated with oil spills, strong El Niño Southern Oscillation (ENSO) events, and bycatch in gillnet fisheries (Wilson 1991, Manuwal *et al.* 2001, Thompson *et al.* 2003, Helm *et al.* 2006). In Washington, murres are considered highly vulnerable to oil-spill contamination because they raft on the water in large groups and have relatively small wintering areas that overlap with shipping lanes, particularly within the Salish Sea (Thompson *et al.* 2003). Murres are also highly vulnerable to fisheries bycatch; they were captured and drowned in gillnets more often than any other bird species in Washington, with estimated annual fisheries-related mortality numbering in the thousands in some years (Thompson *et al.*1998, Hamel *et al.* 2009).

Murre populations in the Pacific Northwest are also subject to top-down forcing by recovering Bald Eagle *Haliaeetus leucocephalus* populations (Parrish *et al.* 2001, Hipfner *et al*. 2012). Top-down forcing can cause local declines, shifts in murre colony occupancy, or changes in abundance through a combination of eagle predation of adults and chicks and eagle-induced gull predation on eggs or chicks (Parrish *et al.* 2001, Zador *et al.* 2009, Hipfner *et al.* 2012, Horton 2014). For instance, in Oregon a shift in murre colonies from the northern to the central and southern coast corresponded to an increasing density of eagle nests, particularly along the north coast (Horton 2014). Recent modeling efforts suggest that the recovery of Bald Eagle populations in the Puget Sound region could result in a trophic cascade extending through at least three trophic levels (eagles → seabirds → small fishes or invertebrates), resulting in a fundamental restructuring of the food web (Harvey *et al.* 2012). This dynamic is further complicated by recovering Peregrine Falcon *Falco peregrinus* populations in the Pacific Northwest. Hipfner *et al.* (2012) found that eagle activity around a subcolony on Triangle Island, BC, was effectively eliminated by a pair of falcons protecting an eyrie adjacent to the murre sub-colony during 2003–2006 and resulted in greater murre breeding success (Hipfner *et al.* 2012).

Bottom-up forcing is important as well (Zador *et al.* 2009). Limited availability of forage resources results in delayed lay dates, poor productivity, or colony abandonment, as well as lower recruitment and/or a redistribution of colonies over the long term (Wilson 1991, Parrish & Zador 2003, Zador *et al.* 2009, Ainley & Hyrenbach 2010, Schrimpf *et al.* 2012, Sydeman *et al.* 2014). Changing ocean conditions amplify negative impacts of bottom-up forcing by further reducing ocean productivity at multiple temporal scales.

Given the various factors affecting murre abundance, periodic assessments of trends and distribution are valuable for managing this species and its breeding habitat, particularly in response to changing ocean conditions and predation pressure. We assessed murre population trends and changes in the spatial distribution of breeding colonies on National Wildlife Refuge islands along the outer coast of Washington during 1996–2001 and 2010–2015.

METHODS

Survey area

Our survey area encompassed all historic and current murre colonies as well as regular surveys of other potentially suitable islands on Copalis, Quillayute Needles, and Flattery Rocks National Wildlife Refuges (Fig. 1). These refuges are located along 135 km of coastline between Cape Flattery (48.367°N, 124.730°W) and Point Grenville (47.298°N, 124.268°W), Washington, and up to 8 km offshore. Following Carter *et al*. (2001), we divided colonies into two geographic regions: northern Washington (18 colonies in Clallam and Jefferson counties) and southern Washington (six colonies in Grays Harbor County). The only known colony in Washington outside of our survey area is located on Tatoosh Island

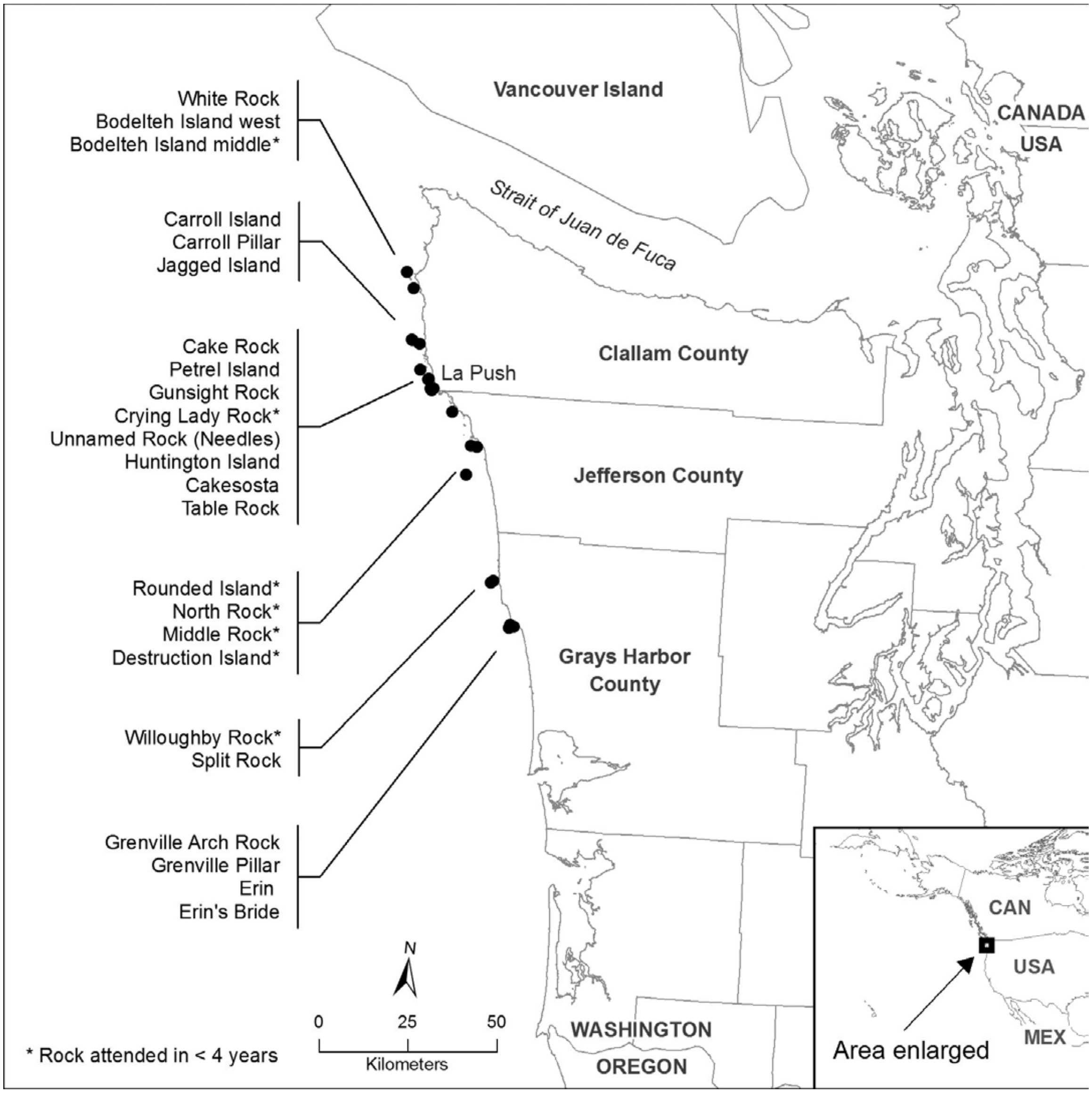

Fig. 1. Location of Common Murre colonies in this analysis, Clallam to Grays Harbor Counties, WA.

(48.391°N, 124.737°W). This island has supported one of the larger nesting aggregations in the state, with an attendance low of 2 750 in 1998 and a high of 5 700 in 2006 (Julia K. Parrish, pers. comm.). This island was not covered due to restrictions on research and monitoring on the island as well as funding limitations.

Murre surveys and photo processing

Murre surveys were conducted during two time periods: 1996–2001 and 2010–2015. Field methods, survey equipment, and procedures to count murres in photographs differed slightly between these two time periods. During 2010–2015, we followed a standardized method established in 2010 for aerial surveys of surface-nesting seabirds in Oregon and Washington, as described here. We conducted surveys during the incubation phase (mid-June through mid-July), when murres are most active on the colony, between 09h00 and 13h00, weather permitting (although surveys in 2010–2012 were flown in early June, slightly earlier than other years). The 12 survey dates and the count data used in this analysis are provided in Appendix 1 (available online at http://www.marineornithology.org/). All flights were conducted onboard a Bell 206 helicopter with the passenger side doors removed. Photographs were taken with Canon EOS 5D Mark II cameras with 100–400 mm f/4.5–5.6 L lenses and image stabilization set to 1.8 m–∞. To optimize overall image quality and account for more vibration in the back seat, we used the following settings: front seat — aperture priority mode with f-stop set to 5.6 and ISO set to 800; back seat — shutter priority mode with shutter speed set to 1/2 500 sec or faster, automatic f-stop, and ISO set to 500–800. We photographed all sections of colony islands above the high-tide line, regardless of whether seabirds were observed. To avoid disturbance, we photographed colonies from an altitude of 200–250 m. The pilot circumnavigated the island at approximately 75–140 km/h and 180–225 m from the island/rock perimeter while we took as many photographs as possible.

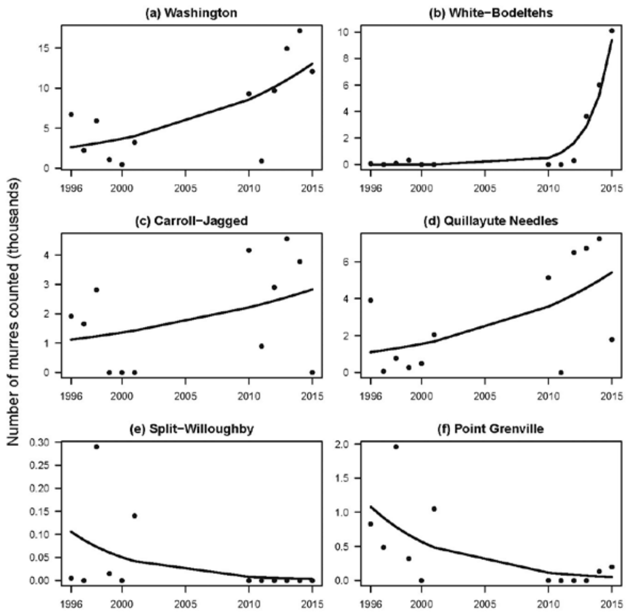

Fig. 2. Changes in whole-colony counts of Common Murres in Washington (except Tatoosh Island), 1996–2015, at (a) all colonies in state combined, (b–d) colony complexes in northern Washington, and (e–f) colony complexes in southern Washington. Solid line in each panel is trend in sum of colony counts estimated with an overdispersed Poisson regression model.

We processed three to six photos into panoramas using Adobe Photoshop (CS4) to reduce the time needed to reconcile overlapping photos and the potential for double counting. We then used ArcMap v10.1 to manually count individual birds as point feature classes within panoramas or individual photos to account for birds in areas that did not appear in the panoramas. Because it was not possible to verify each count, given the time and personnel available, we randomly selected 30% of the counts of colonies photographed during 2013–2015 and counted the entire colony again. If the difference between the verification count and the original count was ≥6% of the original count, we re-counted the colony and reconciled the difference.

Surveys conducted during 1996–1998 followed the photographic and counting methods used in 1995 and described by Carter *et al*. (2001; see also Wilson 1997, 1999). We assumed that the same methods were used in the ensuing years, 1999–2001, but documentation to address this assumption is lacking. There were some differences in methods of the early (1996–2001) and later (2010–2015) years of our study. Specifically, in the early years, flight altitudes varied from 70 to 250 m; photographs were made with a Canon EOS A2 35 mm camera equipped with a Canon EF 70–200 mm f/2.8 L lens and Kodak Ektachrome 400 ASA Elite film; film images were processed into slides; slides were projected; and murres were counted with a tally counter or estimated if the quality of the image was poor. When making inferences about population change over time, we assumed the differences in methodology (i.e., altitude and technological improvements associated with slide vs. digital counting) did not substantially affect our results.

Trend estimation

Our trend estimation is based on the sum of whole-colony counts for each year at five colony complexes and in three geographic regions: northern Washington, southern Washington, and the entire state (Fig. 1). We estimated trends in the sum of whole-colony counts (*N*) for complexes and regions during 1996–2015 with an overdispersed Poisson model:

$$N \sim Poisson(\lambda),\ Var(\lambda)=\lambda\varphi \qquad (1)$$

$$log(\lambda)=\beta_0+\beta(year) \qquad (2)$$

where λ is the mean and variance of the Poisson random variable, φ is the overdispersion parameter, β is log-scale mean abundance, and β is log-scale estimated change in abundance with each year. The overdispersed Poisson model allows for extra-Poisson variation in the counts and is preferred to a negative binomial regression model when modeling overall abundance at multiple sites (Ver Hoef & Boveng 2007). We calculated percent change in population size each year using a back-transformation of the log-scale trend estimate, $(exp(\beta)-1) \times 100$; confidence intervals were calculated using log-scale parameter estimates for β and standard error (β). We made inferences about population change using (1) a test of the null hypothesis that the parameter estimate for change over time (β) was equal to 0, and (2) the confidence intervals for percent change per year (i.e., 95% CI that do not include 0 indicated a significant trend). We tested for serial correlation in the sum of colony counts for Washington (*N*) with the autocorrelation function; we calculated the autocorrelation function for both standardized (*z*-score) counts (Cameron & Trivedi 1998) and standardized residuals from the Poisson model. In addition, we tested for 0 correlation between residuals from the Poisson model and (one-year) lagged residuals. Neither the autocorrelation functions nor the correlation test with lagged residuals indicated significant serial correlation, so we consider the overdispersed Poisson model to be adequate for estimating trend in population size. Statistical analyses were conducted in R (R Core Team 2015); the regression model (Eq. 1) was fitted using the *glm* function and the tests for autocorrelation were conducted with the *acf* and *cor.test* functions.

RESULTS

Overall numbers of murres counted at breeding colonies in Washington increased by 8.8% per year (95% CI 3.0%–14.9%) during 1996–2015 (*t* = 3.02, *P* = 0.01; Table 1). Between 1996 and 2001, the average number of murres counted each year was 3 280 (average sum of whole-colony counts); during 2010–2015, the average number counted was 10 684 (Fig. 2a, Appendix 1). Population change was not consistent among geographic regions, however. The overall increase in populations statewide during 1996–2015 was the result of increases at colonies in northern Washington, which are relatively large compared to those in southern Washington (Fig. 2). Between 1996 and 2015, colonies in northern Washington increased by approximately 11% per year (95% CI 4.5%–17.8%; *t* = 3.41, *P* = 0.007; Table 1). Populations increased at all colony complexes in northern Washington, but increases were most substantial in the White-Bodeltehs complex (West Bodelteh was labeled Unnamed Rock #140 in Carter *et al.* 2001, Appendix F). In the early years, 1996–2001, this complex was unoccupied during three of the six years and did not exceed 325 birds in the remainder. By 2014–2015, however, approximately 6 000–10 000 murres were counted at White-Bodeltehs (Fig. 2b, Appendix 1). Because population changes were small in the early years and counts varied widely in later years, confidence intervals for percent change per year spanned 0 at both the White-Bodeltehs and Carroll-Jagged complexes (Table 1, Fig. 2b, c). Populations at the third colony complex in northern Washington, Quillayute Needles, increased by 8.7% (95% CI 0.7%–17.4%) per year during 1996–2015 (Table 1, Fig. 2d). In southern Washington, populations decreased 15% per year during 1996–2015 (95% CI -25.9 to -2.5%; *t* = -2.3, *P* = 0.04; Table 1). Between 1996 and 2001, an average of approximately 850 murres were counted at colonies in southern Washington; by the later years (2010–2015), colonies in southern Washington were essentially unoccupied during 2010–2013, and the average number counted during 2010–2015 was down to 55 murres (Fig. 2e,f; Appendix 1). The population changes in northern and southern Washington resulted in a shift in distribution during the timeframe of our analysis. At the beginning, 88% of all breeding murres in the state (excluding Tatoosh Island) were found at colonies in northern Washington; by the end, this percentage had increased to 98%.

Population trend estimates vs. timing of assessments

During 2010–2012, our surveys were conducted slightly earlier in the season than in other years in this study. To investigate the sensitivity of our estimated trends to seasonal timing of the surveys, we conducted a separate trend analysis without years 2010–2012. Estimated trends with this restricted dataset were similar to the analysis with all years, differing by <1%. We also conducted separate analyses for early (1996–2001) and late (2010–2015) years of our study to investigate trends in the two periods, which were separated by eight years without data. Not surprisingly (Fig. 2), results of separate analyses showed a decreasing trend in the early years and an increasing trend in the later years, including an increasing trend in southern Washington due to a small increase in 2014–2015 from 0 birds. To suggest there is an increasing trend in southern Washington would be misleading, and we maintain that a better understanding of population change in Washington is provided when all years are included in the analysis.

DISCUSSION

We found that murre populations regularly surveyed via aerial methods in Washington have increased by almost 9% per year between 1996 and 2015, particularly in northern Washington. Yet, despite increasing population trends in recent years, murre

TABLE 1
Trends in whole-colony counts (*N*) at five colony complexes and three geographic regions in Washington, 1996–2015, using an overdispersed Poisson regression model

Region/complex	β = Slope of ln(*N*)						% per year = ($e^{β}$ - 1) × 100		
			95% CI					95% CI	
	Estimate (β)	SE	Lower	Upper	*t*[a]	*P*	Estimate	Lower	Upper
Washington total[b]	0.08	0.03	0.030	0.139	3.02	0.013	8.8	3.0	14.9
Northern Washington[c]	0.10	0.03	0.044	0.164	3.41	0.007	11.0	4.5	17.8
Carroll-Jagged	0.05	0.04	-0.023	0.121	1.32	0.215	5.0	-2.3	12.9
Quillayute Needles	0.08	0.04	0.007	0.160	2.14	0.058	8.7	0.7	17.4
White-Bodeltehs	0.59	0.72	-0.827	2.005	0.82	0.434	80.2	-56.3	642.9
Southern Washington	-0.16	0.07	-0.300	-0.025	-2.32	0.043	-15.0	-25.9	-2.5
Point Grenville	-0.16	0.07	-0.291	-0.031	-2.42	0.036	-14.9	-25.2	-3.0
Split-Willoughby	-0.19	0.13	-0.441	0.070	-1.42	0.185	-16.9	-35.6	7.3

[a] *t* is the ratio of β parameter estimate and SE.

[b] Does not include Tatoosh Island.

[c] Northern Washington includes complexes listed and other locations.

populations in Washington are still roughly only half those in 1979, as compared with the results of Carter *et al*. (2001). Making that comparison, though, required some adjustments. Population estimates in our analysis were based on raw, unadjusted colony counts and therefore do not account for birds at sea during the time of the survey. Based on intensive colony observations at the Farallon Islands, California, Carter *et al*. (2001) recommend an adjustment factor of $k = 1.67$ to account for birds at sea. However, given the high daily and annual variability of attendance, as well as differences between and within colonies (Harris *et al*. 2015), the most appropriate adjustment to raw counts may vary from year to year. Nevertheless, to facilitate comparisons of results with population size estimates from Carter *et al.* (2001), we adjusted raw counts by a factor of 1.67. Thus, an adjusted population size estimate for Washington (excluding Tatoosh Island) in 2014, the year of peak population size during the period under analysis, was approximately 28 650 birds, while the minimum adjusted population estimate was 800 birds in 2000. Interestingly, the 2014 estimate is corroborated by at-sea surveys using distance sampling, which resulted in an estimated population size of 22 839 birds (95% CI 11 637–36 398; S. Pearson, pers. comm.). Close agreement between carefully conducted colony and at-sea counts of murres is not novel (Clarke *et al*. 2003). In contrast to our high count, Carter *et al.* (2001) estimated a population size that was nearly double, at 53 000 murres, during the highest count year of 1979 (minimum estimate of 885 in 1993), including an adjusted estimate of 334 birds for Tatoosh Island.

Changes in spatial distribution of breeding colonies over time

In contrast to the declining trend observed in Washington between 1979 and 1995 (-13.3% per year, 95% CI -20.2 to -5.7%; Carter *et al*. 2001, Table H-2), we found an increasing trend of 8.8% per year. The decline in the 1980s and early 1990s was attributed to a significant decline in the southern colonies of 25% per year, while the northern colonies declined at a slower rate of 4% per year (also excluding Tatoosh Island; Carter *et al*. 2001). We found a continuing decline in southern colonies (-15% per year) but an increase of nearly 11% per year in the northern colonies. During 1996–2015, the northernmost colonies on White Rock and West Bodelteh have shown the greatest increase (approximately 80% and 135% annually for the entire period and recent years, respectively), and ranked among the top three colonies in recent years. During 1979–1995, these colonies had been sparsely attended (Carter *et al.* 2001). The largest southern colonies in 1979 (Grenville Arch, Radio Stack, Split, Willoughby, and Rounded) have been inactive between 2010 and 2015, and only a small number of murres have been in attendance at the smaller colonies (Erin and Erin's Bride) during that time. Although methods have varied over time, and not all colonies in the survey area were monitored regularly, historic data from Washington provide further evidence for great variability in the spatial distribution of colonies over time (Ainley *et al*. 2002; see also Manuwal & Campbell 1979, Speich & Wahl 1989, Carter *et al.* 2001 for further details).

Murre colonies shift in response to changes in availability of prey and/or predation pressure (Carter *et al.* 2001, Parrish *et al.* 2001, Hipfner 2005, Horton 2014). Zador *et al.* (2009) found that, during years with favorable environmental conditions, new subcolonies formed on Tatoosh Island. Horton (2014) found that murre colonies declined by 50% on the north coast of Oregon, where Bald Eagle nest density was highest; colonies along the central coast, where eagle nest density was moderate, increased by 104%. Hipfner (2005) suggested that the increase in the number of eagle nests on Triangle Island, BC, could account for the abandonment of satellite colonies and an increase of the main colony. On the Olympic Peninsula of Washington, the number of eagle nesting territories increased from 54 to 259 between 1980 and 1998, and by 2005 the population appeared to have reached carrying capacity (Watson *et al.* 2002, Stinson *et al.* 2007). However, Zador *et al.* (2009) cautioned that abundance of eagles is a poor measure of predation pressure, since a substantial number of predation attempts may originate from relatively few individuals.

Population trend estimates, anthropogenic threats, and environmental stressors

It is unclear why populations continue to decline in southern Washington. No large oil spills have occurred along the Washington coast since the previous analysis of Carter *et al.* (2001). Estimated mortality following the New Carissa oil spill off the Coast of Oregon in February of 1999 was limited (i.e., <200 murres). Injury assessment is ongoing for the Mystery Spill, which occurred in March of 1999 off the coasts of northern Oregon and southern Washington; estimated murre mortality is anticipated to be less than that of the New Carissa oil spill (Cindy Schexnider, pers. comm.). Although monitoring over the past 15 years has been limited spatially and temporally, seabird bycatch appears to be ongoing. Hamel *et al.* (2009) analyzed available data from bycatch-observer programs on limited fisheries, years, and/or areas in the Salish Sea as well as marine bird stranding data associated with fisheries during 1967–2007. They found that murres are the species most affected by net fisheries, with a conservative estimate of murre mortality, primarily adults, from bycatch of 0.2%–2.9% per year. Because murres from Oregon and British Columbia overwinter in the Salish Sea, the proportion of murres affected from colonies in Washington is unknown (Hamel *et al.* 2009). In the absence of observer programs, it is also unknown whether this proportion has been reduced following declines in fishing effort and bycatch mitigation measures on specific fisheries introduced in 1999. Additional efforts to decrease stress on murres from anthropogenic threats such as oil spills and disturbance will further enhance population resiliency. Olympic Coast National Marine Sanctuary (Sanctuary) and National Wildlife Refuge managers are working together to restrict access to refuge islands and low overflights. In addition, Sanctuary staff have developed an "Area to be Avoided," which routes oil tankers and large freighters offshore of refuge islands and the Sanctuary (ONMS 2008).

Interpretation of trends in the survey area is further complicated by variable ocean conditions. The early years (1996–2001) were dominated by the 1997/98 El Niño, with elevated sea surface temperature (SST) measured locally while Multivariate ENSO Index (MEI) values included both strong negative (2010/11 La Niña) and positive (2015 El Niño) conditions during the later years of the analysis (2010–2015; NOAA 2016). The longer-term, Pacific Decadal Oscillation (PDO) shifted to a predominantly cool phase in 1998, while the North Pacific Gyre Oscillation (NPGO) grew stronger, indicating more nutrient-rich, cooler waters from the north Pacific were likely present in our study area (Sydeman *et al.* 2014). Conditions characterized by increased SST and a delay in spring transition — the annual switch to a more productive cycle dominated by upwelling — have been shown to affect murres in multiple ways, in both the current and subsequent breeding season

(Parrish *et al*. 2003). Declining trends during periods characterized by warm SST are due in part to declining food resources, but species-specific responses vary, based on energetic demand, season, breeding habitat availability, and intra- or inter-specific competition (Irons *et al.* 2008, Ainley & Hyrenbach 2010, Cavole *et al.* 2016). In the northern CCS, murre diet varies regionally, and adults appear to modify chick diets in response to changes in prey availability during years with elevated SST (Schrimpf *et al.* 2012, Ainley *et al*. 2014). Yet, there is a point at which adults cannot compensate for low-quality resources; as a result, productivity may be reduced and chicks may suffer poor body condition (Schrimpf *et al*. 2012).

Climate scientists are currently studying how MEI, PDO, NPGO, and other aggregations of oceanic and atmospheric processes influence conditions in the Pacific Ocean, but their interrelationships are not well understood and often combine to produce unanticipated consequences (Leising *et al*. 2015, Newman *et al*. 2016). One unexpected outcome was a warm-water anomaly found offshore of the northern CCS from 2014 through 2015 (referred to as the "blob" by Bond *et al.* 2015). Once the spring upwelling cycle began to subside in the late summer of 2015, the blob was rapidly advected into the coastal zone, and SST measures spiked (Leising *et al.* 2015). In August 2015, the Coastal Observation and Seabird Survey Team reported significantly higher rates of carcass-encounters, the majority of which were of murres, than usual along Washington's beaches (COASST 2015). This mortality was an order of magnitude higher than any observed during the post-breeding period in Washington since 2001 and included emaciated adults and chicks. Given the uncertainties in forecasting changing ocean conditions, understanding how they affect seabird trends poses an even greater challenge. However, incorporating long-term trends of seabird species into climate analysis and hindcasting may help climate scientists, managers, and seabird biologists better understand the range of impacts of changing ocean conditions and the time horizon of this impact (Cavole *et al.* 2016, Newman *et al.* 2016).

Breeding biology and implications for monitoring

Daily, seasonal, and annual variability in colony attendance (Hatch & Hatch 1989, Halley *et al.* 1995) make it difficult to determine the best time for surveys each year and to develop effective monitoring strategies for murre populations. There were a number of years in our study that we conducted multiple surveys in June and July. We explored these data to identify the optimal timing of survey effort in relation to within-season colony dynamics. Unfortunately, sample sizes (number of years with multiple surveys) were small, the data were extremely variable from survey-to-survey within each season, and there were no clear patterns indicating optimal sampling time. Although available data from past repeated surveys each season were insufficient to inform future monitoring efforts, effective monitoring may be derived from more systematic and strategic repeated surveys each year and from a model-based estimation procedure. Movements to and from the colony by foraging breeders and prospecting non-breeders represent a "temporary emigration" from the colony; a portion of the population is not available to be counted on any given survey day. One way to account for bias created by temporary emigration is to conduct repeated counts each season and use a hierarchical model to estimate population size and probability of detection during surveys (Lyons *et al.* 2012). Royle's (2004) *N*-mixture model is a hierarchical model that accounts for random temporary emigration and probability of detection conditional on presence at the colony. The modeling and estimation framework requires spatial and temporal replication (i.e., a set of colonies that are visited multiple times during the breeding season). If implemented as part of a coordinated monitoring effort, the framework can be used to track population trends over large areas (Kery *et al.* 2009). For monitoring murre populations, repeated counts would be most effective if conducted during a relatively short period of time (e.g., 10 d) around the peak of diurnal attendance, which for Washington may be at the end of June and beginning of July. Given that multiple surveys are often conducted in Washington each year, relatively small, strategic changes in the way these surveys are conducted (e.g., shorter time window centered on peak diurnal attendance) may provide large benefits in modeling abundance and detection as a function of survey covariates and producing reliable population estimates for long-term monitoring in a way that accounts for known biases in count data.

In closing, we provide the most comprehensive analysis of murre trends in the state to date, comparable to Carter *et al.* (2001). Both assessments provide evidence of high spatial variability within the survey area. Despite an increasing trend of murres, abundance and distribution are still not comparable to that of the late 1970s. Colonies in southern Washington, which historically were the most abundant, are no longer active or minimally so, whereas colonies in the north, which were rarely active in the early 1970s, are now most abundant. This dynamic spatial distribution indicates the need for coordinated monitoring efforts throughout the CCS as well as additional movement studies to understand dispersal, colonization, and changes in spatial distribution. Long-term monitoring of trends and further study of the dynamics between changing ocean conditions, prey resources, and murre breeding success will help expand our understanding of how a nearshore coastal seabird can adapt to changing ocean conditions within the northern CCS (Cavole *et al*. 2016). Finally, these data highlight the importance of conserving historic and current colony locations to accommodate spatial variability. Protection of all known colonies may provide enough of a buffer to allow murres to maintain population resilience despite the limiting factors they face.

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