Received: 13 December 2018 | Accepted: 29 March 2020

DOI: 10.1111/mms.12696

ARTICLE

Marine Mammal Science

Maximizing surveillance through spatial characterization of marine mammal stranding hot spots

Jennifer K. Olson[1] | **John Aschoff**[1,2] | **Alice Goble**[2] | **Shawn Larson**[1] | **Joseph K. Gaydos**[2]

[1]The Whale Museum, Friday Harbor, Washington

[2]The SeaDoc Society, University of California Davis Karen C. Drayer Wildlife Health Center, Orcas Island Office, Eastsound, Washington

Correspondence
Current affiliation: Ocean Research College Academy, Everett Community College, 2000 Tower Street, Everett, WA 98201.
Email: jenolson@everettcc.edu

Abstract

Spatial analyses of marine mammal stranding data can be used to identify stranding patterns and improve surveillance and monitoring. Using ArcGIS and SaTScan, we analyzed 12 years (2002–2014) of dead beachcast marine mammals from San Juan County, Washington, to better understand patterns of carcass deposition. We plotted the locations for 631 dead marine mammals and aggregated strandings into 1,000 m segments of shoreline. "Hot spots" included beach segments with significantly higher carcass deposition according to the Getis-Ord Gi* statistic in ArcGIS or beach segments that were encompassed by significant spatial clusters using the discrete Poisson model in SatTScan. We identified 52 hot spots for harbor seals (*Phoca vitulina*) in ArcGIS and 62 hot spots in SaTScan with 81% agreement between methods. Carcass deposition showed a strong relationship with locations having high proximity to public pedestrian access points, suggesting increased reporting effort in those areas. Stranding frequency was also higher on beach segments with longer fetch and nearly level or gentle slopes. Beaches with these geomorphic characteristics, especially those without high proximity to public access, would be ideal locations to actively survey to improve high quality carcass collection during times of high expected mortality and limited resources.

KEYWORDS

baseline data, GIS, harbor seal, *Phoca vitulina*, Salish Sea, SaTScan, spatial analysis, stranding

1 | INTRODUCTION

Stranded marine mammals provide a unique opportunity to learn more about the biology of aquatic species and serve as a source of information about life history, population dynamics, and ecosystem health. Marine mammal stranding data have vastly improved our understanding of the causes of morbidity and mortality in marine mammals including infectious diseases, parasites, toxins, and environmental disturbances (Geraci & Lounsbury, 2005). Furthermore, stranding data are one of the primary sources of evidence for how human activities, such as vessel strikes, gunshot wounds, or fishery interactions, affect marine mammals (Leeney et al., 2008). These data can be used to inform policy and management decisions (Geraci & Lounsbury, 2005).

Spatial analysis of marine mammal strandings can be a useful tool for identifying existing patterns, as well as predicting carcass deposition during future mortality events and enhancing surveillance and monitoring programs (Norman, 2008). Many previous applications of geographic information system (GIS) analyses of stranding data have been used to study mortality patterns and disease outbreaks (Grieg, Gulland, & Kreuder, 2005; Miller et al., 2004; Morris et al., 2015). Others have shown correlations between stranding patterns and human influences such as fishing effort (Byrd et al., 2014; Crosti, Arcangeli, Romeo, & Andaloro, 2017). In the Northeast Pacific, spatial analyses of harbor porpoise (*Phocoena phocoena*) strandings detected clusters during an Unusual Mortality Event (UME) in 2006–2007 and have highlighted the importance of consistent stranding response effort and protocols (Huggins et al., 2015). Pinniped hot spots have also been characterized in the Northeast Pacific to highlight areas in the region with high numbers of human interaction cases (Warlick et al., 2018). While most existing studies have focused on temporary changes in stranding trends as seen in UMEs or disease outbreaks, few have used spatial patterns to establish baseline trends in an effort to improve surveillance and monitoring and to plan for future response efforts.

Discrepancies exist between the total number of dead marine mammals and those that are ultimately found on shore and reported. A 20% recovery rate of the well-studied Southern Resident killer whale (*Orcinus orca*) population represents a “best case scenario,” but recovery rates are likely much lower for less conspicuous species (Barbieri et al. 2013). Many factors influence the movement of marine mammals at sea and carcass deposition on shore, including but not limited to: ocean currents, winds, tides, ocean topography, shoreline geography, upwelling and downwelling, proximity to shore at time of death, and even scavenging at sea (Evans et al., 2005; Faerber & Baird, 2010; Long & Jones, 1996; Norman et al., 2004). Because of the opportunistic nature of procuring stranding data, patterns are also biased by factors that influence human behavior, such as weather conditions, shoreline characteristics, human population size, and accessibility to a given coastal area (Evans et al., 2005; Maldini, Mazzuca, & Atkinson, 2005; Norman et al., 2004). Though effort by a stranding network or human presence to encounter strandings can complicate the interpretation of stranding data, results of analyses may still be useful when human bias is taken into consideration and data are restricted to a subset of the time series during which monitoring effort is considered to be relatively constant (Peltier et al., 2019; Saavedra et al., 2017).

Within the National Oceanographic and Atmospheric Administration's (NOAA) designated West Coast Region, marine mammals in subregions can vary in abundance, reproductive patterns, and even genetic distinctness (Caretta et al., 2016; Huber, Dickerson, Jeffries, & Lambourn, 2012; Seekins, 2009). Because of these differences, along with the unique logistical challenges faced in different subregions, Washington State shoreline is broken into defined networks for stranding response. While the value of looking for large scale patterns across regions and states is indisputable from a population monitoring perspective, it is also important to examine patterns on a subregional scale relative to local response logistics. Analyzing patterns at this scale can reveal important local patterns that might be missed when looking at a larger region-wide scale (Norman et al., 2012).

San Juan County, Washington, is situated in the center of the highly productive Salish Sea Ecosystem and encompasses more than 400 miles of shoreline with varying accessibility (Figure 1). More than ten different marine mammal species are dependent on the waters of the Salish Sea and are frequently sighted in San Juan County (Gaydos & Pearson, 2011). San Juan County is centrally located in the designated critical habitat for the Endangered Species Act-listed Southern Resident killer whales (*Orcinus orca*) (U.S. Federal Register, 2005). Furthermore, San Juan

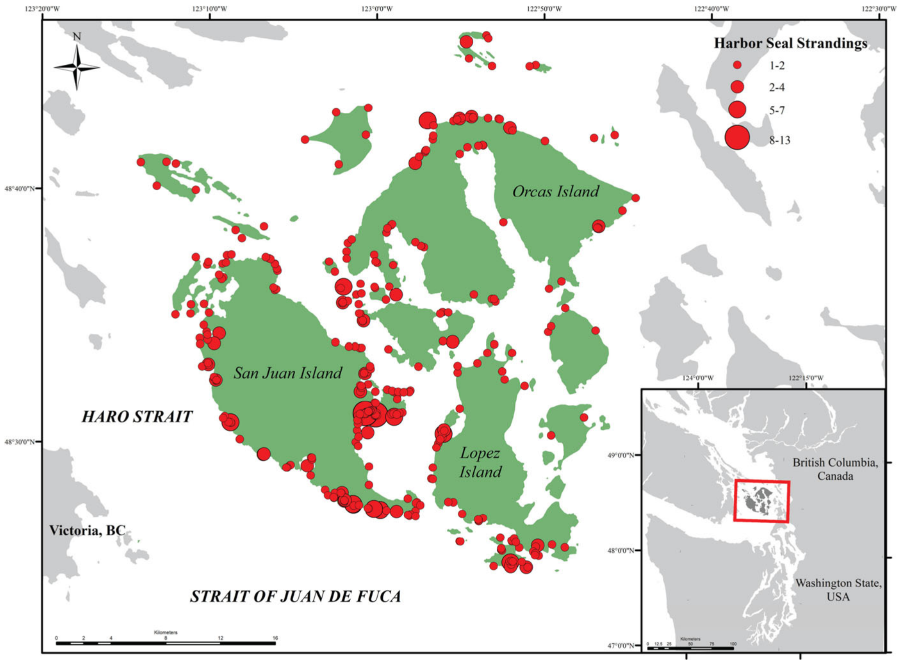

FIGURE 1 Harbor seal carcass deposition frequency in San Juan County, Washington (2002–2014).

County is home to one of the densest harbor seal (*Phoca vitulina*) populations in the world, with over 5,000 animals living in an area of 1,160 km^2 of marine habitat and reaching carrying capacity in the late 1990s (Jeffries, Huber, Calambokidis, & Laake, 2003). Over 145 known haul-out sites have been identified in the county, ranging in size from a few animals to over 200 at a given site (Jeffries, Gearin, Huber, Saul, & Pruett, 2000). The productive ecosystem and extensive shoreline contribute to San Juan County having the highest marine mammal stranding frequency of any other county in Washington State (NOAA, 2018).

We conducted a retrospective spatial analysis of marine mammal stranding data from 2002 to 2014 to identify areas of high carcass deposition in San Juan County. We also investigated the most important factors influencing stranding frequency in order to predict potential stranding hot spots that might be overlooked due to lack of public use or accessibility. Because of the robust harbor seal population in San Juan County and abundance of stranding records, analyses focused on this important indicator species for which data on contaminant, disease, and fishery interactions are valuable indicators of Salish Sea ecosystem health (Ross et al., 2013). This proof of concept study targets a unique island county with more shoreline than any other county in the United States and should serve as a model for similar studies in other regions. Knowing key stranding locations will improve routine collection of quality carcasses for disease surveys and allow Stranding Network responders to prioritize response efforts and improve data collection during times of limited response capability, such as oil spills or Unusual Mortality Events (UMEs).

2 | METHODS

2.1 | Stranding data

The San Juan County Marine Mammal Stranding Network (SJCMMSN) has been a program of The Whale Museum since 1982 and has maintained a close partnership with wildlife veterinarians from the SeaDoc Society since 2002. A

Stranding Agreement with the National Marine Fisheries Service (NMFS) provides legal authority for staff and volunteers to investigate and document sick, injured, distressed, and dead animals. Basic "Level A" data were collected for dead stranded animals from 1982 to the present, including date, species, location, carcass condition, age class, sex, length, weight, signs of human interaction, and carcass disposition.

Consistent funding, expanded partnerships, and systematic improvements beginning in 2002 led to increased capabilities and response effort. Stranding response effort in San Juan County has been relatively stable since this time; thus, only data collected after 2002 was used in this analysis. We also restricted our analysis to dead stranded animals that were reported on beaches, and all cases of animals floating offshore were removed from the data set. All strandings that did not have latitude or longitude recorded in the database were assigned coordinates based on the descriptive location.

2.2 | Features and mapping

All strandings were plotted using GIS (ArcGIS, Version 10.5). All data were plotted using the World Geodetic System 1984 and were then projected using Projection Coordinate System NAD 1983 Stateplane Washington South (meters). The shoreline for San Juan County, Washington, was divided into 1,000 m segments by plotting the centroids of each segment using the Create Points Along Lines Tool. All strandings were spatially joined to the designated beach segments and aggregated into counts per segment.

We used a Washington Coastal Digital Elevation Model (DEM) for the San Juan Islands obtained from the U.S. NOAA National Geophysical Data Center (NGDC) to evaluate beach characteristics for slope and aspect (the direction in which the beach was facing). The DEM was converted from a 10 m resolution to a 100 m resolution before analysis to account for variation in accuracy of location reporting. We used the aspect and slope calculation tools within Spatial Analyst Extension Pack to create aspect and slope rasters and used the extract tool to assign a slope and aspect value to each beach segment. Aspect values were divided into eight categories according to degree: North (337.5°–222.5°), North East (22.5°–67.5°), East (67.5°–112.5°), South East (112.5–157.5), South (157.5°–202.5°), South West (202.5°–247.5°), West (247.5°–292.5°), and North West (292.5°–337.5°). Slope was divided into four categories by gradient: Nearly Level (0%–3%), Gently sloping (4%–8%), Moderately sloping (9%–16%), and Steep (≥17%).

Fetch was defined as the distance over water from the centroid of each beach segment to the nearest point of land and was calculated for each segment in the same direction as the aspect. It was estimated by using a string of evenly spaced points extending from the centroid of each beach segment (generated using program R) and then calculating the length of strings through uninterrupted water. The distance between generated points was 100 m up to a maximum of 30,000 m. We ended the string when two contiguous points were on land. Final fetch distances were divided into categories spanning 1,500 m each.

We used data from the Washington State Department of Ecology (https://ecology.wa.gov/Research-Data) to assess the relationship between stranding locations and distance to public beach access points (Figure S1). We plotted the public beach access points and then eliminated all locations that could not be accessed by pedestrians. We used a spatial join in GIS to join the centroid of each beach segment to the nearest public pedestrian beach access point and thus calculated the distance between the two. We made the assumption that the closer a beach segment is to a pedestrian access point, the more human use it will experience.

Known local haul-out sites (Jeffries et al., 2000) were used to determine distance from a stranding location to a haul-out (Figure S1). Similar to methods used for public beach access, we used a spatial join from the centroid of each beach segment to the nearest haul-out and thus calculated the distance between them.

We also used shoreform data, obtained from the Friends of the San Juans (http://sanjuans.org/nearshorestudies-htm/), to determine if carcass deposition and maintenance were more likely to occur on a particular type of coastal geomorphic shoreline (Figure S2). Shoreform data were divided into eight types for San Juan County: artificial shoreline (breakwater and armoring), barrier beaches (sand ridge that rises slightly above the surface of the sea and runs parallel to shore), embayments (including bays, estuaries, and lagoons), feeder bluffs (an eroding coastal bluff that delivers sand and gravel to a beach over time), pocket beaches (a small beach, between two headlands), rocky shoreline (intertidal area where solid rock is dominant), and transport zones (areas where sand, sediment, and flotsam are transported; Shipman, 2008). Each stranding event was assigned to the nearest shoreform using a spatial join. We then used a chi-square test to assess differences between the predicted and actual strandings according to shoreform and used adjusted standardized residuals to assess departure of observed values from expected values.

2.3 | Hot spot analysis

Spatial patterns were determined using two methods. First, we ran a Hot Spot Analysis based on Getis-Ord Gi* statistics using the Arc GIS Spatial Statistics Toolbox. This tool identifies significant spatial clusters of high values (hot spots) and low values (cold spots). The results are shown in three levels of confidence bins; we considered all clusters that were significant within the 90% confidence level (±1 bin) as hot spots.

Second, we also looked for clusters in our data using the program SaTScan and compared the different approaches. We used a discrete Poisson model for spatial clusters. We limited our cluster radius to a maximum of 5 km and considered clusters significant if $p < .05$. We also included covariates of sex, age class (adult, subadult, weaned pup, dependent pup), and season. We considered all beach segments encompassed by a significant cluster as hot spots.

2.4 | Determining significant driving factors

The most important factors influencing spatial stranding clusters were determined for harbor seals using a zero-inflated negative binomial model (ZINB) in R. This model allows for imperfect detection by coping with overdispersion of zeros as well as overdispersion of count data. Zero-inflated negative binomial models are mixture models with a negative binomial used to model the count data and a binomial GLM used to model the probability of measuring false zeros (Authier et al., 2014; Zuur, Ieno, Walker, Saveliev, & Smith, 2009). The ZINB was chosen as a better fit over a zero-inflated Poisson model using a likelihood ratio test. Our response variable was the frequency of strandings at each beach segment. Our initial model contained our GIS assigned beach characteristics as independent variables for the count model (fetch, slope, aspect, distance to haul-out, distance to public pedestrian access) and factors we suspected might influence missed detections as independent variables in the logistic model (distance to public pedestrian access and slope). The model of best fit was selected by dropping the least significant variable sequentially and comparing resulting models using AIC values.

Differences between the actual and predicted strandings according to shoreform were compared using a separate chi-square test, as shoreform was not consistent across beach segments. Adjusted standardized residuals were reported as a measure of departure of actual values from expected values.

One hot spot with a known bias was not considered significant for analysis. Stranded harbor seals that are candidates for relocation are frequently taken to a haul-out site at Yellow Island (48°35′29.6″N, 123°02′05.2″W), a property of The Nature Conservancy with an on-site caretaker; thus, the harbor seal stranding rate is artificially inflated at this location.

To assess the potential for hot spots to be detected irrespective of human-related factors, we identified all segments that shared the characteristics of statistically significant factors regardless of pedestrian beach access and plotted these predictive hot spots using ArcGIS.

3 | RESULTS

3.1 | General patterns

From 2002 to 2014, 631 confirmed dead stranded marine mammals of all age classes were recorded in San Juan County, Washington. Of these, 84% (n = 530) were pinnipeds and 16% (n = 101) were cetaceans. Harbor seals were the most commonly stranded species representing 78% (n = 491) of all strandings (Figure 1). Of these harbor seal strandings, 41% were nursing pups (n = 201) and 10% were weaned pups (n = 51). We found no difference in stranding frequency by sex for harbor seals (χ^2 = 1.06, df = 1, p = .30). There was, however, a difference in stranding frequency by age class (χ^2 = 160.74, df = 3, p < .001). An increase of observed pups had the strongest deviation from expected values (Pearson's adjusted residual of 9.44.)

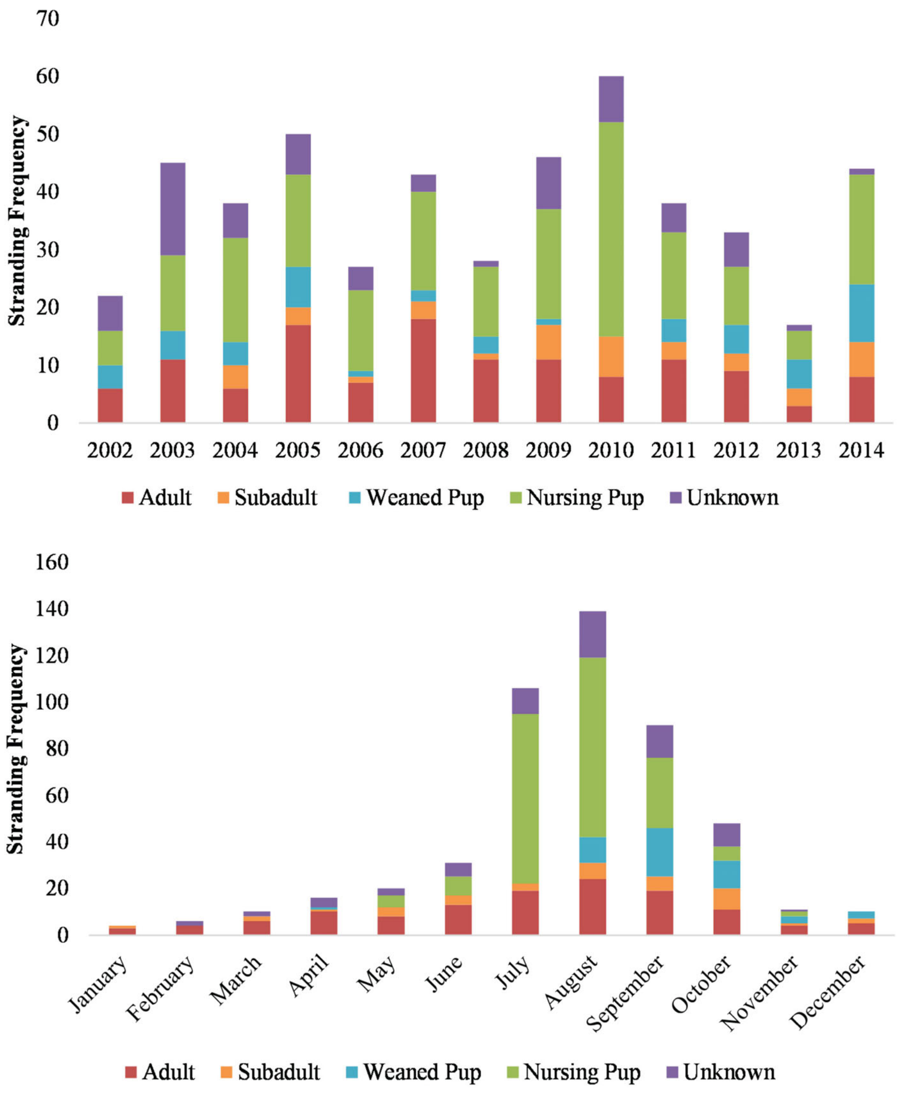

FIGURE 2 Counts of beach-cast, dead harbor seal strandings by age class and year (top) and month (bottom) in San Juan County, Washington from 2002 to 2014.

Stranding rates for dead harbor seals in San Juan County, Washington, were relatively stable from 2002 to 2014 with an average of 38 strandings per year. Harbor seals showed peak stranding frequency occurring during the summer months of July–September (Figure 2). Strandings of harbor seal pups sharply increased during these months compared to other age classes with nursing and weaned pups representing 73% (n = 212) of all harbor seal strandings of known age class from July to September (n = 290).

3.2 | Spatial patterns

Multiple areas of high carcass deposition (i.e., hot spots) were identified along the shoreline of San Juan County. Of the 496 beach segments, the ArcGIS Hot Spot Analysis identified 52 significant hot spots for harbor seal strandings (Figure 3). These hot spots were concentrated on the three islands with the highest human populations: San Juan Island, Orcas Island, and Lopez Island. SaTScan analysis also identified 11 significant primary clusters and 3 secondary clusters, which encompassed 62 different beach segments (henceforth also referred to as hot spots). There was an 81% agreement between the two methods with a few unique segments encompassed by SaTScan clusters on the northeast side of San Juan Island and the west side of Lopez Island (Figure 3). While taking into account the covariates of sex, age class, and season in SaTScan, these significant clusters did not change.

The most important factors contributing to stranding frequency for harbor seals were distance to public access, fetch distance, and slope (Table 1, Figure 4). Beach segments that were closer to public beach access points were significantly more likely to have higher stranding counts. High carcass deposition was also more likely to occur at beaches with a large fetch (30,000 m) compared to categories of shorter fetch (Table 2). Despite being included in the model of best fit, slope had weaker effect on stranding frequency; nevertheless, stranding frequency increased with nearly level and gently sloping beaches, while decreasing with moderately or strongly sloping beaches. Beach aspect and proximity to haul-out sites were not included in the model of best fit (Table 1).

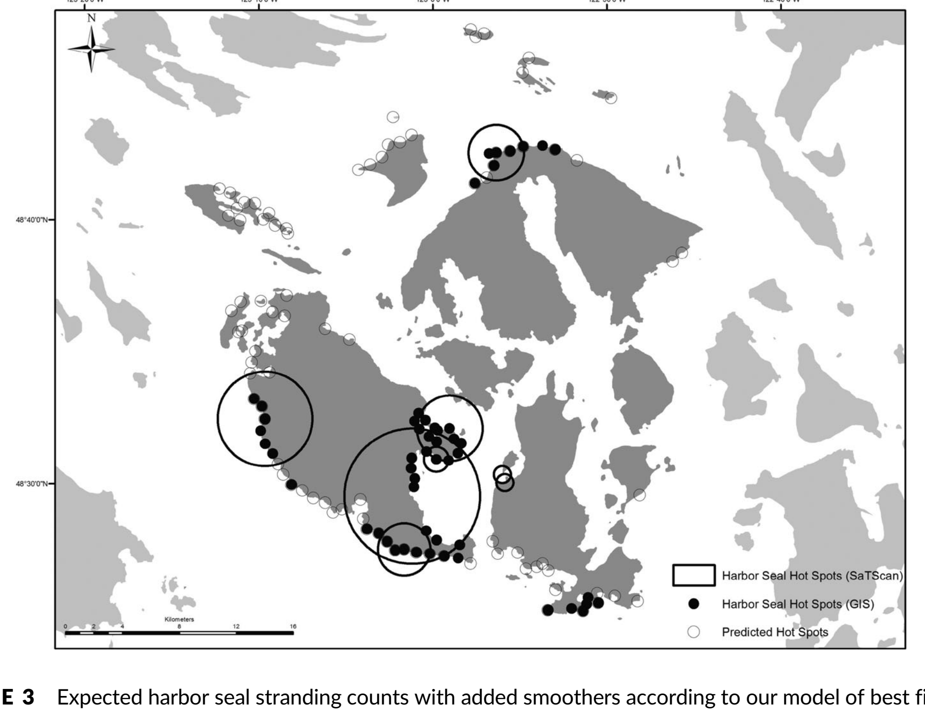

FIGURE 3 Expected harbor seal stranding counts with added smoothers according to our model of best fit.

TABLE 1 Results of the zero-inflated negative binomial model (ZINB) model selection process. Variables for the count part of the model are expressed on the left half of the model and variables for the binomial part of the model are expressed on the right half and are separated by the symbol "|".

Model	*df*	AIC
Frequency ~ Access + Haul-out + Aspect + Slope + Fetch | Access + Slope	34	1,222
Frequency ~ Access + Haul-out + Aspect + Slope + Fetch | Access	31	1,220
Frequency ~ Access + Aspect + Slope + Fetch | Access	30	1,218
Frequency ~ Access + Slope + Fetch | Access	23	1,214
Frequency ~ Access + Slope + Fetch | 1[a]	22	1,212
Frequency ~ Access + Fetch | 1	19	1,216

[a]Final selected model.

TABLE 2 Variable, coefficients, standard errors (*SE*), and *p* values for the zero-inflated negative binomial model (ZINB) model of best fit.

Variable	Level	Coefficient	*SE*	*p*
Distance to Pedestrian Beach Access		−0.6512	0.1099	<.001
Slope	Nearly Level	Baseline		
	Gently Sloping	0.2051	0.2096	.33
	Moderately Sloping	−0.5408	0.3014	.07
	Strongly Sloping	−1.0668	0.6804	.12
Fetch	0–1,500 m	Baseline		
	1,600–3,000 m	−0.10	0.28	.72
	3,100–4,500 m	0.41	0.34	.23
	4,600–6,000 m	0.69	0.32	.03
	6,100–7,500 m	0.66	0.51	.19
	7,600–9,000 m	0.82	0.59	.16
	9,100–10,500 m	0.49	0.99	.62
	10,600–12,000 m	−1.89	1.15	.10
	12,100–13,500 m	−1.07	0.93	.25
	13,600–15,000 m	0.78	0.80	.33
	15,100–16,500 m	−1.08	1.20	.37
	16,600–18,000 m	1.74	0.67	.01
	18,100–19,500 m	−12.15	1,570.85	.99
	19,600–21,000 m	0.36	1.48	.81
	21,100–22,500 m	—	—	—
	22,600–24,000 m	−13.32	963.35	.99
	24,100–25,500 m	—	—	—
	25,600–27,000 m	—	—	—
	27,100–28,500 m	—	—	—
	28,600–30,000 m	1.02	0.26	<.001

TABLE 3 Harbor seal carcass deposition according to geomorphic shoreform.

Shoreform type	Actual strandings	Expected strandings	Total length (km)	Total length (%)	Adjusted residuals
Artificial	10	3.078	4.111	0.627	3.945
Barrier Beach	86	30.212	40.345	6.153	10.150
Embayments (estuary and lagoon)	8	20.756	27.718	4.227	−2.800
Feeder Bluff	28	31.881	42.575	6.493	−0.687
Pocket Beach	142	58.283	77.831	11.870	10.966
Rocky Shoreline	183	311.897	416.511	63.523	−7.299
Transport Zone	34	34.893	46.597	7.107	−0.151
Total	491	491	655.687	100	

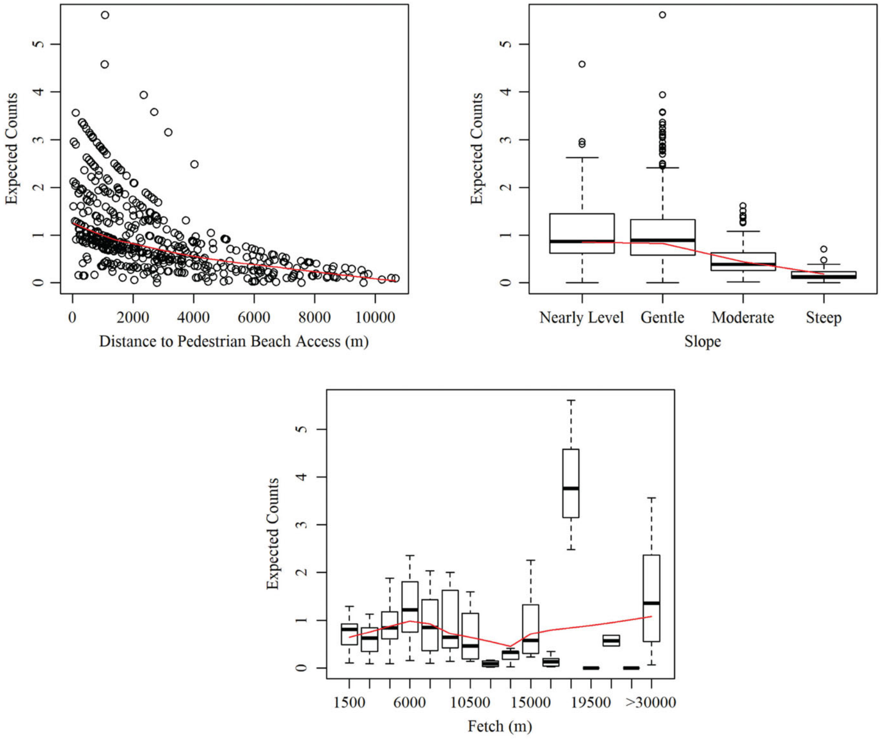

FIGURE 4 Established and predicted hot spots for harbor seal carcass deposition in San Juan County, Washington.

Carcass deposition was also significantly different than predicted according to shoreform ($\chi^2 = 300.44$, $p < .001$). Adjusted standardized residuals were largest for pocket beaches with an increased number of carcasses than expected for this type of shoreline. There were also relatively strong residual scores indicating increased deposition on barrier beaches and decreased deposition along rocky shorelines (Table 3).

3.3 | Predicted hot spots

After eliminating the influence of public accessibility, an additional 65 beach segments with large fetch (30,000 m) and gentle or nearly level slopes were identified as potential hot spots for San Juan County (Figure 3). Many of these newly predicted hot spots were concentrated on the outer shorelines of San Juan County, including north facing beaches on Stuart and Waldron Islands and southeast facing beaches on Orcas and Decatur Islands.

4 | DISCUSSION

Though many spatial analyses of marine mammal stranding patterns are approached with the aim of estimating abundance or distribution at sea as opposed to improving surveillance on shore, recent studies have begun to investigate potential driving factors of stranding patterns. Saavedra et al. (2017) investigated the impact of oceanographic, meteorological, prey, and fishing-related variables on small cetacean strandings in Spain. Others have focused on the influence of cause of mortality, such as fishery activity or by-catch (Crosti et al., 2017; Lopez, Pierce, Santos, Gracia, & Guerra, 2003). In this study, we utilized records of a nonmigratory species with high site fidelity and stable population counts to reduce the need to account for factors such as prey availability and population growth. The stranding data reported here also cover a time window with consistent effort from the stranding network and take into account reporting bias by incorporating human access to the shoreline.

Given the best available data and adjusting for factors that could influence the location of stranding clusters, several beach segments with high carcass deposition were identified. Stranding patterns were not uniform in San Juan County and were more likely to be observed on beaches near public access points with large fetch and nearly level or gently sloping beaches. Taking into account the covariates of sex, age class, and season did not change our results, which suggests that these covariates were not associated with the location of the significant SaTScan clusters. The stranding hot spots identified by our two methods (ArcGIS and SaTScan) had a strong degree of overlap, the importance of which has been stressed by others (Norman et al., 2012) and increases confidence in our results.

Despite the inherent value of data collected from dead marine mammals, interpretation must be approached cautiously due to biases in reporting. Many of the identified high deposition areas were situated near public pedestrian access points, such as National Parks or public marinas, and the distance to these access points was the most significant factor in predicting stranding hot spots. This was not surprising due to the fact that SJCMMSN relies entirely on the public for stranding reports, and these hot spots strongly corresponded with anecdotal observations and predictions. Beaches nearer to public access points are likely to allow increased human presence and thus increase the odds of observations and reporting. It is well known that marine mammal strandings are influenced by reporting bias and these findings were consistent with our predictions (Authier et al., 2014; Evans et al., 2005; Maldini et al., 2005; Norman et al., 2004). Proximity to pedestrian beach access did not, however, appear to influence the probability of missed detections as reflected by the binomial part of our ZINB.

Fetch was also a significant factor in predicting stranding hot spots. Beach segments with longer fetch (30,000 m) were more likely to have a higher frequency of marine mammal strandings. Fetch is defined as the distance over water that the wind blows in a single direction, and controls wave height along with wind speed and wind duration. Large waves, particularly those that occur during high tides, are more likely to result in a landward transport of sediment or debris (Johannessen & MacLennan, 2007). Thus, it is possible that beach segments with longer fetch were more likely to have carcass deposition due to the greater distance available for the generation of waves and thus greater energy available for transport of heavy carcasses to shore. Our results are somewhat consistent with other studies that have shown that beaches with moderate exposure to waves were more likely to accumulate wrack (i.e., organic material cast onto beaches by winds and tides) than beaches with low exposure (Orr, Simmer, Jelinski, & Mews, 2005). However, this effect may vary according to beach sediment as well as the buoyancy or type of material collecting on shore (Barreiro, Gómez, Lastra, López, & de la Huz, 2011; Orr et al., 2005).

Slope was also included as a predictor in the best fit final model. Despite having a weaker effect than other variables, higher stranding frequency occurred on beaches in the nearly level or gently sloping categories compared to moderately or strongly sloping beaches. Slope affects the degree of wave energy dissipation that occurs along a beach, and beaches with gentle slope create favorable conditions for wrack accumulation (Jackson & Nordstrom, 1992; Ruiz-Delgado et al., 2014). Our findings are consistent with studies of whale strandings in New Zealand that showed slopes were significantly gentler at known herd-stranding sites compared to random control sites (Brabyn & McLean, 1992). Other studies have pointed to steeper slopes as an important factor for stranding occurrence; however, this may be influenced by prevalence of geographical features in an area and/or habitat preference of a particular species. (Faerber & Baird, 2010; Moura et al., 2016).

Our results did not show a significant effect of aspect. It is possible that patterns according to aspect may be seasonally influenced. In the inland waters of Washington State, winds from the south tend to be the strongest and most prevailing during winter months while northerly winds are more common during the summer (Hickey, 1979; Johannessen & MacLennan, 2007). Nevertheless, season did not influence the designation of beach segments as hot spots in our spatial analysis, and our results are consistent with other studies that have shown the importance of wave exposure to wrack accumulation regardless of shoreline aspect (Orr et al., 2005).

Distance to haul-out site, a proxy for distribution at sea, was also not considered to be a significant predictor of high frequency deposition areas. This was a surprising result given the high site fidelity of adult harbor seals (Hardee, 2008; Suryan & Harvey, 1998); however, the ubiquitous presence of near shore haul-out sites in the study area may have weakened the explanatory power of this variable (Figure S1). It is important to note that our study focused on passive (dead) strandings as opposed to live stranded animals, which may be more influenced by distance to haul-out site than animals that have died at sea.

When human-related factors were eliminated, an additional 65 predicted hot spots were identified. These were primarily located on the most exposed shorelines of San Juan County, particular north or south facing shores, and included several nonferry serviced islands such as Waldron Island, Stuart Island, and Decatur Island. Strandings in these areas may have been previously overlooked due to limited reporting effort.

We also considered the possibility that shoreform may affect carcass deposition and maintenance. Pocket beaches (i.e., natural sand or gravel beaches along a bedrock shoreline) in particular had significantly more strandings than expected given the amount of coastline made up of this shoreform type. Because of their protected structure and consistent exposure to wind in a single direction, it is possible that carcasses may be more likely to remain in pocket beaches after having been deposited there. Furthermore, the protected structure of pocket beaches can also lead to isolation from active tidal circulation cells in the intertidal, resulting in water flow that is exclusively influenced by waves (Dehouck, Dupuis, & Sénéchal, 2009). Pocket beaches may not have as much traffic by people relative to larger beaches as they are small and adjacent to nonsandy areas; however, given that they may have more strandings than expected based on our analysis, it may also be beneficial to incorporate pocket beaches into surveillance or monitoring efforts in addition to beaches with longer fetch and nearly level slopes.

There are likely additional factors that contribute to stranding frequency beyond what we accounted for in this study. Some factors (e.g., proximity to shore upon death, carcass buoyancy, scavenging at sea) may vary by individual stranding and were not feasible for us to include. Furthermore, there are likely additional site- and region-specific factors that may result in different findings in different systems. Beaked whales, for example, may be more likely to strand in areas with steeper slopes due to habitat preferences (Faerber & Baird, 2010). Similarly, species with a greater degree of prey preference may warrant the inclusion of prey abundance and distribution in a spatial model (Saavedra et al., 2017). Considering the logistical goals of our study, however, we believe that the use of a long-term data set with relatively consistent effort and a stable study population establishes reasonable baseline spatial patterns for harbor seal strandings in San Juan County over the past decade. Future research efforts should attempt to quantify shoreline use to better tease out the effect of reporting bias. Incorporating the complexities of drift or causes of mortality may also improve the predictive capacity of future models (Peltier et al., 2012).

Understanding spatial patterns of marine mammal strandings on a subregional scale will help responders increase collections of freshly deceased specimens and ultimately permit scientists to learn more about marine mammal life history and diseases. Understanding these patterns could also prove to be valuable for emergency planning and focusing response resources during large scale oil spills, which are a known risk for the region, or for Unusual Mortality Events (UMEs) that may occur with outbreaks such as phocine distemper virus. These patterns may also come in handy during large scale rescue or recovery operations. When critically endangered Southern Resident killer whales are known to be in poor health or missing from their pod, for example, studies like this could effectively inform shoreline or aerial surveys towards common stranding locations and potentially overlooked areas. For San Juan County, targeting the identified hot spots as well as beaches with longer fetch distances and nearly level slopes could maximize carcass recovery with limited effort. Other stranding networks in the region could use this methodology to identify local marine mammal hot spots and improve stranding response, especially during times of high expected mortality and low response resources. Such improved knowledge for emergency preparation could prove to be critically important in areas with ESA-listed species.

ACKNOWLEDGMENTS

We would like to thank all San Juan County Marine Mammal Stranding Network volunteers and interns. S. Norman provided guidance on spatial analysis using SaTScan and valuable improvements to the manuscript. Friends of the San Juan Islands provided data for San Juan County beaches and guidance for assignment of shoreline features. All stranding response was conducted under authorization by the National Marine Fisheries Service Marine Mammal Health and Stranding Response Program. This work would not have been possible without funding from the John H. Prescott Marine Mammal Rescue Assistance Grant Program and in-kind contributions from The Whale Museum and the SeaDoc Society, a program of the Karen C. Drayer Wildlife Health Center at UC Davis.

AUTHOR CONTRIBUTIONS

John Aschoff: Formal analysis; methodology; visualization; writing-review and editing. **Alice Goble:** Data curation; writing-original draft. **Shawn Larson:** Writing-review and editing. **Joseph Gaydos:** Conceptualization; funding acquisition; investigation; supervision; writing-review and editing.

ORCID

Jennifer K. Olson https://orcid.org/0000-0002-7584-7578

REFERENCES

Authier, M., Peltier, H., Dorémus, G., Dabin, W., Van Canneyt, O., & Ridoux, V. (2014). How much are stranding records affected by variation in reporting rates? A case study of small delphinids in the Bay of Biscay. *Biodiversity and Conservation*, *23*, 2591–2612.

Barbieri, M.,. M., Raverty, S., Hanson, M. B., Venn-Watson, S., Ford, J. K. B., & Gaydos, J. K. (2013). Spatial and temporal analysis of killer whale (*Orcinus orca*) strandings in the North Pacific Ocean and the benefits of a coordinated stranding response protocol. *Marine Mammal Science*, *29*, E448–E462.

Barreiro, F., Gómez, M., Lastra, M., López, J., & de la Huz, R. (2011). Annual cycle of wrack supply to sandy beaches: Effect of the physical environment. *Marine Ecology Progress Series*, *433*, 65–74.

Brabyn, M. W., & McLean, I. G. (1992). Oceanography and coastal topography of herd-stranding sites for whales in New Zealand. *Journal of Mammalogy*, *73*, 469–476.

Byrd, B. L., Hohn, A. A., Lovewell, G. N., Altman, K. M., Barco, S. G., Friedlander, … Thayer, V. G. (2014). Strandings as indicators of marine mammal biodiversity and human interactions off the coast of North Carolina. *Fisheries Bulletin*, *112*, 1–23.

Caretta, J., Forney, K. A., Oleson, E. M., Weller, D. W., Lang, A. R., Baker, J., … Brownell, R. L. (2016). *U.S. Pacific marine mammal stock assessments: 2015*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Crosti, R., Arcangeli, A., Romeo, T., & Andaloro, F. (2017). Assessing the relationship between cetacean strandings (*Tursiops truncatus* and *Stenella coeruleoalba*) and fishery pressure indicators in Sicily (Mediterranean Sea) within the framework of the EU Habitats Directive. *European Journal of Wildlife Research*, *63*, 55.

Dehouck, A., Dupuis, H., & Sénéchal, N. (2009). Pocket beach hydrodynamics: The example of four macrotidal beaches, Brittany, France. *Marine Geology*, *266*, 1–17.

Evans, K., Thresher, R., Warneke, R. M., Bradshaw, C. J. A., Pook, M., Thiele, D., & Hindell, M. A. (2005). Periodic variability in cetacean strandings: Links to large-scale climate events. *Biology Letters*, *1*, 147–150.

Faerber, M. M., & Baird, R. W. (2010). Does a lack of observed beaked whale strandings in military exercise areas mean no impacts have occurred? A comparison of stranding and detection probabilities in the Canary and Hawaiian Islands. *Marine Mammal Science*, *26*, 602–613.

Gaydos, J. K., & Pearson, S. R. (2011). Birds and mammals that depend on the Salish Sea: A compilation. *Northwestern Naturalist*, *92*, 79–94.

Geraci, J. R., & Lounsbury, V. J. (2005). Perspectives on strandings and response programs. In *Marine mammals ashore: A field guide for strandings* (2nd ed., pp. 1–8). Baltimore, MD: National Aquarium in Baltimore.

Grieg, D. J., Gulland, F. M. D., & Kreuder, C. (2005). A decade of live California sea lion (*Zalophus californianus*) standings along the central California coast: Causes and trends, 1991–2000. *Aquatic Mammals*, *31*, 11–22.

Hardee, S. (2008). *Movements and home ranges of harbor seals* (Phoca vitulina) *in the inland waters of the Pacific Northwest* (M.S. thesis). Western Washington University, Bellingham, WA.

Hickey, B. M. (1979). The California Current system—hypotheses and facts. *Progress in Oceanography*, *8*, 191–279.

Huber, H. R., Dickerson, B. R., Jeffries, S. J., & Lambourn, D. M. (2012). Genetic analysis of Washington State harbor seals (*Phoca vitulina richardii*) using microsatellites. *Canadian Journal of Zoology*, *90*, 1361–1369.

Huggins, J. L., Raverty, S. A., Norman, S. A., Calambokidis, J., Gaydos, J. K., Duffield, D. A., … Barre, L. (2015). Increased harbor porpoise mortality in the Pacific Northwest, USA: Understanding when higher levels may be normal. *Diseases of Aquatic Organisms*, *115*, 93–102.

Jackson, N. L., & Nordstrom, K. F. (1992). Site specific controls on wind and wave processes and beach mobility on estuarine beaches in New Jersey, U.S.A. *Journal of Coastal Research*, *8*, 88–98.

Jeffries, S. J., Gearin, P. J., Huber, H. R., Saul, D. L., & Pruett, D. A. (2000). *Atlas of seal and sea lion haulout sites in Washington*. Olympia, WA: Washington Department of Fish and Wildlife, Wildlife Science Division.

Jeffries, S., Huber, H., Calambokidis, J., & Laake, J. (2003). Trends and status of harbor seals in Washington State: 1978–1999. *Journal of Wildlife Management*, *67*, 208–219.

Johannessen, J., & MacLennan, A. (2007). *Beaches and bluffs of Puget Sound* (Puget Sound Nearshore Partnership Report No. 2007-04). Seattle, WA: Seattle District, U.S. Army Corps of Engineers.

Leeney, R., Amies, R., Broderick, A. C., Witt, M. J., Loveridge, J., Doyle, J., & Godley, B. J. (2008). Spatio-temporal analysis of cetacean strandings and bycatch in a UK fisheries hotspot. *Biodiversity and Conservation*, *17*, 2323–2338.

Long, D. J., & Jones, R. E. (1996). White shark predation and scavenging on cetaceans in the eastern North Pacific Ocean. In A. P. Klimley & D. G. Ainley (Eds.), *Great white sharks: The biology of* Carcharodon carcharias (pp. 293–307). San Diego, CA: Academic Press.

Lopez, A., Pierce, G. J., Santos, M. B., Gracia, J., & Guerra, A. (2003). Fishery by-catches of marine mammals in Galacian waters: Results from on-board observations and an interview survey of fishermen. *Biological Conservation*, *111*, 25–40.

Maldini, D., Mazzuca, L., & Atkinson, S. (2005). Odontocete stranding patterns in the main Hawaiian Islands (1937–2002): how do they compare with live animal surveys? *Pacific Science*, *59*, 55–67.

Miller, M. A., Grigg, M. E., Kreuder, C., James, E. R., Melli, A. C., Crosbie, P. R., … Conrad, P. A. (2004). An unusual genotype of *Toxoplasma gondii* is common in California sea otters (*Enhydra lutris nereis*) and is a cause of mortality. *International Journal for Parasitology*, *34*, 275–284.

Morris, S. E., Zelner, J. L., Fauquier, D. A., Rowles, T. K., Rosel, P. E., Gulland, F., & Grenfell, B. T. (2015). Partially observed epidemics in wildlife hosts: Modelling an outbreak of dolphin morbillivirus in the northwestern Atlantic, June 2013–2014. *Journal of the Royal Society Interface*, *12*, 20150676.

Moura, J. F., Acevedo-Trejos, E., Tavares, D. C., Meirelles, A. C. O., Silva, C. P. N., Oliveira, L. R., … Merico, A. (2016). Stranding events of *Kogia* whales along the Brazilian Coast. *PLoS ONE*, *11*(1), e0146108.

National Oceanic & Atmospheric Administration (NOAA). (2018). Marine mammal health and stranding response database. Retrieved from https://mmhsrp.nmfs.noaa.gov/mmhsrp/jsp/mms_mainpage.jsp (Level A records from 2002 to 2017 accessed 9 March 2018)

Norman, S. A. (2008). Spatial epidemiology and GIS in marine mammal conservation medicine and disease research. *EcoHealth*, *5*, 257–267.

Norman, S. A., Bowlby, C. E., Brancato, M. A., Calambokidis, J., Duffield, D., Gearin, P. J., … Scordino, J. (2004). Cetacean strandings in Oregon and Washington between 1930 and 2002. *Journal of Cetacean Research and Management*, *6*, 87–99.

Norman, S. A., Huggins, J., Carpenter, T. E., Case, J. T., Lambourn, D. M., Rich, J. M., … Klope, M. (2012). The application of GIS and spatiotemporal analyses to investigations of unusual marine mammal strandings and mortality events. *Marine Mammal Science*, *28*, E251–E266.

Orr, M., Simmer, M., Jelinski, D. E., & Mews, M. (2005). Wrack deposition on different beach types: spatial and temporal variation in the pattern of subsidy. *Ecology*, *86*, 1496–1507.

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., & Ridoux, V. (2012). The significance of stranding data as indicators of cetacean populations at sea: Modelling the drift of cetacean carcasses. *Ecological Indicators*, *18*, 278–290.

Peltier, H., Beaufils, A., Cesarini, C., Dabin, W., Dars, C., Demaret, F., … Spitz, J. (2019). Monitoring of marine mammal strandings along French coasts reveals the importance of ship strikes on large cetaceans: A challenge for the European Marine Strategy Framework Directive. *Frontiers in Marine Science*, *6*, 486.

Ross, P. S., Noël, M., Lambourn, D., Dangerfield, N., Calambokidis, J., & Jeffries, S. (2013). Declining concentrations of persistent PCBs, PBDEs, PCDEs, and PCNs in harbor seals (*Phoca vitulina*) from the Salish Sea. *Progress in Oceanography*, *115*, 160–170.

Ruiz-Delgado, M. C., Vieira, J. V., Veloso, V. G., Reyes-Martínez, M. J., Sallorenzo, I. A., Borzone, C. A., … García, F. J. G. (2014). The role of wrack deposits for supralittoral arthropods: An example using Atlantic sandy beaches of Brazil and Spain. *Estuarine, Coastal and Shelf Science*, *136*, 61–71.

Saavedra, C., Pierce, G. J., Gago, J., Jusufovski, D., Cabrero, Á., Cerviño, S., … Santos, M. B. (2017). Factors driving patterns and trends in strandings of small cetaceans. *Marine Biology*, *164*, 165.

Seekins, B. (2009). Harbor seal pupping timeframes in Washington State. NOAA Fisheries Northwest Regional Office. Retrieved from https://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/marine_mammals/seal-pups-timing.pdf (accessed 4 December 2018)

Shipman, H. (2008). *A geomorphic classification of Puget Sound nearshore landforms* (Puget Sound Nearshore Partnership Report No. 2008-01). Seattle, WA: Seattle District, U.S. Army Corps of Engineers.

Suryan, R. M., & Harvey, J. T. (1998). Tracking harbor seals (*Phoca vitulina richardsi*) to determine dive behavior, foraging activity, and haul-out site use. *Marine Mammal Science*, *14*, 361–372.

U.S. Federal Register. 2005. Endangered and threatened wildlife and plants: Endangered status for Southern Resident killer whales, Final Rule. FR 70(222):69903–69912 (18 November 2005). National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Department of Commerce, Washington, DC.

Warlick, A. J., Duffield, D. A., Lambourn, D. M., Jeffries, S. J., Rice, J. M., Gaydos, J. K., … Norman, S. (2018). Spatio-temporal characterization of pinniped strandings and human interaction cases in the Pacific Northwest, 1989–2016. *Aquatic Mammals*, *44*, 299–318.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009). *Mixed effect models and extensions in ecology with R*. New York, NY: Springer.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

How to cite this article: Olson JK, Aschoff J, Goble A, Larson S, Gaydos JK. Maximizing surveillance through spatial characterization of marine mammal stranding hot spots. *Mar Mam Sci*. 2020;1–14. https://doi.org/10.1111/mms.12696