Predictive Marine Mammal Modeling for Queen Charlotte Basin, British Columbia

Technical Report

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RAINCOAST
CONSERVATION FOUNDATION

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Introduction to the Project

Characterizing the distribution and abundance of protected species is essential to more objectively assess the risk factors related to potentially adverse interactions with human activities. Although information on animal distribution and abundance is integral to wildlife conservation and management, surprisingly few data on marine mammal distribution and abundance have been collected in the waters of Canada's Pacific coast (Ban et al. 2008). Accordingly, spatial distribution and abundance of marine mammals in this region is poorly described. In addition, there has been little work examining and comparing the habitat preferences of different species. Although killer whales in this region are particularly well studied, abundance estimates and distribution information is not available for most cetacean and pinniped species inhabiting the region, including those species that were heavily depleted by commercial exploitation.

Our goal is to build regional multi-species models that help to understand the environmental factors that influence marine mammal abundance and distribution. The habitat preferences of a species represent an important part of a species niche (Guisan and Zimmermann 2000). Although a species distribution may change in the short-term as local conditions change, its niche is likely to remain unchanged (Martínez-Meyer, Townsend Peterson, and Hargrove 2004). Therefore, an understanding of a species niche can be used to predict how a species will react to changes in its local environment over time. This is particularly important considering the likelihood of increased anthropogenic impacts in our study area, which heightens the urgency to collect baseline data on marine mammal distribution and abundance. In recent years, for example, there has been considerable discussion about lifting existing moratoria on offshore oil and gas exploration and extraction off the north and central coasts of BC (Royal Society of Canada 2004). Such changes have implications for how species interact with human activities and for determining the best approaches to the conservation and management of marine mammal populations and species.

The nearshore waters of British Columbia are home to a diverse suite of marine mammals, some of whose endangered status is threatened by human activities. Whereas the Species at Risk Act (SARA) has afforded these animals protections and some conservation efforts are in place, existing and expanding activities from transportation, oil, wind, and fisheries industries may adversely impact these species (Government of British Columbia 2006; Ban & Alder 2008). Quantifying the temporal trends and spatial distribution of their abundance is essential for conservation management.

The Raincoast Conservation Foundation has conducted marine mammal surveys in the nearshore BC waters between 2004 and 2008, predominately during summer, but also during spring and fall. Using the first two years of data, Williams and Thomas (2007) first characterized the abundance of marine mammals across 4 strata using design-based estimates, which assumes homogenous density within strata. Herein, we update these stratum-based estimates with subsequent years of survey effort and look at abundance across years and seasons. We then employ density surface modeling with environmental covariates to produce maps that describe the spatially heterogeneous distribution of animal density across the study area. These density surfaces are then composited into a single marine mammal hotspot map, which can be used to reduce risk for site-specific activities. This composite map is finally used as a cost surface for suggesting a framework to route vessel traffic around sensitive areas.

Survey Design

Marine mammal surveys were conducted across the inner waters of British Columbia during the summers of 2004, 2005, 2006, and 2008 and during spring and fall for 2007. The surveys were designed to maximize coverage and minimize off-effort time over 4 strata (Figure 1) for the purposes of design-based multi-species density estimation according to Thomas et al. (2007). Zigzag configurations were applied over the open strata (1 and 2), with sub-stratification for the more topographically complex strata 2. For the narrower strata (3 and 4), parallel lines oriented perpendicular to the long axis minimized edge effects. The inlet strata (4) were further subdivided into primary sampling units (PSUs) so that for a given season, a random subsample of PSUs was selected for surveying. Total effort by length and number of transects per year and season of survey is given in Table 1. To estimate density, effort-weighted means were used for all strata, except stratum 4 which was derived from the unweighted mean of the PSUs.

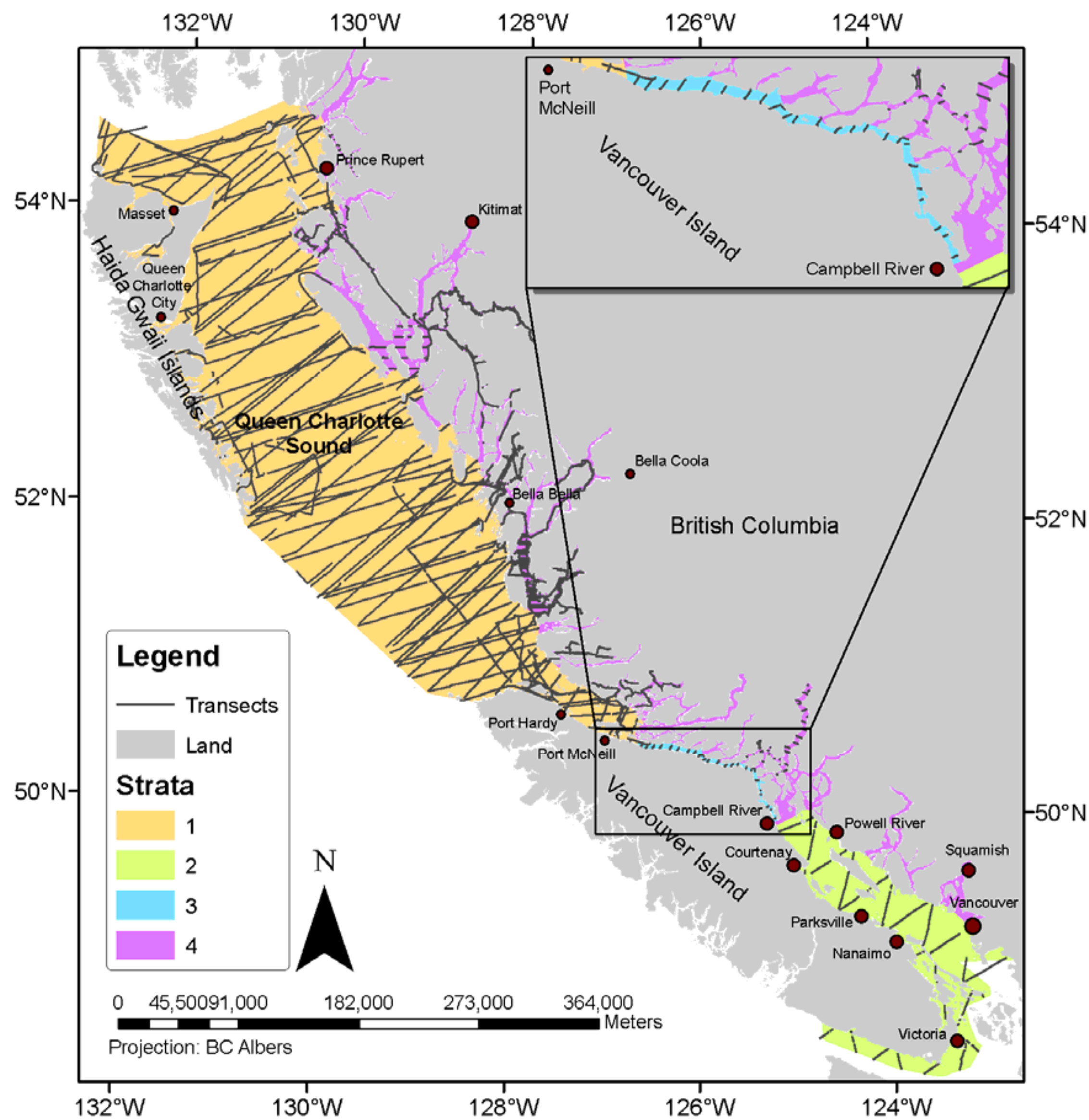

Figure 1. Stratum ID and on-effort transects, including transit legs between design-based transects, for all years, corresponding to Queen Charlotte Basin (1), Straits of Georgia and Juan de Fuca (2), Johnstone Strait (3), and mainland inlets (4).

Table 1. Stratum summary of realized survey effort with on-survey on-effort transects. Note the sample unit for Stratum 4 is based on PSU, not number of transects (not displayed).

Stratum	PSU	Year	Season	Length (nm)	Length (km)	# of Transects	Area (nm2)	Area (km2)
1		2004	sum	903	1,672	17	18,361	62,976
		2005	sum	914	1,693	18		
		2006	sum	327	605	9		
		2007	spr	915	1,694	17		
			fal	485	897	13		
		2008	sum	914	1,692	17		
2		2004	sum	259	479	24	2,387	8,186
3		2005	sum	40	74	29	122	420
4							3,489	11,965
	4	2004	sum	13	24			
	10			45	84			
	17			25	47			
	21			53	98			
	29			43	79			
	17	2006	sum	24	44			
	21			56	104			
	7	2007	spr	26	49			
	13			21	39			
	17			17	32			
	21			64	119			
	23			7	13			
	7	2007	fal	27	51			
	13			21	39			
	17			18	33			
	21			66	123			
	23			7	13			
	8	2008	sum	11	20			
	14			25	46			
	17			16	30			
	21			67	125			
	25			21	39			

Species Observed and Conservation Status

Observations used for density estimation are mapped in Appendix 2. Maps of Observations for the nine marine mammal species with sufficient sightings for analysis: harbour porpoise, Dall's porpoise, Pacific white-sided dolphin, killer whale (residents and transients), humpback whale, common minke whale, fin whale, harbour seal, Steller sea lion and elephant seal. Sighting and density estimation of pinnipeds were further separated into “haul-out” or “in-water” categories. The spatial distribution of observations per species across all surveys are shown in Figure 2.

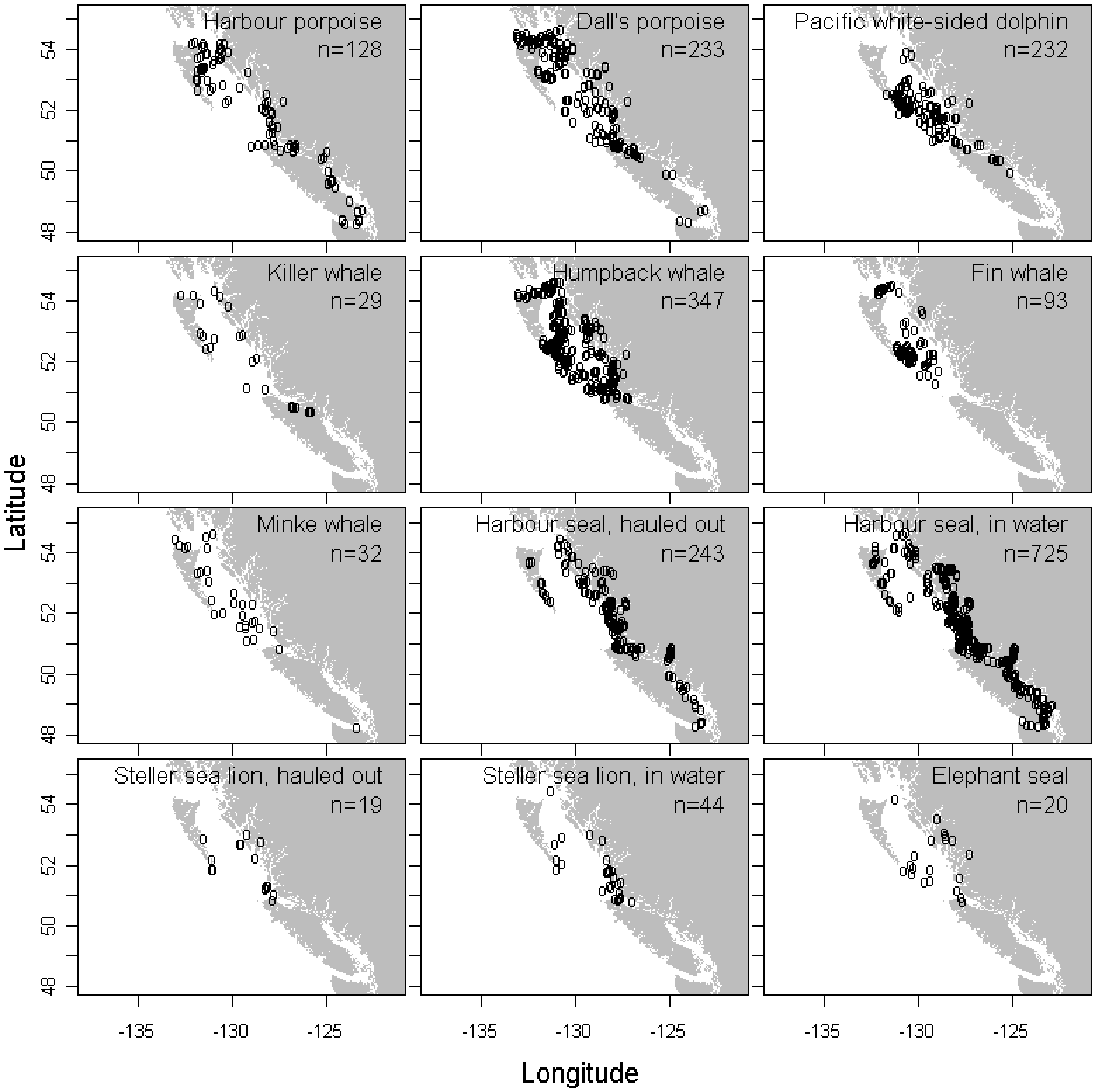

Figure 2. Observations by species from all surveys.

The conservation status of species are determined globally by the United Nations body the International Union for Conservation of Nature (IUCN) and within Canada by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). The government of British Columbia also designates conservation status locally within the province. The status of the marine mammal species assessed in this study are listed in

. Many of the threats to marine mammals are shared across species: low populations from historical hunting, incidental catch from fishing gear, depletion of prey from overfishing, chemical pollution, vessel strikes, and ship noise (Rice 1998).

Table 2. Conservation status of marine mammals in British Columbia waters. Provincial BC status ranges from imperiled (S2) to secure (S5), and not applicable (SNA). Additional criteria, such as subpopulations for killer whales or breeding versus non-breeding status are specified by this criteria in some instances. National COSEWIC status ranges from Endangered (E) to Threatened (T) to Special Concern (SC) to not at risk (NAR). The IUCN status ranges from Endangered (EN) to Least Concern (LC), and data deficient (DD). Year assessed in paranthesis.

	Provincial	National	Global
	BC Status	COSEWIC	IUCN Status
Harbour porpoise	S3 (2006)	SC (2003)	LC (2008)
Dall's porpoise	S4S5 (2006)	NAR (1989)	LC (2008)
Pacific white-sided dolphin	S4S5 (2006)	NAR (1990)	LC (2008)
Humpback whale	S3 (2006)	T (2003)	LC (2008)
Fin whale	non-breeding S2 (2006)	T (2005)	EN (2008)
Killer whale (res+trans)	offshores S3 (2006)	T (2008)	DD (2008)
	transients S2 (2006)	T (2008)	
	S residents S2 (2006)	E (2008)	
	N residents S3 (2006)	T (2008)	
Minke whale	non-breeding S4 (2006)	NAR (2006)	LC (2008)
Harbour seal	S5 (2006)	NAR (1999)	LC (2008)
Steller sea lion	breeding S2S3 (2006)	SC (2003)	EN (2008)
	non-breeding S3 (2006)		
Elephant seal	SNA (2006)	NAR (1986)	LC (2008)

Harbour Porpoise

The Harbour porpoise (*Phocoena phocoena*) is listed as *Vulnerable* by the IUCN with a global population estimate of about 700,000 individuals (Hammond et al. 2008a). Within Canadian Pacific waters, it is recognized as a species of *Special Concern* (COSEWIC 2003). Found predominantly in shallow waters less than 200m in the Northern Hemisphere, 4 subspecies have been genetically identified globally (Rice 1998). Despite continuous distribution alongshore from Point Conception around the Pacific rim to the northern islands of Japan and as far north as Barrow, Alaska, many small populations appear genetically distinct, suggesting the need to consider small subpopulation management units (Chivers et al. 2002). Prior to the study conducted by Williams & Thomas (2007), the only distribution information estimated 3,000 individuals for the southern inshore portion of BC based on 1996 surveys (Baird 2003a). To the south, the stock in the coastal waters of Washington and Oregon were estimated to be around 40,000 in 1997, and to the north in southeastern Alaska to be 10,000 animals (Baird 2003a).

Dall's porpoise

The Dall’s porpoise (*Phocoenoides dalli*) is globally abundant with an estimated population of over 1.2 million individuals and listed as a species of *Least Concern* by the IUCN (Hammond et al. 2008b) and not at risk within Canada. They are distributed within the North Pacific Ocean, generally in deeper waters between 30°N and 62°N (Jefferson 1988). Considered either a subspecies or color-morph the most common *dalli*-type resides in the NW Pacific.

Pacific white-sided dolphin

The Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) is listed by the IUCN as a species of *Least Concern* with global populations estimated to be over 1 million (Hammond et al. 2008c) and not at risk in Canadian waters. They are distributed along the temperate coastal shelf waters and some inland BC waterways of the North Pacific from roughly 35°N to 47°N (Stacey and Baird 1991; Heise 1997).

Humpback whale

Humpback whales (*Megaptera novaeangliae*) were down-listed by the IUCN in 2008 to a species of *Least Concern* status since current global estimates now exceed 60,000 individuals. This population level exceeds the 50% threshold of the 1940 population needed to retain its former *Vulnerable* status (Reilly et al. 2008a). Combined mark-recapture and photo-id analysis conducted under the "Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific" (SPLASH) project estimate the population in the region to be just under 20,000, which is appoximately double the previous population estimates (Calambokidis et al. 2008). These increasing numbers have been heralded as a sign of post-whaling recovery (Dalton 2008). In southeastern Alaska, Dahlheim et al. (2009) found that humpback whale abundance over the period 1991 to 2007 increased annually by 10.6% (SE=0.015). The Canadian and Provincial designations, *Special Concern* (S3) and *Threatened* (T) respectively (Baird 2003b), have not been updated since 2006 and 2003.

Fin whale

The *Endangered* fin whales (*Balaenoptera physalus*) are found globally, largely in offshore waters and less so in warm tropical regions (Reilly et al. 2008b). They are noted to occur on the middle (50–100 m) and outer shelves (100–200 m) in the eastern Bering Sea (Moore et al. 2002). In the waters off of western Alaska and the central Aleutian Islands, Zerbini et al. (2006) compared surveys from 1987 with those from 2001-2003 and found a 4.8% (95% CI = 4.1-5.4%) annual rate of increase (Zerbini et al. 2006) with population levels at 1652 (95% CI = 1142-2389) individuals. Since the 1975 north Pacific estimate of roughly 17,000 fin whales, down from an estimated 44,000 preceding intensive whaling, there has been a lack of sufficient survey data and abundance estimates (Reilly et al. 2008b) to develop estimates for the entire regional population of fin whales.

Killer whale

Killer whales (*Orcinus orca*) occur globally in highly productive, often cooler waters, and are listed by the IUCN as *Data Deficient (Taylor et al. 2008)*. In British Columbia there are four designated units of killer whales (with 2006 population estimates based on photo-id): 1) Northern Resident (244), 2) Southern Resident (87), 3) West Coast Transient (198), and 4) Offshore (COSEWIC 2008). All of these subunit populations are designated within Canadian waters as *Threatened*, except the southern residents, which are listed as *Endangered*. These subunit populations generally do not interact with each other, have unique habitats, forage for different prey, and can generally be identified by dorsal fin morphology (Ford et al. 2009). The residents feed on fishes, especially salmon, while the transients prey on marine mammals. The less understood offshore type probably also feed on fish, although different varieties than the transients based on stable isotope analysis (Ford et al. 2009).

Minke whale

Minke whales (*Balaenoptera acutorostrata*) are found globally and are listed as a species of *Least Concern* despite no global estimates since estimates for parts of the Northern Hemsiphere alone are over 100,000 (Reilly et al. 2008c). In Canada they are listed as a species *Not at Risk*.

Harbour seal

Harbour seals (*Phoca vitulina*) inhabit the coastal parts of the Northern Hemisphere in temperate and polar areas with a global population between 350,000 to 500,000 individuals (Thompson and Härkönen 2008), and have an IUCN status of species of *Least Concern* and are considered *Secure* in Canada. Of the five subspecies, *P.v. richardii* is found in the eastern Pacific, which has been stable or increasing in population since the early 1990s.

Steller sea lion

Steller sea lions (*Eumetopias jubatus*) inhabit the coastal waters of the North Pacific. They experienced a dramatic 64% decline in their population from 1960 to 1989, with a current estimate of 105,000 to 117,000 animals (Gelatt and Lowry 2008). Recognized as a species of *Special Concern* in British Columbia, they are found near one of three breeding grounds and 21 haul-out sites. The BC breeding population is estimated to be about 19,000 animals, out of the total Eastern population estimated to be 45,000 individuals in 2002 (COSEWIC 2003).

Elephant seal

Elephant seals (*Mirounga angustirostris*) have recovered from virtual extinction with 2005 population estimates around 171,000 and are now listed by the IUCN as a species of *Least Concern*. Elephant seals range from throughout the northeastern Pacific (Campagna 2008). Within Canada's waters they are considered *Not at Risk*.

Abundance Estimation

Estimating the abundance of marine mammal species requires specialized techniques to objectively extrapolate from individual point observations of animals collected along line-transect surveys to broader estimates of the expected encounter rate, abundance and density. Since many marine animals are only observed at the surface for short intervals and the ability of observers is significantly influenced by sea conditions, modeling techniques must also specifically estimate the expected probability of detection (Buckland et al. 2001; Burnham et al. 1980).

Traditional estimates of cetacean abundance have relied on design-based surveys covering an entire survey strata at a time and have been based on simple estimates of the detection function (Buckland et al. 2001). More recently, detection functions have been fitted using environmental covariates to provide more precise estimates (Marques and Buckland 2003). For instance, Barlow and Forney (2007) used covariates such as glare, group size and survey vessel as detection function covariates to analyze the most comprehensive set of multi-species cetacean surveys to date for the US West Coast. Even more recently, a spatial modeling component has been in development for inclusion in the *Distance* software (Thomas et al. 2006). Ferguson et al. (2006) used similar techniques in the Eastern Tropical Pacific to predict both encounter rates and group sizes of cetacean species using generalized additive models to link with environmental covariates. They tested for geographic pattern in the unexplained model residuals to control for potential autocorrelation. These methods help provide more precise and object estimates of habitat preferences.

Regression-based techniques are the most common methods for determining habitat preferences of cetaceans (Redfern et al. 2006), and of these generalized additive models have the most flexibility for determining non-linear relationships. Here we use the conventional stratum-based abundance estimation with a covariate of group size and compare with the more advanced density surface, or spatial modeling, approaches using generalized additive models to estimate encounter rate.

Tier A. Estimating Abundance

Product 1. Using Conventional Distance Sampling

Methods

The design-based abundance methods of Williams and Thomas (2007) originally performed on 2004 and 2005 data were reproduced for the extended period of survey to include 2006, 2007 and 2008 data for comparison. Abundance is is estimated as the density of animals multiplied by the applicable study area or stratum. To estimate density ($\hat{D}$) the encounter rate ($\frac{n}{L}$), or number of schools seen (n) over the length of the transect (L), is multiplied by twice the truncation distance (w) to obtain an area and the estimated school size ($\hat{s}$).

$$\hat{D} = \frac{n\hat{s}}{L2w\hat{p}} \quad (1)$$

If all animals present within the transect area were assumed to be detected over this area, as with strip transects, we would stop here. However, we can safely assume that the probability of detecting a school decreases with distance. Accounting for this probability of detection ($\hat{p}$) forms the basis of 'conventional distance sampling' (CDS), formally described by Buckland *et al.* (2001) by fitting a detection function (covered in the next section). This $\hat{p}$ term in the denominator allows for the probability of detection to decrease with distance and the estimate of density will be appropriately compensated.

Detection Functions

Detection functions were estimated using the software *Distance* 6.0 Beta 3 (Thomas et al, 2006), which can apply several key functions (uniform, half-normal or hazard rate) and series expansion terms (polynomial or cosine) to estimate the shape of the function. The observers recorded radial distance (d) and angle (θ) during the field surveys. These relative values are then converted to perpendicular distance from the trackline using simple geometry, $sin(\theta)*d$. All on-effort (i.e. periods when the observers were actively observing for animals) sightings, including off-transect observations, were used for detection model fitting. Models were generally selected that minimize the Akaike Information Criterion (AIC) score, which promotes explanation of deviance while penalizing the addition of terms to achieve the most parsimonious model (Akaike 1974). In addition, the Kolmogorov-Smirnov goodness-of-fit test was employed to provide a measure of agreement between the model and data (S. T. Buckland et al. 2004). If species exhibit an attraction to the survey vessel, then a spike is typically seen nearest the trackline which can inflate the density estimates by lowering the $\hat{p}$ over the rest of the strip width. As detectability drops off, inclusion of further distances in the function can similarly inflate the density so a reasonable truncation distance (w), usually excluding the furthest 5% or 10% of the observations is selected and is then rounded to the nearest 100m.

These detection functions all assume perfect detection on the trackline, i.e. g(0)=1. A probability of availability is typically divided by the density to account for the fact that marine mammals are often below the water surface and not detected even when directly on the trackline of the observer vessel. Estimating this probability requires tracking of individuals to estimate proportion of time spent underwater (Laake et al. 1997) or multiple platforms of simultaneous, independent observation. These time and cost-intensive estimates have not yet been conducted for these species in these waters, and so are not applied. Therefore the abundance estimates developed may underestimate the true population size.

School Size

School size, or group size, was estimated in *Distance* using the default conventional distance sampling method. We expect that the estimate of school size diminishes with observed distance, e.g. school sizes tend to be underestimated when observed from a further distance. The natural logarithm of group size is regressed on the probability of detection, and the value of $ln(\hat{s})$ at zero distance is back-transformed to obtain the estimated school size $(\hat{s})$.

Results

The final detection models selected are given in Table 3. Observations are truncated to within the perpendicular distance (*w*) used by the conventional distance sampling (CDS) methods for comparison (see Appendix 2. Maps of Observations).

Table 3. Detection function summary statistics. Truncation distance (*w*) with number of sightings (n) before and after truncation. Model described by key function (hazard-rate (hr), half-normal (hn), or uniform (un)) with optional series expansion terms (polynomial (poly), or cosine (cos)). The *p*-value for the goodness-of-fit Kolmogorov-Smirnov (K-S), and the final probability of detection $\hat{p}$ and its percent coefficient of variation (%CV ($\hat{p}$)).

	w (m)	*n* before	*n* after	Model	K-S *p*	$\hat{p}$	%CV ($\hat{p}$)
Harbour porpoise	600	128	118 (-8%)	hr	0.899	0.201	24.29
Dall's porpoise	700	239	221 (-8%)	hn+cos(3)	0.190	0.344	9.86
Pacific white-sided dolphin	1200	233	219 (-6%)	hn+cos(4)	0.001	0.253	8.63
Humpback whale	2300	352	325 (-8%)	hn+cos(1)	0.951	0.421	6.43
Fin whale	3900	91	82 (-10%)	hn+cos(2)	0.375	0.270	11.01
Killer whale (res+trans)	1300	29	25 (-14%)	hn	0.302	0.558	16.71
Minke whale	400	32	29 (-9%)	un+cos(1)	0.641	0.620	13.81
Harbour seal (haul-out)	700	244	212 (-13%)	un+cos(1)	0.326	0.728	6.69
Harbour seal (in-water)	500	774	732 (-5%)	hn+cos(1)	0.030	0.477	4.74
Steller sea lion (haul-out)	1300	20	17 (-15%)	un+cos(1)	0.639	0.686	21.57
Steller sea lion (in-water)	500	123	114 (-7%)	hn	0.047	0.548	7.71
Elephant seal	500	20	18 (-10%)	un	0.572	1.000	0.00

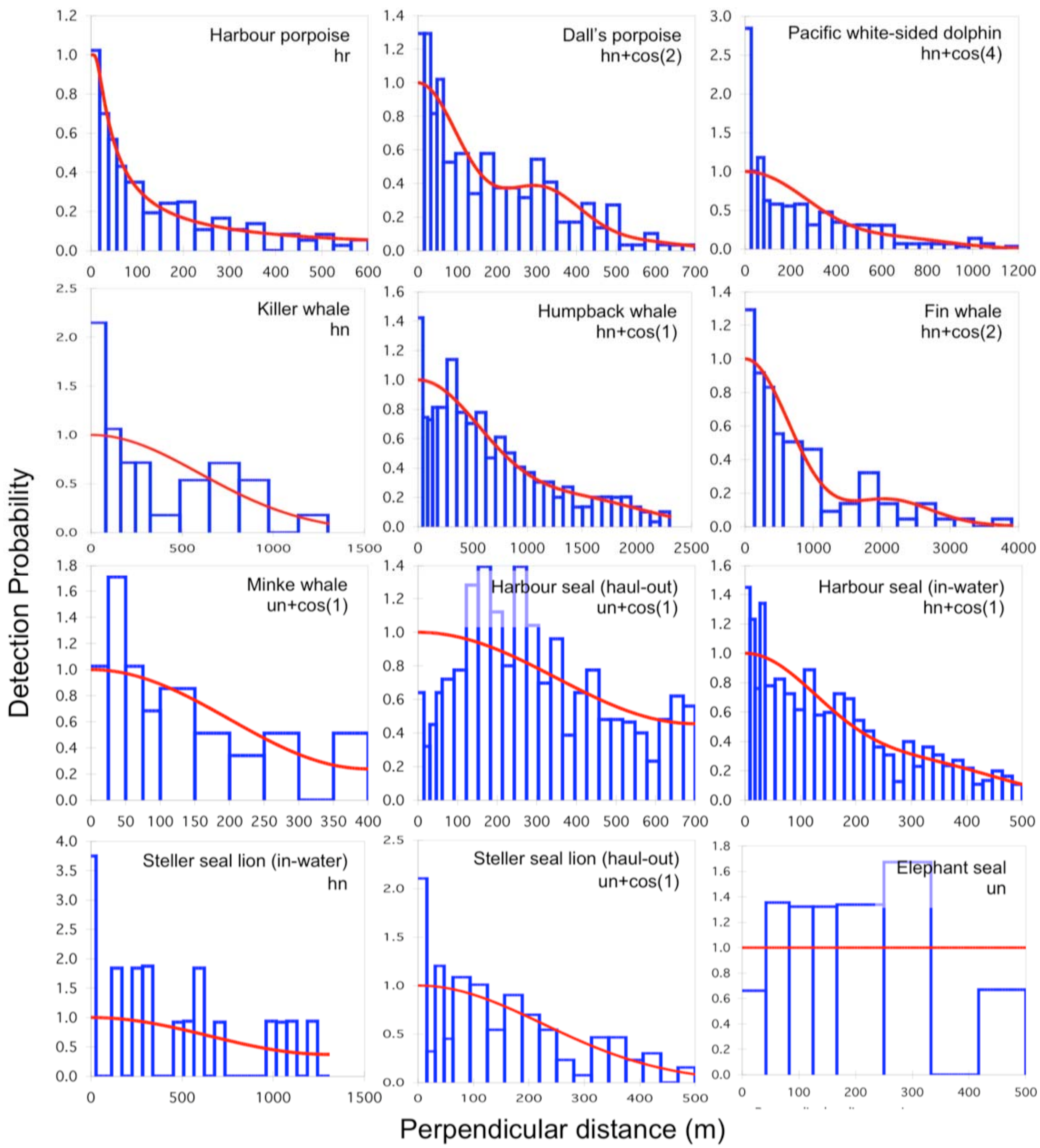

Figure 3. Detection functions for conventional distance sampling (CDS) analysis.

Table 4. Estimated school size.

	Estimated school size		Observed school size		
	$\hat{s}$	%CV ($\hat{s}$)	Mean	%CV	Maximum
Harbour porpoise	1.67	4.56	1.81	4.53	5
Dall's porpoise	2.41	4.54	2.43	5.55	15
Pacific white-sided dolphin	13.53	14.77	38.27	20.41	1200
Humpback whale	1.51	2.79	1.57	3.75	8
Fin whale	1.78	6.86	1.99	12.73	20
Killer whale (res+trans)	3.67	18.26	3.80	15.27	28
Minke whale	0.99	2.36	1.03	3.33	2
Harbour seal (haul-out)	5.58	9.51	6.82	9.77	90
Harbour seal (in-water)	1.11	1.20	1.20	2.83	18
Steller sea lion (haul-out)	70.29	66.86	37.77	49.25	300
Steller sea lion (in-water)	6.11	20.31	14.41	26.89	370
Elephant seal	1	0	1	0	1

Cetaceans

Harbour porpoise

Combining all surveys, 128 harbour porpoise schools were sighted (Table 3). They are distributed widely across the northern and southern extents of the study area, and are found to be more common nearshore and within inlets (Figure 2). The vast majority (122/128=95%) exhibited traveling/foraging behaviour, and only 2 feeding and 2 avoiding, so no obvious response to the observer vessel is indicated with this data..

Restricting the observations to a truncation distance of 600m excluded 10 observations, or 8% of the data collected (Table 3). The most parsimonious detection function as determined by the lowest AIC was the hazard rate model without adjustment terms (Figure 3). The data show a spike near zero, which is most appropriately fit with a hazard rate model. These spikes are typically of concern with attractive movement, but none was noted in the field, so alternate models that removed the spike were not chosen over the lower AIC criteria. The bias towards zero in the detection function may accurately reflect the small size and cryptic nature of this species. All the other models tested with higher AIC values produced smoother fits than the data or the hazard rate model, which produced a higher $\hat{p}$ and lower abundance estimate. For instance, the next lowest-AIC model (ΔAIC=8.53), uniform with 5 cosine adjustments, produced a $\hat{p}$ 41% larger (0.284 vs 0.201).

Dall's porpoise

Of the 239 Dall's porpoise school sightings (Table 3), most occurred in the northern and southern ends of Queen Charlotte Basin, often offshore within the basin, with relatively few schools within the inlets or the southern straits (Figure 2). Whereas most observations (212/239 = 88.7%) were traveling/foraging, a noteworthy portion (11/239 = 4.6%) were approaching and the same number feeding. Other behaviours included schooling (2/239 = 0.8%), avoiding (1/239 = 0.4%) and unknown (2/239 = 0.8%).

A truncation distance of 700m excluded 18 observations, or 8% of the observations from model fitting. The hazard rate function with 1 cosine adjustment fit the data best according to the AIC crietria, but exhibited a sharp spike near zero. Given that Dall's porpoise were recorded with attractive behaviour and are known to bow-ride in front of boats, the next lowest-AIC model (ΔAIC=7.02), half-normal with 3 cosine adjustments, was chosen because of it's broader shoulder which minimizes the bias near zero due to attractive behaviour (Figure 3). Turnock and Quinn (1991) also found that a half-normal corrects most for the attractive aspects, using simulations and data from Dall's porpoises in Alaska. To further quantify a correction factor, a secondary platform of observation is recommended.

Pacific white-sided dolphin

Of the 233 schools of Pacific white-sided dolphin, the majority were seen throughout the southern half of the Queen Charlotte Basin, particularly near Haida Gwaii, as well as a few observations in the inlets and northern end of the southern straits (Figure 2). This species exhibits the strongest approaching behaviour (47/233=20.2%). Other behaviours include: traveling/foraging (151/233=64.8%), feeding (18/233=7.7%), breaching (13/233=6%), socializing (1/233=0.4%), avoidance (1/233=0.4%) and uncertain (2/233=0.8%).

Using a truncation distance of 1200m (Table 3), the lowest-AIC models used a hazard rate model, which followed the spike of the data near zero distance. To minimize the bias of attractive movement, the next lowest-AIC (ΔAIC=23.89) half-normal model with 4 cosine adjustments was chosen (Figure 3). This is a similar strategy for model selection as used with Dall's porpoise.

Humpback whale

The highest number of cetacean school sightings (n=352) were attributed to humpback whale (Table 3). These sightings occurred exclusively in Queen Charlotte Sound and the inlets, and not in the southern straits (Figure 2). Most Sound sightings were in deep water, with some preference towards the southern Haida Gwaii region and the northeastern Sound. Only one observation was noted for approaching behaviour (1/352=0.2%), and the rest included: traveling/foraging (265/352=75.3%), feeding (41/352=11.6%), breaching (25/352=7.1%), socializing (3/352=8.5%), and unknown (5/352=1.4%). Using a 2300m truncation distance, the lowest-AIC model was chosen using a half-normal model with one cosine adjustment term (Figure 3).

Fin whale

All of the 91 school sightings of fin whale were found in Queen Charlotte Basin, with the exception of a couple of observations in the Grenville Channel inlet. Historical records reveal that fin whales were once one of the most abundant and heavily whaled marine mammals within the inshore waters British Columbia (Gregr et al. 2000). Most these sightings are in the southern end of the Queen Charlotte Islands, with another large cluster of sightings are in the north of the Sound (Figure 2). The behaviours of sightings include: traveling/foraging (73/91=80.2%), feeding (3/91=3.3%), socializing (1/91=1.1%), and other/uncertain (4/91=4.4%). A 3,900m truncation distance was applied (Table 3). The hazard rate model obtained the lowest AIC, but exhibited a spike near zero, so a half-normal model with two cosine adjustment terms (ΔAIC=1.4) was used instead (Figure 3).

Killer whale

At 29 school sightings, the killer whale is the least common of the observed whale species (Table 3), but one of the most studied species. Most targeted killer whale studies differentially treat the resident versus transient ecotypes (Zerbini et al. 2007), but data constraints forced us to lump the two types together for this analysis. Sightings occurred in both Queen Charlotte Basin and Johnstone Strait, most commonly near shore (Figure 2). Observed behaviours include: traveling/foraging (24/29=82.7%), feeding (2/29=6.7%), socializing (1/29=3.4%) and other (2/29=6.7%). A truncation distance of 1300m was applied to provide a monotonically decreasing tail, while retaining as many observations as possible (25/29=86%). A hazard rate model best fit these data. But to offset the spike near zero, the half-normal model without adjustment terms (ΔAIC=0.53) was chosen.

Minke whale

Only slightly more common (n=32) than killer whales in school sightings is the common minke whale (Table 3). All observations were widely distributed within Queen Charlotte Basin, generally offshore in deeper waters (Figure 2). All sightings were recorded as traveling/foraging behaviour, although minke whales are at surface less than other species so directionality is often difficult to determine. Of the 32 observations only 3 exceeded 400 m in perpendicular distance from the transect line (2377 m, 1888 m, and 1532 m), so a truncation distance of 400m was used. The lowest-AIC model, a uniform model with one cosine adjustment term, was chosen in this case.

Pinnipeds

Harbour seal

The most commonly sighted of all marine mammals (n=1018), the harbour seal was seen most typically nearshore throughout all strata: sound, inlets, and straits (Figure 2). Harbour seals exhibited the following behaviours: traveling/foraging (701/1018=68.9%), socializing (75/1018=7.4%), feeding (13/1018=1.3%), approaching (1/1018=0.1%), and other/unknown (110/1018=10.8%). Detectability is expected to vary as a function of whether the pinniped is in or out of water, hence the separation between in-water and haul-out observations for truncation distances and detection functions (Table 3). For in-water observations, a truncation distance of 500m was used and the lowest-AIC model selected was a half-normal model with one cosine adjustment term (Figure 3). For haul-out observations, a 700m truncation was used, indicative of greater visibility when out of water, and the lowest-AIC model selected was a uniform model with one cosine adjustment. The distance data for haul-out observations exhibit a peak around 200m rather than monotonically increasing towards zero. Because most haul out sightings are to the side during along-shore transects, this off-zero peak is understandable. Roughly one quarter of the sightings were haul-out versus three quarters in-water.

Steller sea lion

A total of 123 Steller sea lion schools were sighted in-water and 20 on land, all generally in the nearshore and inlet environments of the southern Queen Charlotte Basin (Figure 2). They exhibited slight responsiveness to the ship (avoidance: 3/123=24.4%; approach: 2/123=1.6%), otherwise found traveling/foraging (67/123=54.5%), socializing (10/123=8.1%), feeding (3/123=2.4%), or other/unknown (38/123=30.9%). For in-water observations, a 500m truncation distance was used and the lowest-AIC model selected was a half-normal model. For haul-out observations, a 1300m truncation distance was used and the lowest-AIC model selected was a uniform model with one cosine adjustment.

Elephant seal

The least numerous of all the marine mammal species analyzed here (# school sightings=20), the elephant seal was observed in the open waters of Queen Charlotte Basin as well as the southern and central inlets (Figure 2). A 500m truncation distance was used, and the final model selected was a uniform model, which corresponds to a strip transect, i.e. density is assumed to not vary with distance from transect. In this case there were too few observations to construct a robust distance detection function, further evidenced by the unrealistic *p* value of a solid 1 (Table 3).

Abundance Estimates

Abundance estimates were calculated across all surveyed seasons and strata by species, as summarized by Tables 5 through 8. During the 2006 survey, observer effort within the inlet stratum 4 was not part of a designed survey and only included effort while on passage, so this data was excluded for estimation of abundance estimates.

Comparing this analysis with previous estimates (Williams and Thomas 2007), which used only survey data from 2004 and 2005; we see closer confidence intervals for the results of the overall surveyed region with the addition of recent survey data. Steller sea lions and elephant seals were included in this analysis and not in Williams and Thomas (2007) due to limited sample size.

Table 5. Conventional distance sampling estimates for cetaceans in Stratum 1.

	Stratum 1							
Estimate	2004 Summer	2005 Summer	2006 Summer	2007 Spring	2007 Fall	2008 Summer	Average All	Average Summers
Harbour porpoise								
D	0.157	0.309	0	0.056	0.027	0.211	0.153	0.202
95%CI(D)	0.036 - 0.675	0.108 - 0.887	0	0.014 - 0.221	0.003 - 0.216	0.044 - 1.023	0.066 - 0.355	0.083 - 0.492
N	2,874	5,677	0	1,032	487	3,874	2,806	3,704
95%CI(N)	667 - 12,391	1,980 - 16,279	0	263 - 4,054	60 - 3,964	799 - 18,785	1,209 - 6,514	1,518 - 9,040
%CV	79.4%	54.9%	0	73.4%	125.1%	87.5%	43.2%	45.9%
Dall's porpoise								
D	0.492	0.354	0.113	0.081	0.115	0.182	0.247	0.318
95%CI(D)	0.248 - 0.978	0.109 - 1.152	0.039 - 0.332	0.027 - 0.240	0.035 - 0.379	0.085 - 0.391	0.147 - 0.416	0.180 - 0.560
N	9,038	6,507	2,083	1,487	2,105	3,350	4,540	5,838
95%CI(N)	4,549 - 17,956	2,001 - 21,159	711 - 6,098	503 - 4,399	638 - 6,950	1,562 - 7,184	2,700 - 7,632	3,313 - 10,289
%CV	33.8%	60.9%	50.1%	55.1%	59.7%	37.7%	25.5%	27.8%
Pacific white-sided dolphin								
D	2.196	1.762	3.415	0.858	0.085	1.582	1.566	2.013
95%CI(D)	1.048 - 4.600	0.544 - 5.705	1.107 - 10.536	0.387 - 1.901	0.007 - 1.041	0.745 - 3.361	0.928 - 2.642	1.152 - 3.517
N	40,316	32,345	62,708	15,755	1,565	29,054	28,759	36,958
95%CI(N)	19,243 - 84,464	9,988 - 104,747	20,327 - 193,448	7,111 - 34,905	128 - 19,113	13,680 - 61,706	17,047 - 48,517	21,153 - 64,573
%CV	37.2%	61.1%	53.9%	40.0%	166.2%	37.8%	26.4%	28.1%
Humpback whale								
D	0.049	0.046	0.026	0.132	0.06	0.112	0.078	0.065
95%CI(D)	0.020 - 0.121	0.022 - 0.095	0.012 - 0.059	0.086 - 0.204	0.027 - 0.131	0.075 - 0.167	0.059 - 0.103	0.045 - 0.092
N	909	839	486	2,431	1,093	2,057	1,431	1,186
95%CI(N)	373 - 2,213	406 - 1,737	219 - 1,081	1,577 - 3,747	496 - 2,405	1,382 - 3,062	1,085 - 1,888	835 - 1,684
%CV	44.1%	35.7%	36.2%	21.0%	37.7%	19.3%	13.6%	17.0%
Fin whale								
D	0.012	0.045	0	0.024	0.026	0.024	0.024	0.024
95%CI(D)	0.005 - 0.030	0.017 - 0.120	0	0.010 - 0.060	0.010 - 0.068	0.010 - 0.057	0.014 - 0.041	0.012 - 0.047
N	223	820	0	441	476	442	446	443
95%CI(N)	91 - 548	305 - 2,199	0	176 - 1,108	182 - 1,242	188 - 1,040	262 - 760	229 - 859
%CV	44.9%	50.0%	0	46.2%	47.1%	42.7%	26.4%	32.7%
Killer whale								
D	0.026	0.01	0.014	0.015	0.01	0.005	0.014	0.014
95%CI(D)	0.007 - 0.100	0.002 - 0.045	0.003 - 0.073	0.005 - 0.050	0.001 - 0.138	0.001 - 0.025	0.006 - 0.032	0.005 - 0.036
N	476	188	263	282	177	94	251	253
95%CI(N)	124 - 1,829	43 - 829	51 - 1,346	86 - 921	12 - 2,527	19 - 463	107 - 585	96 - 666
%CV	72.0%	81.3%	83.6%	62.1%	186.3%	88.6%	43.4%	49.9%
Minke whale								
D	0.029	0.02	0.045	0.02	0	0.02	0.022	0.025
95%CI(D)	0.014 - 0.058	0.007 - 0.055	0.015 - 0.136	0.007 - 0.061	0	0.005 - 0.078	0.013 - 0.037	0.014 - 0.045
N	526	371	830	371	0	371	396	466
95%CI(N)	258 - 1,071	136 - 1,013	275 - 2,505	123 - 1,119	0	96 - 1,431	231 - 678	261 - 829
%CV	35.4%	51.1%	52.1%	56.5%	0	71.2%	26.7%	28.6%

Table 6. Conventional distance sampling estimates for cetaceans in Strata 2,3,4, and entire region.

	Stratum 2	Stratum 3	Stratum 4				Entire Region
	2004	2005	2004	2007	2008		
Estimate	Summer	Summer	Summer	Fall&Spring	Summer	Average	Average
Harbour porpoise							
D	1.342	0	0.24	0.049	0.247	0.178	0.272
95%CI(D)	0.540 - 3.334	0	0.006 - 9.643	0.001 - 1.616	0.006 - 9.903	0.012 - 2.709	0.138 - 0.536
N	3,203	0	838	170	861	622	6,631
95%CI(N)	1,289 - 7,957	0	21 - 33,641	5 - 5,639	21 - 34,546	41 - 9,449	3,366 - 13,065
%CV	47.4%	0	225.0%	317.2%	225.0%	213.6%	34.9%
Dall's porpoise							
D	0.358	0.695	0.252	0.335	0.028	0.216	0.256
95%CI(D)	0.289 - 0.443	0.562 - 0.860	0.009 - 7.159	0.015 - 7.550	0.001 - 1.125	0.011 - 4.390	0.171 - 0.383
N	855	85	879	1,168	96	752	6,232
95%CI(N)	691 - 1,058	69 - 105	31 - 24,973	52 - 26,339	2 - 3,926	37 - 15,315	4,165 - 9,324
%CV	10.9%	10.9%	182.0%	238.4%	223.9%	267.7%	20.0%
Pacific white-sided dolphin							
D	0.16	20.675	0	0.151	0.916	0.277	1.34
95%CI(D)	0.114 - 0.223	14.803 - 28.875	0	0.005 - 4.997	0.030 - 27.527	0.011 - 7.093	0.825 - 2.177
N	381	2,532	0	525	3,195	965	32,637
95%CI(N)	273 - 533	1,813 - 3,536	0	16 - 17,433	106 - 96,029	38 - 24,744	20,087 - 53,029
%CV	17.2%	17.1%	0	316.7%	189.2%	322.8%	24.6%
Humpback whale							
D	0	0	0.062	0.004	0.047	0.031	0.063
95%CI(D)	0	0	0.002 - 1.711	0.000 - 0.145	0.004 - 0.615	0.002 - 0.436	0.049 - 0.082
N	0	0	216	15	164	110	1,541
95%CI(N)	0	0	8 - 5,967	0 - 505	13 - 2,146	8 - 1,521	1,187 - 2,000
%CV	0	0	178.5%	316.3%	117.1%	199.0%	12.9%
Fin whale							
D	0	0	0	0	0	0	0.018
95%CI(D)	0	0	0	0	0	0	0.011 - 0.031
N	0	0	0	0	0	0	446
95%CI(N)	0	0	0	0	0	0	263 - 759
%CV	0	0	0	0	0	0	26.4%
Killer whale							
D	0	0.469	0	0	0	0	0.013
95%CI(D)	0	0.287 - 0.766	0	0	0	0	0.006 - 0.027
N	0	57	0	0	0	0	308
95%CI(N)	0	35 - 94	0	0	0	0	146 - 649
%CV	0	24.8%	0	0	0	0	38.2%
Minke whale							
D	0.014	0	0	0	0	0	0.018
95%CI(D)	0.011 - 0.019	0	0	0	0	0	0.011 - 0.029
N	34	0	0	0	0	0	430
95%CI(N)	26 - 45	0	0	0	0	0	259 - 712
%CV	14.0%	0	0	0	0	0	25.2%

Table 7. Conventional distance sampling estimates for pinnipeds in Stratum 1.

Estimate	2004 Summer	2005 Summer	2006 Summer	2007 Spring	2007 Fall	2008 Summer	Average All	Average Summer
Harbour seal, hauled out								
D	0.09	0.089	0.093	0.022	0.042	0.067	0.066	0.083
95%CI(D)	0.019 - 0.428	0.041 - 0.192	0.009 - 0.954	0.002 - 0.215	0.008 - 0.212	0.025 - 0.175	0.033 - 0.133	0.039 - 0.176
N	1,651	1,630	1,712	407	769	1,224	1,212	1,523
95%CI(N)	347 - 7,863	753 - 3,527	167 - 17,517	42 - 3,939	152 - 3,894	467 - 3,209	600 - 2,450	717 - 3,236
%CV	85.2%	38.4%	133.4%	146.7%	86.5%	48.4%	34.6%	37.2%
Harbour seal, in water								
D	0.119	0.047	0.026	0.09	0.089	0.047	0.074	0.066
95%CI(D)	0.051 - 0.282	0.018 - 0.121	0.004 - 0.155	0.048 - 0.167	0.015 - 0.528	0.018 - 0.125	0.046 - 0.118	0.038 - 0.117
N	2,192	866	485	1,644	1,634	866	1,350	1,217
95%CI(N)	929 - 5,172	336 - 2,227	82 - 2,849	880 - 3,072	275 - 9,690	326 - 2,300	839 - 2,172	690 - 2,145
%CV	42.3%	47.2%	89.8%	30.3%	97.5%	48.7%	22.8%	27.3%
Harbour seal, total								
D	0.209	0.136	0.12	0.112	0.131	0.114	0.14	0.149
95%CI(D)	0.089 - 0.491	0.075 - 0.246	0.019 - 0.750	0.053 - 0.235	0.035 - 0.495	0.057 - 0.226	0.093 - 0.210	0.092 - 0.241
N	3,842	2,496	2,197	2,052	2,403	2,090	2,562	2,740
95%CI(N)	1,638 - 9,016	1,379 - 4,516	350 - 13,778	974 - 4,323	635 - 9,087	1,052 - 4,154	1,704 - 3,852	1,697 - 4,426
%CV	43.8%	29.9%	105.8%	37.9%	71.9%	34.8%	20.3%	24.0%
Steller sea lion, hauled out								
D	0	0	0	0	0.301	0.24	0.082	0.072
95%CI(D)	0	0	0	0	0.024 - 3.821	0.039 - 1.462	0.013 - 0.497	0.012 - 0.438
N	0	0	0	0	5,530	4,399	1,503	1,314
95%CI(N)	0	0	0	0	436 - 70,158	721 - 26,845	248 - 9,119	215 - 8,036
%CV	0	0	0	0	179.6%	109.6%	108.9%	109.6%
Steller sea lion, in water								
D	0.16	0.135	0.063	0.316	0.17	0.158	0.18	0.142
95%CI(D)	0.038 - 0.664	0.060 - 0.307	0.012 - 0.334	0.132 - 0.758	0.052 - 0.553	0.059 - 0.422	0.098 - 0.333	0.067 - 0.297
N	2,936	2,485	1,160	5,797	3,126	2,901	3,314	2,601
95%CI(N)	706 - 12,200	1,096 - 5,634	219 - 6,129	2,415 - 13,914	963 - 10,145	1,087 - 7,746	1,796 - 6,116	1,239 - 5,460
%CV	76.7%	41.8%	85.1%	44.7%	60.0%	50.4%	31.3%	37.7%
Steller sea lion, total								
D	0.16	0.135	0.063	0.316	0.471	0.398	0.262	0.213
95%CI(D)	0.038 - 0.664	0.060 - 0.307	0.012 - 0.334	0.132 - 0.758	0.071 - 3.117	0.114 - 1.388	0.121 - 0.567	0.091 - 0.498
N	2,936	2,485	1,160	5,797	8,656	7,301	4,817	3,915
95%CI(N)	706 - 12,200	1,096 - 5,634	219 - 6,129	2,415 - 13,914	1,309 - 57,235	2,092 - 25,483	2,230 - 10,403	1,675 - 9,153
%CV	76.7%	41.8%	85.1%	44.7%	116.8%	69.0%	40.2%	44.5%
Elephant seal								
D	0.008	0.004	0	0.004	0	0	0.003	0.004
95%CI(D)	0.003 - 0.021	0.001 - 0.014	0	0.001 - 0.012	0	0	0.002 - 0.006	0.002 - 0.008
N	151	74	0	74	0	0	61	67
95%CI(N)	58 - 391	21 - 260	0	24 - 228	0	0	31 - 119	30 - 146
%CV	47.4%	64.9%	0	56.8%	0	0	32.0%	38.3%

Table 8. Conventional distance sampling estimates for pinnipeds in Strata 2,3,4, and entire region.

	Stratum 2	Stratum 3	Stratum 4				Entire Region
Estimate	2004 Summer	2005 Summer	2004 Summer	2007 Fall&Spring	2008 Summer	Average	Average
Harbour seal, hauled out							
D	1.217	0	1.567	0.3	1.437	0.844	0.29
95%CI(D)	0.968 - 1.529	0	0.090 - 27.386	0.033 - 2.773	0.059 - 34.745	0.067 - 10.642	0.225 - 0.374
N	2,904	0	5,467	1,047	5,014	2,944	7,060
95%CI(N)	2,311 - 3,649	0	313 - 95,538	113 - 9,673	207 - 121,210	233 - 37,126	5,477 - 9,101
%CV	11.7%	0	138.8%	128.1%	166.4%	185.0%	12.9%
Harbour seal, in water							
D	1.934	0.647	1.631	1.225	0.902	1.246	0.427
95%CI(D)	1.754 - 2.133	0.588 - 0.713	0.134 - 19.808	0.220 - 6.830	0.088 - 9.234	0.240 - 6.480	0.375 - 0.485
N	4,617	79	5,689	4,275	3,145	4,348	10,394
95%CI(N)	4,187 - 5,090	72 - 87	468 - 69,099	767 - 23,827	307 - 32,212	836 - 22,606	9,143 - 11,816
%CV	5.0%	4.9%	111.8%	88.4%	101.2%	93.6%	6.5%
Harbour seal, total							
D	3.151	0.647	3.198	1.526	2.339	2.09	0.717
95%CI(D)	2.832 - 3.506	0.588 - 0.713	0.553 - 18.492	0.373 - 6.237	0.303 - 18.047	0.424 - 10.309	0.631 - 0.814
N	7,521	79	11,156	5,322	8,159	7,292	17,454
95%CI(N)	6,760 - 8,367	72 - 87	1,929 - 64,510	1,302 - 21,757	1,057 - 62,957	1,479 - 35,964	15,362 - 19,831
%CV	5.4%	4.9%	88.7%	75.4%	109.4%	93.2%	6.5%
Steller sea lion, hauled out							
D	0	0	0.323	0	0	0.073	0.072
95%CI(D)	0	0	0.009 - 11.304	0	0	0.002 - 2.923	0.013 - 0.391
N	0	0	1,126	0	0	256	1,759
95%CI(N)	0	0	32 - 39,433	0	0	6 - 10,196	324 - 9,534
%CV	0	0	234.4%	0	0	474.3%	99.9%
Steller sea lion, in water							
D	0	0	0.261	0.43	0	0.271	0.175
95%CI(D)	0	0	0.013 - 5.410	0.022 - 8.249	0	0.015 - 4.804	0.101 - 0.301
N	0	0	910	1,499	0	946	4,260
95%CI(N)	0	0	44 - 18,874	78 - 28,777	0	53 - 16,760	2,472 - 7,341
%CV	0	0	155.6%	213.7%	0	240.4%	27.9%
Steller sea lion, total							
D	0	0	0.583	0.43	0	0.345	0.247
95%CI(D)	0	0	0.051 - 6.618	0.022 - 8.249	0	0.024 - 4.860	0.125 - 0.487
N	0	0	2,035	1,499	0	1,202	6,019
95%CI(N)	0	0	179 - 23,087	78 - 28,777	0	85 - 16,956	3,056 - 11,853
%CV	0	0	147.1%	213.7%	0	214.5%	35.3%
Elephant seal							
D	0	0	0	0.003	0	0.001	0.003
95%CI(D)	0	0	0	0.000 - 0.093	0	0.000 - 0.051	0.001 - 0.005
N	0	0	0	10	0	4	65
95%CI(N)	0	0	0	0 - 324	0	0 - 176	35 - 121
%CV	0	0	0	316.2%	0	469.0%	29.9%

Discussion

We would generally expect to reduce the uncertainty of our abundance estimates by collecting more data. This is certainly true for a constant population, but natural variability exist amongst species and sampling periods. In this section, we compare abundance estimates across years, seasons, and past estimates (Williams and Thomas, 2007) to evaluate shifts in the mean population sizes and confidence intervals.

Compared with past estimates using 2004 and 2005 survey data (Williams and Thomas, 2007), the 95% confidence intervals for the average over the entire study area are in fact all narrowed with the addition of subsequent surveys from 2006 to 2008 (Figure 4). The coefficient of variation (CV) is a useful single measure of the confidence interval for comparison. In some cases means for the entire study area (Table 5, Table 7) are lower than earlier estimates as with the harbour porpoise (6,631 and 34.9% CV vs. 9,120 and 40.5% CV), fin whale (446 and 26.4% CV vs. 496 and 45.8% CV), and harbour seal (in-water) (10,394 and 6.5% CV vs. 13,524 and 15.3% CV). The rest are higher as with Dall's porpoise (6,232 and 20.0% CV vs. 4,913 and 29.2% CV), Pacific white-sided dolphin (32,637 and 24.6% CV vs. 25,906 and 35.3% CV), humpback whale (1,541 and 12.9% CV vs. 1,313 and 27.5% CV), killer whale (308 and 38.2% CV vs. 161 and 67.4% CV), minke whale (430 and 25.2% CV vs. 388 and 26.8% CV), harbour seal (haul-out) (7,060 and 12.9% CV vs. 5,852 and 25.9% CV).

Per Equation 1, abundance estimates are a product of school size ($\hat{s}$), the inverse of the detection probability ($\hat{p}$), and the encounter rate ($\frac{n}{L}$), all of which were updated with the additional surveys. The greatest difference in mean abundance between estimates is with the most populous of species, the Pacific white-sided dolphin (32,637 / 25,906 = 1.26%). The detection probability ($\hat{p}$) is much lower than earlier estimates (0.344 vs. 0.551), which increases the abundance. Williams and Thomas (2007) chose a half-normal detection function over the lower-AIC hazard rate model to avoid following the spike of data near zero distance, which assumes an attraction of the animals to the observer vessel. With the additional surveys, a half-normal detection function with automatic series expansion selects for a model with 4 cosine adjustment terms. Whereas this model was still preferable to the lower-AIC hazard rate model with the subsequent surveys of data for the same reason, it did more closely account for the spike near zero distance, resulting in a lower $\hat{p}$ and higher abundance. The estimated school size ($\hat{s}$) (Table 4) was also higher (13.53 and 14.77% CV vs. 12.49 and 17.79% CV) which results in higher abundance. The other species similarly have a difference in detection probability and/or estimated school size incongruent with the shift in overall abundance. Harbour porpoise is an exception which has a slightly lower $\hat{p}$ (0.201 vs. 0.212) in contrast with a lower abundance (6,631 vs. 9,120), but congruous with a lower estimated school size $\hat{s}$ (1.67 vs. 1.79). We can also infer from the mean abundance estimates in Table 5 for Stratum 1 (*N*) and the corresponding survey effort (*L*) in Table 1, which provides a weighting, that the overall encounter rate ($\frac{n}{L}$) was also lowered by the zero harbour porpoises seen in the summer of 2006 (*N*=0 and *L*=327nm) and far fewer in the 2007 spring (*N*=1,032 and *L*=915nm) and fall (*N*=487 and *L*=485nm) seasons, with a rebound in

2004 (N=2,874 and L=903) and 2005 (N=5,677 and L=914) levels for the summer of 2008 (N=3,874 and L=914). The other two exceptions to differences in overall abundances that are not in the same direction as the differences in detection probability and/or estimated school size are with the slightly less abundant fin whale (446 vs. 496) and slightly more abundant humpback whale (1,541 vs. 1,313). These minor differences can again be attributed to fewer sightings in subsequent surveys for fin whales (eg. zero in 2006), and more with humpback whales (eg. N> 1,000 for 2007 and 2008 vs. N < 1,000 for 2004 and 2005).

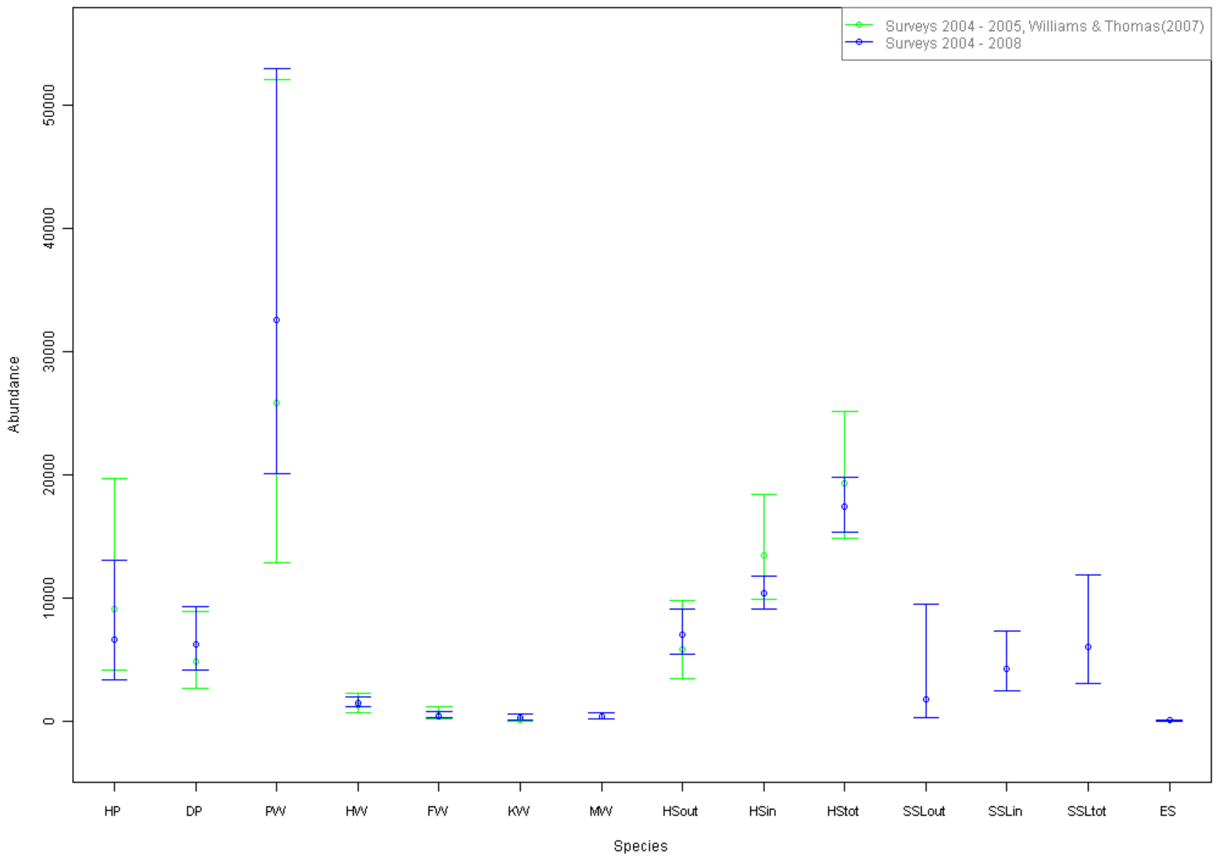

Figure 4. Abundance estimate comparisons between Williams & Thomas (2007) from the 2004-2005 surveys with the updated 2004-2008 survey data pooled across all strata and seasons. Note that with more data (eg 2004-2008), the confidence intervals consistently shrink. Species abbreviations are for harbour porpoise (HP), Dall's porpoise (DP), Pacific white-sided dolphin (PW), killer whale (KW), humpback whale (HW), common minke whale (MW), fin whale (FW), harbour seal (HS), Steller sea lion (SSL) and elephant seal (ES). Further abbreviations for pinnipeds denote haul-out (out), in-water (in) and total combined (tot).

Detection of changes in population requires long-term data, especially relative to the lifespan of the animal (Taylor et al. 2007). Based on plotting the abundances from Tables 5 through 8 in Figure 5, we see that almost none of the surveys have non-overlapping confidence intervals. This suggests no significant population changes between these sampling periods. When zero animals are seen, a confidence interval is not calculated.

The only clearly non-overlapping confidence interval is with humpback whales which are lowest in summer 2006 (486 and 95% CI 219 – 1,081) and highest the next survey season spring 2007 (2,431 and 95% CI 1,577 – 3,747). This could be due to a seasonal difference between summer and spring, or demarking an overall increase in the local population since summer of 2006. The next highest summer season survey which happened in 2008 was still higher (2,057 and 95% CI 1,382 – 3,062) than summer 2006. It is worth noting that summer 2006 had the least amount of realized survey effort at 605 km versus nearly 1,700 km for all other summer surveys (Table 1). More sophisticated methods exist for estimating trends using linear and spline models (S. T. Buckland et al. 2004, 71-91), but there must be sufficient data and suggested pattern to employ these. In the case of humpback whale, a simple linear trend is non-significant, either by summer surveys (p=0.276) or inclusive of 2007 fall and spring (p=0.204). Still, the mean abundance estimates are appreciably higher more recently in 2007 to 2008 versus the earlier period 2004 to 2006.

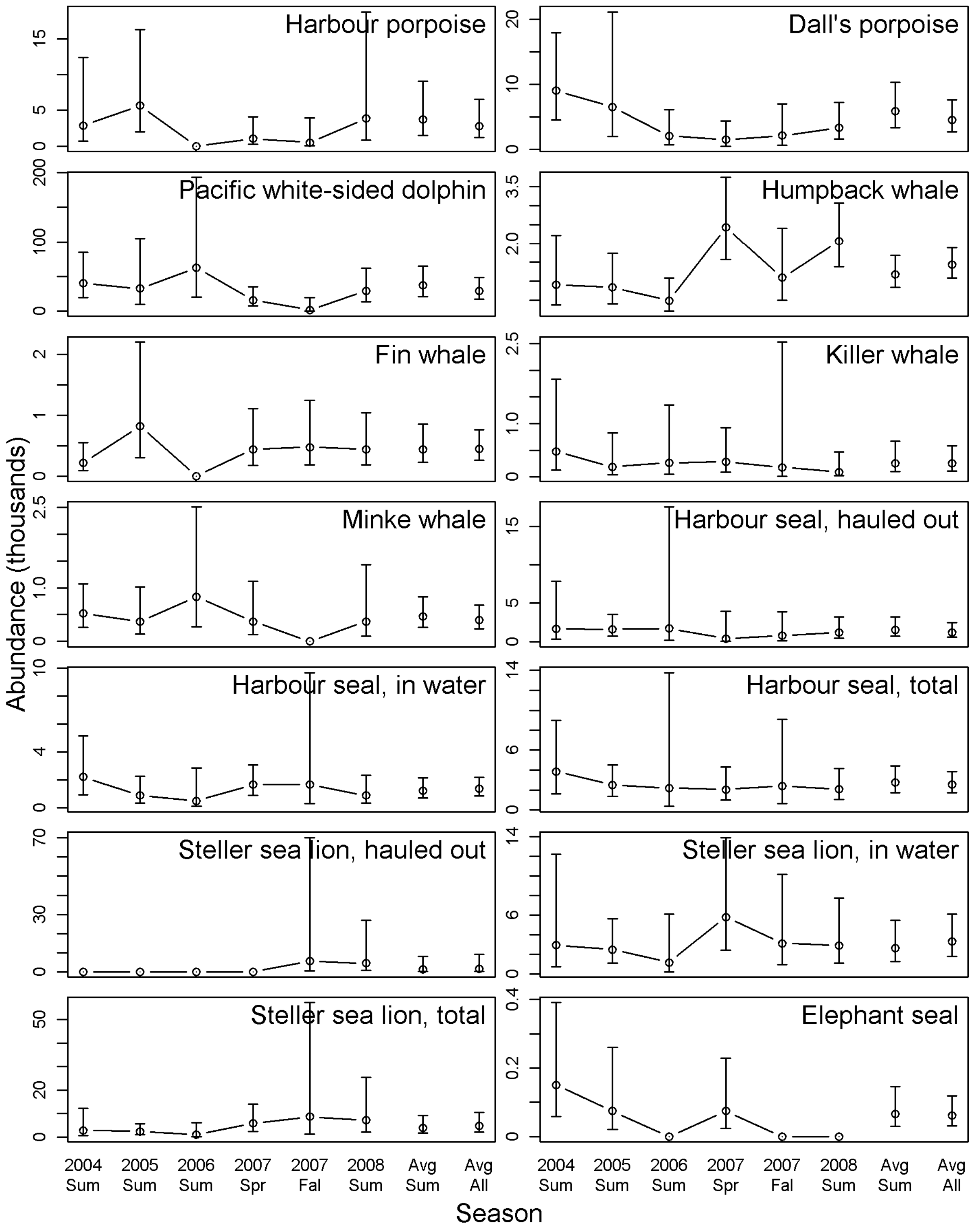

Figure 5. Abundance with 95% confidence intervals over surveyed years and seasons in Stratum 1, including the average of all seasons. Summer averages are included for seasonal comparison with 2007 fall and spring. No significant differences were found across seasons or years per species.

Product 2. Using Density Surface Models and Identifying Hotspots

The conventional distance sampling (CDS) estimates of abundance from the previous section assume a homogenous density of animals across the study area. Rather than assuming an even density, desnity can be modeled to vary spatially by explicitly linking environmental predictors. This technique has been described as "density surface modeling" (S. T. Buckland et al. 2004). More specifically, we use generalized additive models to fit the environment to the observations. It can improve the precision on the final estimate (De Segura et al. 2007), include on-effort data when off the designed transect, and can help identify hotspots of density important for spatially managing natural resources.

Methods

Transects were segmented into 1 nautical mile (1852 m) segments which were then associated with underlying environmental data. The response variable in this analysis is the estimated number of schools encountered per segment *i*, $\hat{N}_i$, given by the Horvitz-Thompson estimator (Horvitz and Thompson, 1952):

$$\hat{N}_i = \sum_{j=1}^{n_i} \frac{1}{\hat{p}_{ij}}, \quad i = 1,\ldots,v \quad (2)$$

where the inverse of the detection probability for the *j*th detected school in the *i*th segment is summed across all detected schools, n_i, per segment. These data were then merged with segments without sightings (N=0). Equipped with this response data, we then fit a generalized additive model (GAM) using a logarithmic link function to relate *N* to the environmental predictor variables (the environmetal predictor variables are described in the next section):

$$\hat{N}_i = \exp\left[\alpha + \sum_{k=1}^{q} s_k(z_{ik}) + \log(a_i)\right] + e_i \quad (3)$$

where the predictor variables, z_{ik} are fitted by a smoothing function s_k, and then are summed with an intercept β_0 and an offset a_i, which represents the segment's area ($2wL_i$). We used the software DISTANCE 6.0 Beta with the R 2.8.0 statistics package for this analysis. The estimation of the smoothing functions was performed by the R library `mgcv` (Wood , 2001).

Once the model is fitted to the observed environmental conditions, a prediction was made over the entire study area based on a snapshot of the input environmental data (*z*). Thus far, the response $\hat{N}$ is the number of schools detected over the area, or the school density. To obtain an estimate of abundance ($\hat{A}$), we must then multiply by the estimated school size ($\hat{s}$).

Variance on this estimate is calculated with the Delta method (Seber, 1982) to combine the variance of the school density ($CV(\hat{N})$) with that of the detection function ($CV(\hat{p})$) and the mean school size ($CV(\hat{s})$).

$$\hat{A} = \sum_{i=1}^{n} \hat{N}_i \hat{s}$$
$$CV(\hat{A}) = \sqrt{CV(\hat{p})^2 + CV(\hat{N})^2 + CV(\hat{s})^2} \qquad (4)$$

To estimate variance of $\hat{N}$, the *Distance* software uses a moving block bootstrap resampling technique. Even for only 400 replicates, this technique can be very time consuming and often failed before reaching completion. Instead, we generated bootstraps using a multivariate normal sampler on the Bayesian posterior covariance matrix. This method is described by Wood & Augustin (2002) and also in the R documentation for the `predict.gam` function (see example code using `Vp` of the GAM model object).

The set of covariates used in the final model is selected to explain the greatest deviance while minimizing unnecessary addition of parameters. Many criteria exist that variously weight these two factors against each other such as Akaike information criteria (AIC) or Bayesian information criteria (BIC). For GAMs that have a dispersion term, as with the quasipoisson response dispersion used in these models, the lowest generalized cross-validation GCV value is the preferred model selection tool (Wood 2008). The number of knots, or allowed wiggles, are further reduced in most of these models by using the non-default thin-plate spline with smoothing (ts), which adds a small penalty to additional knots, so that the whole term can be shrunk to zero, removing any contribution from the predictor. Term plots were inspected and any terms whose confidence bounds spanned 0 were then removed to allow the process to test for a model with a still lower GCV score. Models would sometimes fail to converge using this approach. In this case, attempts were then made to limit the possible number of knots to 5, and to implement the default thin-plate spline (tp) without the shrinkage term.

In addition to environmental covariates, the longitude-latitude (lon,lat) bivariate term provides a spatial estimator which can identify geographic hotspots not accounted for by the other predictors. Categorical variables such as season (summer, fall, or spring) and inlet (in or out) were also tested using this approach.

Detection Functions

We could not use the same detection functions for estimating $\hat{p}$ as with the conventional distance sampling (CDS), because the density surface model (DSM) module in the software *Distance* is currently only compatible with the multiple covariate distance modeling (MCDS) engine which only allows for half-normal (hn) and hazard rate (hr) key functions. Based on the CDS analysis the same truncation distances were used with the half-normal key function, unless a hazard rate model was used. The logic of this process is that as the school size increases, the school should become easier to detect. We account for this by adding a covariate of size with detection function, as is possible with the MCDS engine and not with CDS approach. When using the size covariate was not possible, we used the CDS detection function instead.

Environmental Variables

The manipulation of spatial data was performed with ESRI ArcGIS 9.2 ArcInfo license with the Spatial Analyst toolbox (ESRI 2009). Midpoints of the transect segments were used to extract the values of the environmental layers, and then sampled for use in the generalized additive model. To predict across the seascape with the fitted model, a 5km prediction raster rectangular grid was generated using the NAD 83 BC Environment Albers projection. The raster grid was converted to a polygon vector layer and the cells were clipped to the coastline and strata areas. Area was calculated per cell to be used as the offset value during prediction. The centroid location of each cell was used to extract values from the environmental layers.

Static environmental variables included bathymetric depth, slope, and distance to shore. Latitude and longitude were also used, as separate variables and as a covarying term. Shoreline data was extracted from the Global Self-consistent, Hierarchical, High-resolution Shoreline (GSHHS) database (Wessel and Smith 1996) [1]. Bathymetry data was extracted from the SRTM30 Plus 30-arc second resolution dataset (J. J. Becker et al. 2009) [2]. Euclidean distance from shore and the local slope of the bathymetry surface were calculated in ArcGIS. The final static GIS layers were sampled into the 5km prediction grid and are depicted in Figure 35. Static covariate bathymetric depth at 5km grid resolution in meters.Figure 35 (depth), Figure 36 (slope), and Figure 37 (distcoast).

The marine environment is highly dynamic, requiring us to capture this variability over the survey periods to build more temporally meaningful models. These models represent proxies for physiological or biological constraints (e.g. sea-surface temperature) and foraging patterns (e.g. primary productivity) of the species. However, attempts to incorporate dynamic variables, such as sea surface temperature (SST) and Chlorophyll-a (Chl_a), into the predictive model proved unsuccessful. Due to continuous cloud cover and close proximity to shore, sufficient satellite data matched to the specific observation periods of this analysis were not available for this study. So seasonally averaged approaches were attempted instead.

A principal goal in this study was to predict abundance and density over the entire study area. In addition we also set out to compare the abundance estimates for the given stratum between the sum of the density surface modeling cells with the single abundance estimate from the conventional distance sampling method. To compensate for cells with missing data, several approaches were implemented: 1) kriging the data to fill in missing values with neighbours before fitting, 2) obtaining dynamic layers with less cloud cover and averaging them over the survey period, and 3) back-filling cells missing dynamic values with prediction from a model using just the static variables. First, the error with the kriging on monthly values was high, making predicted values unrealistic, especially because kriging ignores the influence of land around points and inlets. Some attempts to incorporate wrapping around land using "soap film smoothing" (Wood et al. 2008) may in the future make this more feasible. Secondly, geostationary satellites have fewer no-data values over polar-orbiting satellites due to more

[1] http://www.ngdc.noaa.gov/mgg/shorelines/gshhs.html
[2] http://topex.ucsd.edu/marine_topo/

constant temporal coverage, so the NOAA Geostationary-orbiting Operational Environmental Spacecraft (GOES) satellite data were used for SST. The polar-orbiting NASA MODIS/Aqua satellite data supplied Chl_a. All satellite monthly data were extracted with custom Python scripts using the OPeNDAP protocol from the web services provided by Ocean Watch Live Access Server[3] from the Pacific Fisheries Environmental Laboratory. However, because of cloud cover issues and proximity to land, many no-data values persisted (more so with Chl_a than SST). Therefore, seasonal averages for each year were calculated. The seasons correspond to the months used over the range of all surveys: spring (April-May), summer (June-September), and fall (October-November) . Finally, by accepting the missing data (averaged but not kriged) as input to fit the model, model convergence became more difficult with less data. Even if the fitted model converged, the prediction grid often contained values outside the range of the originally fitted data making prediction and variance calculations untenable. Models inclusive of dynamic variables were attempted for all species, but in the end the static variables proved to have more explanatory power in the final model selection.

Results

Final models selected are shown in Table 9. In the case of harbour seal (in-water), separate models had to be fitted for stratum 1 and the other strata 2,3,4. The Humpback whale model had to be fitted with separate models for stratum 1 and stratum 4, excluding strata 2 and 3 where no observations were made. For Steller sea lion (haul-out), the log terms of depth and slope had to be used to get a fitted model.

Table 9. Generalized Additive Model (GAM) formulation with truncation distances (w) and distance model type (Detection) for given stratum with generalized cross-validation score (GCV), deviance explained and GAM model terms.

*** terms were limited to 5 knots**

† terms were used with default thin-plate (tp) without smoothing (ts)

Species Name	*w*	Detection	Strata	GCV	Dev. Expl.	GAM Model Terms
Harbour porpoise	600	hr	1-4	0.486	25.6%	lon,lat + depth + distcoast + slope + season + inlet
Dall's porpoise	700	hn	1-4	0.345	19.1%	lon,lat + depth + distcoast + slope + season + inlet
Pacific white-sided dolphin	1200	hn	1-4	0.377	33.4%	lon,lat + depth + distcoast + slope + season + inlet
Humpback whale	2300	hn	1	0.521	19.0%	lon,lat + depth + distcoast + slope + season
			4	0.265	11.0%	depth† + distcoast† + slope†
Fin whale	3900	hn	1-4	0.145	41.8%	lon,lat + depth + distcoast + slope
Killer whale (res + trans)	1300	hn	1-4	0.04	39.5%	lon,lat†
Minke whale	400	hn	1-4	0.045	32.5%	lon,lat + depth + distcoast + slope
Harbour seal (haul-out)	700	hn	1-4	0.188	35.6%	lon,lat + depth + distcoast + slope + season + inlet
Harbour seal (in-water)	500	hn	1	0.157	43.4%	lon,lat + depth + distcoast + slope + season
			2,3,4	1.366	23.9%	lon,lat + depth + distcoast + slope + inlet
Steller sea lion (haul-out)	1300	hn	1-4	0.025	26.5%	log(depth)* + distcoast* + log(slope)*
Steller sea lion (in-water)	500	hn	1-4	0.062	47.1%	lon,lat + depth + distcoast + slope + inlet
Elephant seal	500	hn	1-4	0.018	51.8%	lon,lat + depth + distcoast + slope + season

[3] http://las.pfeg.noaa.gov/oceanWatch

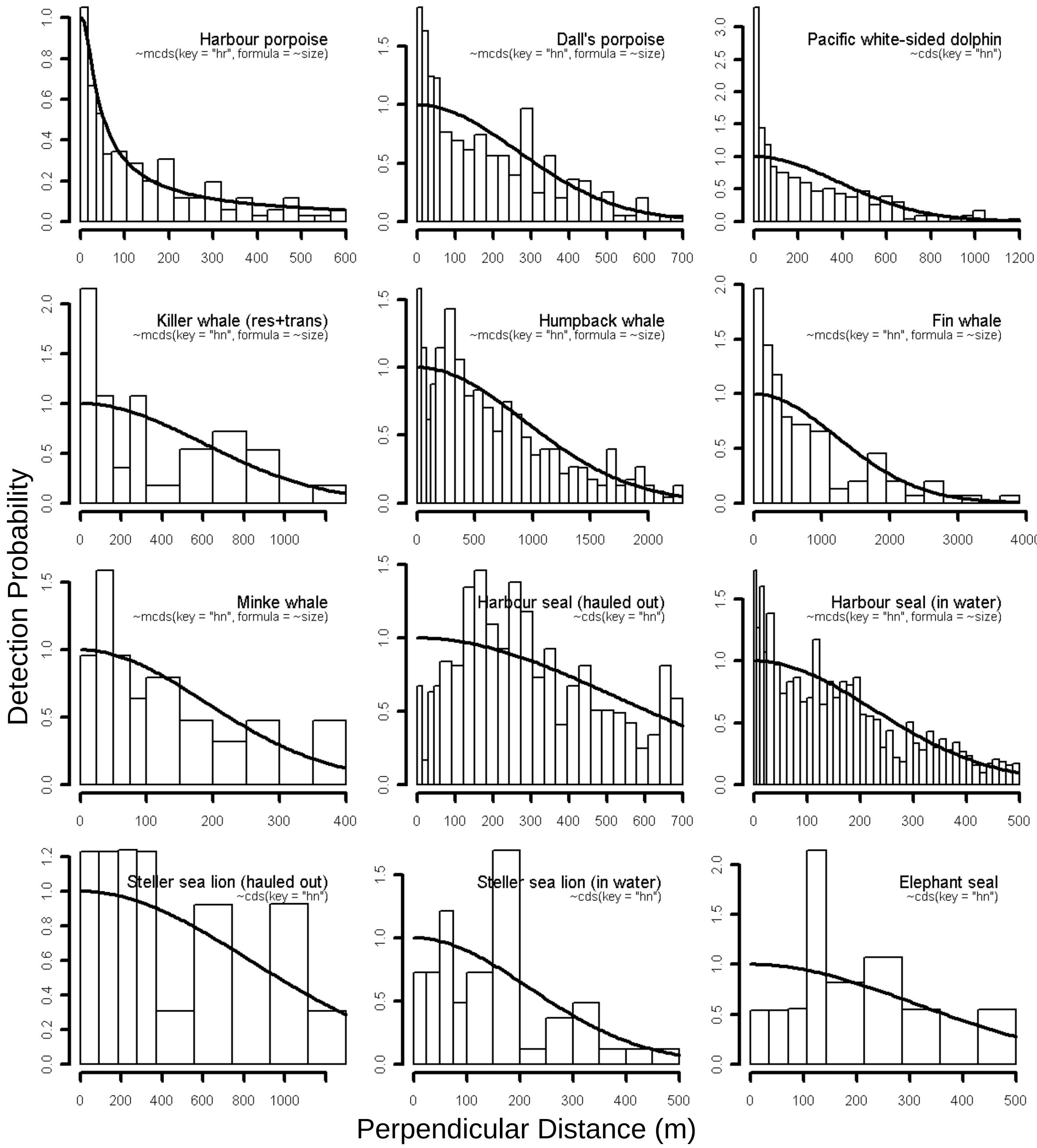

Figure 8. Average detection probabilities for density surface modeling (DSM) using the Multiple Covariate Distance Sampling (mcds) engine with the covariate size where possible, otherwise using Conventional Distance Sampling (cds) without a covariate. The detection function uses either a half-normal("hn") or hazard rate ("hr") key function.

Abundance Estimates

With the fitted model, a prediction is made over the entire region based on the environmental values of the cells. For models using the categorical variable of season, abundances for the summer season are presented below. The inlet categorical variable allows stratum 4 to differ from the other strata by a fitted coefficient. Abundance estimates were summed across all cells within each strata and the entire region for Cetacea (Table 10) and pinnipeds (Table 11). A graphical summary of the abundance estimates is presented in Figure 9, along with the previous conventional distance sampling estimates for comparison.

Table 10. Cetacean abundance estimates derived from the density surface models.

	Strata				Entire Region
Estimate	1	2	3	4	Average
Harbour porpoise					
D	0.058	0.368	0.21	0.11	0.097
95%CI(D)	0.035 - 0.096	0.222 - 0.610	0.127 - 0.350	0.066 - 0.183	0.058 - 0.160
N	3,647	3,053	92	1,298	8,091
95%CI(N)	2,202 - 6,041	1,843 - 5,058	55 - 153	783 - 2,153	4,885 - 13,401
%CV	26.2%	26.2%	26.4%	26.2%	26.2%
Dall's porpoise					
D	0.067	0.063	0.17	0.041	0.063
95%CI(D)	0.059 - 0.077	0.055 - 0.072	0.143 - 0.203	0.035 - 0.046	0.055 - 0.073
N	4,232	518	75	478	5,303
95%CI(N)	3,701 - 4,839	452 - 595	63 - 89	418 - 548	4,638 - 6,064
%CV	6.9%	7.0%	9.0%	7.0%	6.8%
Pacific white-sided dolphin					
D	0.313	0.001	2.704	0.106	0.265
95%CI(D)	0.233 - 0.419	0.000 - 0.002	1.996 - 3.664	0.079 - 0.144	0.198 - 0.356
N	19,715	7	1,183	1,256	22,160
95%CI(N)	14,699 - 26,441	3 - 18	873 - 1,603	931 - 1,693	16,522 - 29,721
%CV	15.1%	52.6%	15.6%	15.3%	15.1%
Humpback whale					
D	0.016			0.008	0.013
95%CI(D)	0.014 - 0.017			0.007 - 0.009	0.012 - 0.014
N	995			97	1,092
95%CI(N)	905 - 1,094			87 - 107	993 - 1,200
%CV	4.8%			5.3%	4.8%
Fin whale					
D	0.005	0	0	0.001	0.004
95%CI(D)	0.004 - 0.006	0	0	0.001 - 0.002	0.003 - 0.005
N	314	0	0	15	329
95%CI(N)	262 - 377	0	0	11 - 19	274 - 395
%CV	9.3%	0	0	12.5%	9.3%
Killer whale					
D	0.004	0	0.118	0.005	0.004
95%CI(D)	0.003 - 0.007	0	0.071 - 0.199	0.003 - 0.008	0.003 - 0.007
N	264	0	52	55	371
95%CI(N)	158 - 442	0	31 - 87	33 - 93	222 - 621
%CV	26.7%	0	26.8%	27.2%	26.7%
Minke whale					
D	0.008	0.003	0	0	0.006
95%CI(D)	0.004 - 0.014	0.001 - 0.005	0	0.000 - 0.001	0.004 - 0.011
N	498	21	0	4	522
95%CI(N)	281 - 883	11 - 39	0	2 - 7	295 - 927
%CV	29.9%	32.3%	0	38.9%	29.9%

Table 11. Pinniped abundance estimates derived from the density surface models.

	Strata				Entire Region
Estimate	1	2	3	4	Average
Harbour seal, haul-out					
D	0.048	0.436	0.051	0.387	0.134
95%CI(D)	0.038 - 0.060	0.348 - 0.547	0.040 - 0.066	0.308 - 0.485	0.107 - 0.168
N	3,040	3,613	22	4,558	11,233
95%CI(N)	2,423 - 3,815	2,881 - 4,530	18 - 29	3,635 - 5,715	8,965 - 14,076
%CV	11.6%	11.6%	12.7%	11.6%	11.5%
Harbour seal, in-water					
D	0.018	0.441	0.352	0.741	0.164
95%CI(D)	0.017 - 0.019	0.413 - 0.471	0.304 - 0.407	0.680 - 0.807	0.152 - 0.176
N	1,141	3,652	154	8,736	13,683
95%CI(N)	1,068 - 1,219	3,420 - 3,900	133 - 178	8,017 - 9,520	12,734 - 14,703
%CV	3.4%	3.4%	7.5%	4.4%	3.7%
Harbour seal, total					
D	0.066	0.877	0.403	1.128	0.298
95%CI(D)	0.052 - 0.084	0.693 - 1.110	0.302 - 0.537	0.885 - 1.436	0.235 - 0.378
N	4,181	7,265	176	13,294	24,916
95%CI(N)	3,301 - 5,296	5,740 - 9,195	132 - 235	10,439 - 16,930	19,666 - 31,569
%CV	12.1%	12.1%	14.8%	12.4%	12.1%
Steller sea lion, haul-out					
D	0.042	0.128	0.023	0.023	0.048
95%CI(D)	0.012 - 0.147	0.037 - 0.442	0.007 - 0.081	0.007 - 0.080	0.014 - 0.166
N	2,673	1,057	10	273	4,014
95%CI(N)	771 - 9,262	305 - 3,664	3 - 36	79 - 948	1,158 - 13,908
%CV	70.3%	70.4%	70.4%	70.3%	70.3%
Steller sea lion, in-water					
D	0.0003	0	0	0.0004	0.0003
95%CI(D)	0.000 - 0.000	0	0	0.000 - 0.001	0.000 - 0.000
N	19	0	0	4	23
95%CI(N)	12 - 30	0	0	3 - 7	15 - 37
%CV	24.1%	0	0	24.4%	24.0%
Steller sea lion, total					
D	0.043	0.128	0.023	0.024	0.048
95%CI(D)	0.012 - 0.157	0.035 - 0.468	0.006 - 0.086	0.006 - 0.087	0.013 - 0.177
N	2,692	1,057	10	278	4,037
95%CI(N)	733 - 9,882	288 - 3,876	3 - 38	76 - 1,021	1,100 - 14,815
%CV	74.4%	74.3%	74.3%	74.5%	74.3%
Elephant seal					
D	7E-05	5E-06	0	0.0004	0.0001
95%CI(D)	0.000 - 0.000	0.000 - 0.014	0	0.000 - 0.065	0.000 - 0.015
N	5	0	0	4	9
95%CI(N)	3 - 7	0 - 116	0	0 - 770	0 - 1,248
%CV	22.2%	411476.5%	0	3497.3%	2452.4%

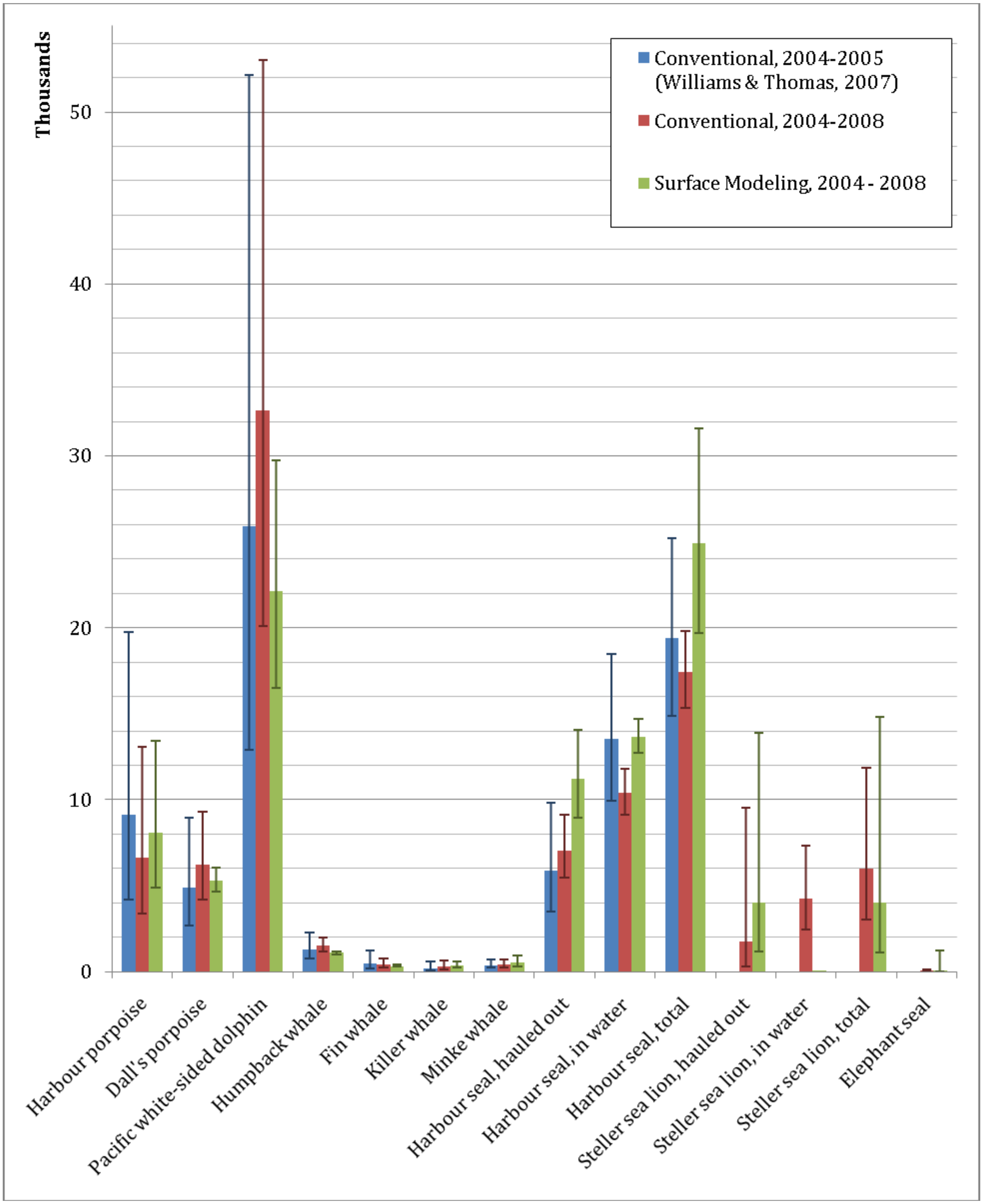

Figure 9. Comparison of abundance estimates for the entire region between conventional distance sampling from 2004 to 2005 (Williams and Thomas, 2007), conventional with the additional survey years to 2008, and density surface modeling, with 95% CI error bars.

Discussion

Because density surface models can account for spatial heterogeneity over the strata, they can theoretically narrow the confidence interval on abundance estimates over the conventional distance sampling methods (Burt & Paxton 2006; De Segura et al. 2007; Hedley & Buckland 2004). Comparing the CVs between methods for the entire region (Figure 9) this holds true for almost all of the individually modeled species: harbour porpoise (26.2% *vs.* 34.95%), Dall's porpoise (6.8% *vs*. 20.0%), Pacific white-sided dolphin (15.1% *vs.* 24.6%), humpback whale (4.8% *vs.* 12.9%), fin whale (9.3% vs. 26.4%), killer whale (26.7% *vs.* 38.2%), harbour seal haul-out (11.5% *vs.* 12.9%), harbour seal in-water (3.7% *vs.* 6.5%), Steller sea lion haul-out (70.3% *vs.* 99.9%), Steller sea lion in-water (24.2% *vs.* 27.9%). Species modeled for which this was not the case are the minke whale (29.9% *vs.* 25.2%), and the elephant seal (2452.4% *vs*. 29.9%). The CV for the elephant seal is wildly high because of the high variance being divided by a very small mean value. Because so few observations were made (n=20) while so many more segments were zero, the density surface model is less reliable in the case of this most rare species in the survey.

The density surface models are most useful for identifying potential high-use areas or hotspots. Comparing the observations (Figure 2) we see general agreement with the distribution of the density surface models (Appendix 4. Density Surface Model Outputs). Where density is predicted to be high, so is the standard error. Much of the predictive power from the models is derived from the bivariate spatial location predictor (*i.e.* latitude, longitude). Harbour porpoise is distributed heavily in the southern strata and some northern areas of Queen Charlotte Basin near Prince Rupert. Because only encounter rate is spatially modeled, observations with larger groups are not more heavily weighted for the density surface. Dall's porpoise is most highly concentrated in the northeastern corner of the study region and the model influenced most positively by medium range depths (Figure 42). The Pacific white-sided dolphin dominates the southern and central portion of the basin, with another hotspot in Johnstone Strait (Figure 45). Distance to coast is a dominant term positively influencing density, offset by the negative contribution of depth and slope (Figure 47). Distribution of humpback whale (Figure 48) is positively influenced by distance to coast and less-so depth (Figure 50, Figure 51), most prominently found off the southern portion of Haida Gwaii island and up through the deep channel of the basin. Fin whales are also clustered at the southern portion of Haida Gwaii Island and the northernmost bit of the basin (Figure 52). Killer whales are found in coastal pockets in the south and central basin (Figure 55). For this species, only spatial location (latitude, longitude) was a selected predictor (Figure 57). Minke whale is spread throughout the basin on at a low density (Figure 58). Harbour seals haul-out are found most in the south central portion of the nearshore basin and inlet waters (Figure 61). In-water harbour seals (Figure 64) are also distributed nearshore and in the southern strata. Steller sea lions haul-out (Figure 67) and in-water (Figure 70) are also found nearshore, but more widely throughout the basin. Elephant seals are very thinly distributed throughout the basin, more so in the inlets (Figure 73).

Tier B. Composite Risk Map and Vessel Routing

Introduction

Reducing risk to endangered wildlife while maintaining human resource uses and maritime operations requires objective assessment of species habitats and human uses in both space and time. In order to better assess management options for separating endangered species and potentially harmful interactions we need to develop a synthetic, composite valuation of our marine species and then apply explicit optimization methods. A specific application of this type of approach is the development of cost-surface models based on expected species distributions that are then applied to optimize the routing of vessels in the region to potentially reduce the risk of adverse interactions.

One method to conduct this type of assessment is to conflate the marine mammal density surfaces into a single layer that will serve as a synthetic environmental cost surface for the analysis. To create this synthetic cost surface, individual species density surfaces were first relativized, and then weighted by conservation score before adding them up into a single cost surface layer. The cost surface was then used to determine alternate navigation routes for ships, which have the potential for striking animals or fouling habitat from potential spills. These methods employ least-cost path algorithms as a means to develop vessel paths that follow the most economical path through the environment while avoiding areas of high environmental risk.

While a wide variety of industries are increasingly active in the coastal waters of British Columbia, many environmental groups (PNCIMA, BCMCA, LOS) are seeking conservation-minded solutions for locating safely locating activities. The current ecological data layers in widest use are based on an expert feedback approach delineating important areas. For example, Figure 10 depicts the areas in the latest draft PNCIMA Atlas, originally provided by Clarke et al. (2006). The maps in these atlases identify areas of importance as polygons. These polygon areas are then overlaid and summed to create an index of potential importance and environmental sensitivity. In the absence of observational data, this qualitative approach is the best available science. Given the availability of Raincoast surveys, the density surfaces of each species can be combined to provide a more quantitative layer for planning purposes.

The ability to operationally provide a framework for minimizing impacts on marine animals is especially appealing. For example, ship traffic lanes have been re-routed in Boston Harbor to reduce likelihood of striking right whales (Russell 2001; Ward-Geiger et al. 2005; Fonnesbeck et al. 2008). Global data layers on human impacts in marine systems are being actively developed, including vessel traffic density (Halpern et al. 2008). Here we provide a simple framework for proposing alternative shipping routes to minimize impacts on marine animals. In this framework competing priorities, such as cost of additional travel distance and time versus risk of striking a marine mammal can be more objectively assessed.

Several large oil and gas projects that are currently underway are likely to increase heavy shipping into Kitimat (EnviroEmerg Consulting Services 2008), making this a useful example of the approach.

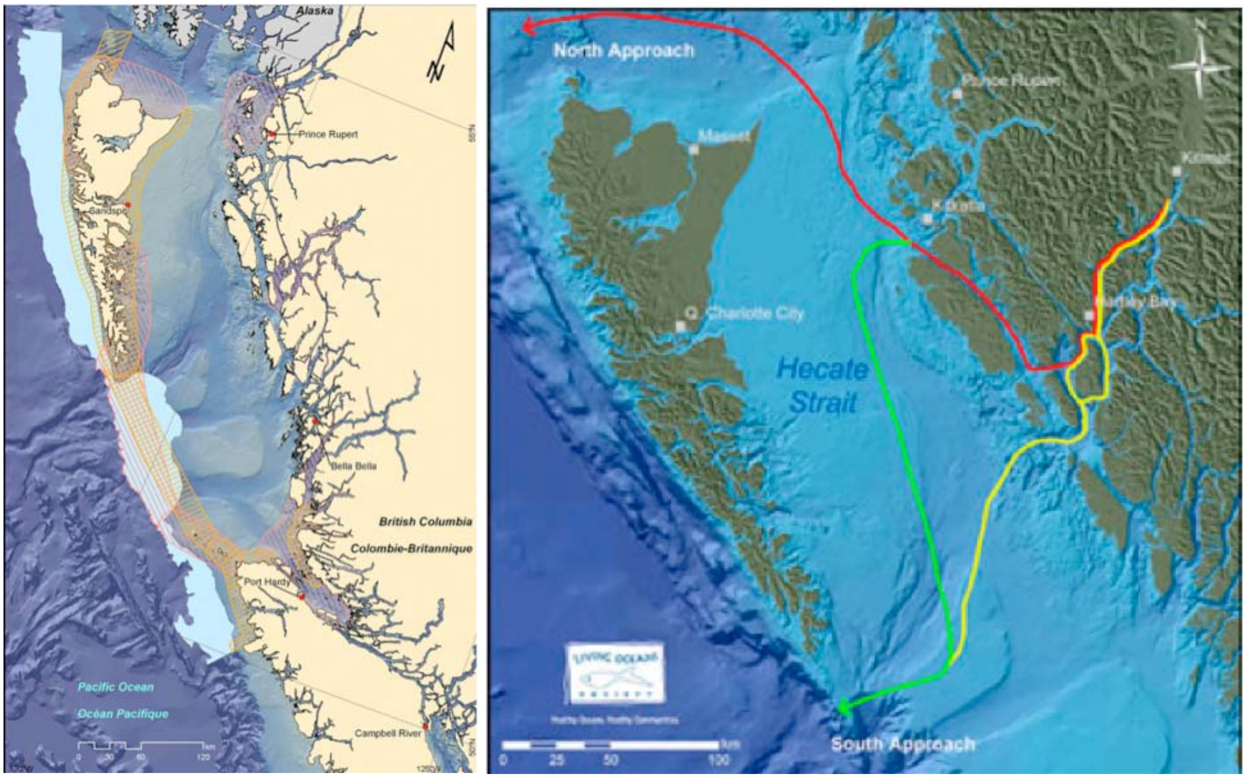

Figure 10. On the left, polygons of important areas for gray, humpback, and sperm whales derived from expert opinion in the PNCIMA Atlas (Draft 2009). On the right, proposed tanker vessel route for servicing the forthcoming Kitimat oil and gas projects (EnviroEmerg Consulting Services 2008).

Methods

Product 3. Composite Risk Map

Density surface model outputs (Appendix 4. Density Surface Model Outputs) were assembled for a marine mammal composite risk (cost-surface) map. Each density surface was normalized in order to highlight areas of high density relative to its average (i.e. the location denotes a relative hotspot). The unitless standard score, or z-value (z_i), per pixel ($_i$) is calculated as the pixel's marine mammal density estimate (x_i) subtracted from the mean of all density estimates for the strata (μ), divided by the standard deviation of those density estimates (σ) and finally multiplied by the species weight (w).

$$z_i = \frac{x_i - \mu}{\sigma} * w$$
$$Z_i = \sum_{j=1}^{n} z_i \qquad (4)$$

An inverse weighting scheme based on species conservation status was applied to favor representation of more endangered species (Louisa Wood and Dragicevic 2007). These rankings were obtained from the Provincial listing status at British Columbia's Endangered Species and Ecosystems website (http://www.env.gov.bc.ca/atrisk). Elephant seal is listed as SNA, species "not applicable", presumably because of its semi-migratory status in BC waters. Given that it's status is S4 in California and Alaska to the south and north of BC, this status was used to conform with the scheme in Figure 11. The values on the y-axis indicate the relativised weight used in the analysis.

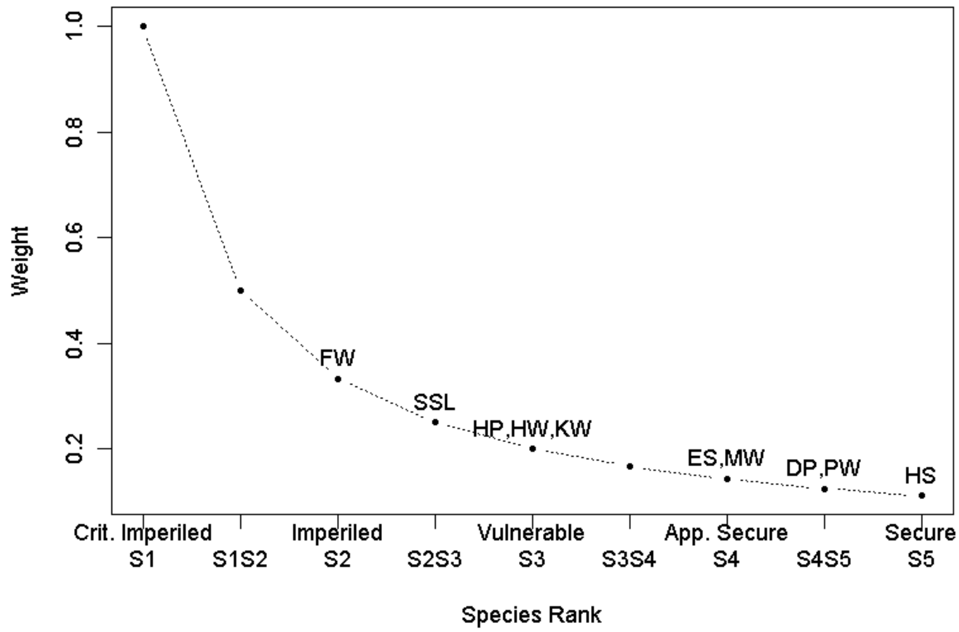

Figure 11. Weighting by BC conservation status for fin whale (FW), Steller sea lion (SSL), harbour porpoise (HP), humpback whale (HW), killer whale (KW), Elephant seal (ES), minke whale (MW), Dall's porpoise (DP), Pacific white-sided dolphin, and harbour seal (HS).

Product 4. Vessel Routing

To avoid encounters with marine mammals, relative hotspots of expected encounter are to be avoided, and routed around. Least-cost algorithms, such as Djikstra's algorithm, are commonly used with the prevalence of online driving directions and many other route-optimization applications. They have also been playing an increasing role in routing corridors of habitat and testing connectivity of habitat patches (Chetkiewicz, Clair, and Boyce 2006) for both terrestrial (Dean Urban and Keitt 2001) and marine applications (Treml et al. 2008).

The cost surface from the composite risk map provides the biological hotspot surface around which to route. The routing was performed with Python scripts using ESRI's ArcGIS ArcInfo version 9.3 with the Spatial Analyst toolbox. The `CostPath` function was used with input cost distance and back-directional raster grids generated from the `CostDistance` function. The 5km original density surface grids were resampled to a 1km resolution for use as the resistance cost surface to provide finer spatial resolution and routing within the inlets. An alternative raster grid in which all cells were assigned a cost value of 1 served as the Euclidean linear distance optimal spatial route providing a comparison of direct routing.

The proposed routes from Figure 10 were digitized and endpoints for north and south approaches used with the exercise to test the framework moving in and out of Kitimat. Routes between all navigation points, originally including other ports (Prince Rupert and Port Hardy), were also calculated. Existing routes may have preference for other factors than efficiency, such as scenic beauty or protection against inclement weather. Given that existing routes are generally preferred, a cost can be associated with movement away from these preferred routes. Here we take the case of cruise routes reported online[4]. Euclidean distance from existing cruise route was relativized by the maximum within the study area and multiplied by the maximum cost surface value. The two surfaces were added to obtain the final cost surface for routing, providing an example of equal weighting to conservation and routing goals.

Results

The composite risk map for all marine mammals (Figure 12) is very similarly distributed as just the large whales (Figure 13) composed of killer, humpback, fin and minke whale. Consistent with Clarke et al. (2006), Hecate Strait was found to be relatively important, along with the Dixon entrance. The highest value hotspot is in Gwaii Hanas Reserve, already under some level of protection. The high levels in stratum 3 Johnstone Strait (see zoom view) may be an anomaly of the environmental conditions there, since few sightings were made by comparison.

4 http://www.empr.gov.bc.ca/OG/offshoreoilandgas/OffshoreMapGallery/Pages/DownloadableShapeFiles.aspx

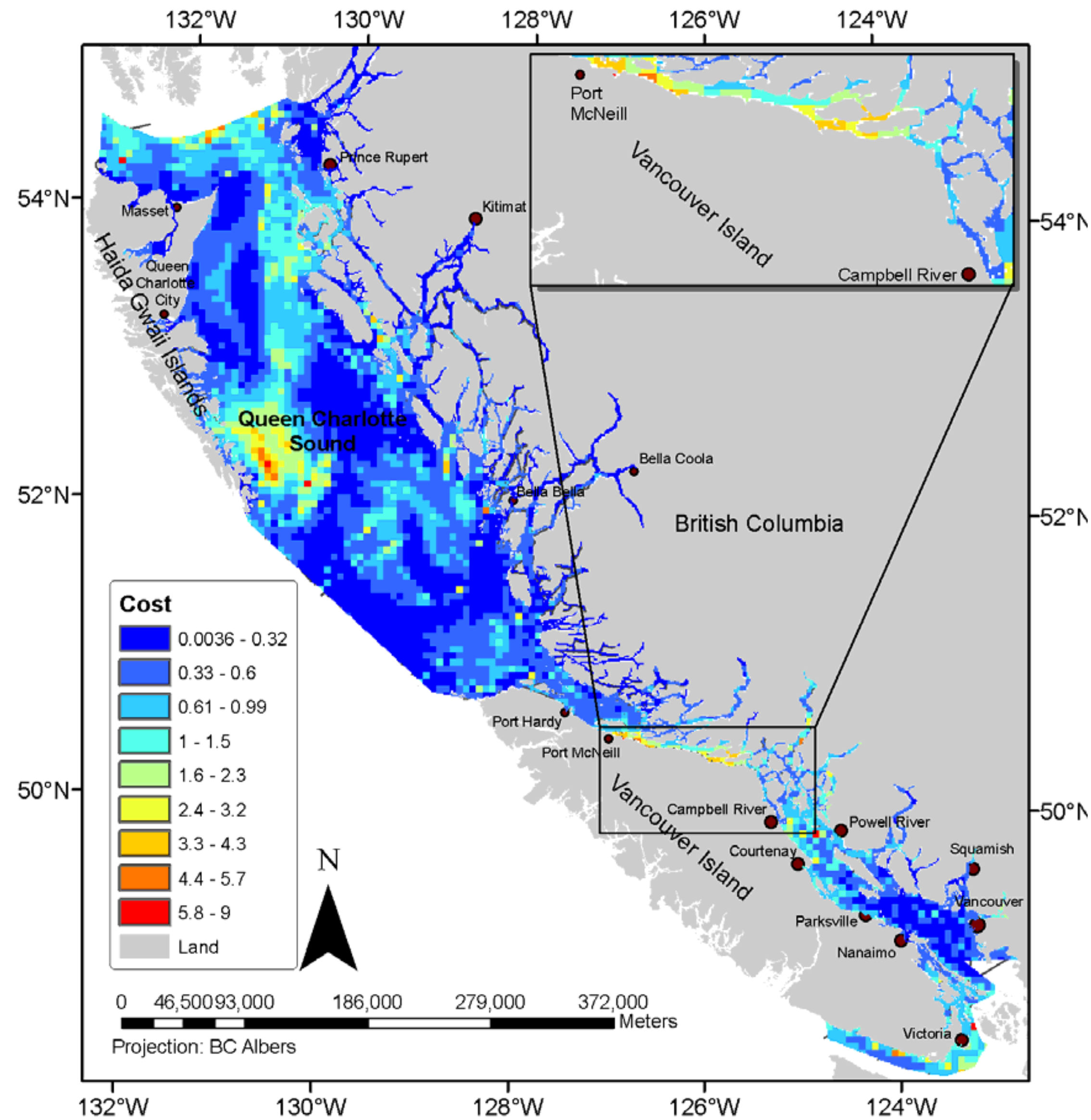

Figure 12. Conservation-weighted composite map of all marine mammal z-scored densities.

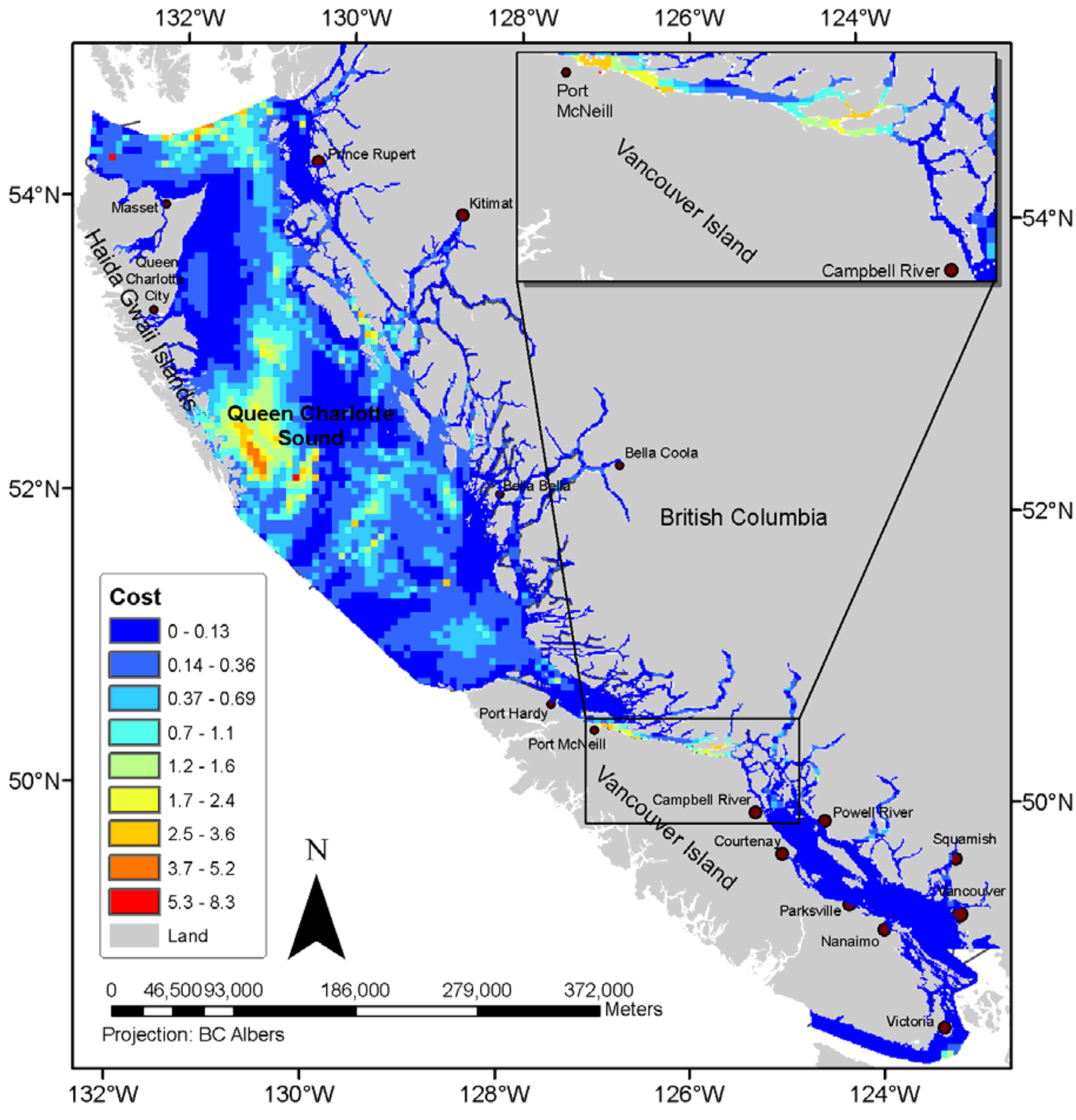

Figure 13. Conservation-weighted composite map of only large whales (killer, humpback, fin and minke) z-scored densities.

The routes do differ markedly (Figure 14). It is not clear whether Grenville Channel is deep and wide enough to support the kind of tanker traffic envisioned. Besides being closer to the northern approach, as evidenced by that inlet choice with the Euclidean path, Grenville Channel exhibits a lower level of potential marine mammal interaction as predicted by the composite marine mammal densities than through the Principe Channel as proposed. By routing through the Grenville Channel the potential interactions in the Hecate Strait could also be avoided. The proposed Southern approach exhibits relatively lower potential interactions with the least-cost route even dipping south around the Gwaii Haanas Reserve.

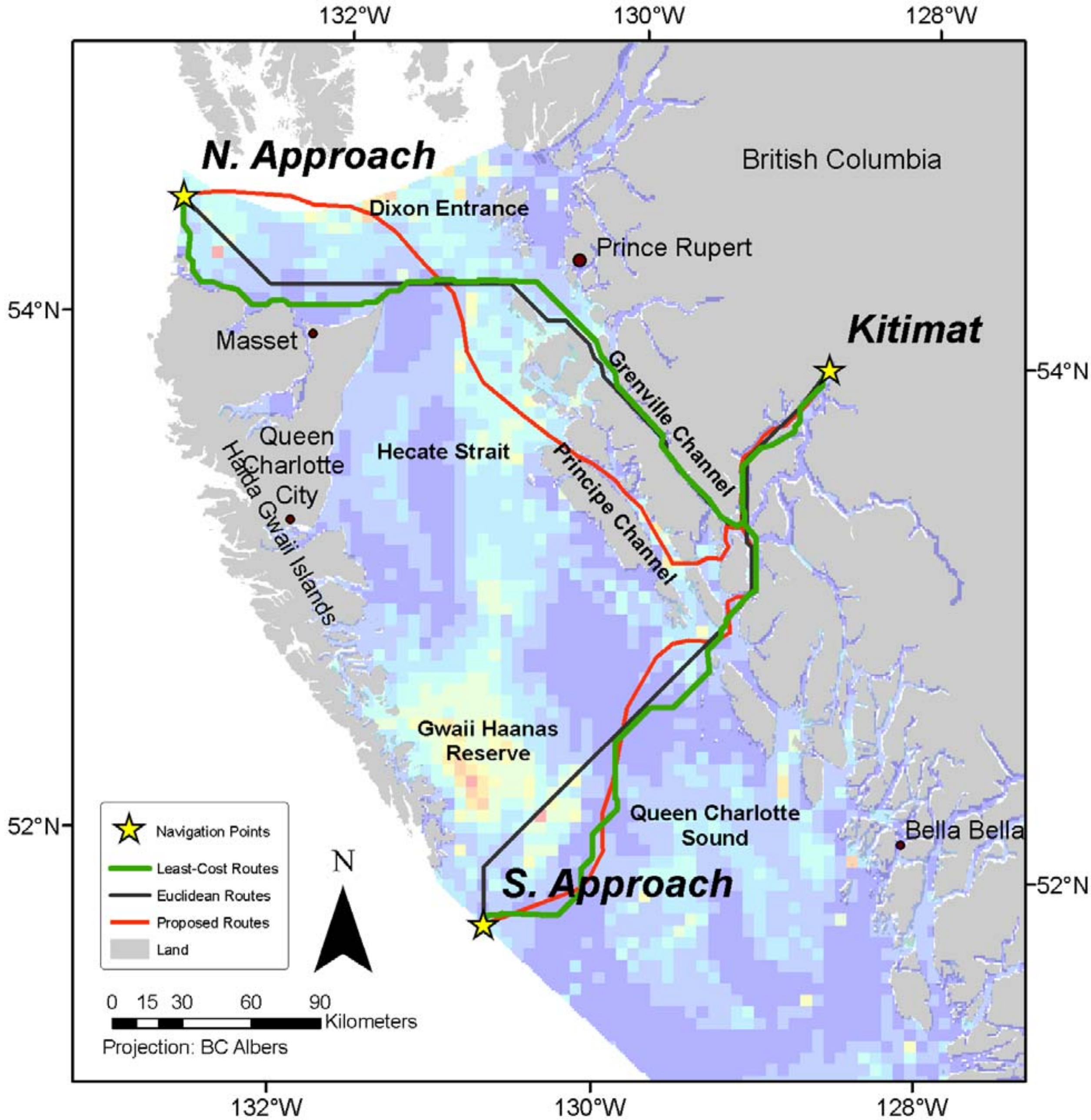

Figure 14. Proposed, linear (Euclidean), and least-cost routes for Kitimat. The least-cost route uses the conservation weighted cost surface from Figure 11 pictured here.

The existing ship routes pass through both the Principe and Grenville Channels. Models developed to conduct least-cost path analysis use raster grid cells. For modeling purposes these cells provide eight possible directions at each step (i.e. 4 side and 4 diagonal directions). These models create uneven turns when compared to the smoother Euclidean route or with the proposed routes (Figure 15). Although the Euclidean path should be the shortest, the existing industry route up the Principe Channel is the shortest (Table 12).

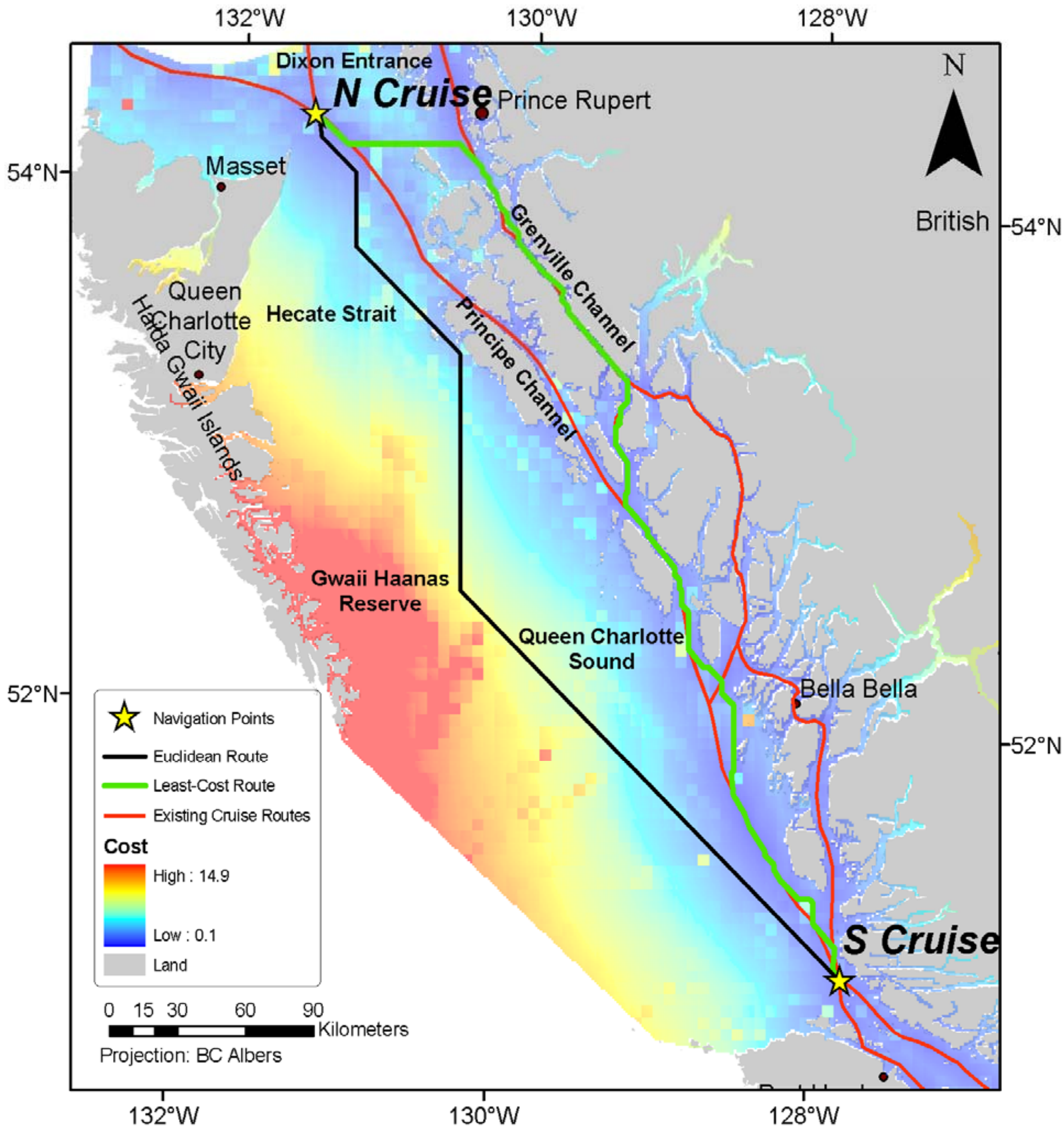

Figure 15. Existing, Euclidean and least-cost routes for cruise ships. The least-cost route uses a cost surface which is the sum of the conservation risk surface and a surface of distance from existing routes scaled to the equivalent range. The least cost-path is chosen which thus avoids biological hotspots while being equally attracted to existing routes.

Table 12. Distances of routes in km. For the oil transportation example in and out of the Kitimat port only the conservation surface is used for generating the least-cost route, compared to the industry-proposed route and Euclidean path. The cruise ship industry path comes from existing routes, the distance from which is summed with the conservation surface to provide the total cost surface with which to route the least-cost path. Since the cruise ship route splits, the distance for the one going up Principe Channel is shown.

Route	Euclidean	Industry	Least-Cost
Kitimat to N. Approach (Conservation)	391	410	425
Kitimat to S. Approach (Conservation)	304	319	336
S. to N. Cruise Route (Conservation + Distance to Route)	470	450	504

Discussion

The composite cost-surface and least-cost path modeling presented here is logically consistent with known hotspots in the region. For example, the highest environmental cost concentration on the composite map is already in an area with protection, the Gwaii Haanas Reserve. The Dixon entrance and a portion of the Hecate Strait also appear as relatively high-cost, hot-spots in this analysis. These areas may warrant further protections, as have already been noted (Clarke et al. 2006). Given further data collection and analysis, risk of encounter above a specified density threshold, could be determined using uncertainty in the underlying density surface model. Still, the general composite risk map provided here offers a useful and intuitive quantitative layer for marine spatial planning in British Columbia.

In order to further develop least cost-path routing and risk assessment, The least-cost routing analysis needs to be targeted towards more specific scenarios, such as risk to individual species and specific routes to be most relevant for planning. Given the expected increase in shipping traffic in the region, this general framework could become increasingly more useful to help develop shipping routes that provide environemtnal protection at the minimum additional cost. Additional criteria such as channel depth, channel width, navigational hazards, percent calm weather and other factors could also be input into this spatial planning process.

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Appendix 1. British Columbia Nautical Charts

The following nautical charts were gathered from Federal Publications Inc without express permission from their website http://www.fedpubs.com/charts/pac_general.htm, so are for internal use only.

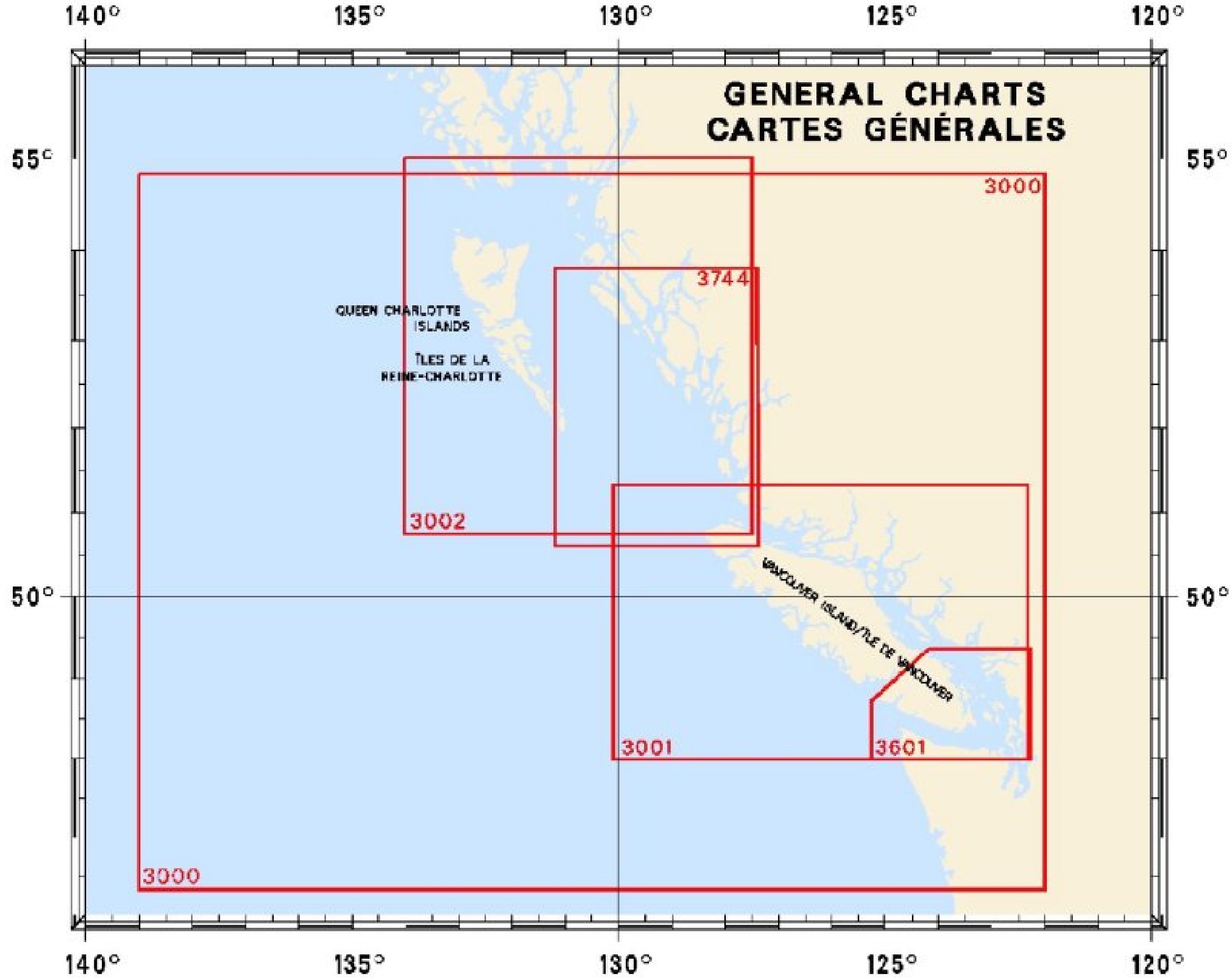

Figure 16. Nautical chart overview of B.C.

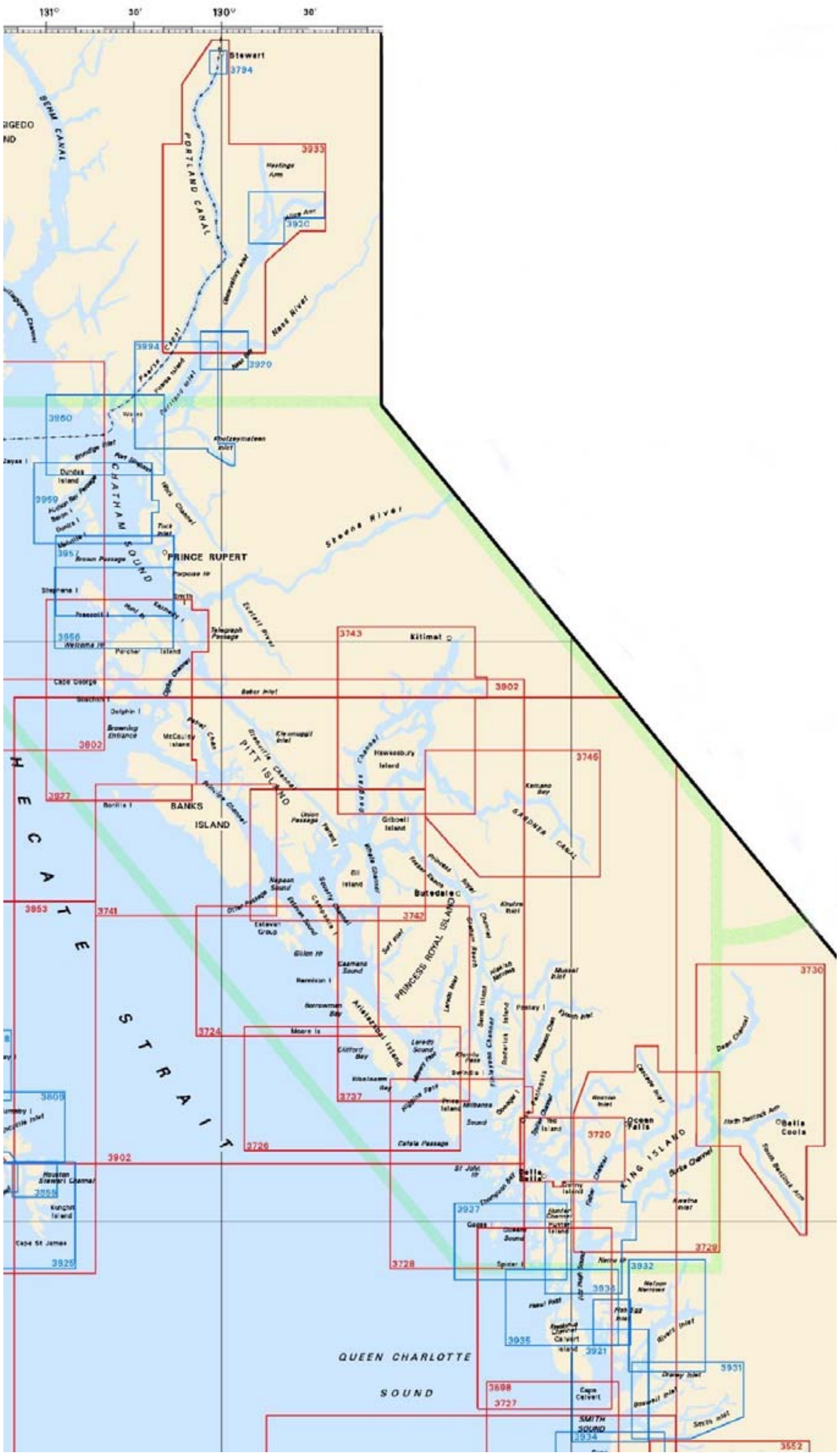

Figure 17. Nautical chart for North Coast of B.C.

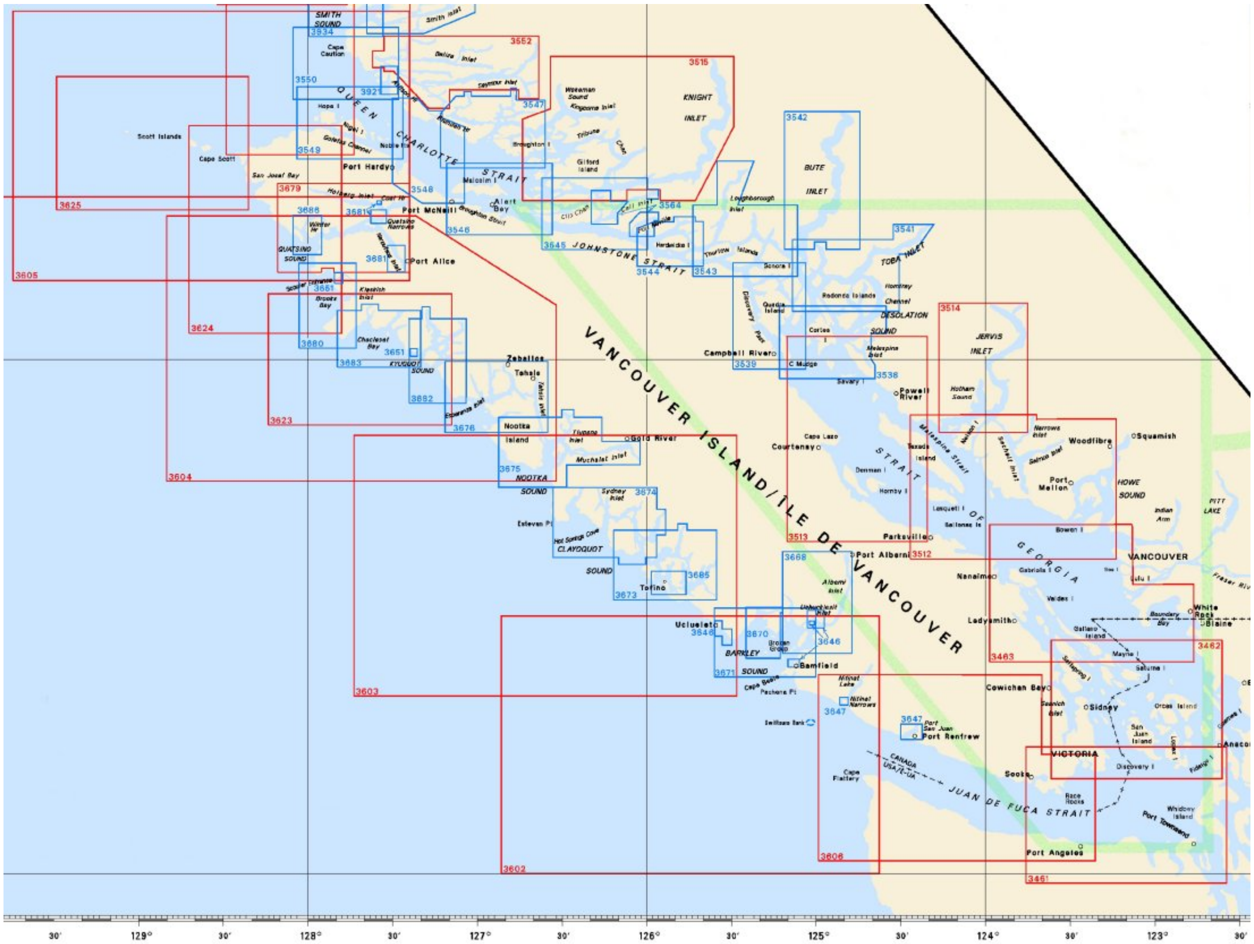

Figure 18. Nautical chart for South Coast of B.C.

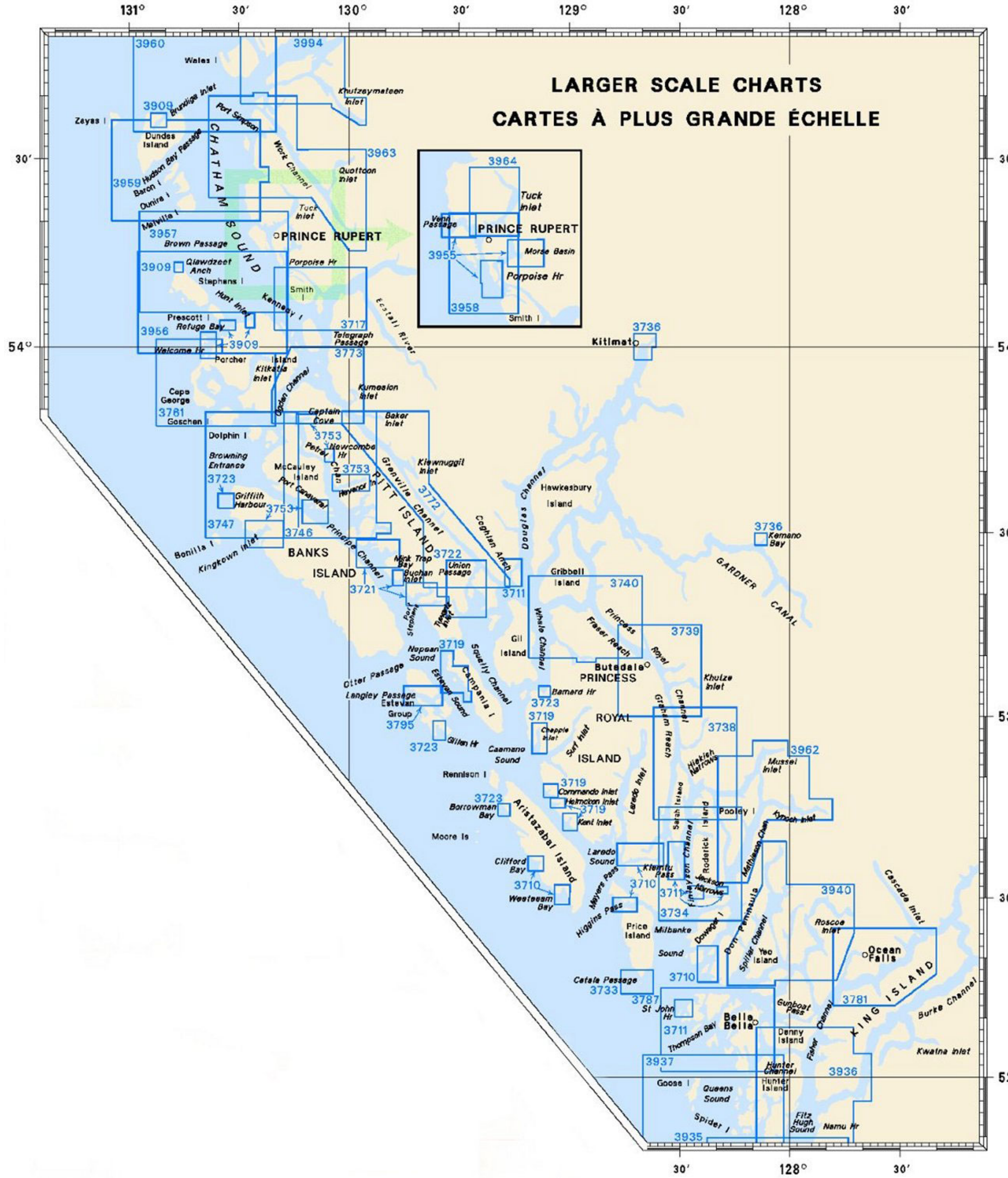

Figure 19. Nautical chart for Large Scale North B.C.

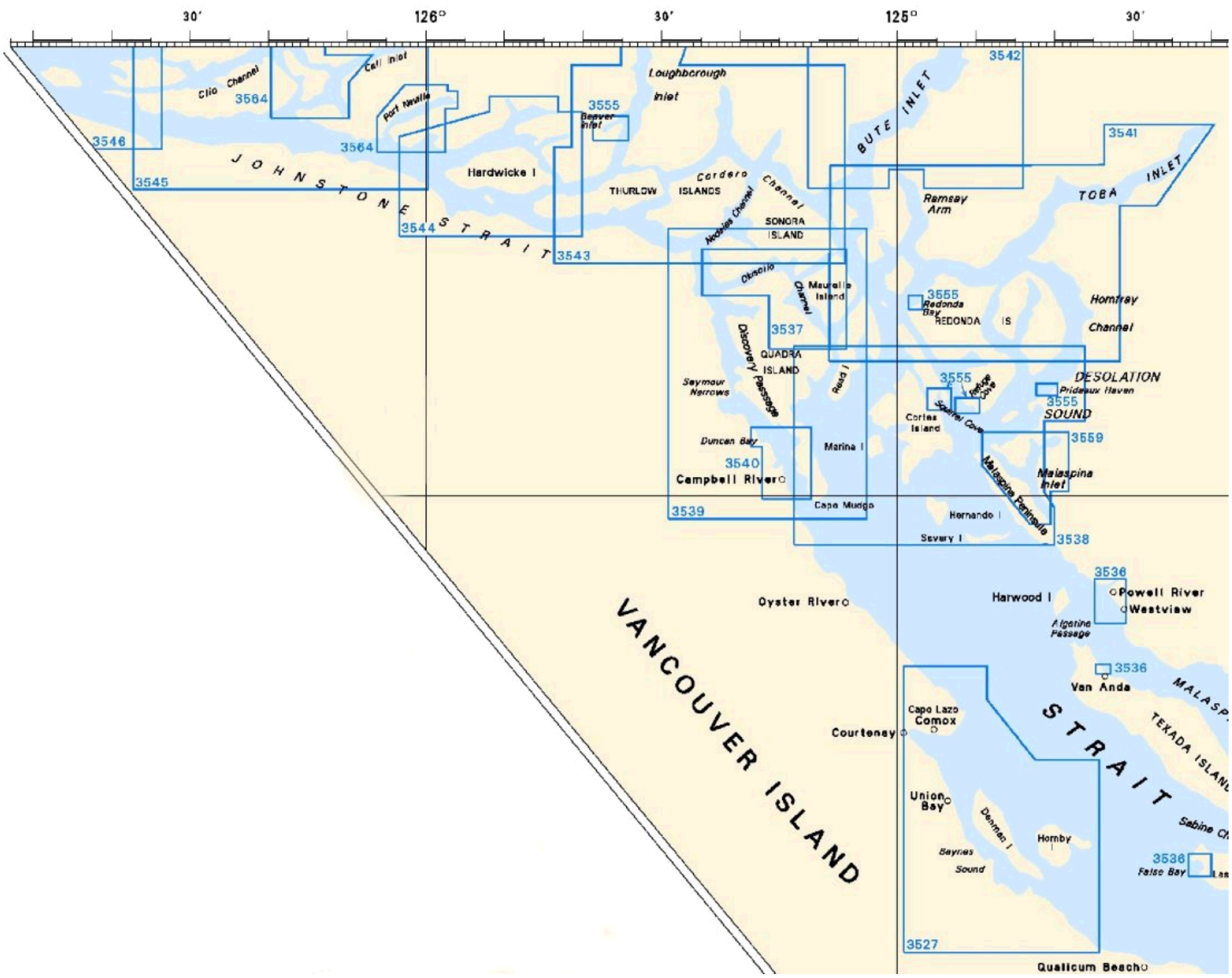

Figure 20. Nautical chart for Large Scale North Vancouver Island.

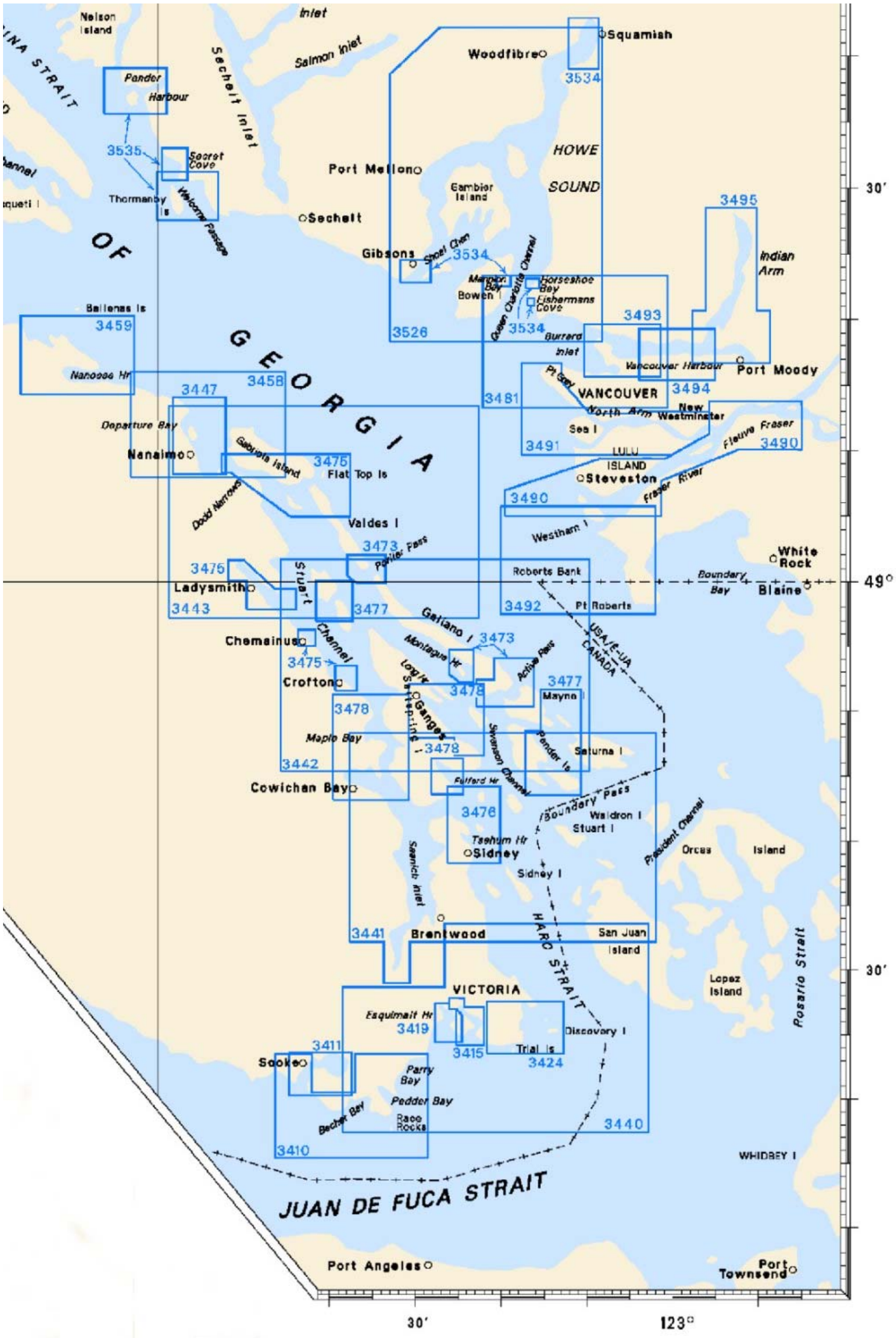

Figure 21. Nautical chart for Large Scale South Vancouver Island.

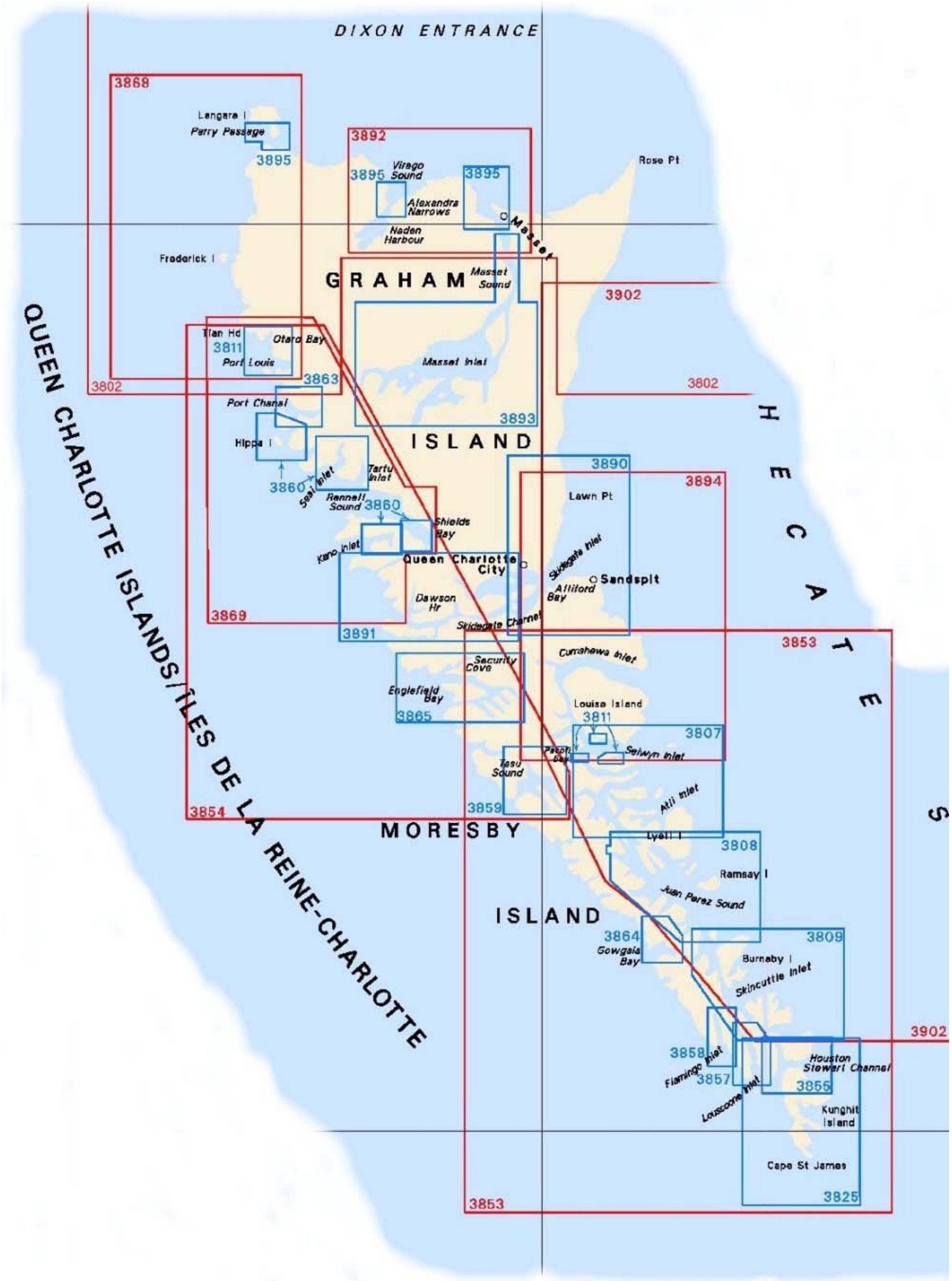

Figure 22. Nautical chart for the Queen Charlotte Islands.

Appendix 2. Maps of Observations

Harbour porpoise

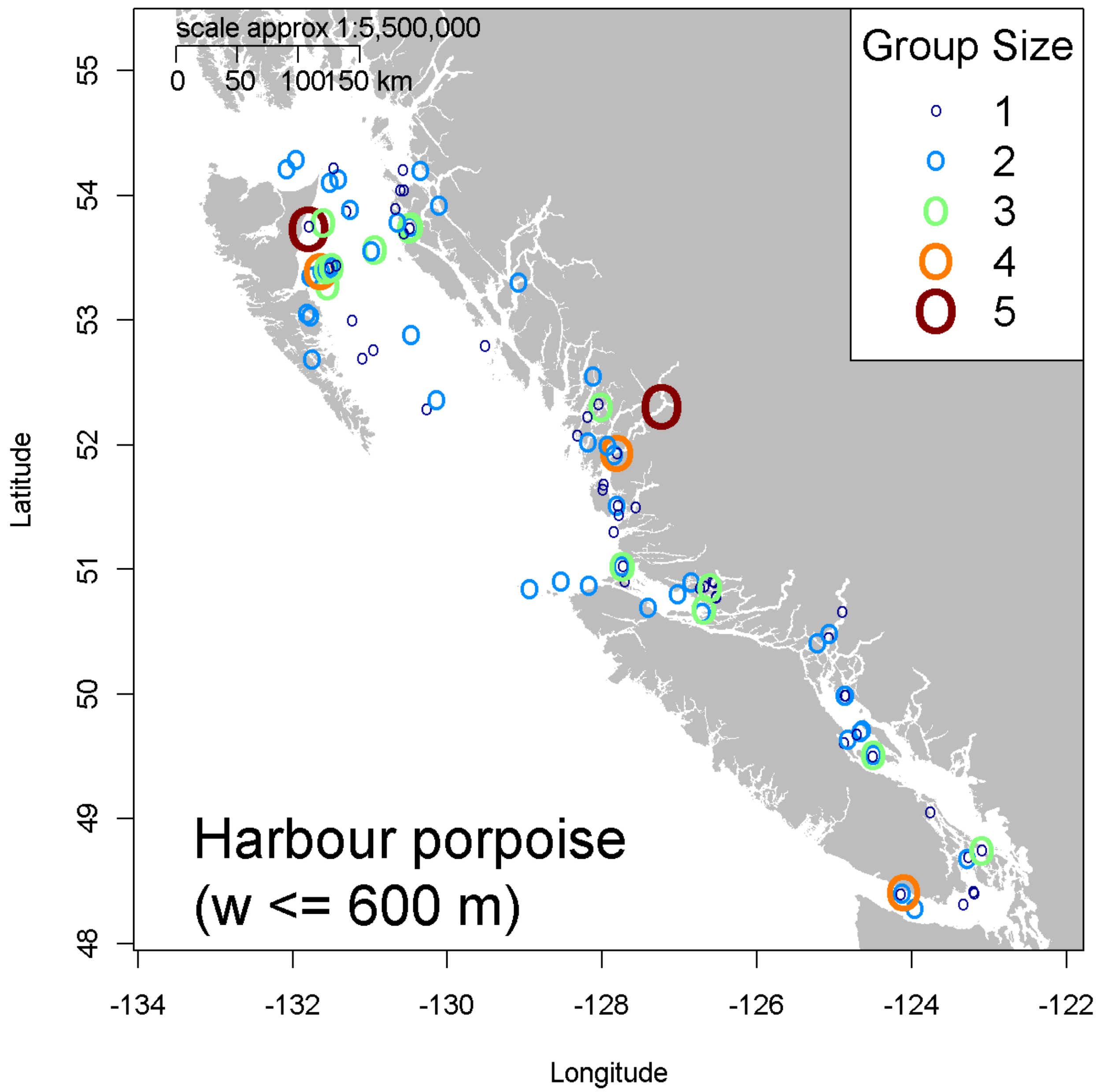

Figure 23. Observations of harbour porpoise by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 600m.

Dall’s porpoise

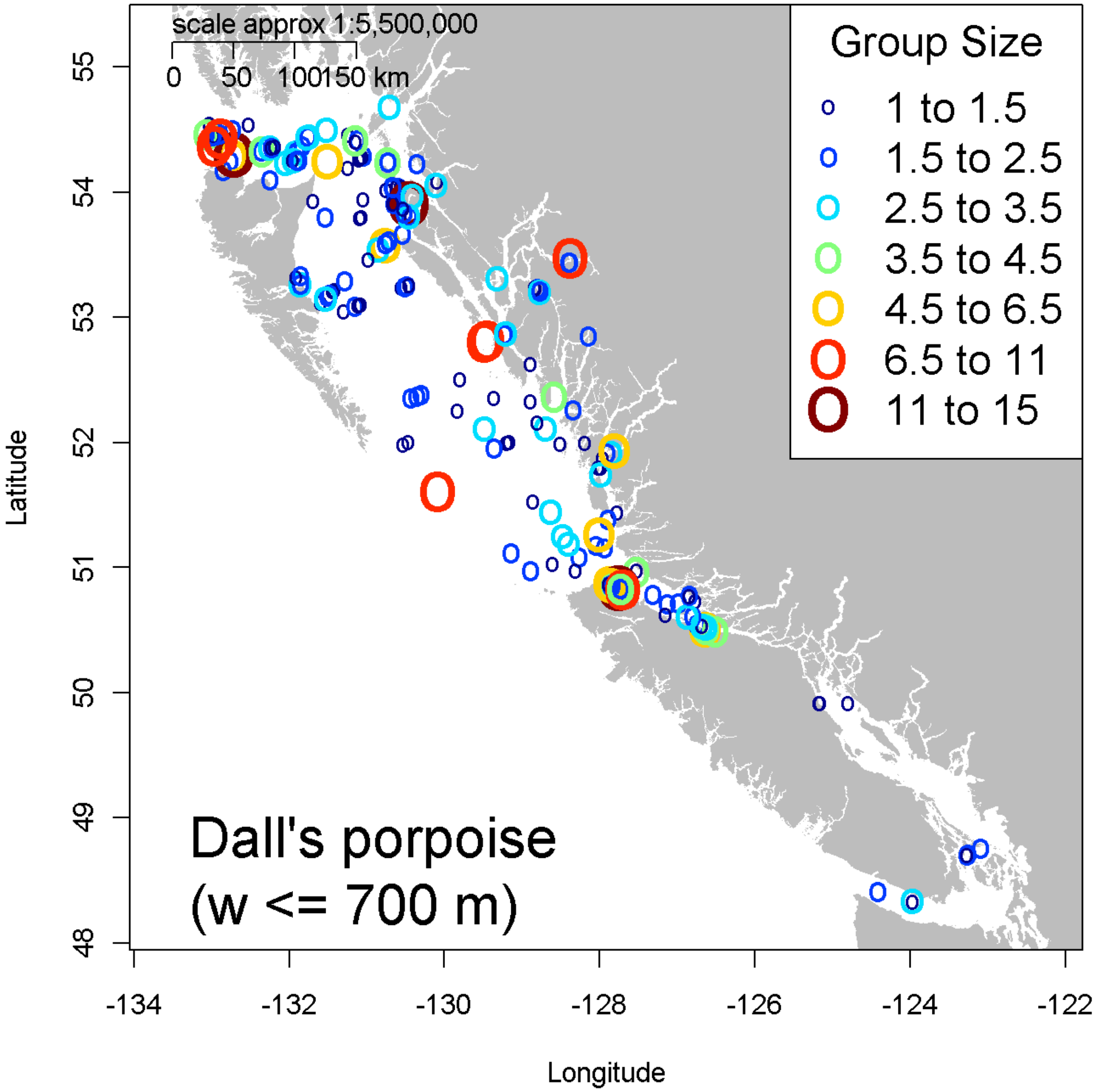

Figure 24. Observations of Dall’s porpoise by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 700m.

Pacific white-sided dolphin

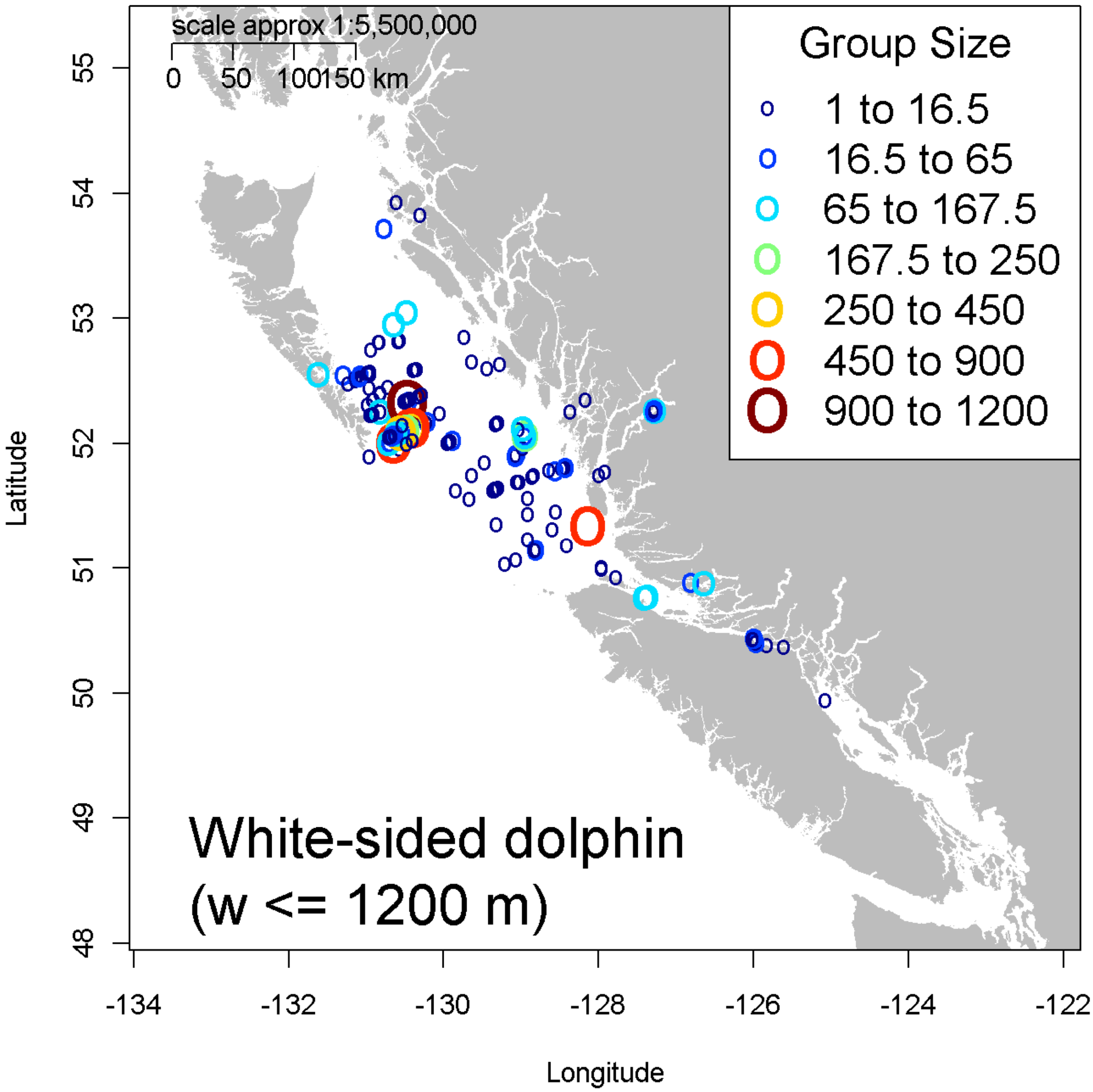

Figure 25. Observations of Pacific white-sided dolphin by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 1200m.

Humpback whale

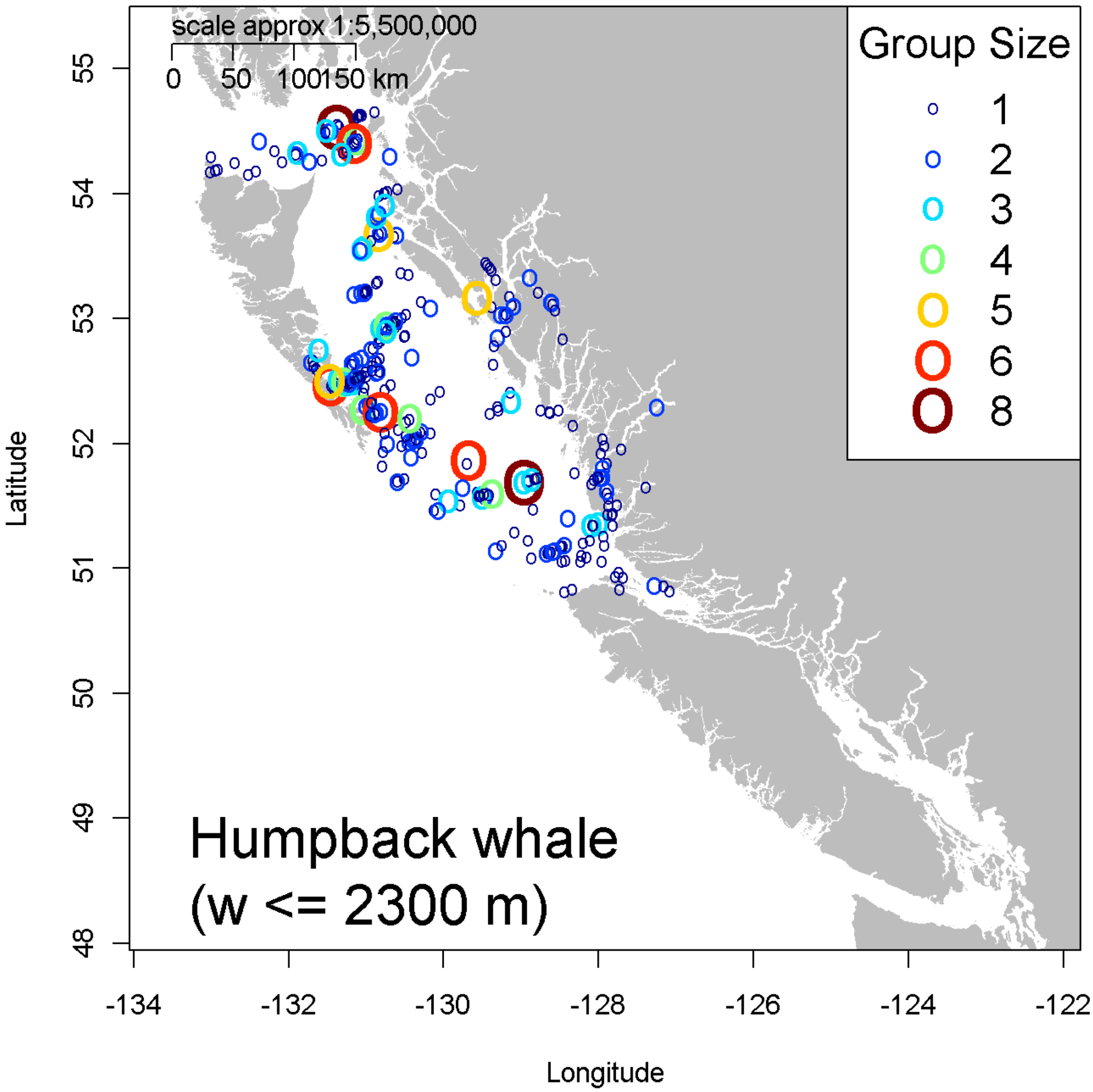

Figure 26. Observations of humpback whale by group size for all surveys (2004-2008), after truncating perpendicular distance (w) to within 2300m.

Fin whale

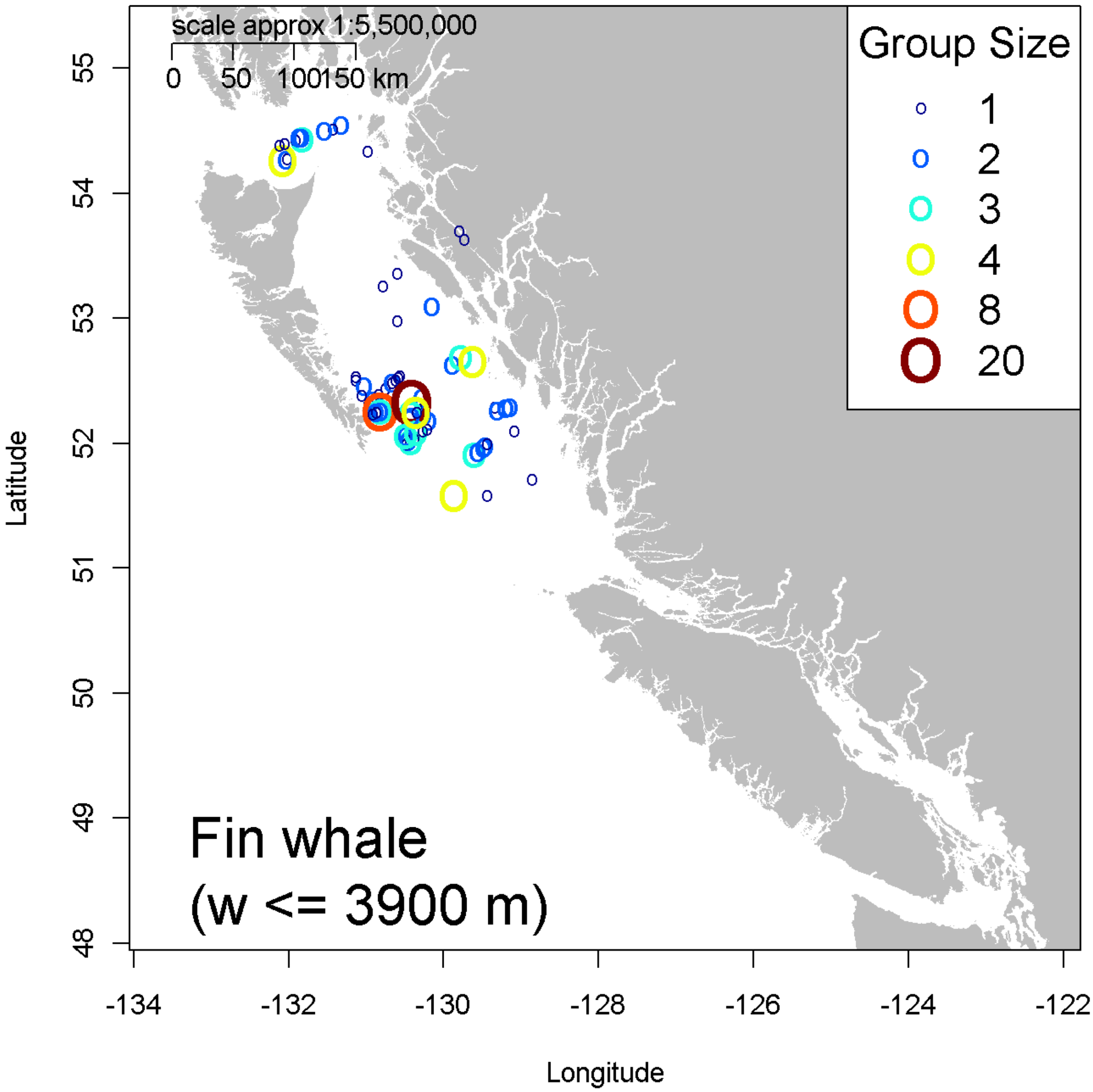

Figure 27. Observations of fin whale by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 3900m.

Killer whale

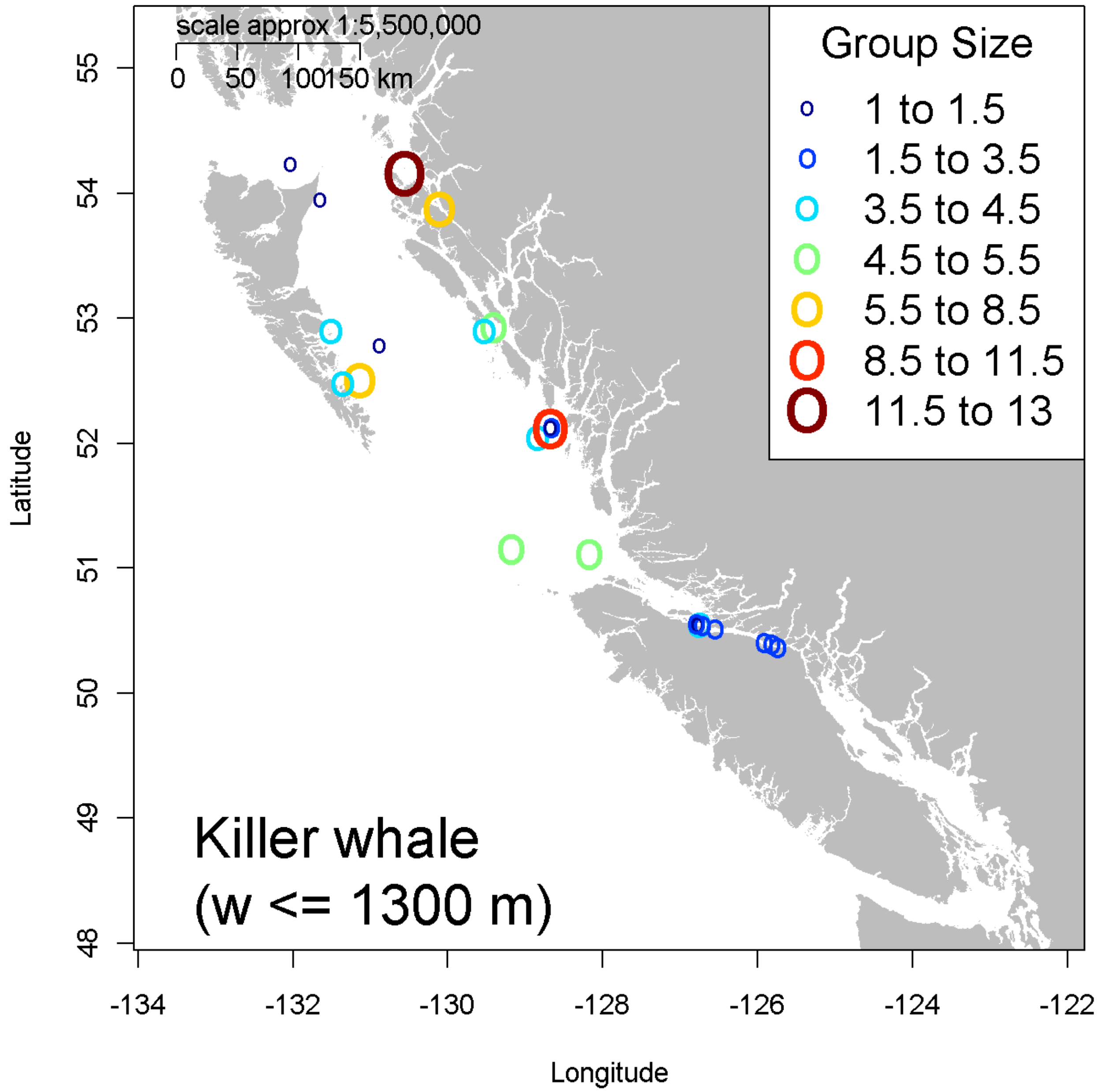

Figure 28. Observations of killer whale by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 1300m.

Minke whale

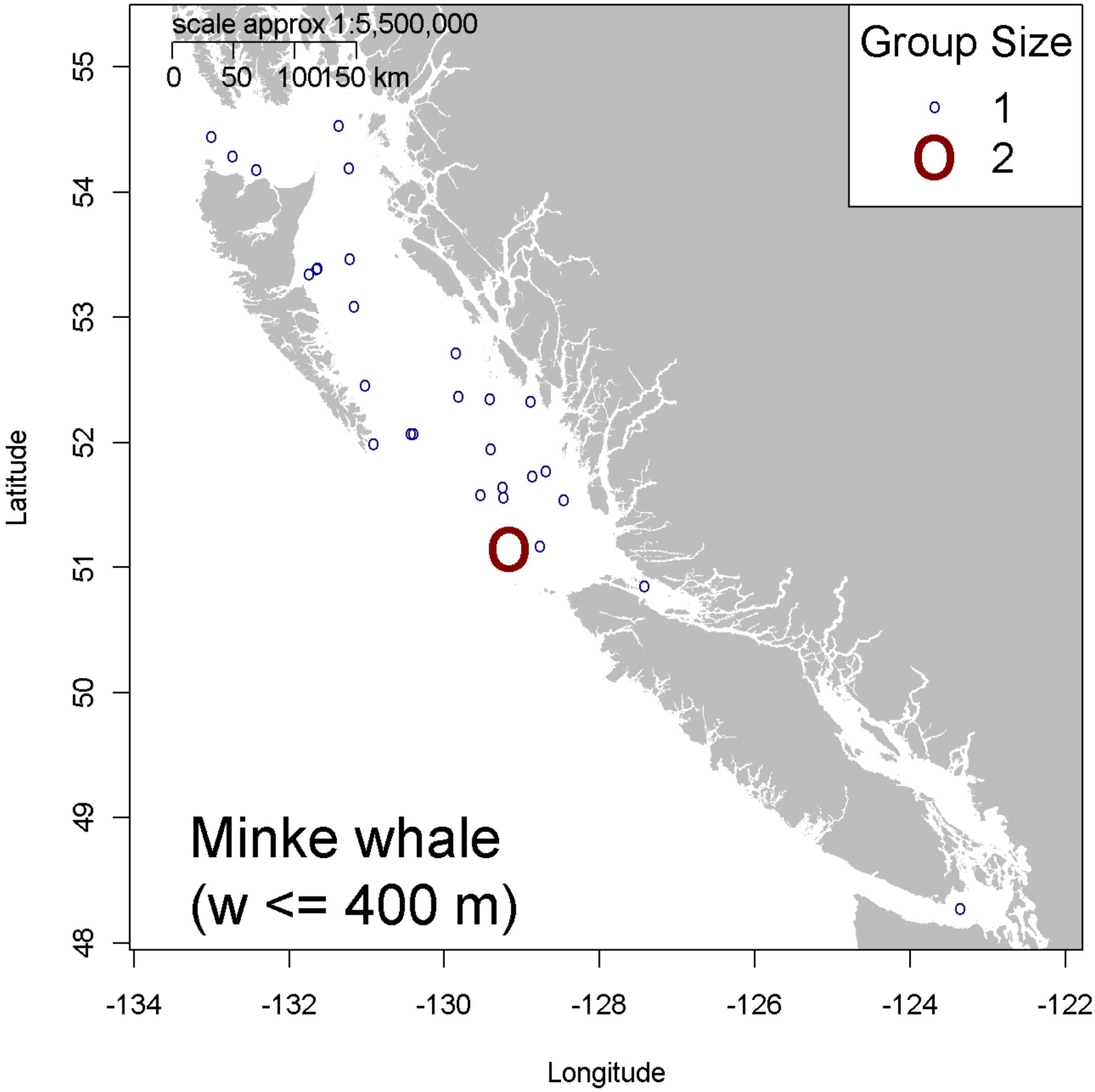

Figure 29. Observations of minke whale by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 400m.

Harbour seal, haul-out

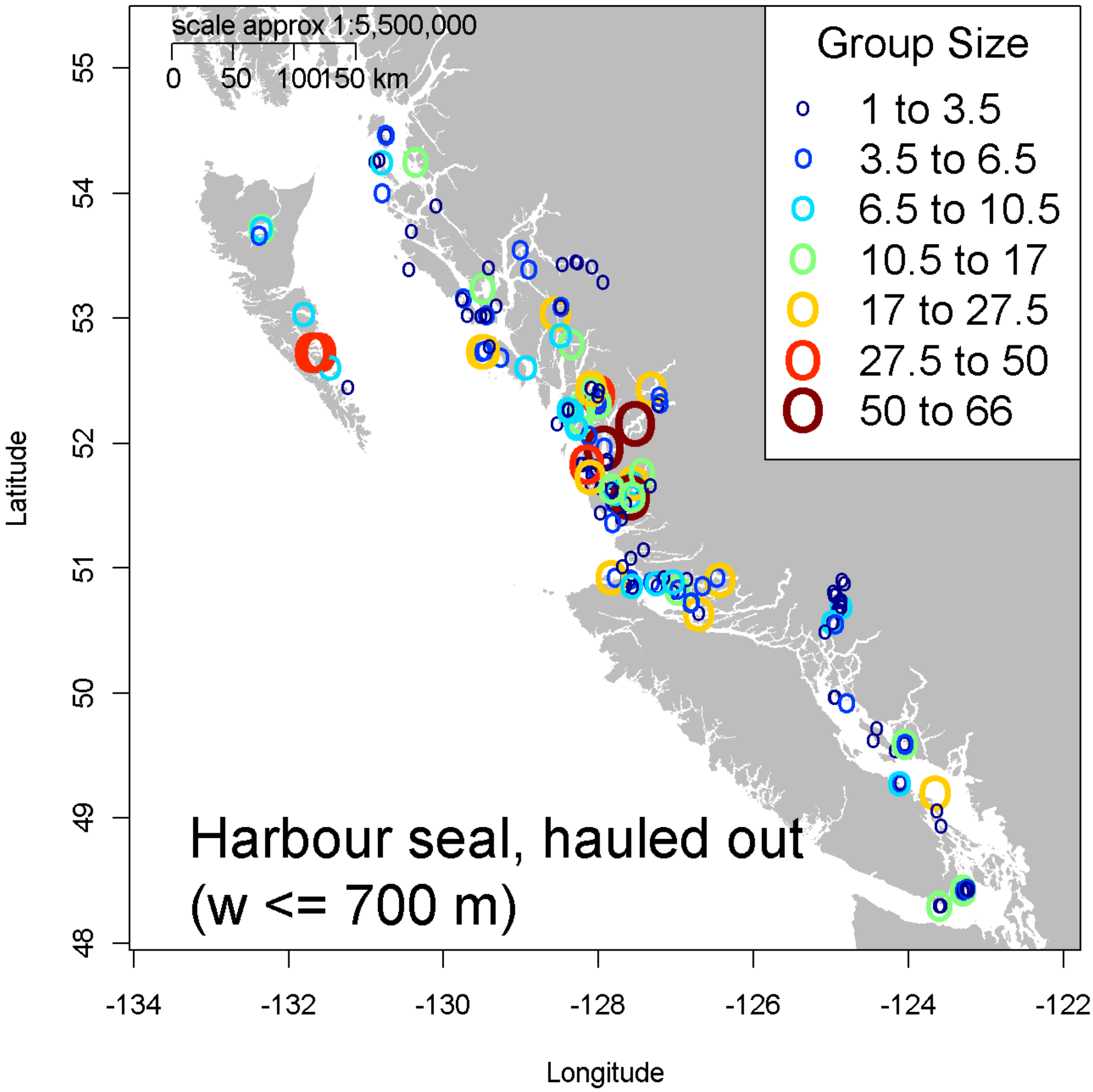

Figure 30. Observations of harbour seal, haul-out, by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 700m.

Harbour seal, in-water

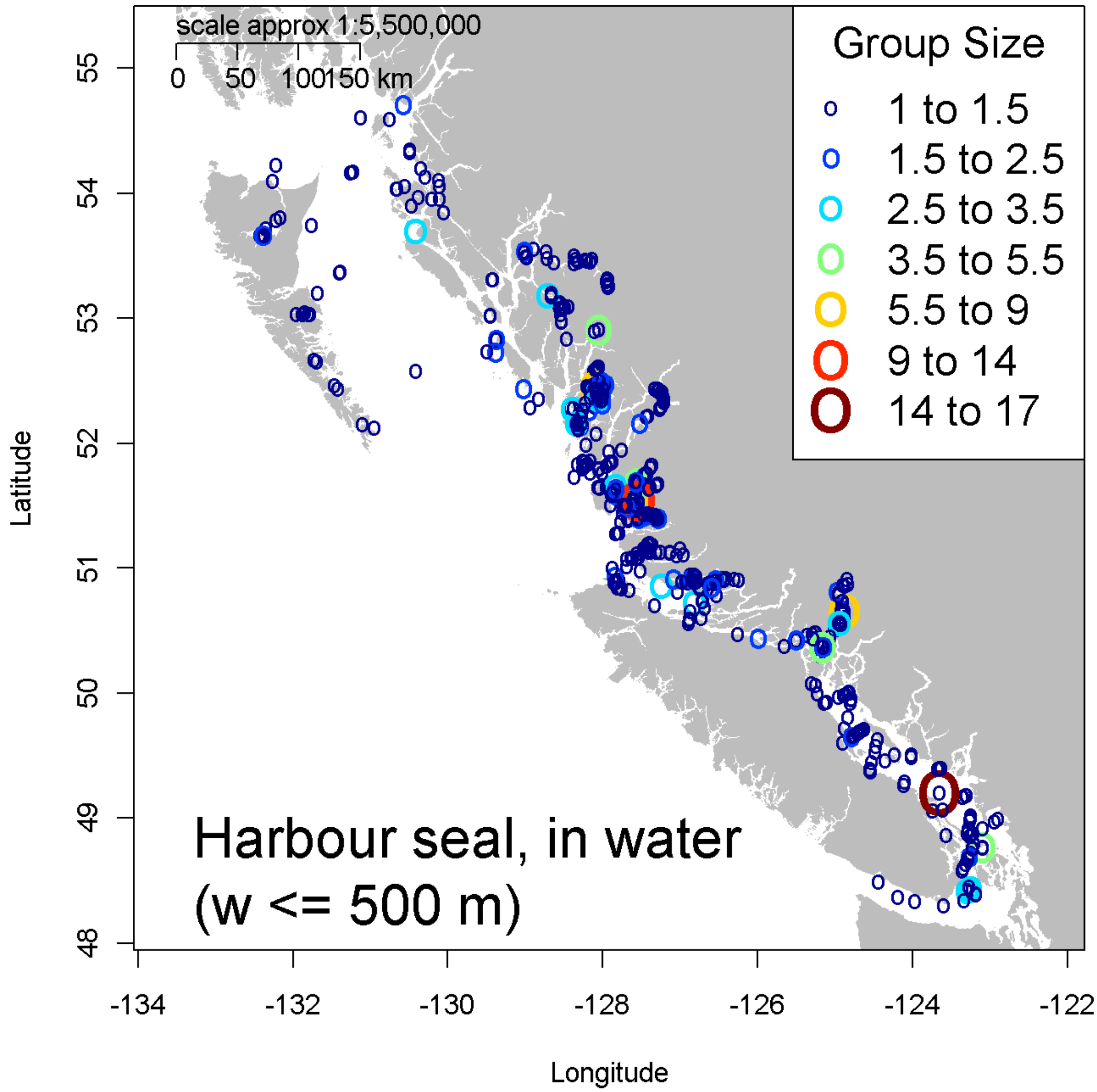

Figure 31. Observations of harbour seal, in-water, by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 500m.

Stellar sea lion, haul-out

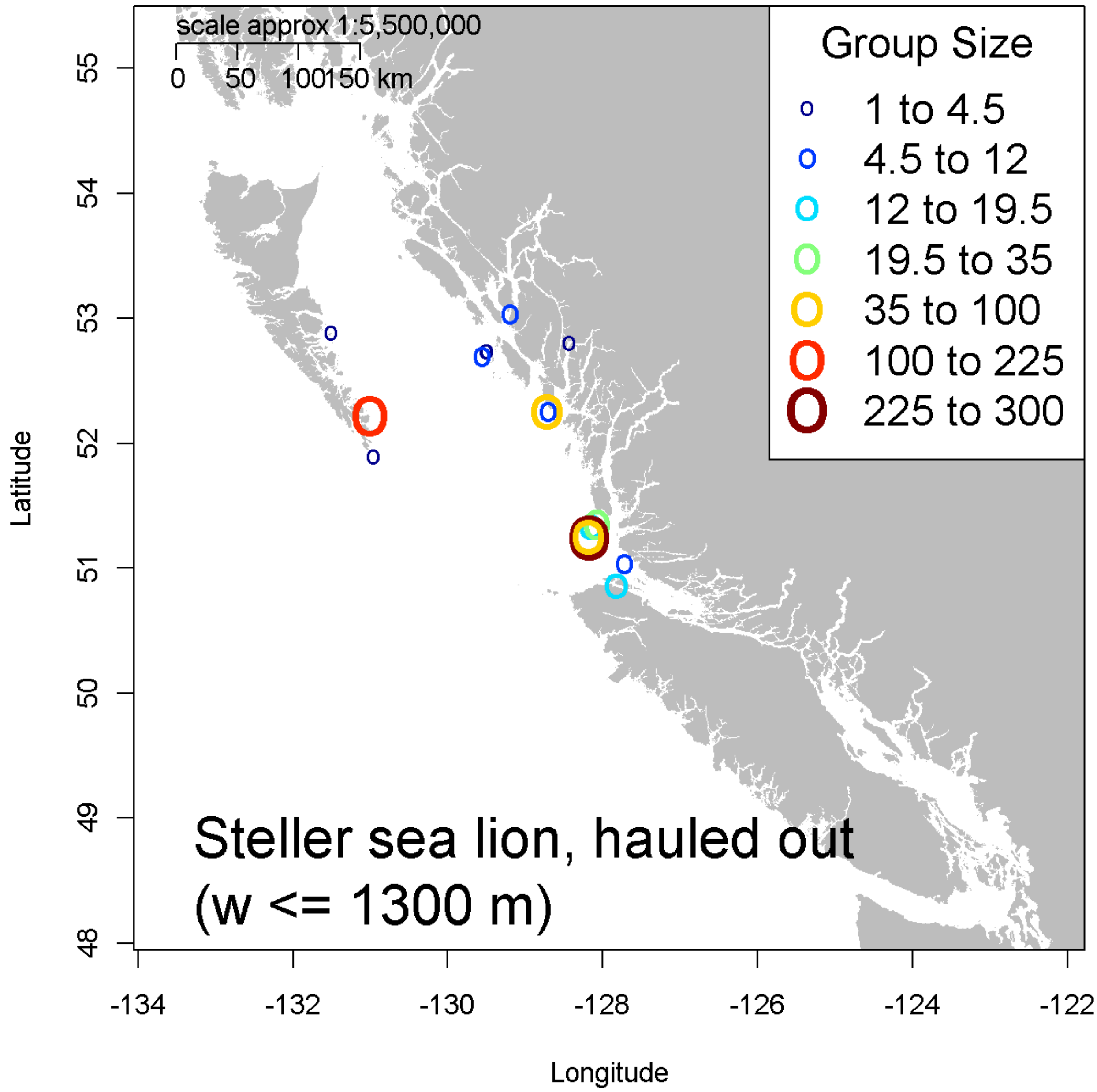

Figure 32. Observations of Steller sea lion, haul-out, by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 1300m.

Steller sea lion, in-water

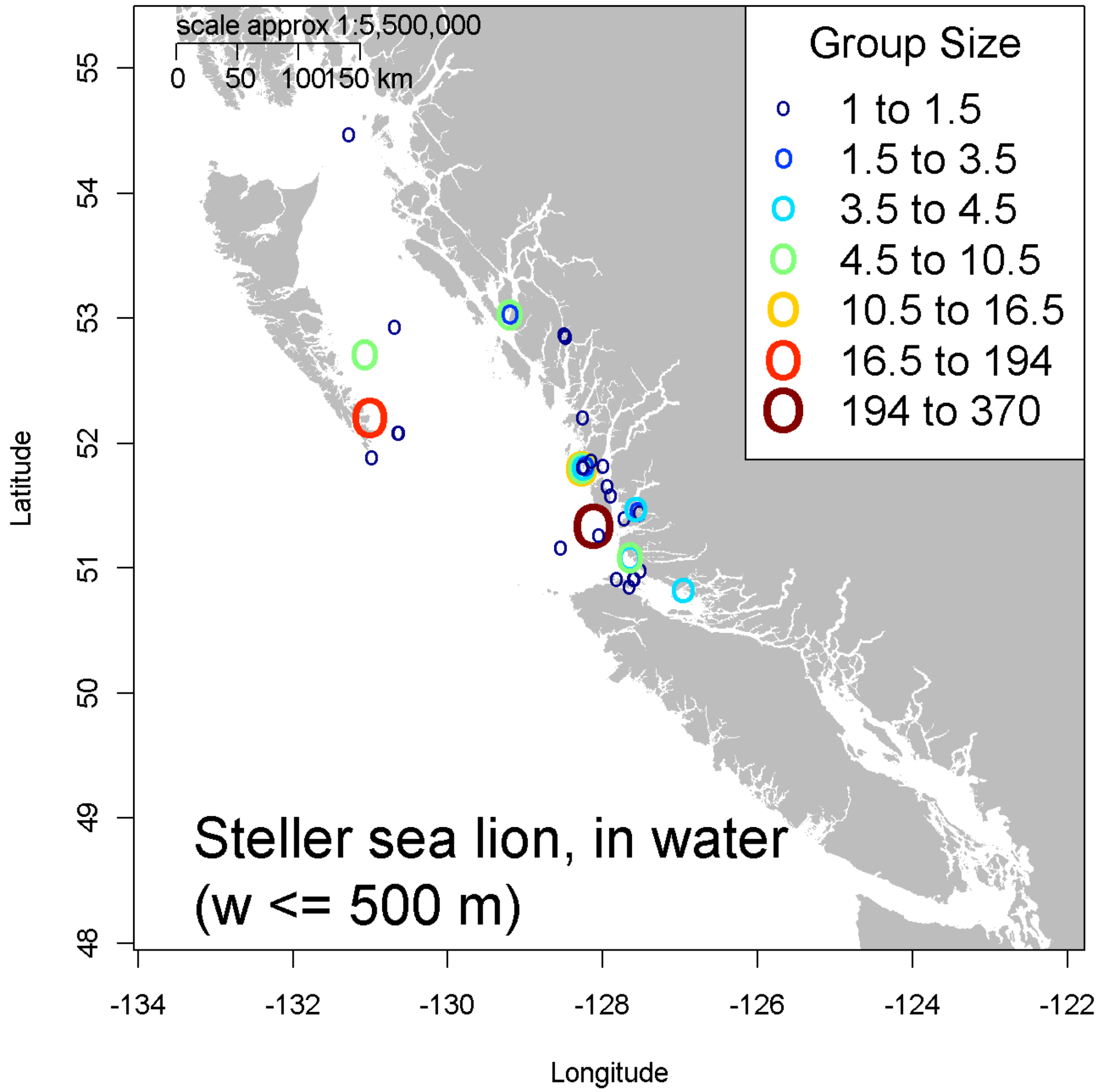

Figure 33. Observations of Steller sea lion, in-water, by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 500m.

Elephant seal

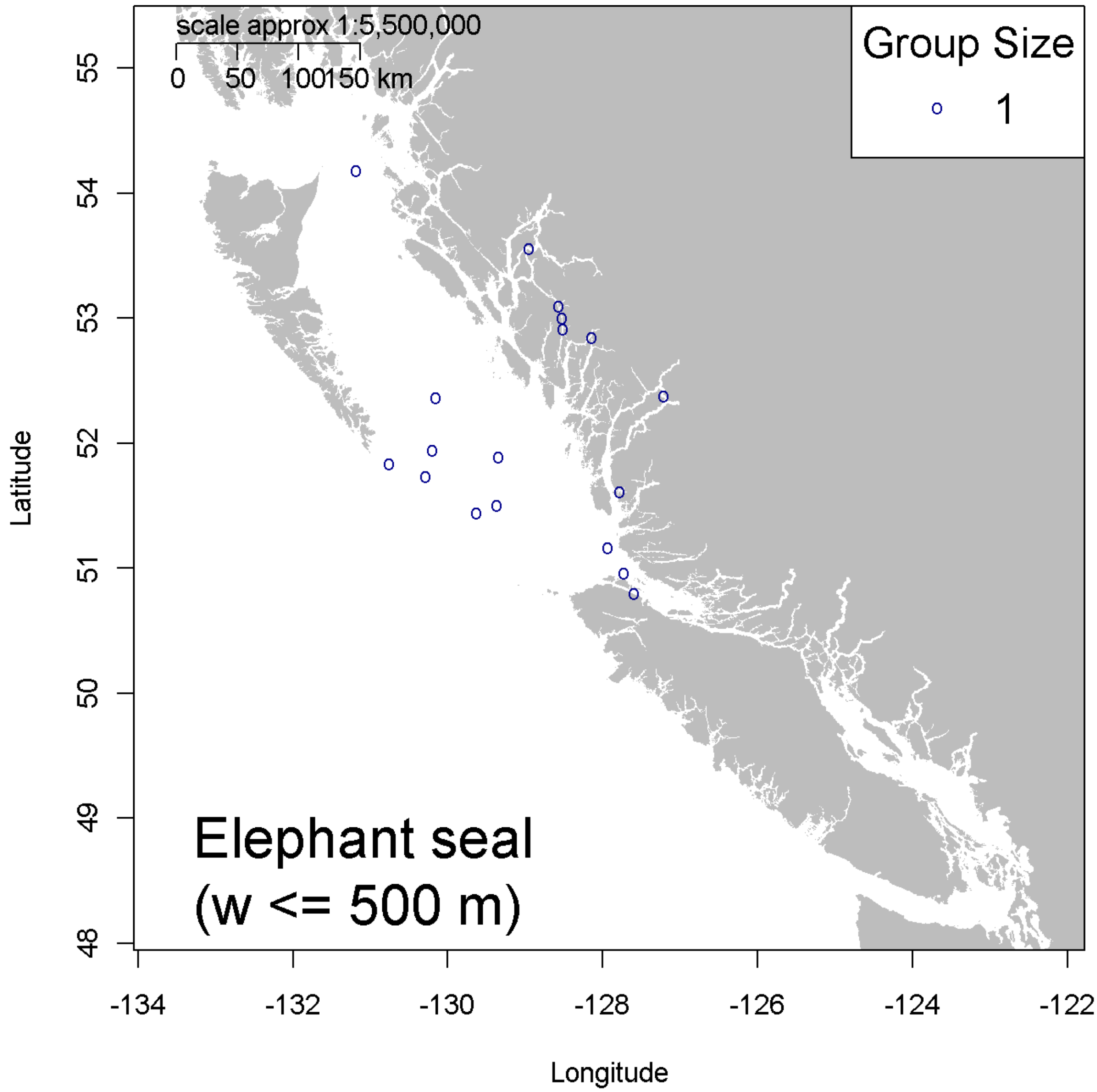

Figure 34. Observations of elephant seal, haul-out, by group size for all surveys (2004-2008), after truncating perpendicular distance (*w*) to within 500m.

Appendix 3. Maps of Environmental Covariates

Depth

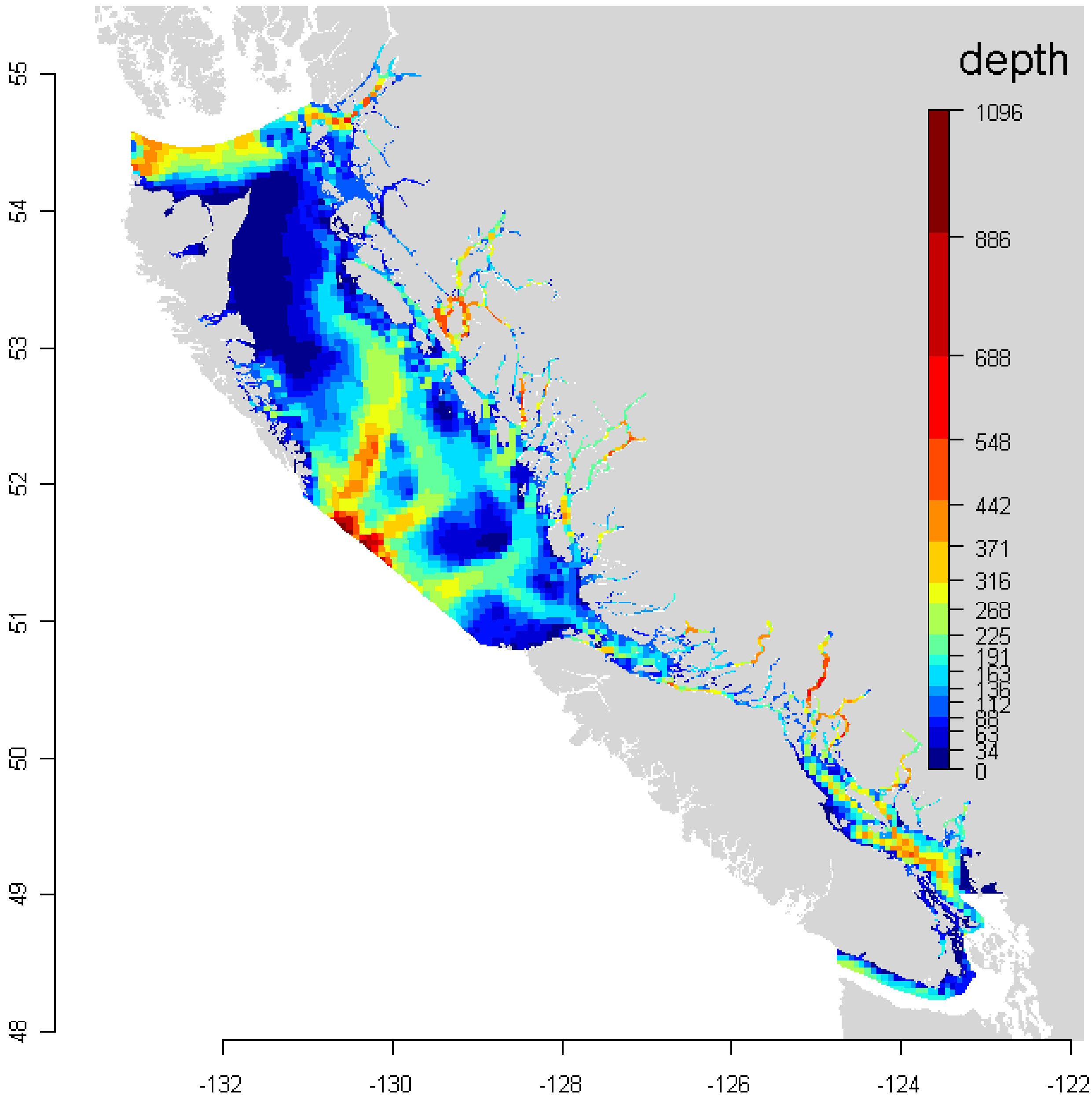

Figure 35. Static covariate bathymetric depth at 5km grid resolution in meters.

Slope

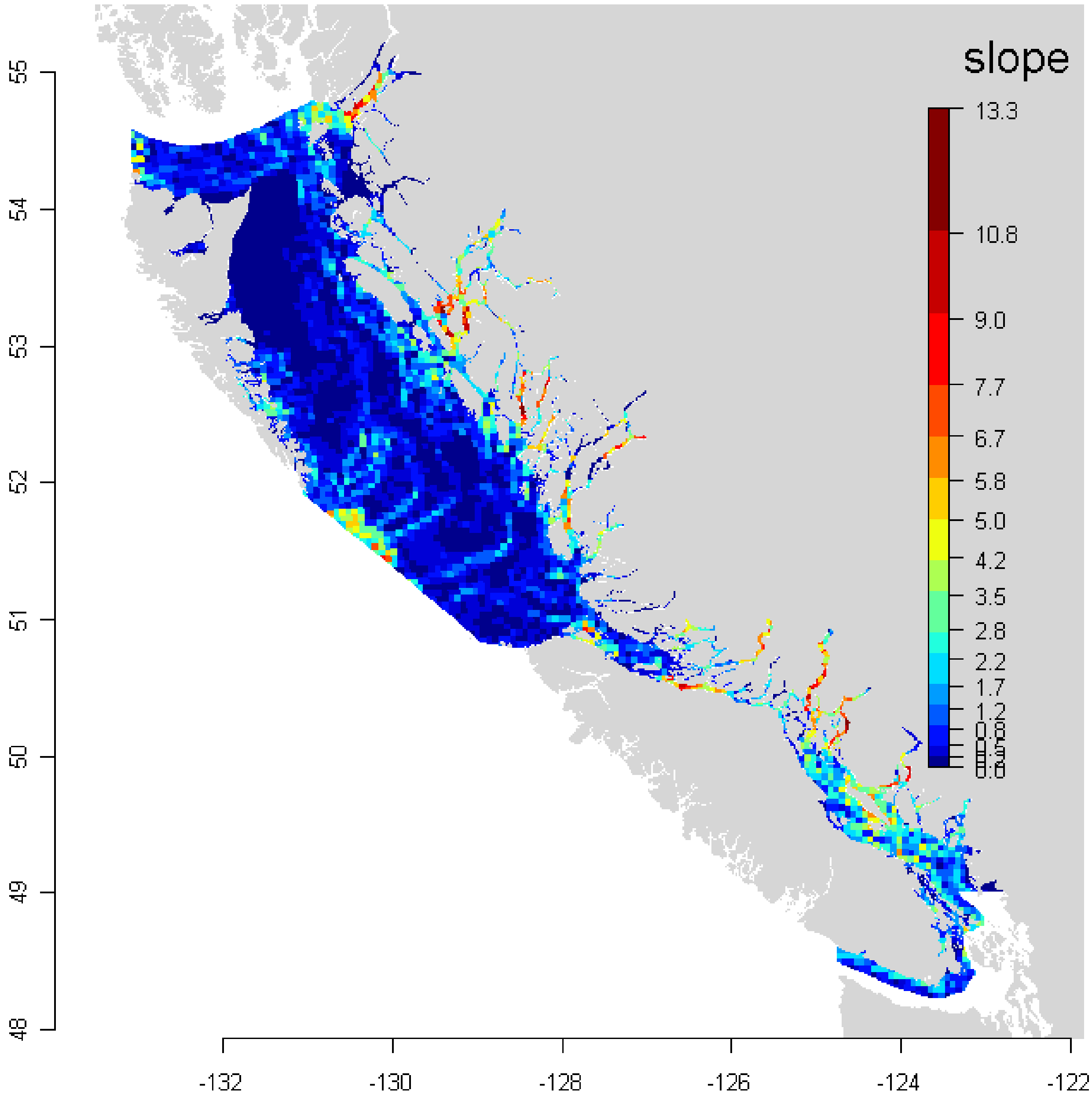

Figure 36. Static covariate slope derived from bathymetric depth in percent degrees at 5km grid resolution.

Distance to Coast

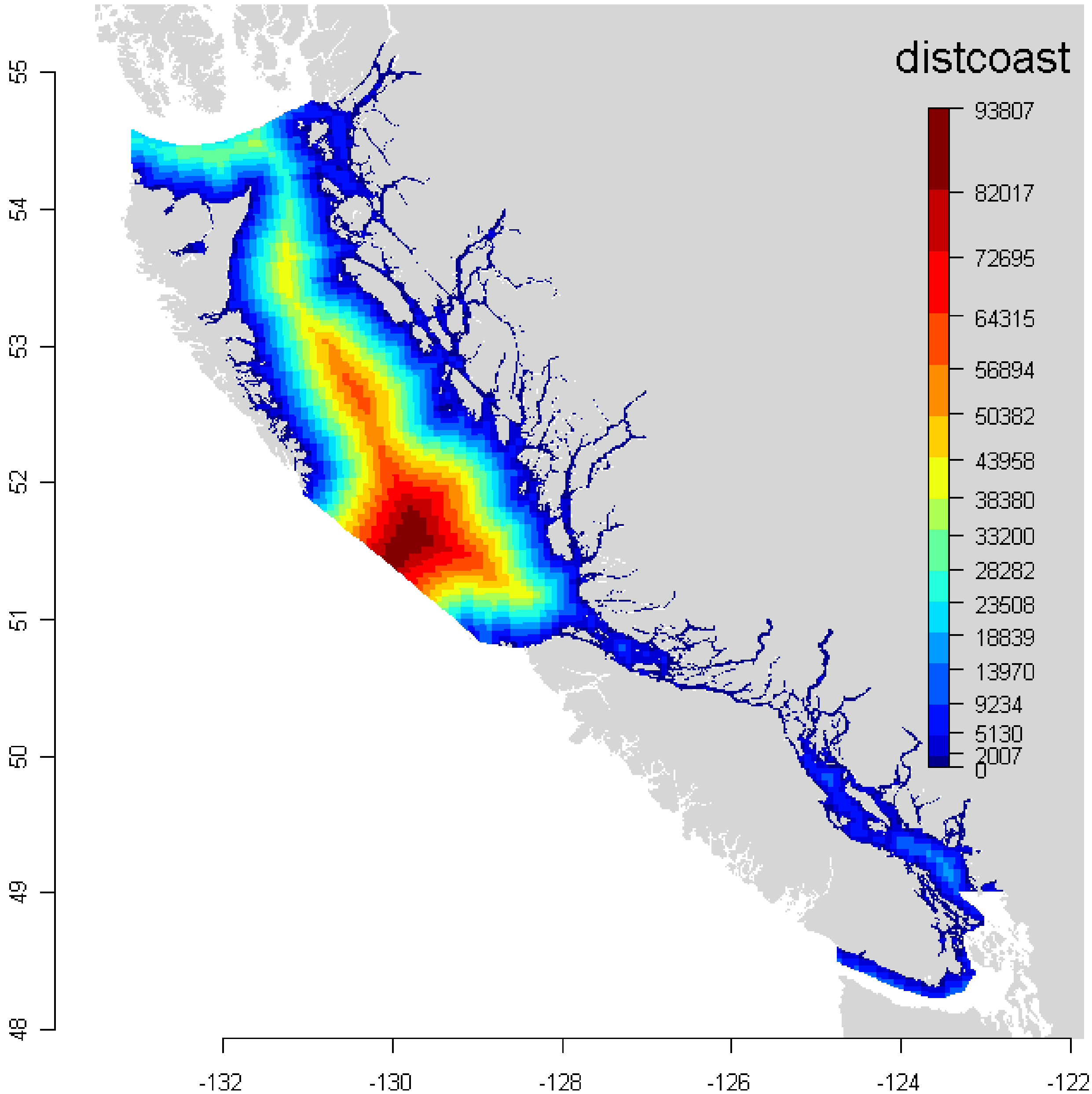

Figure 37. Static covariate distance to coast (distcoast) in meters at 5km grid resolution.

Dynamic Variables: SST, Chl

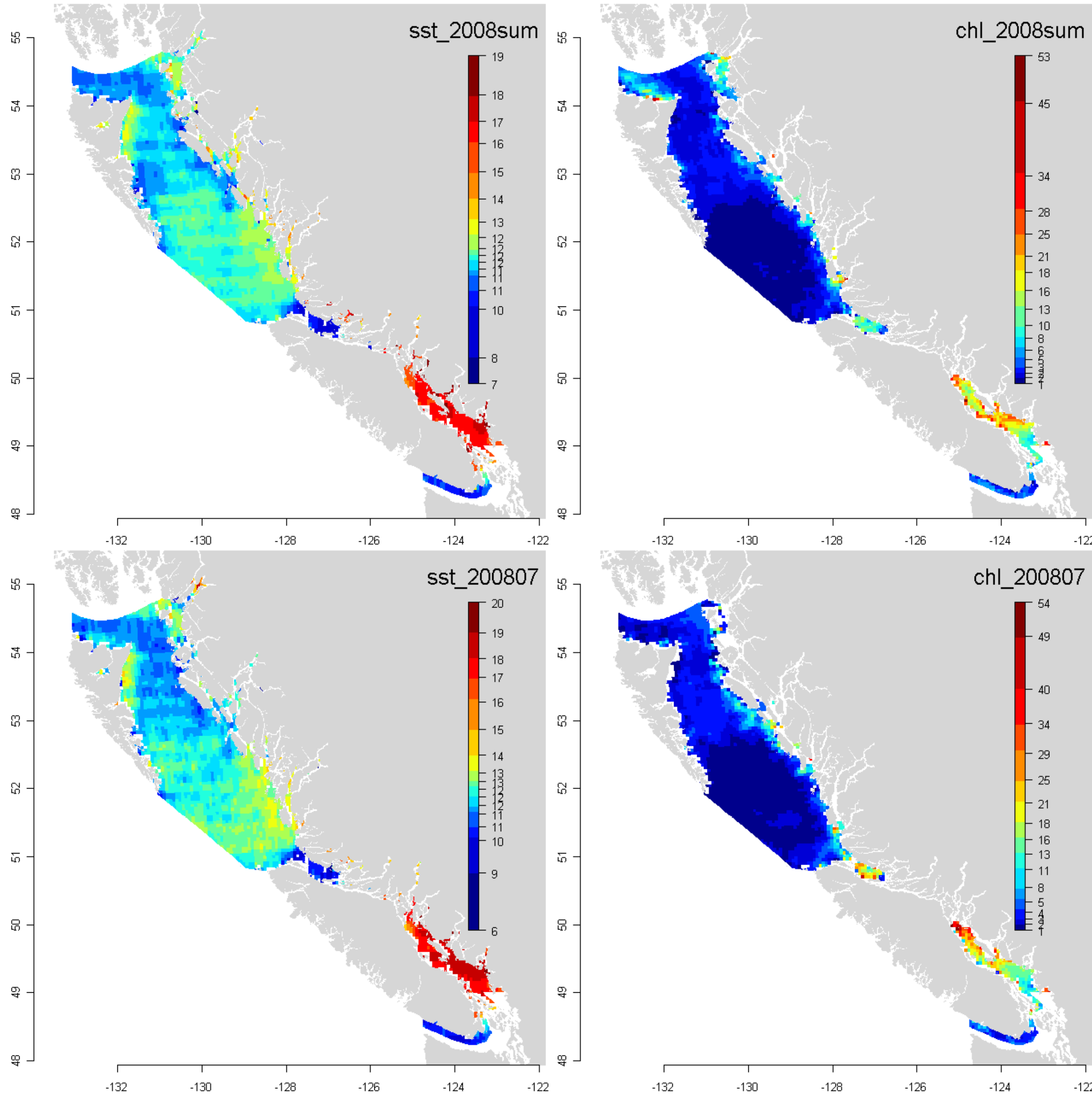

Figure 38. Dynamic predictor variables sea-surface temperature (sst) in degrees Celsius and chlorophyll (chl) in mg/m^3 given for July, 2008 and averaged over the summer months.

Appendix 4. Density Surface Model Outputs

Harbour Porpoise

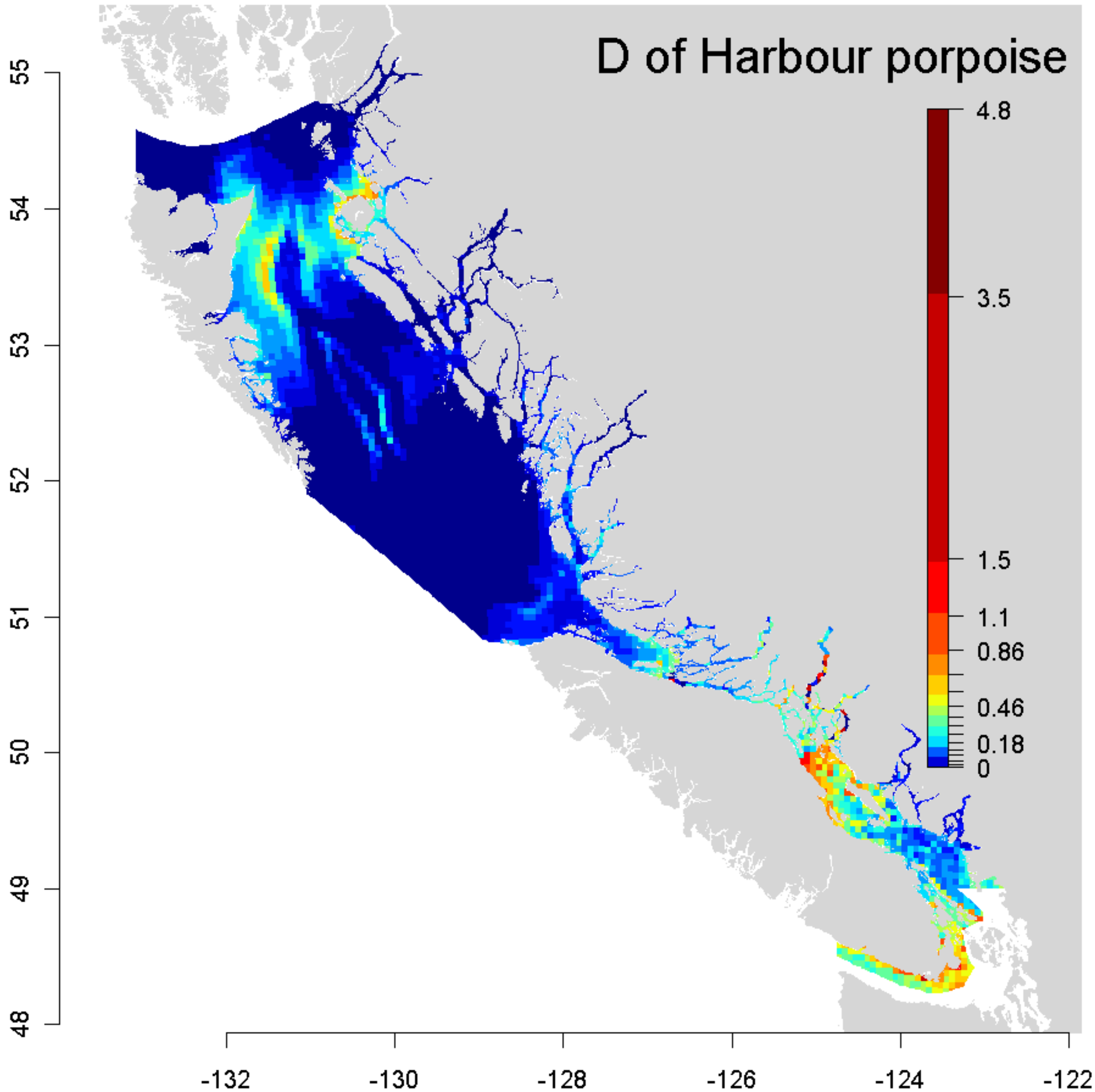

Figure 39. Density surface model of harbour porpoise, in # of individuals per square kilometer.

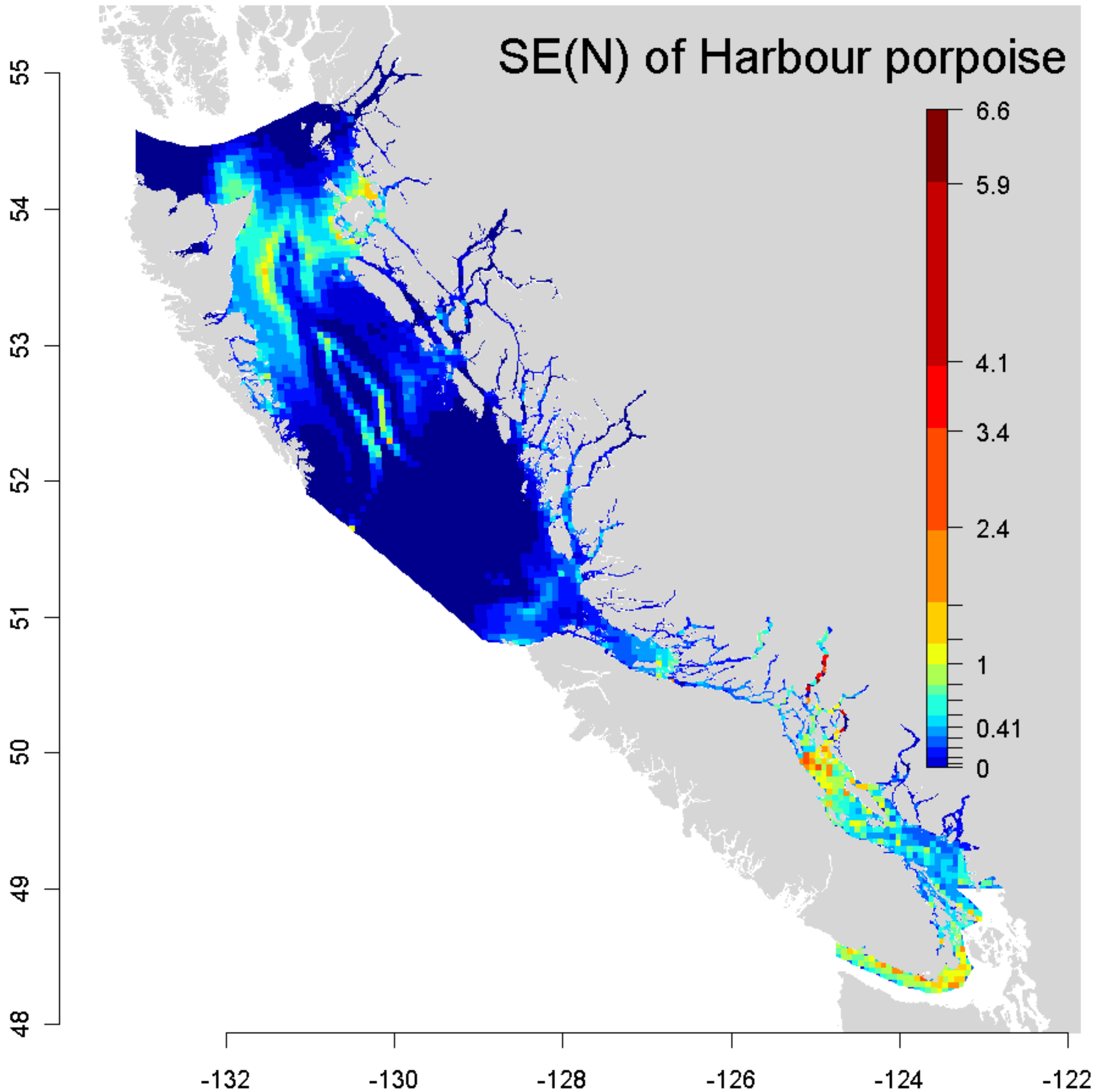

Figure 40. Standard error for density surface of harbour porpoise.

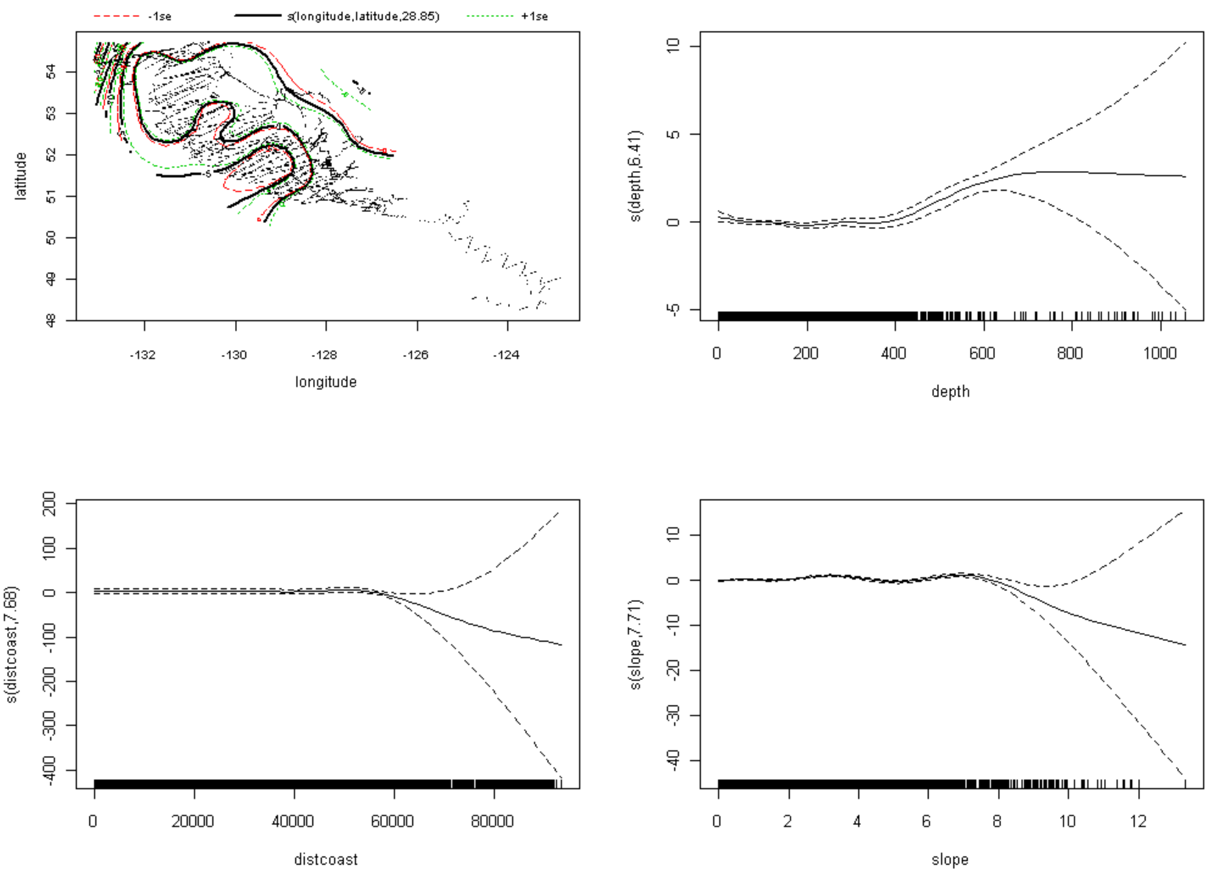

Figure 41. GAM predictor terms for density surface model of harbour porpoise. What is the purpose of these figures- need to be discussed.

Dall’s porpoise

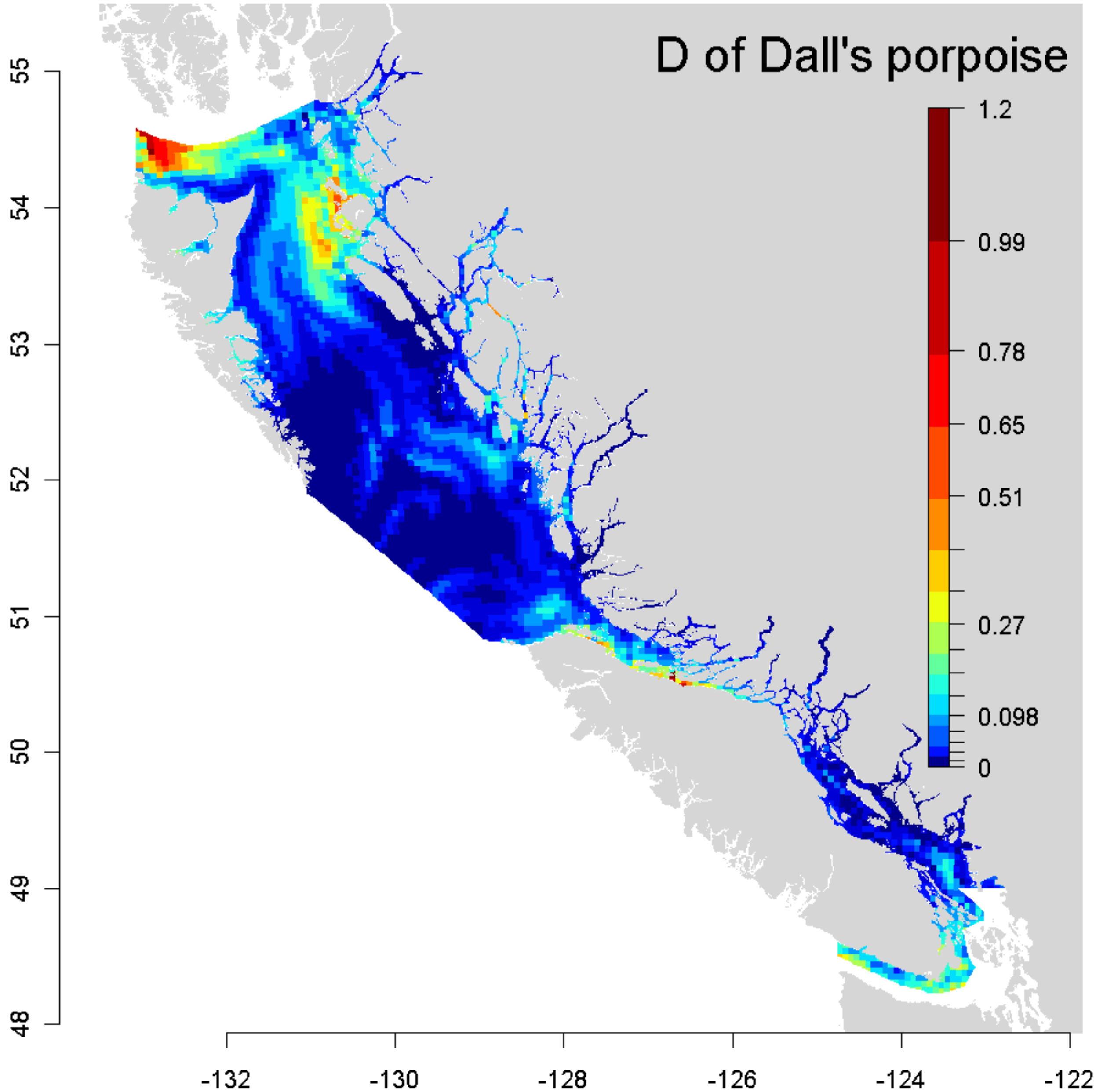

Figure 42. Density surface model of Dall’s porpoise, in # of individuals per square kilometer.

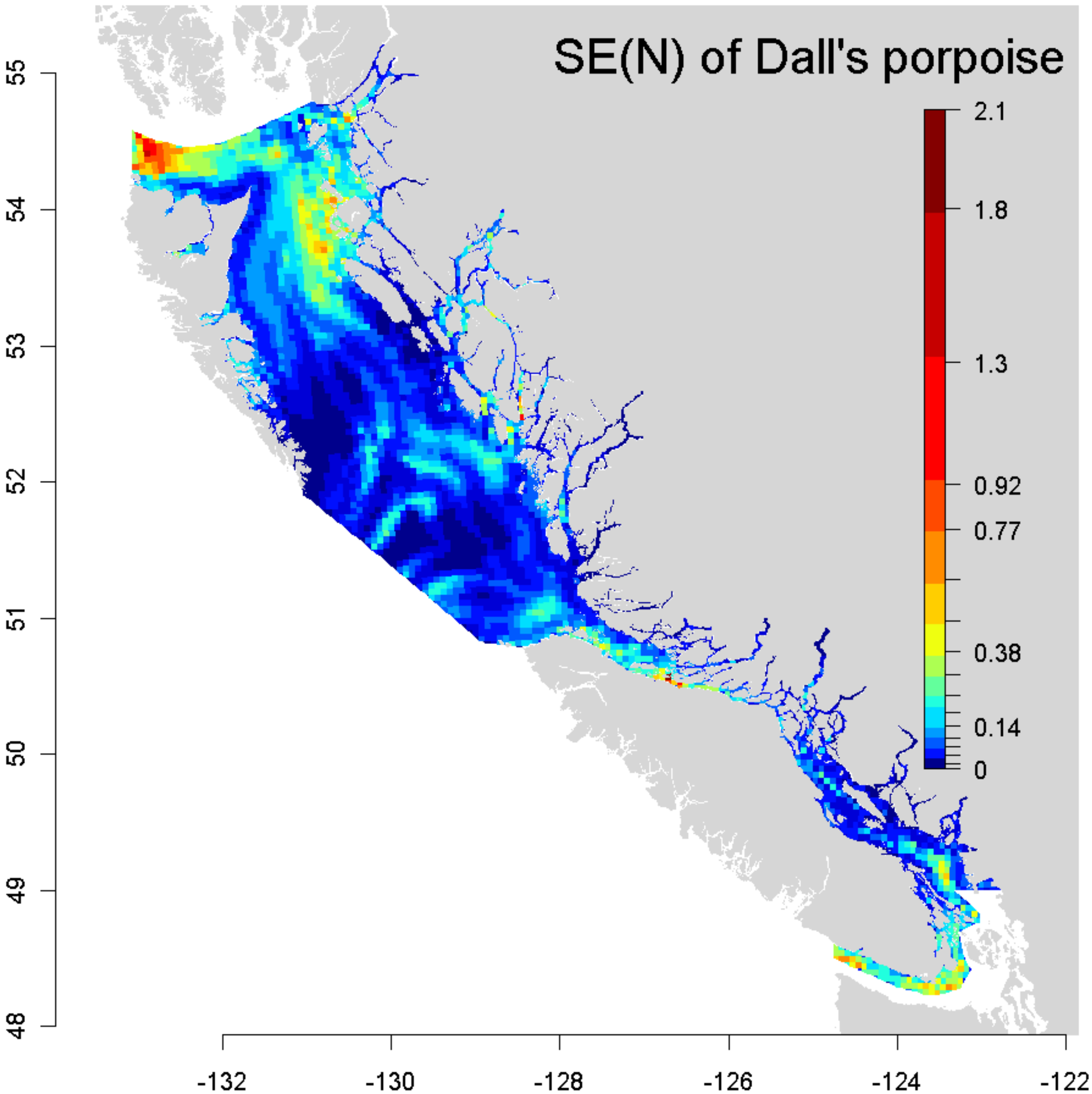

Figure 43. Standard error for distribution model of Dall's porpoise.

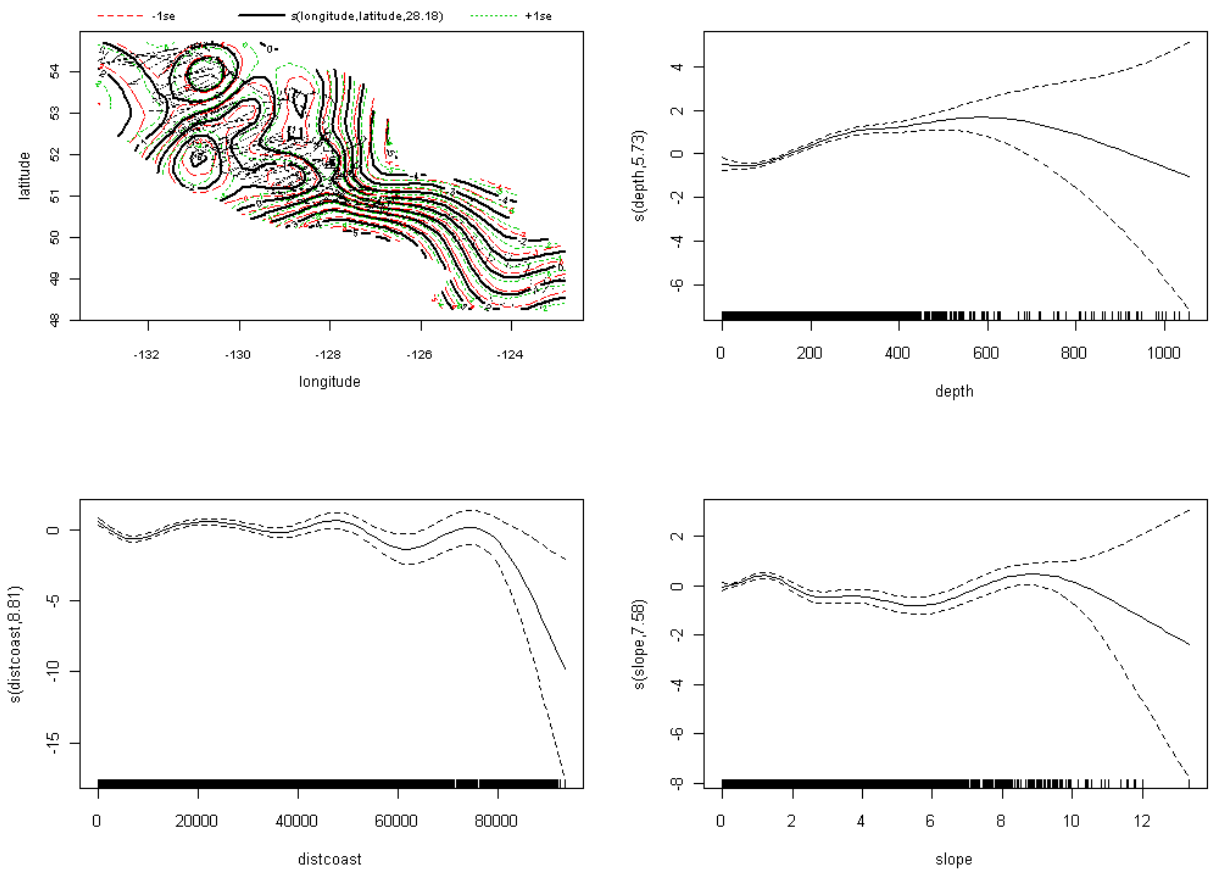

Figure 44. GAM terms plot for density surface model of Dall's porpoise.

Pacific white-sided dolphin

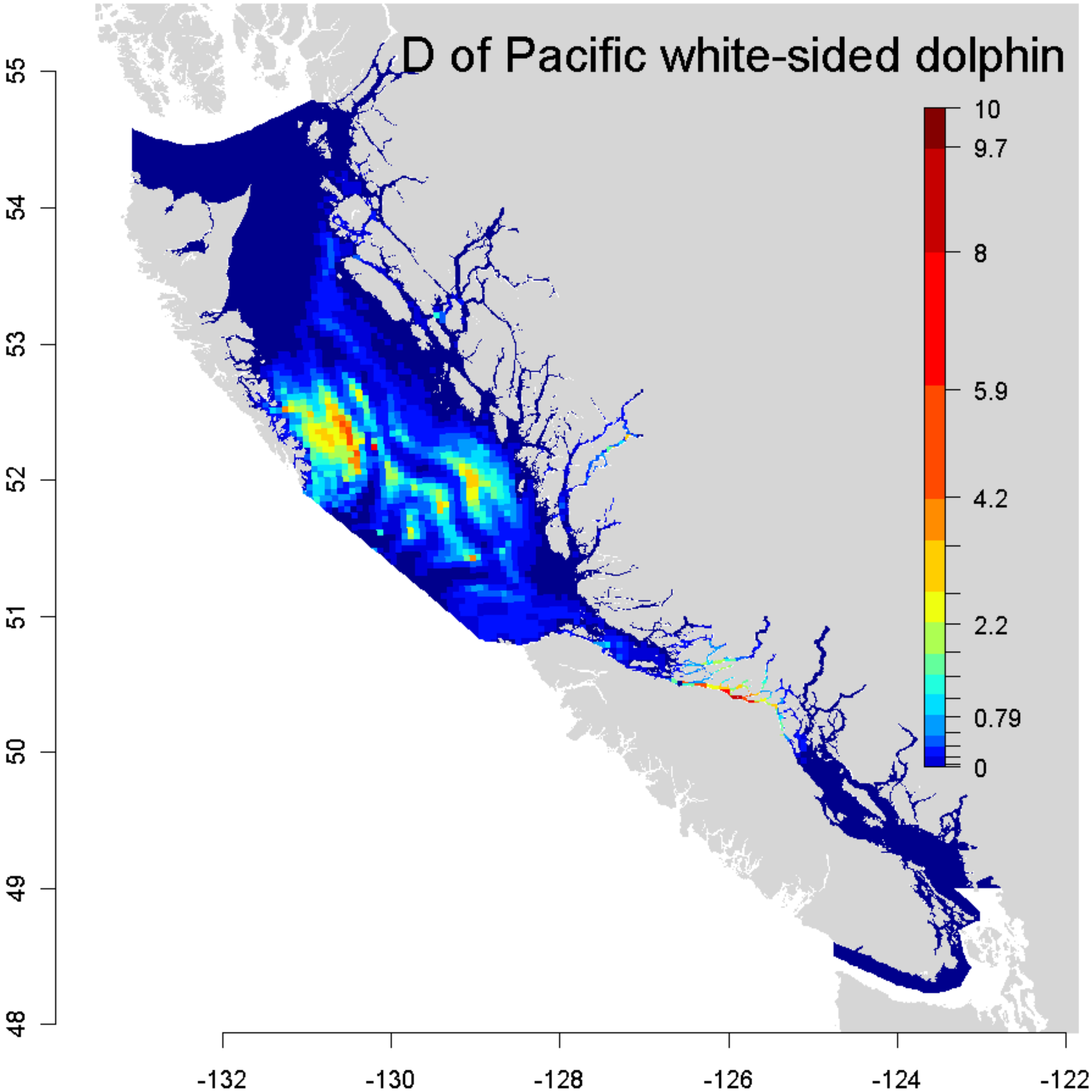

Figure 45. Density surface model of Pacfic white-sided dolphin, in # of individuals per square kilometer.

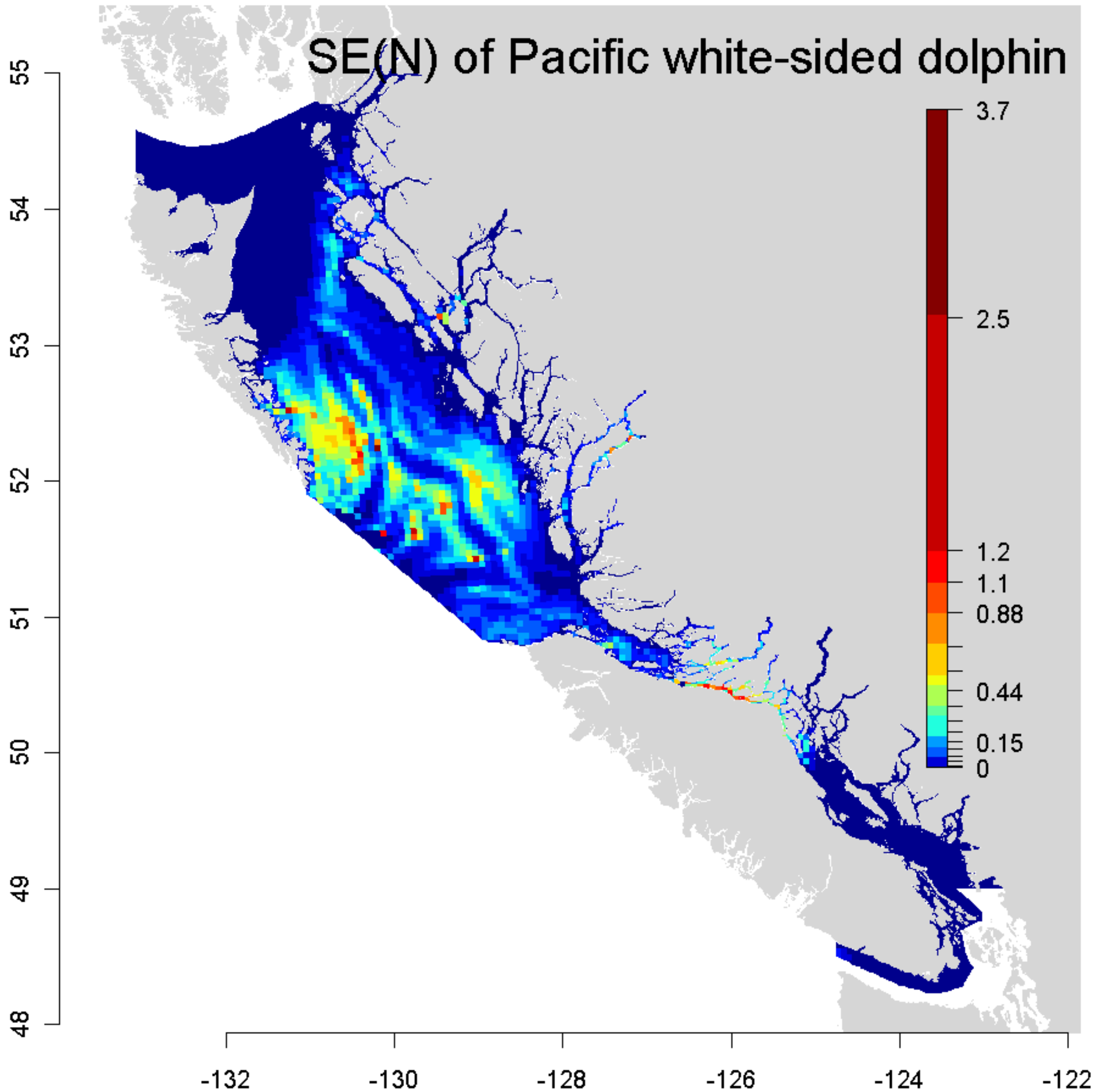

Figure 46. Standard error for density surface model of Pacific white-sided dolphin.

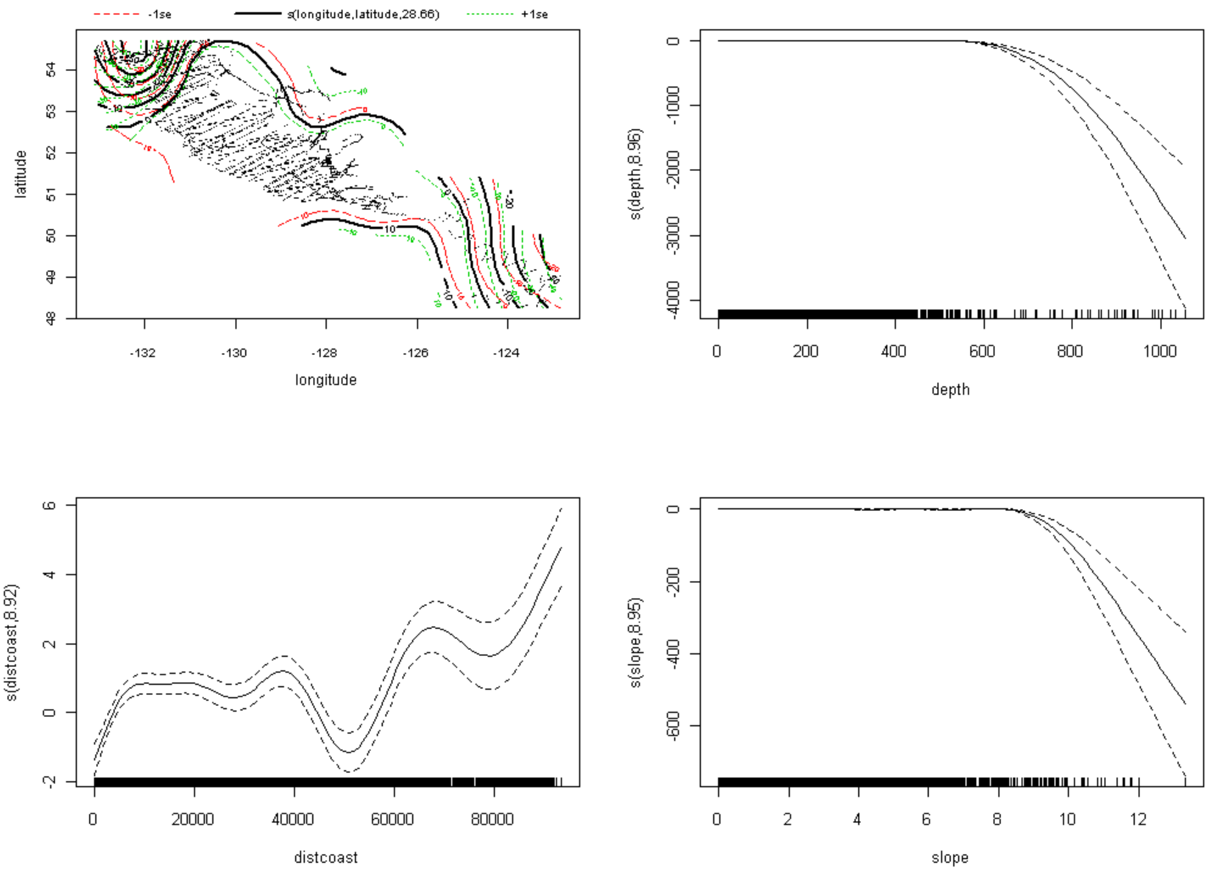

Figure 47. GAM terms plot for density surface model of Pacific white-sided dolphin.

Humpback whale

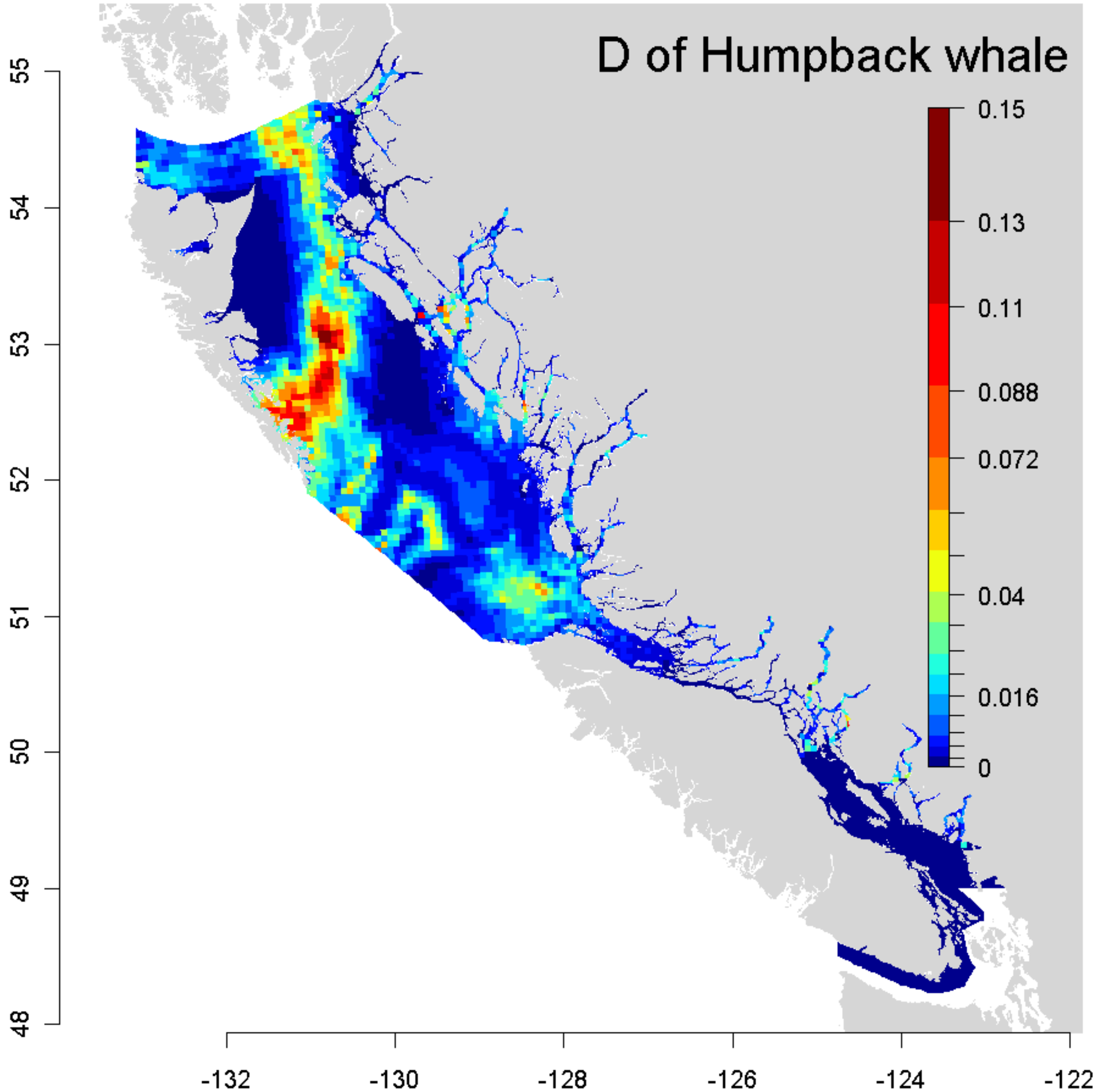

Figure 48. Density surface model for humpback whale, in # of individuals per square kilometer.

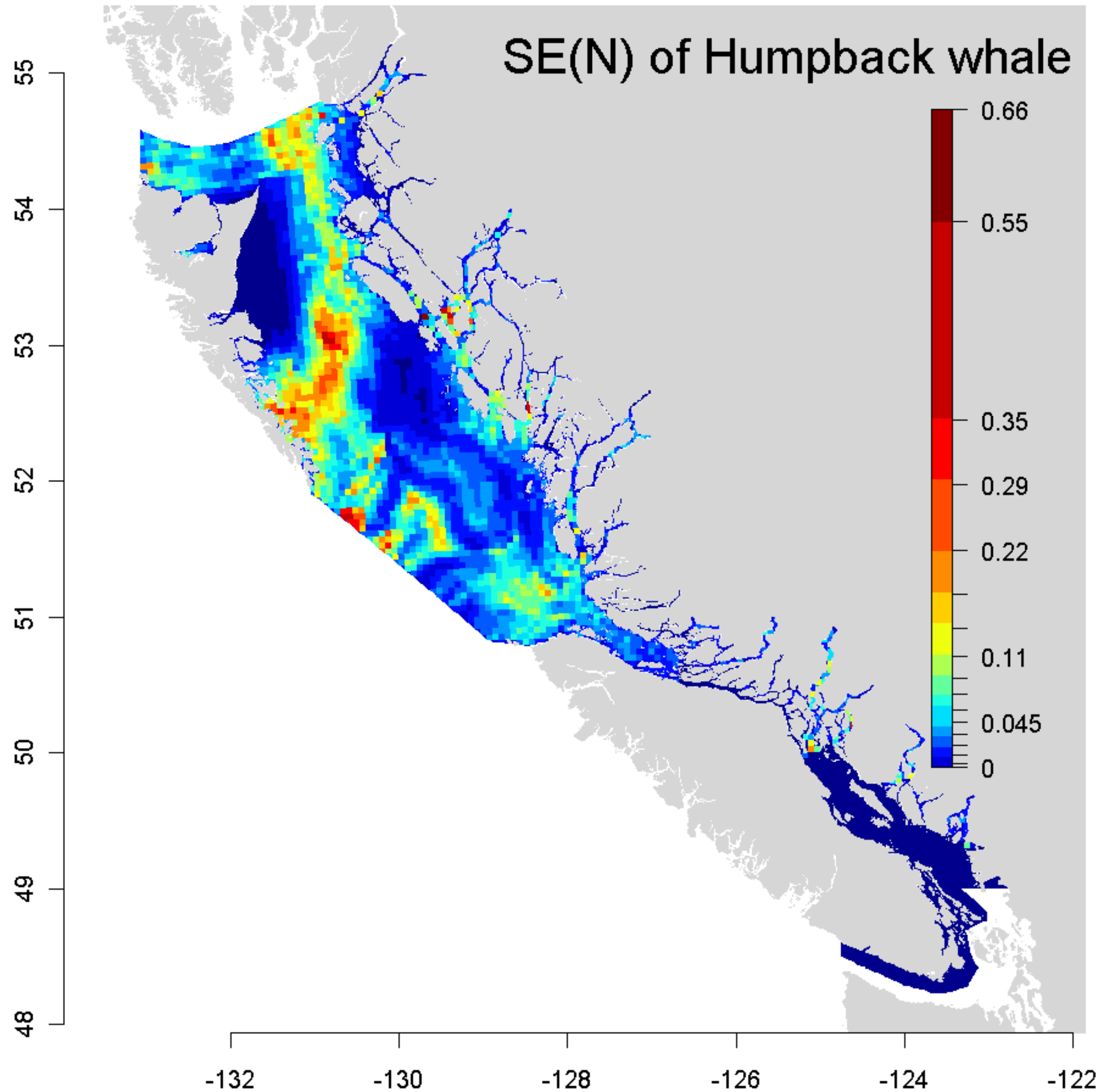

Figure 49. Standard error for density surface model of humpback whale.

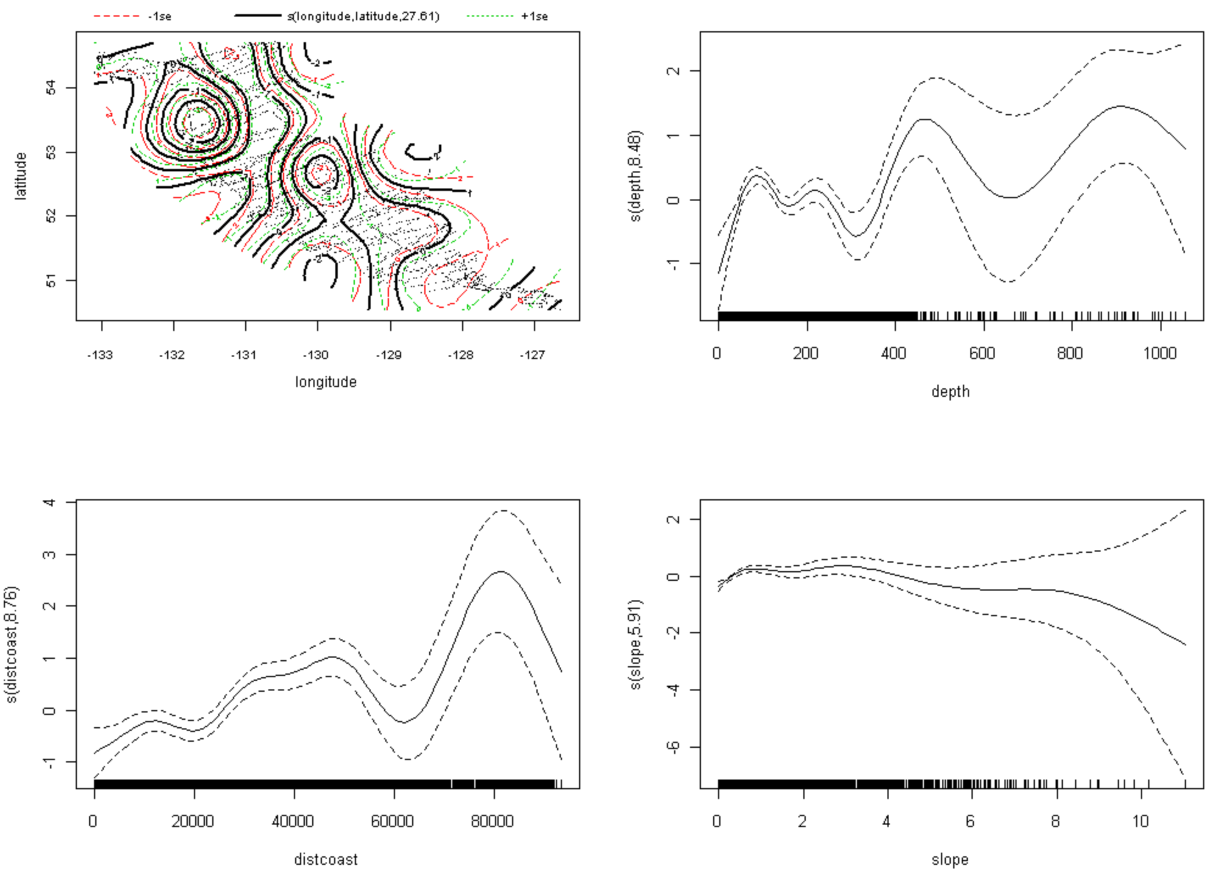

Figure 50. GAM terms plot for humpback whale density surface model in stratum 1.

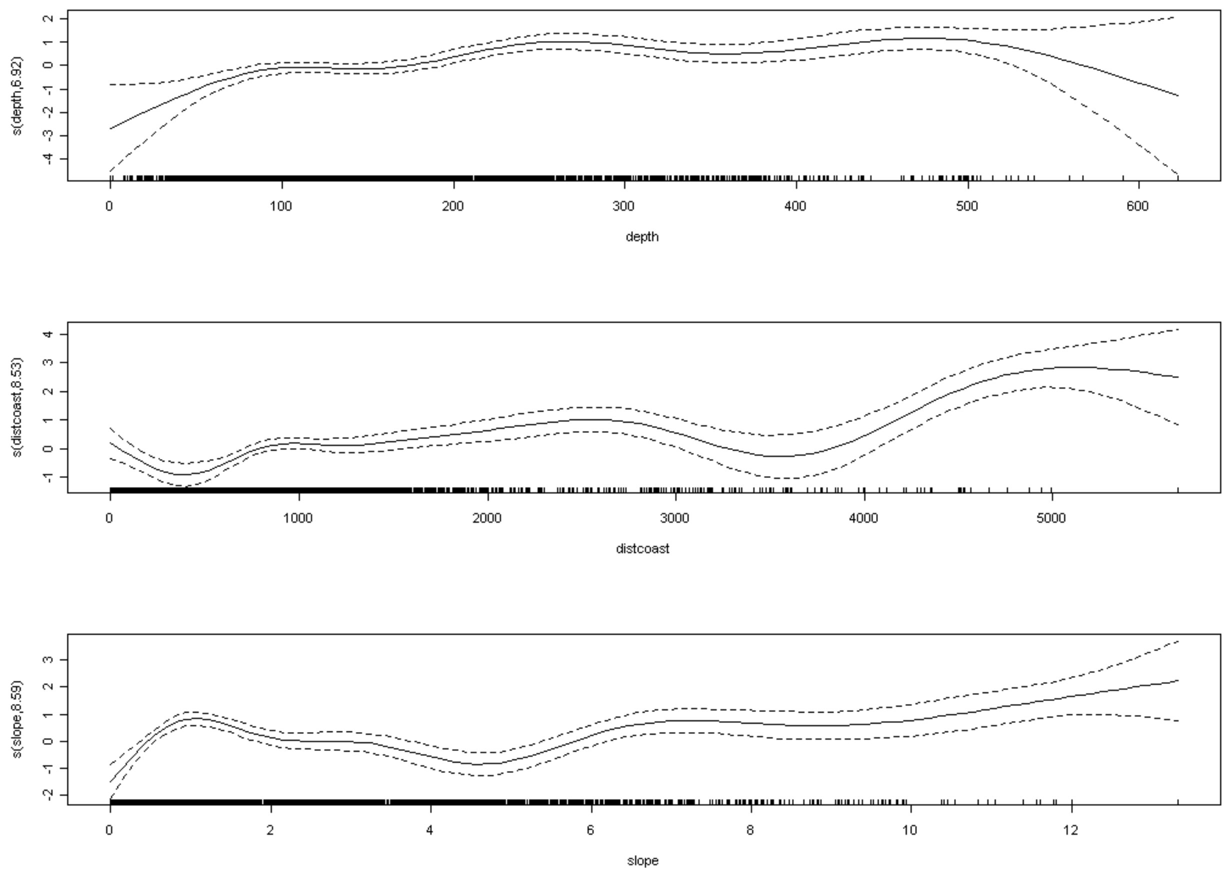

Figure 51. GAM terms plot for humpback whale density surface model in strata 2,3,4.

Fin whale

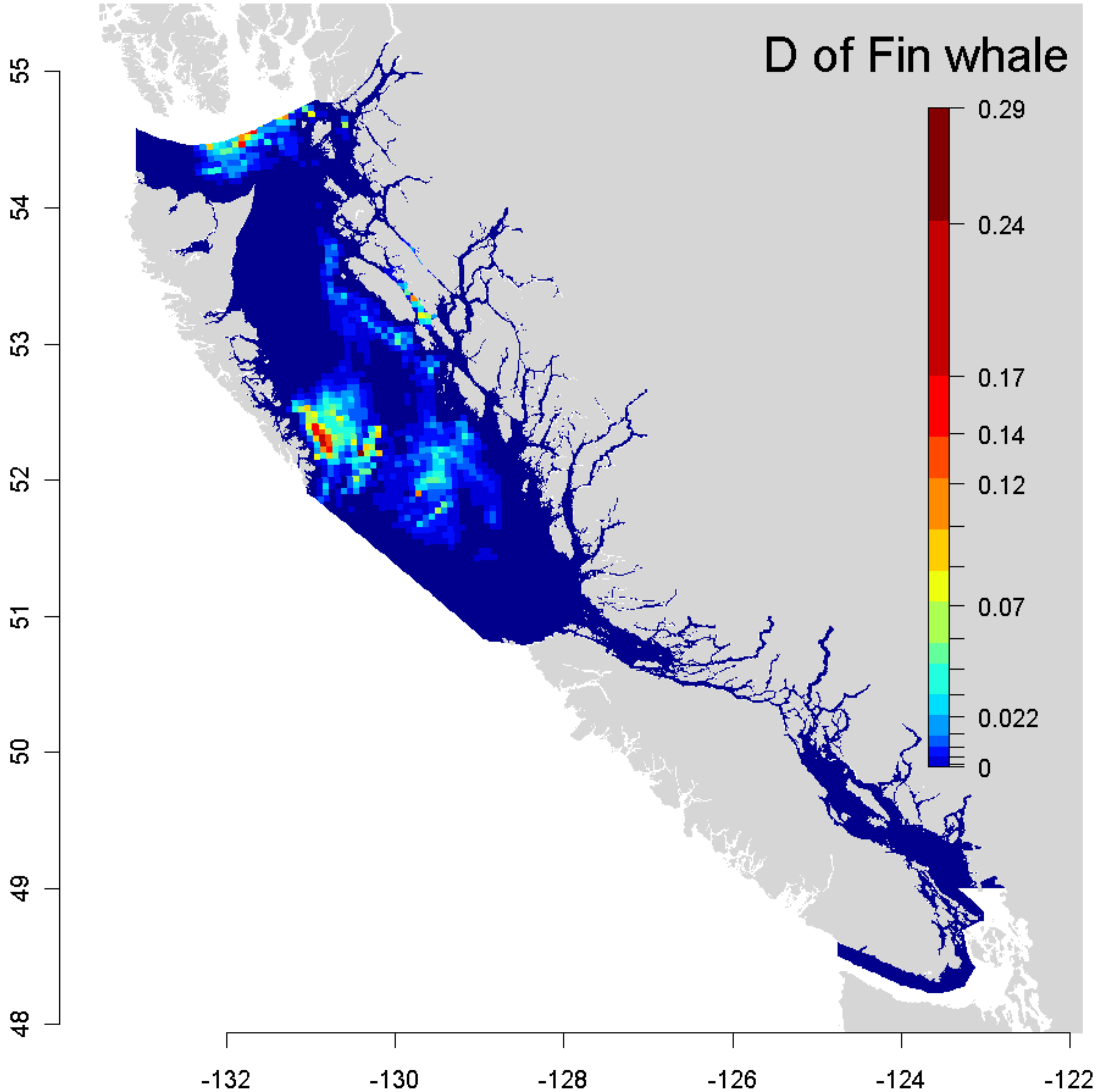

Figure 52. Density surface model for fin whale, in # of individuals per square kilometer.

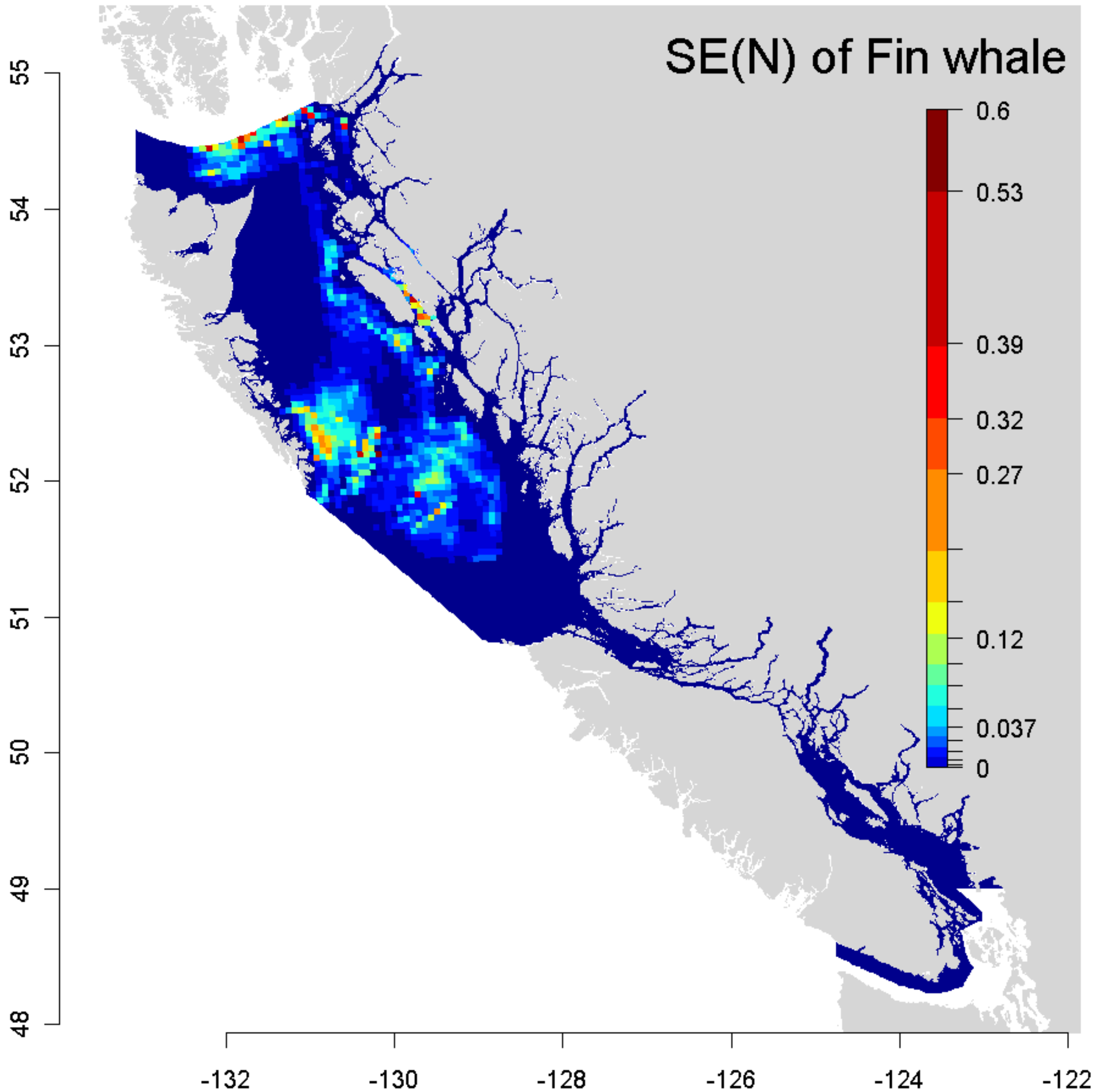

Figure 53. Standard error for density surface model of fin whale.

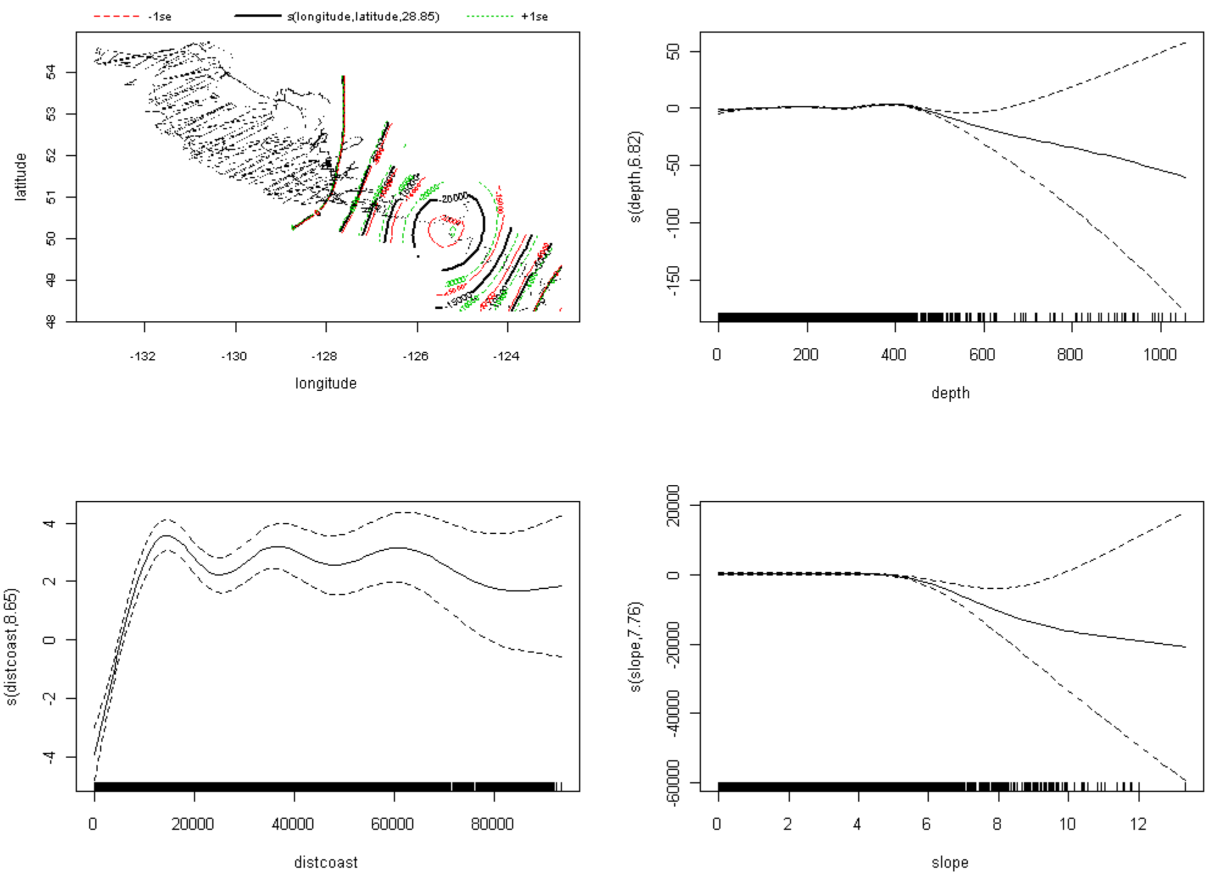

Figure 54. GAM terms plot for density surface model of fin whale.

Killer whale

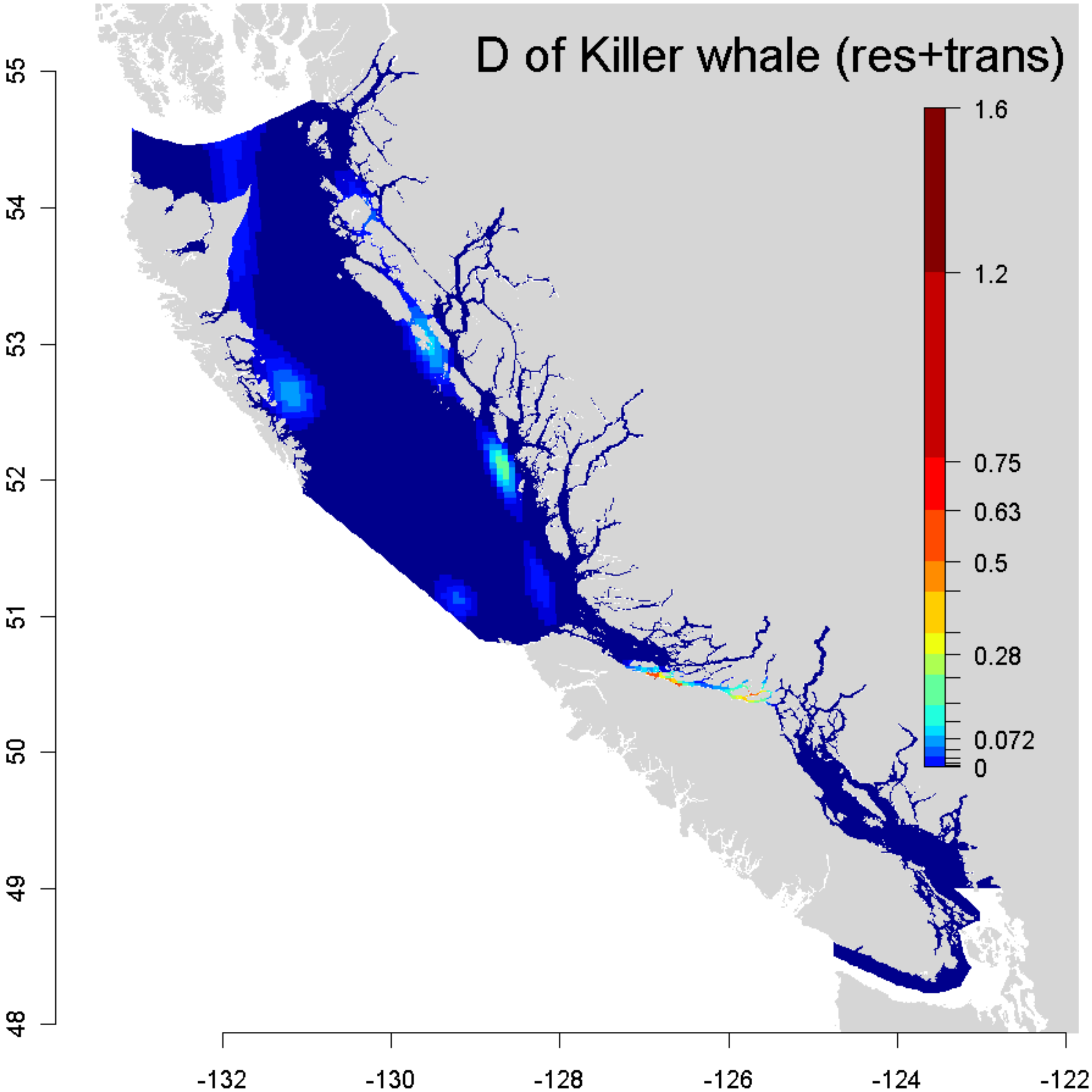

Figure 55. Density surface model of killer whale, in # of individuals per square kilometer.

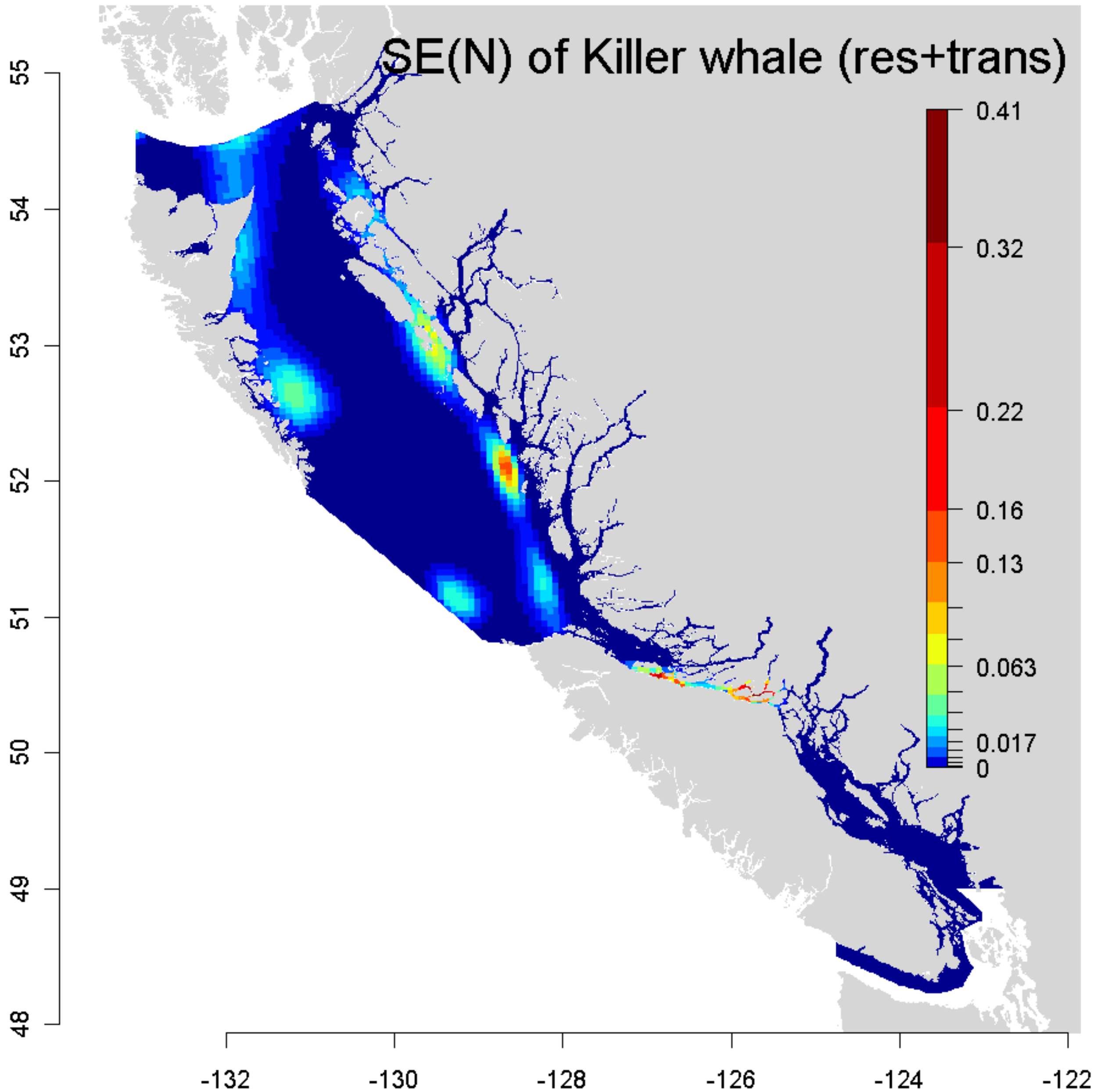

Figure 56. Standard error for density surface model of killer whale.

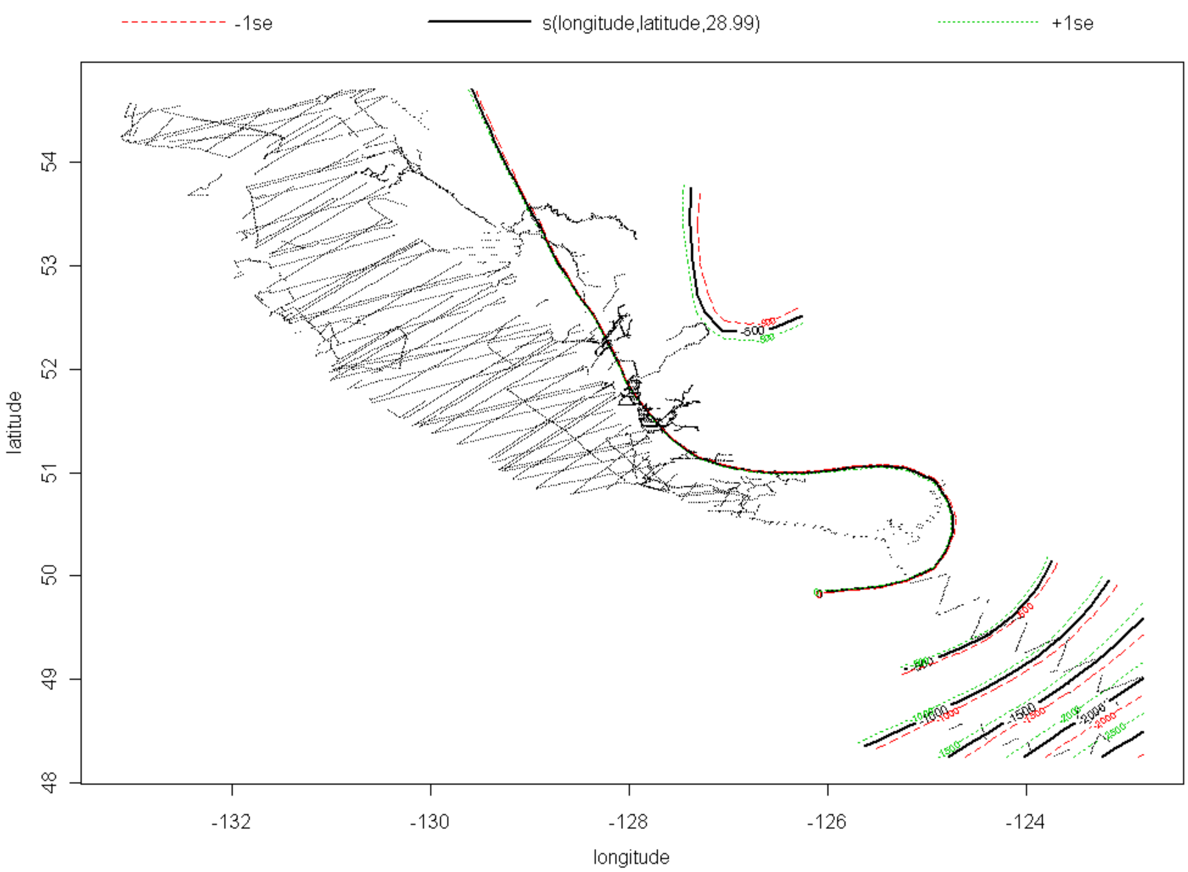

Figure 57. GAM terms plot for density surface of killer whale.

Minke whale

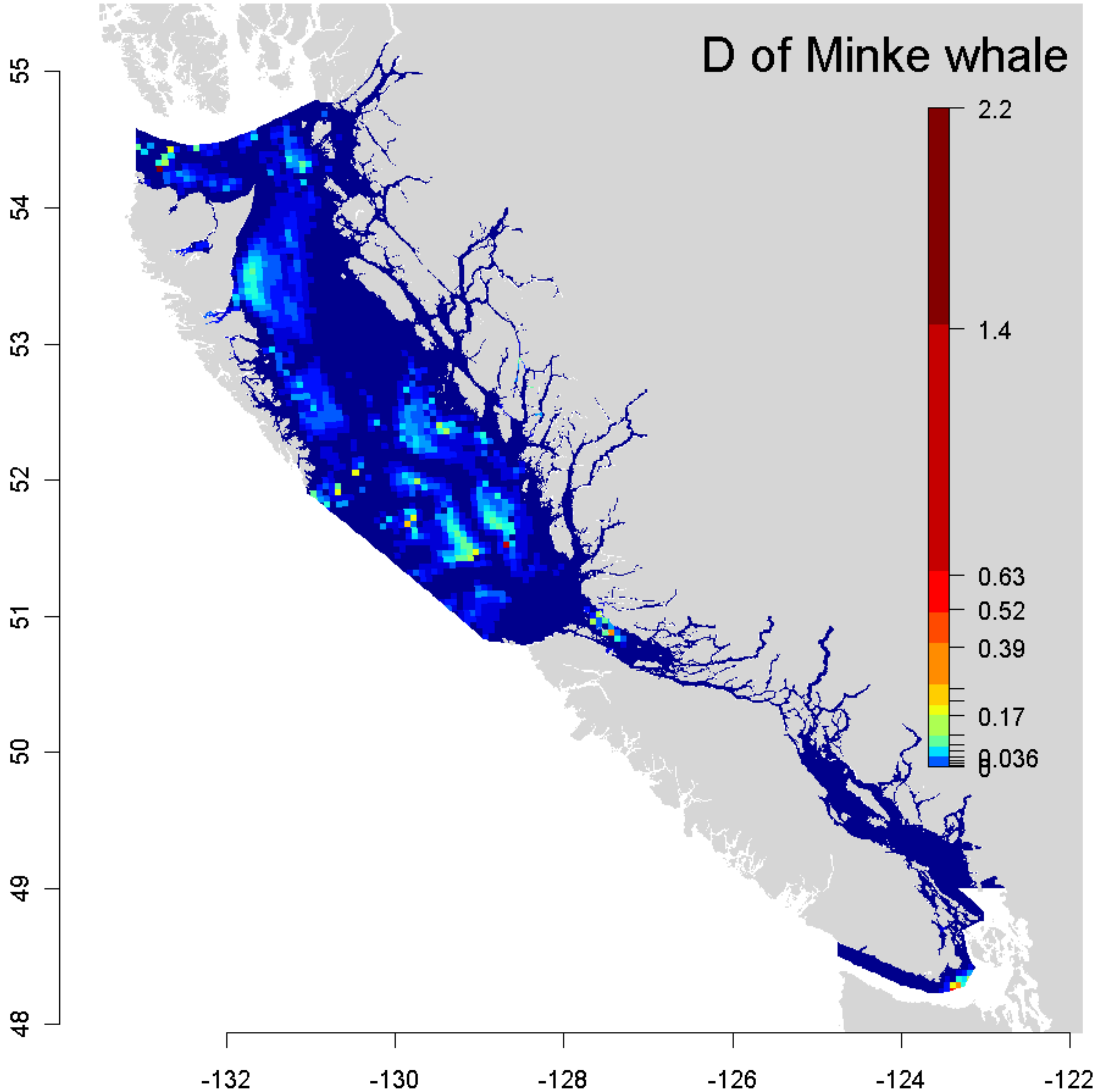

Figure 58. Density surface model for minke whale, in # of individuals per square kilometer.

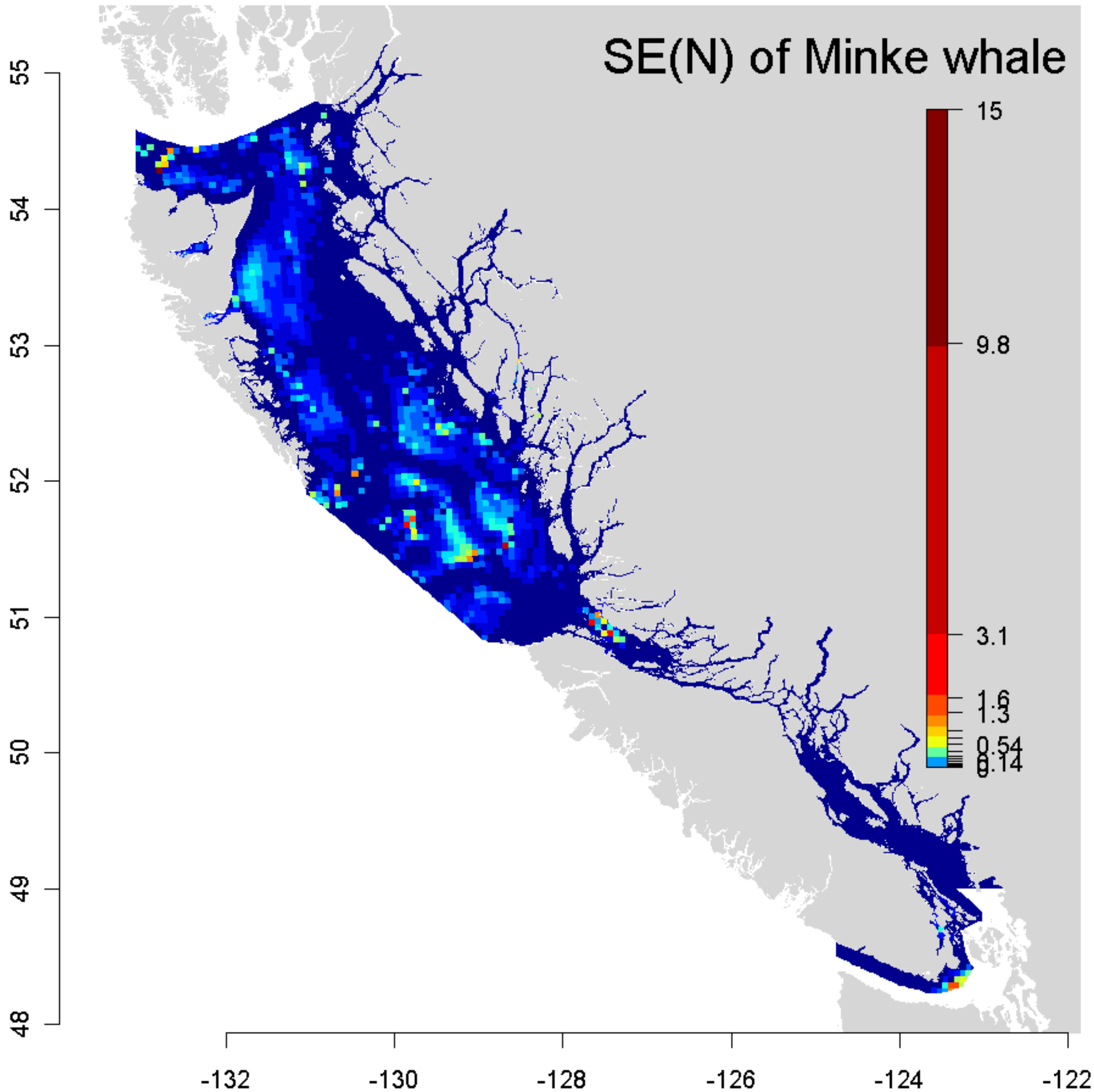

Figure 59. Standard error for density surface model of minke whale.

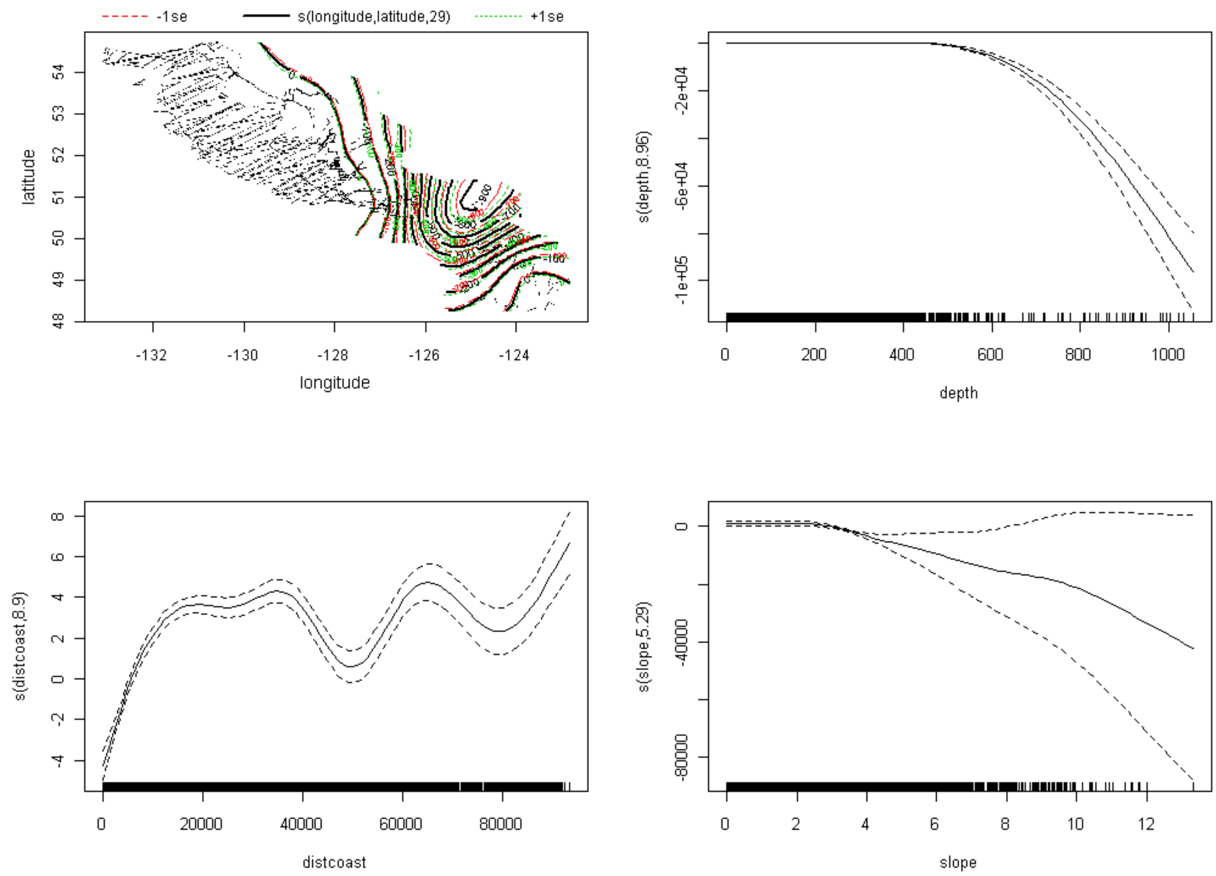

Figure 60. GAM terms plot for density surface model of minke whale.

Harbour seal, haul-out

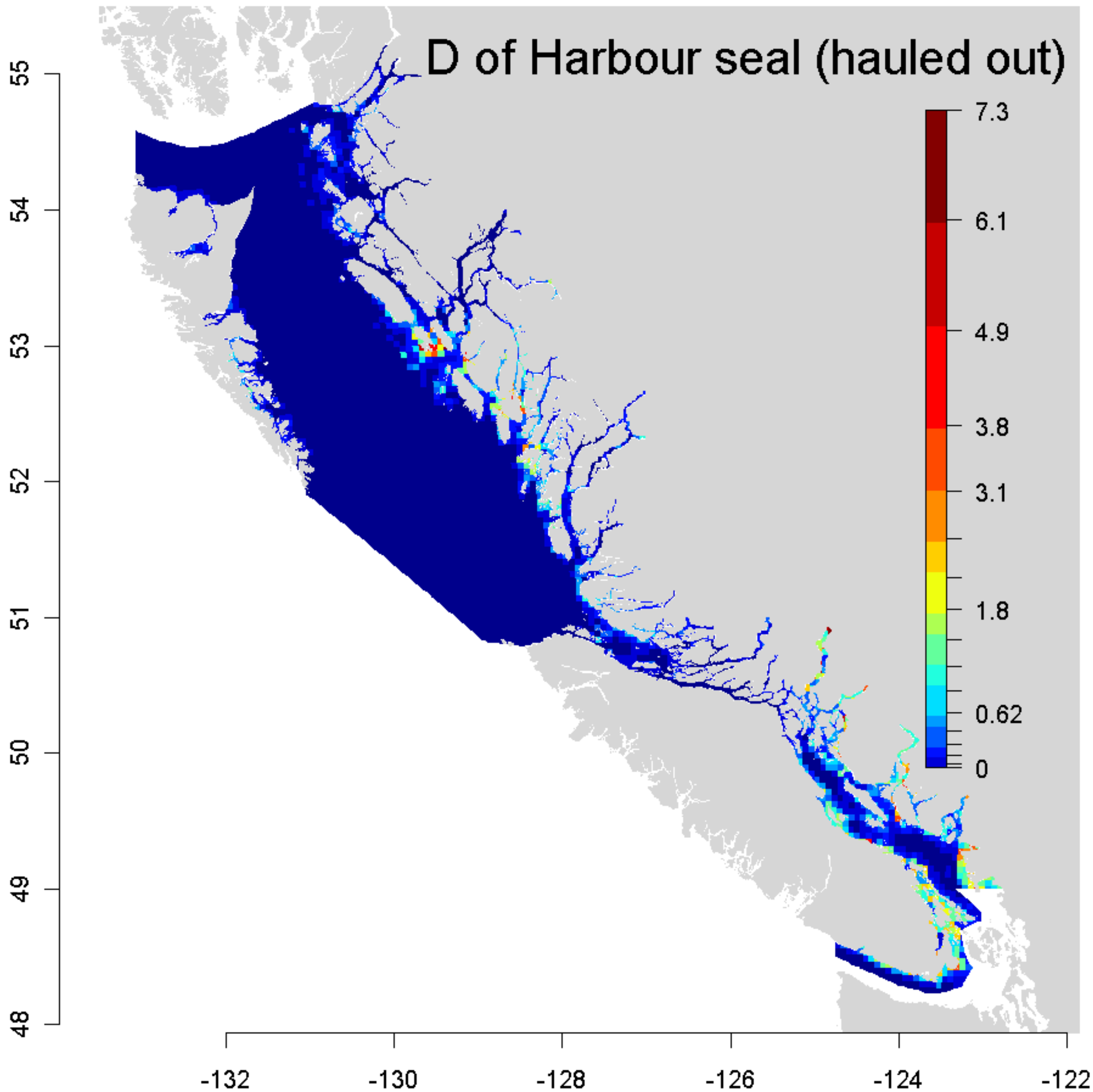

Figure 61. Density surface model of harbour seal, haul-out, in # of individuals per square kilometer.

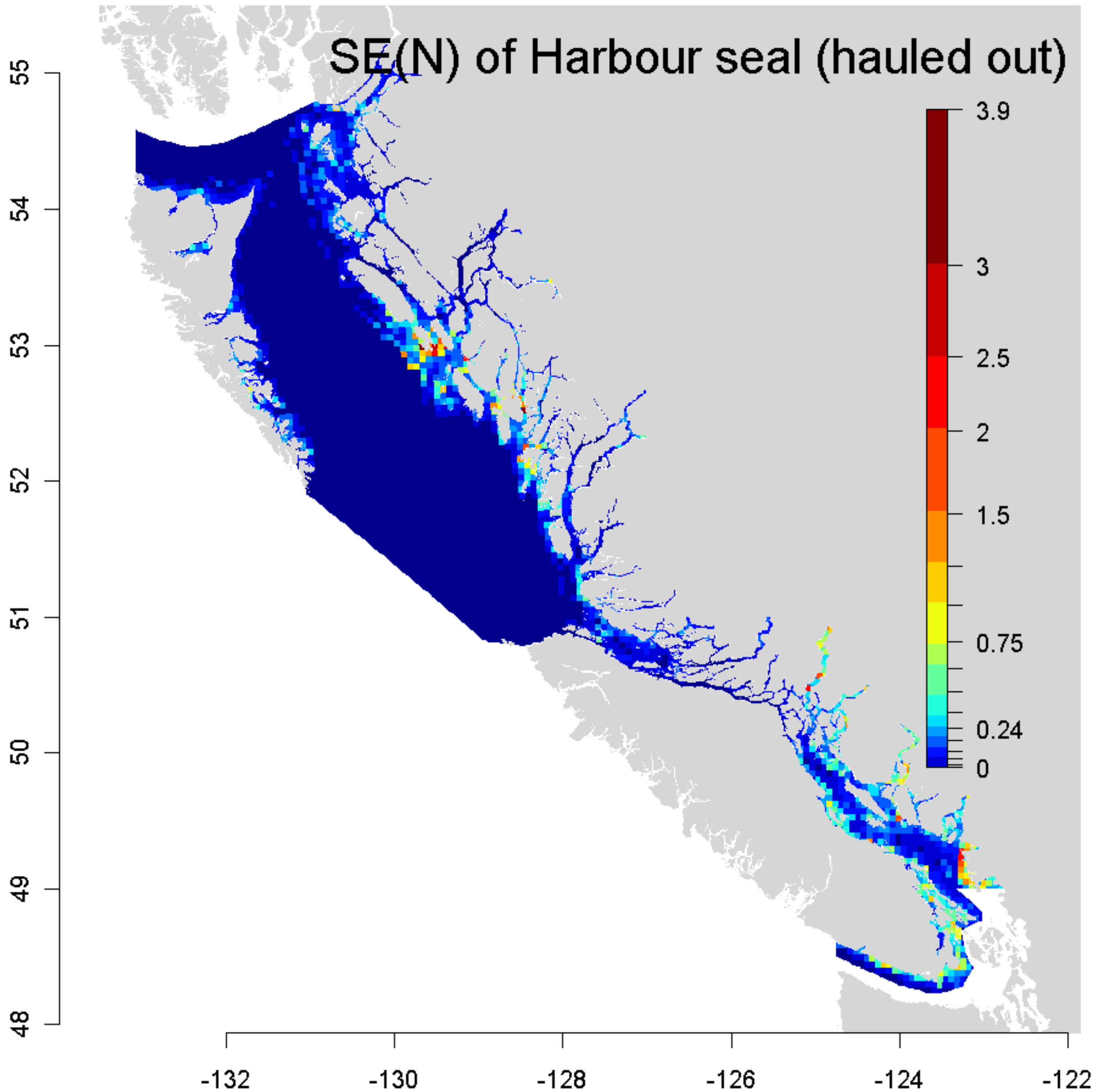

Figure 62. Standard error for density surface model of harbour seal, haul-out.

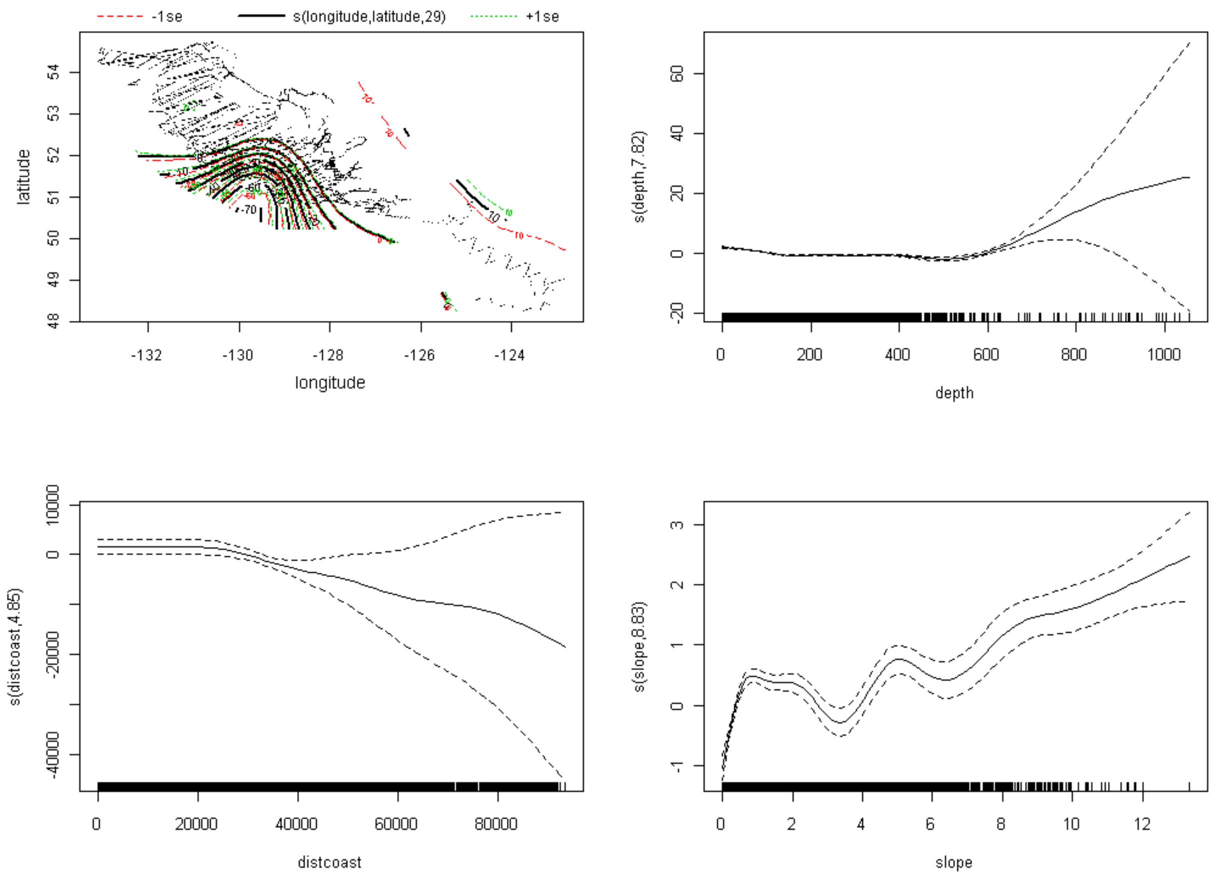

Figure 63. GAM terms plot for density surface model of harbour seal, haul-out.

Harbour seal, in-water

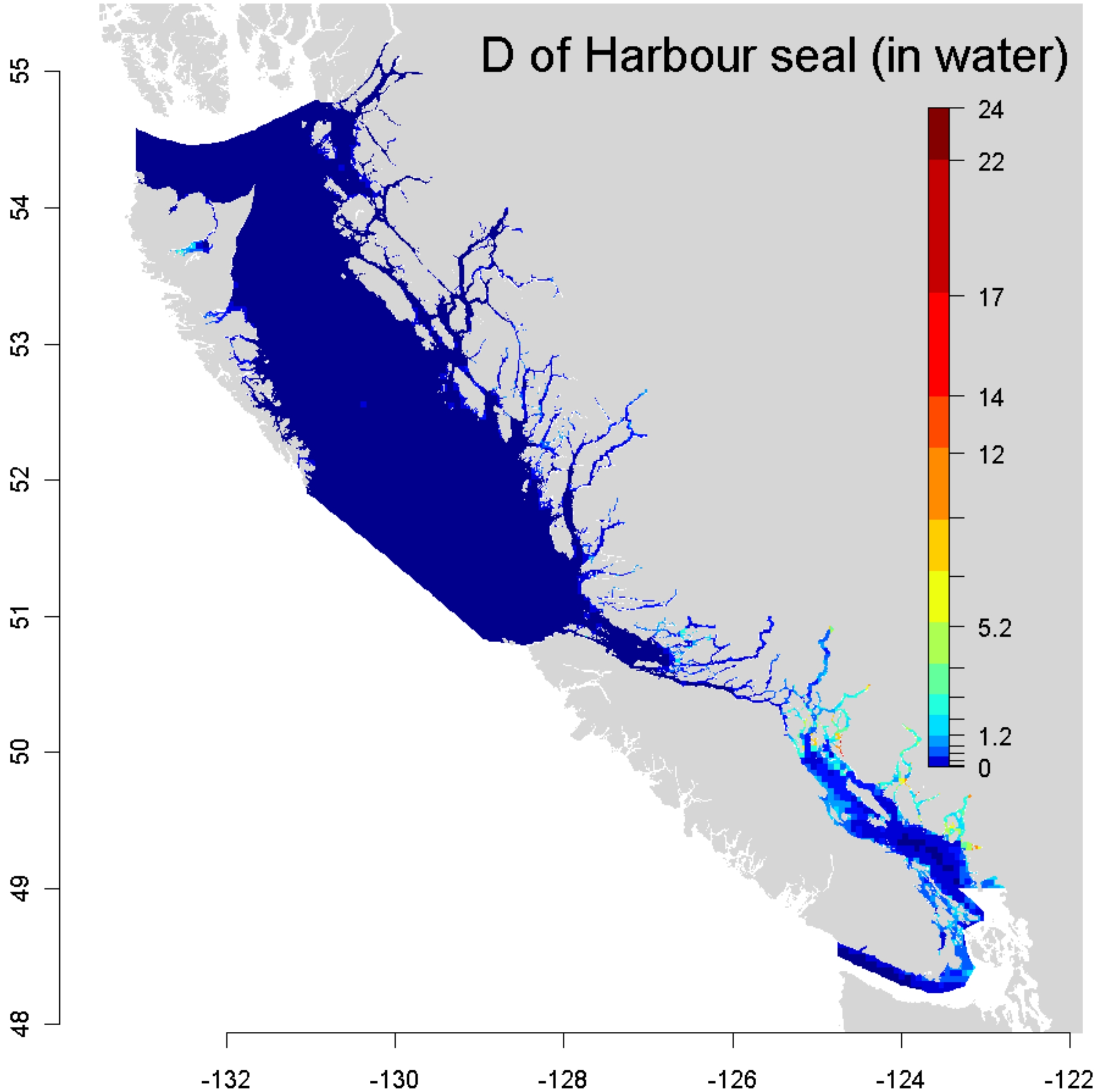

Figure 64. Density surface model for harbour seal, in-water, in # of individuals per square kilometer.

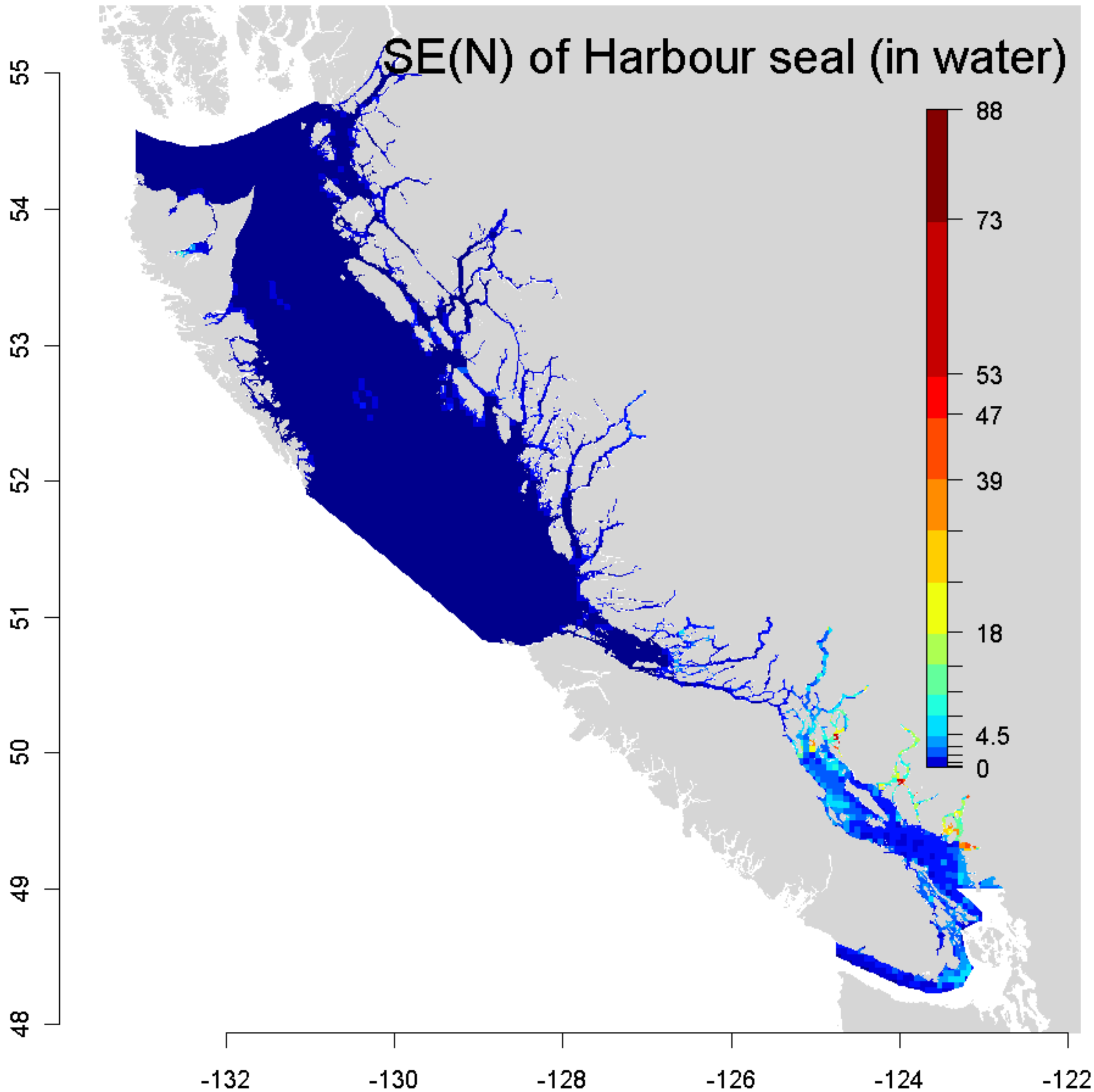

Figure 65. Standard error for density surface model for harbour seal, in-water.

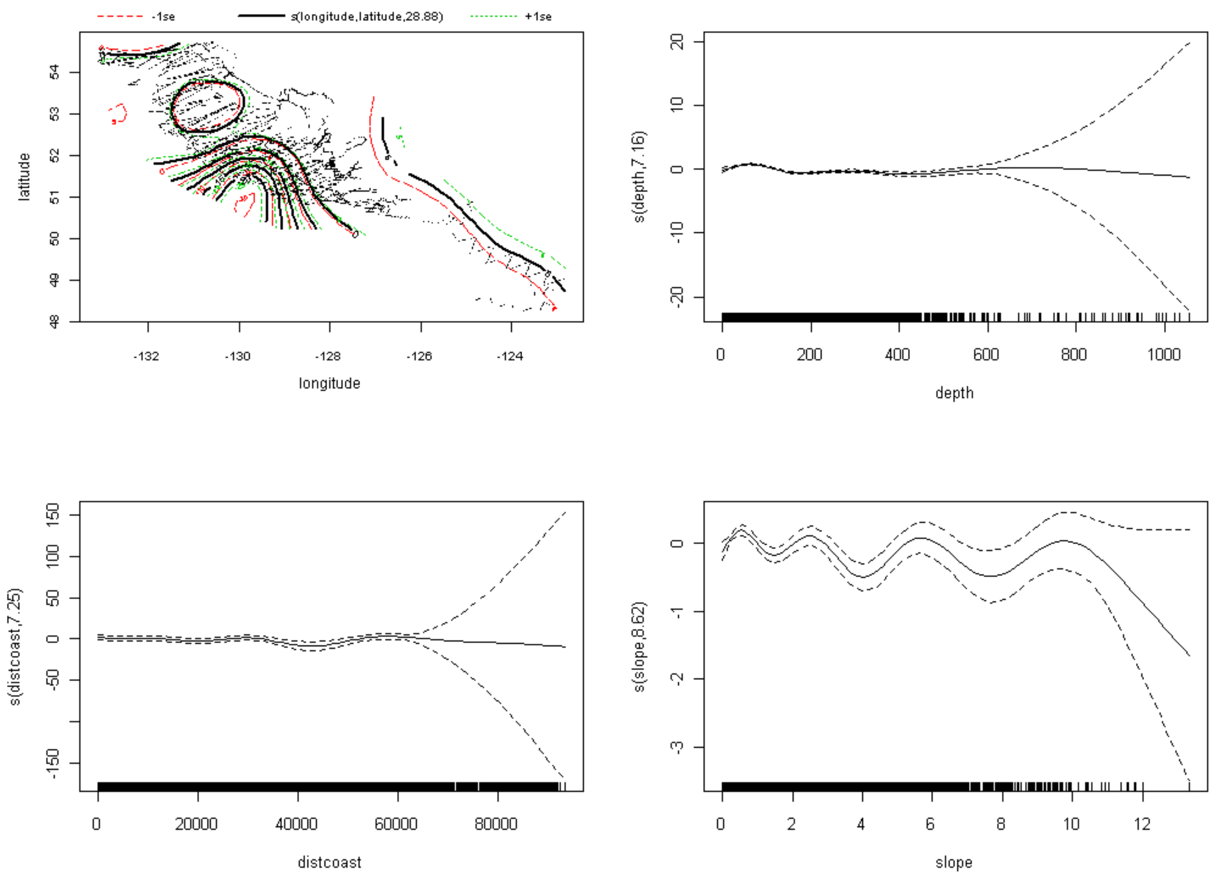

Figure 66. GAM term plots for density surface model for harbour seal, in-water.

__Stellar sea lion, haul-out__

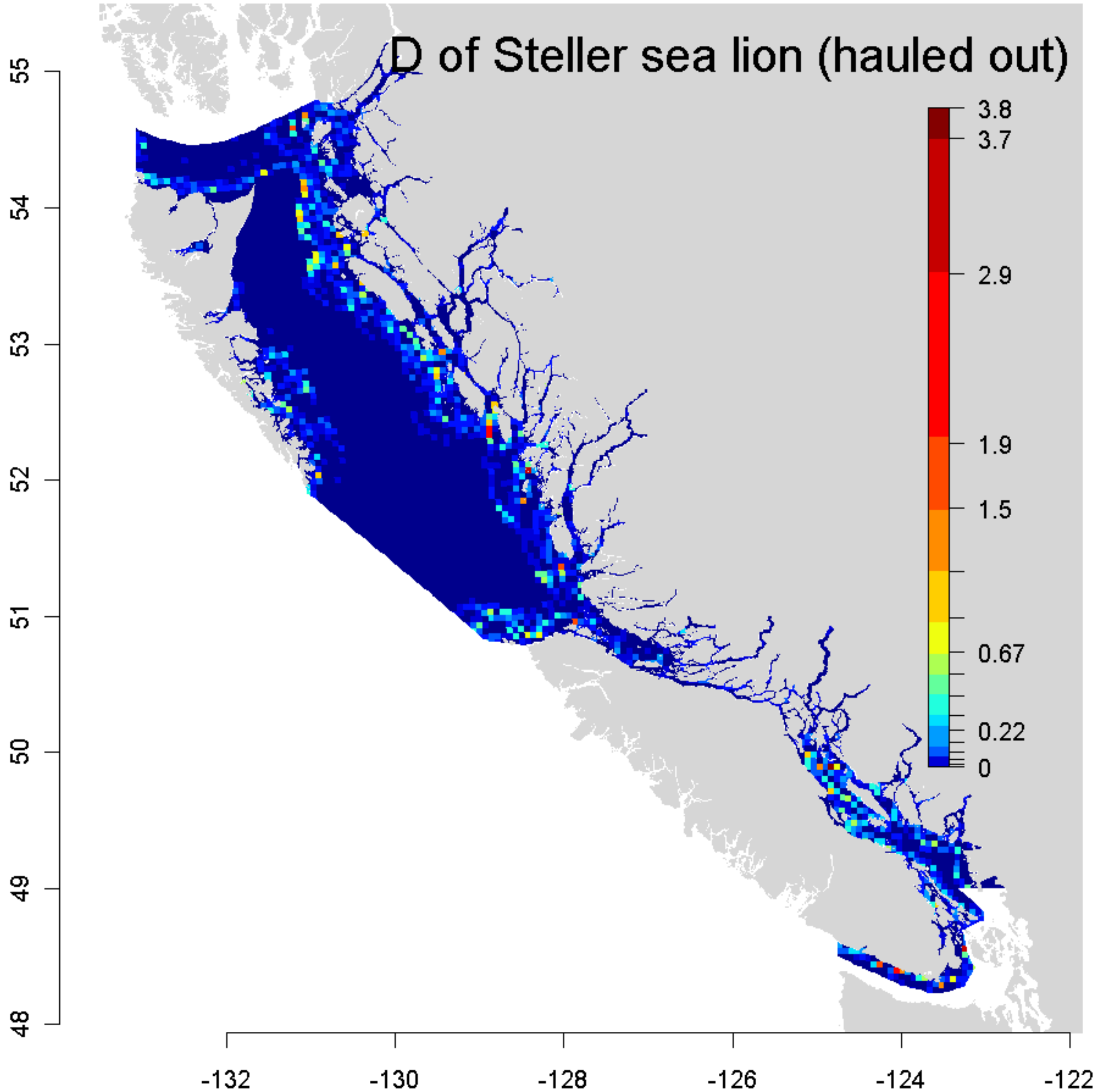

Figure 67. Density surface model for Steller sea lion, in # of individuals per square kilometer.

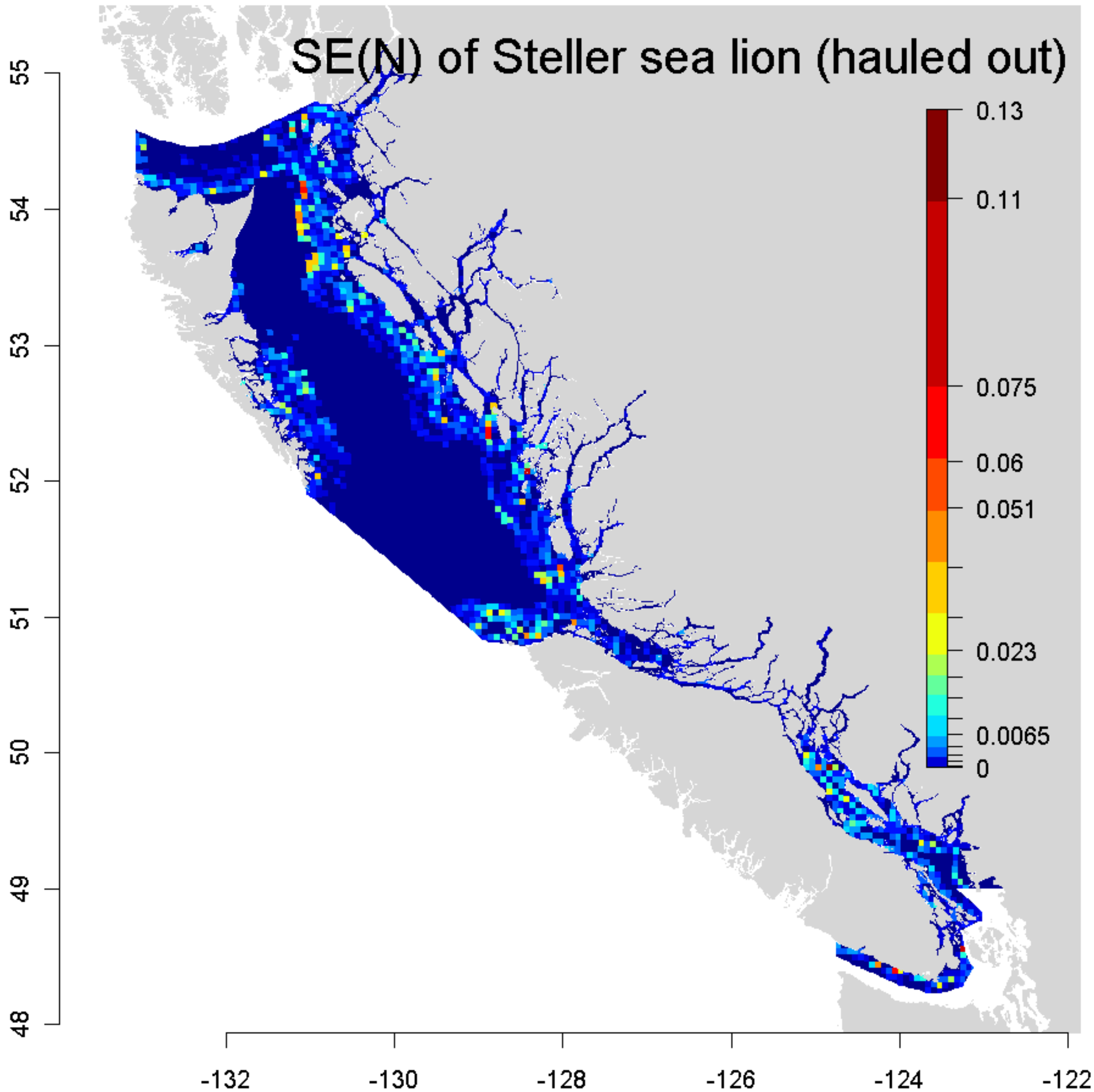

Figure 68. Standard error for density surface model of steller sea lion, haul-out.

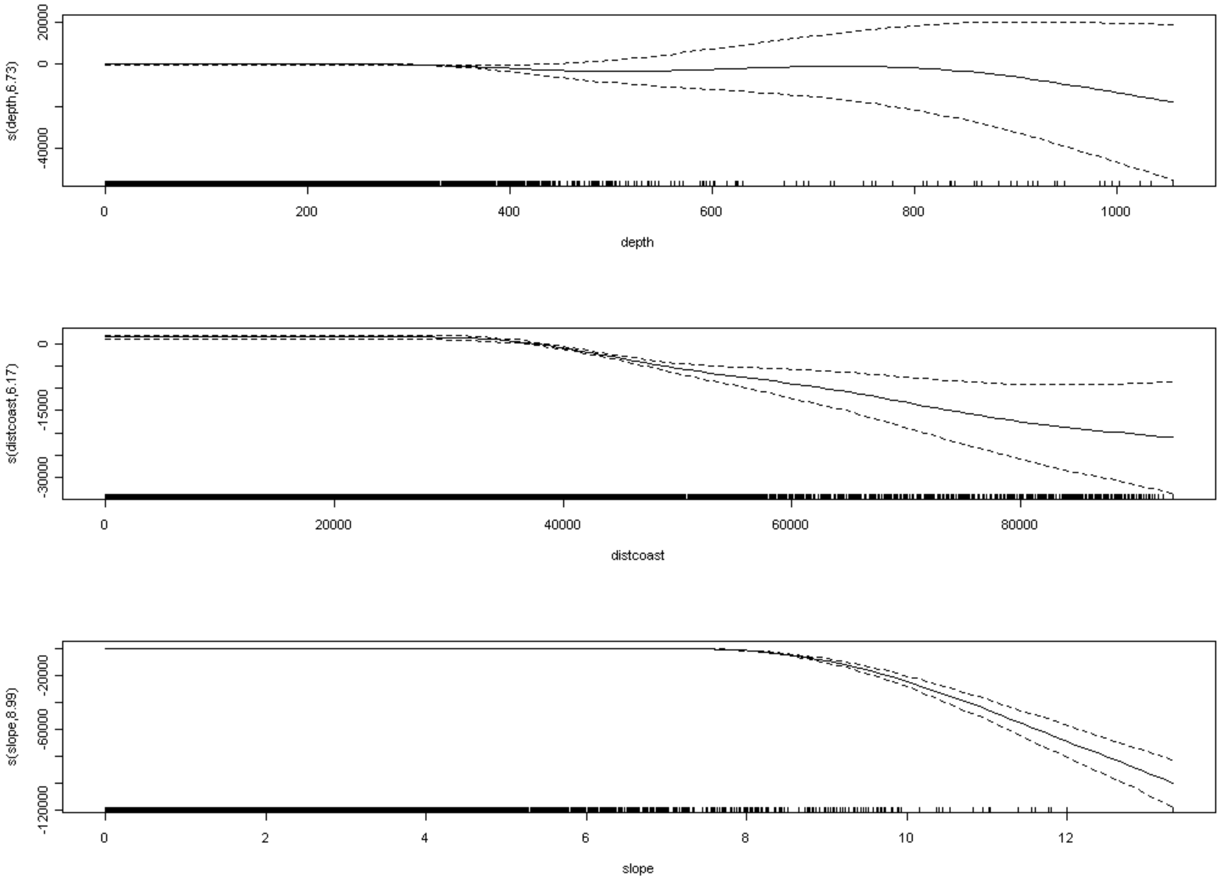

Figure 69. GAM terms plot for density surface model of Steller sea lion, haul-out.

Steller sea lion, in-water

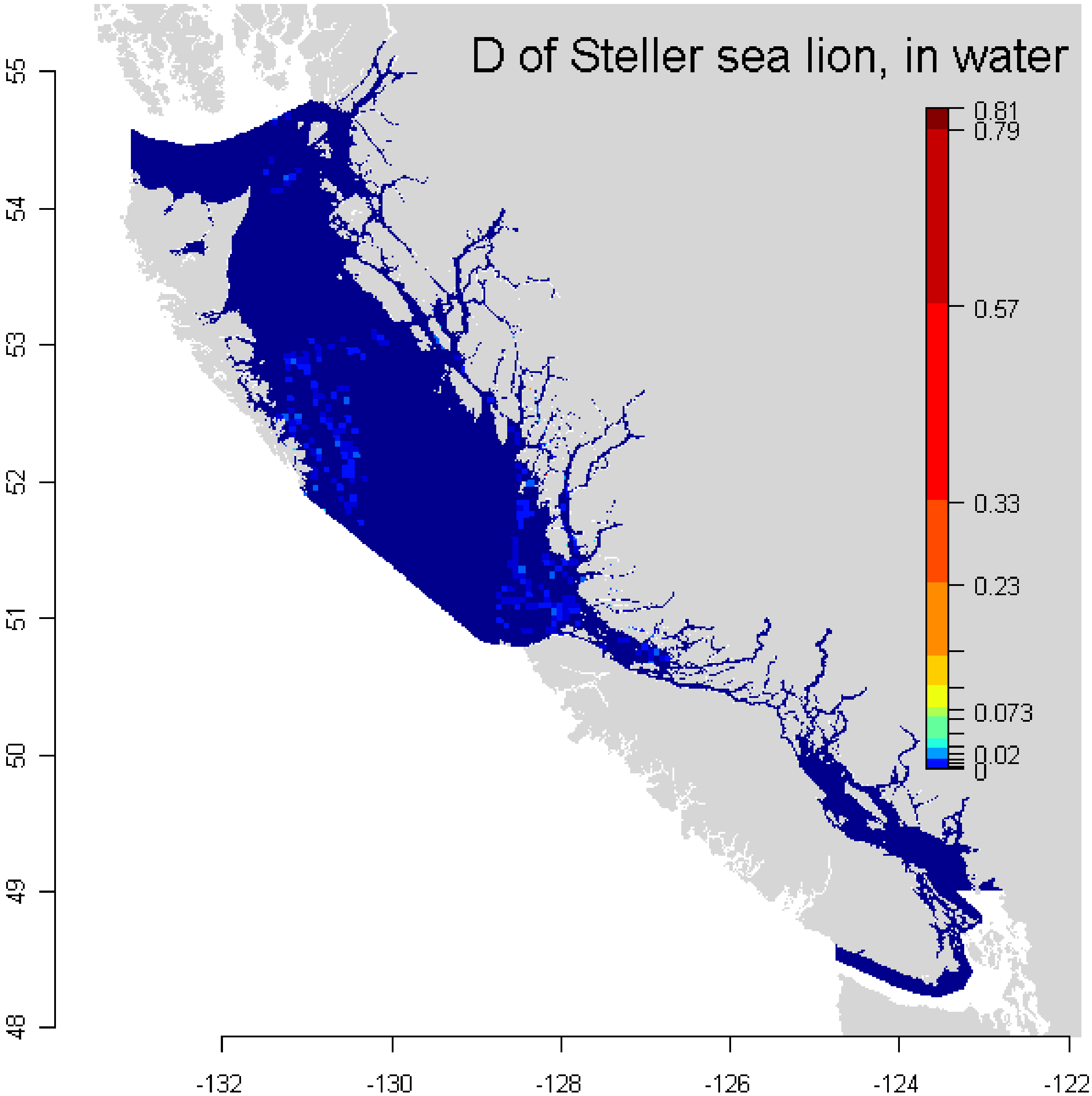

Figure 70. Density surface for Steller sea lion, in-water, in # of individuals per square kilometer.

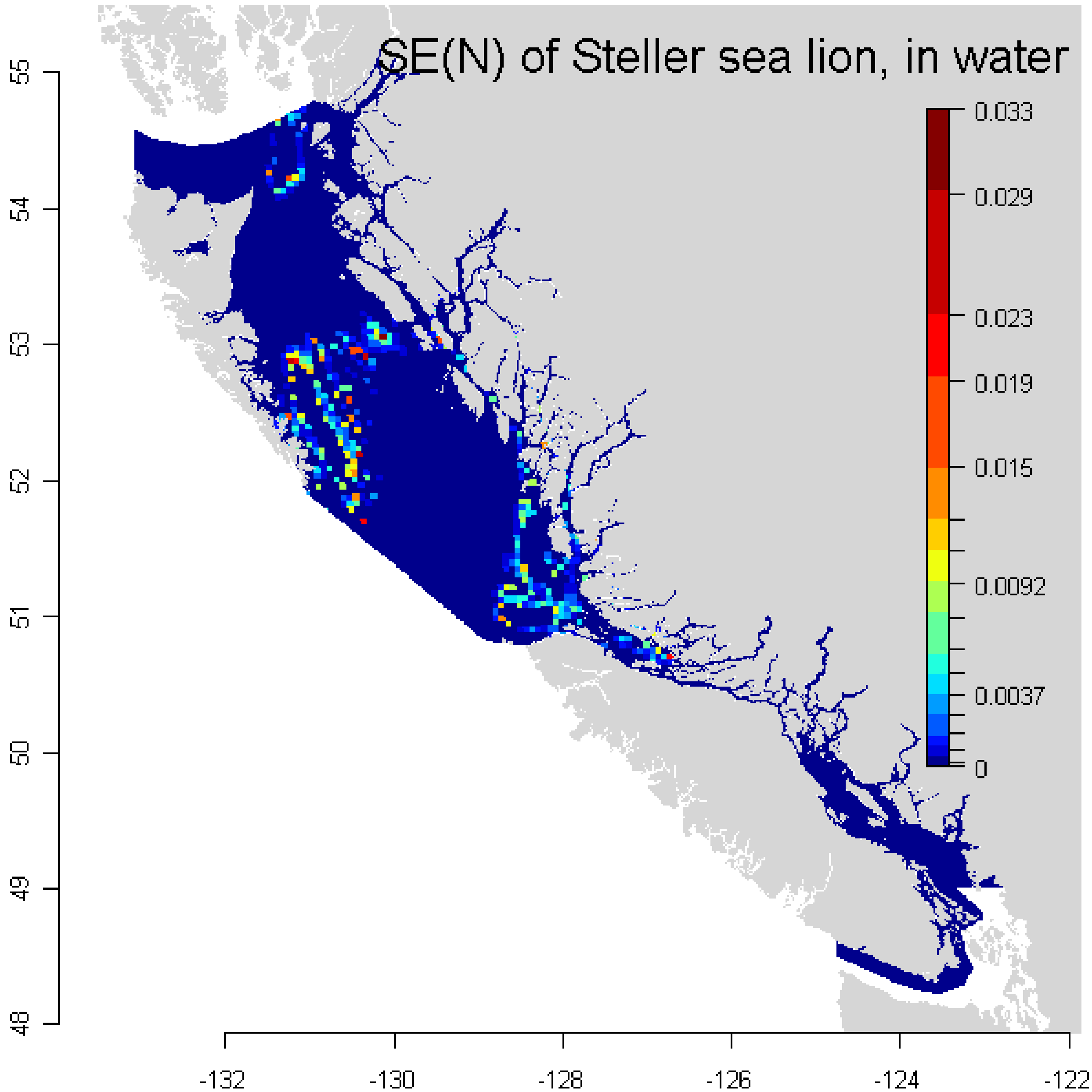

Figure 71. Standard error for density surface model of Steller sea lion.

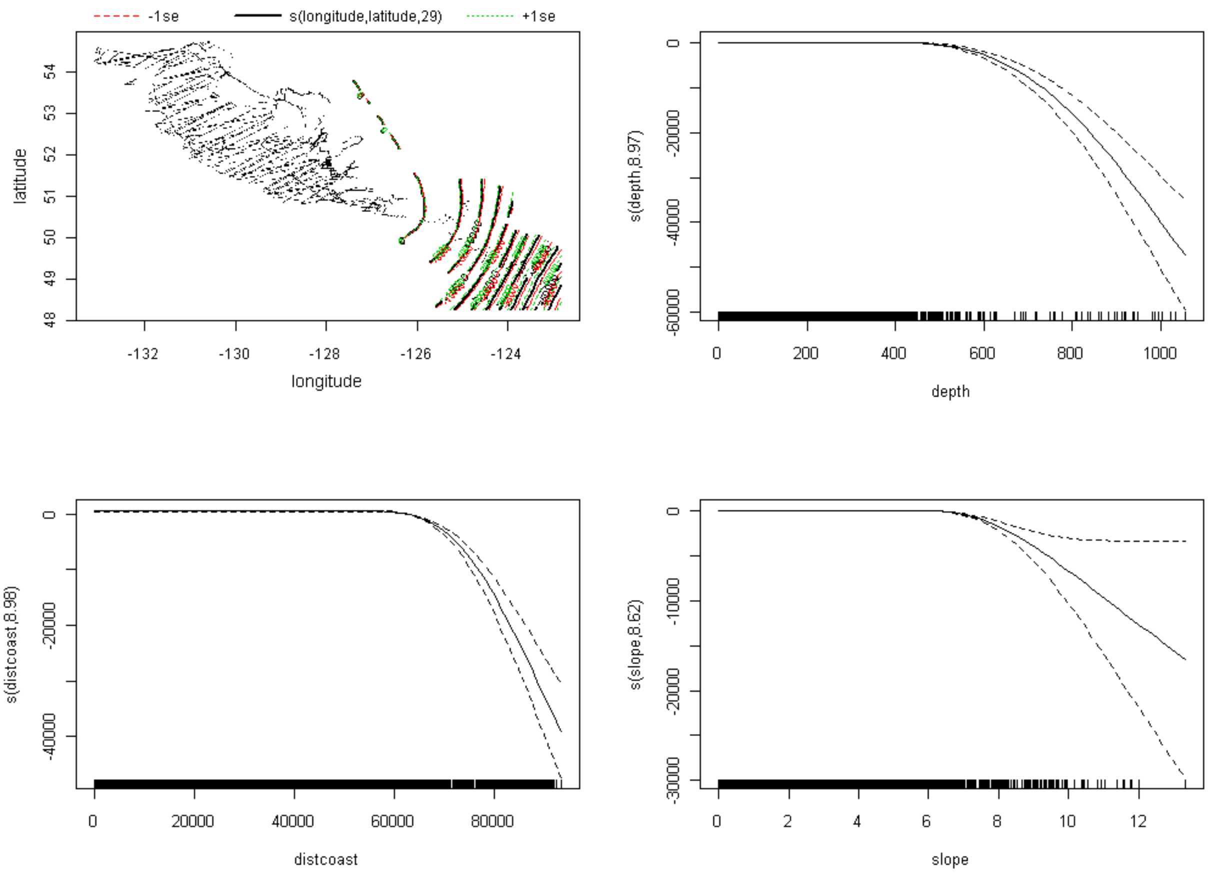

Figure 72. GAM terms plot for density surface model of Steller sea lion, in-water.

Elephant seal

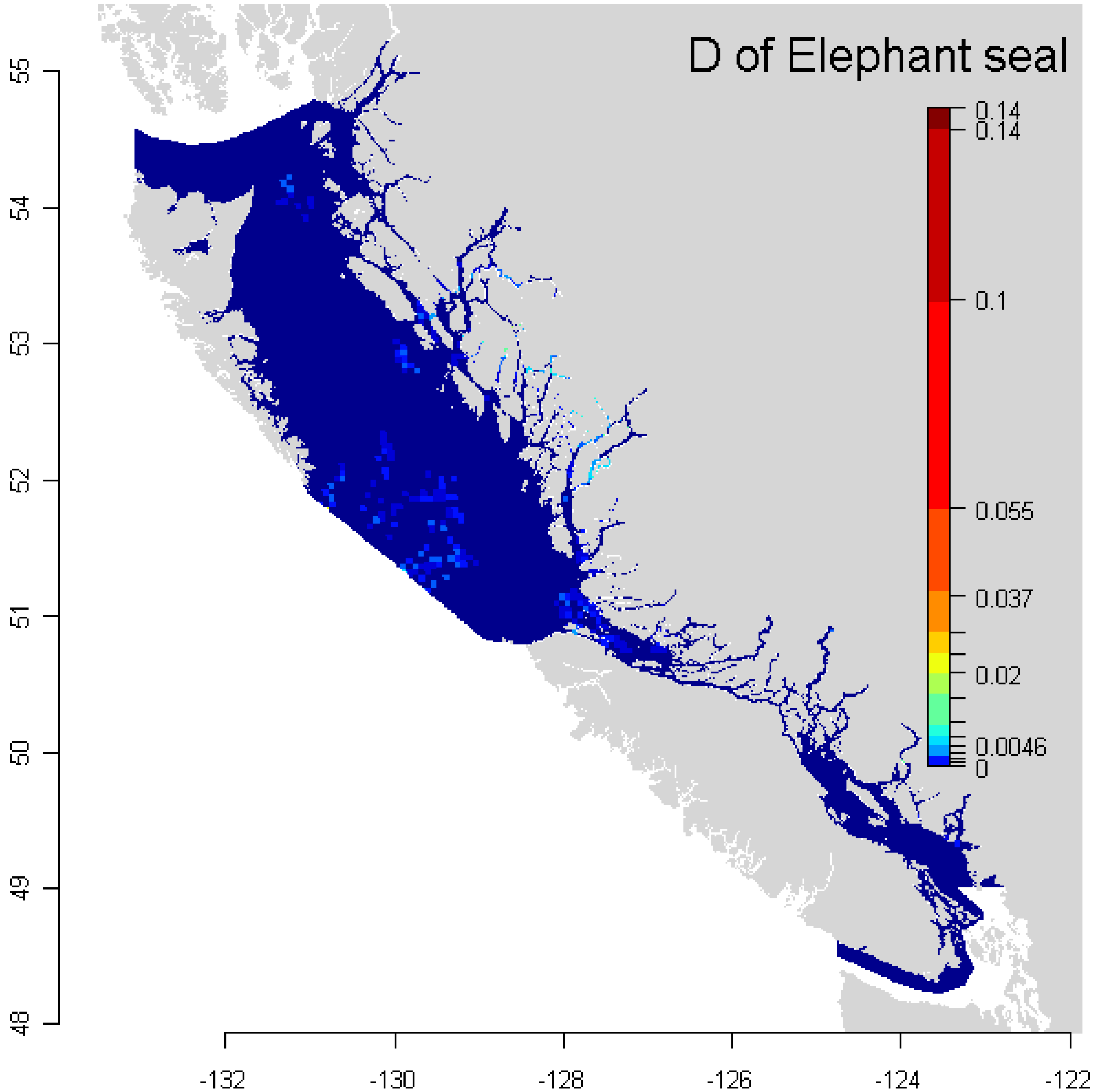

Figure 73. Density surface for elephant seal, in # of individuals per square kilometer.

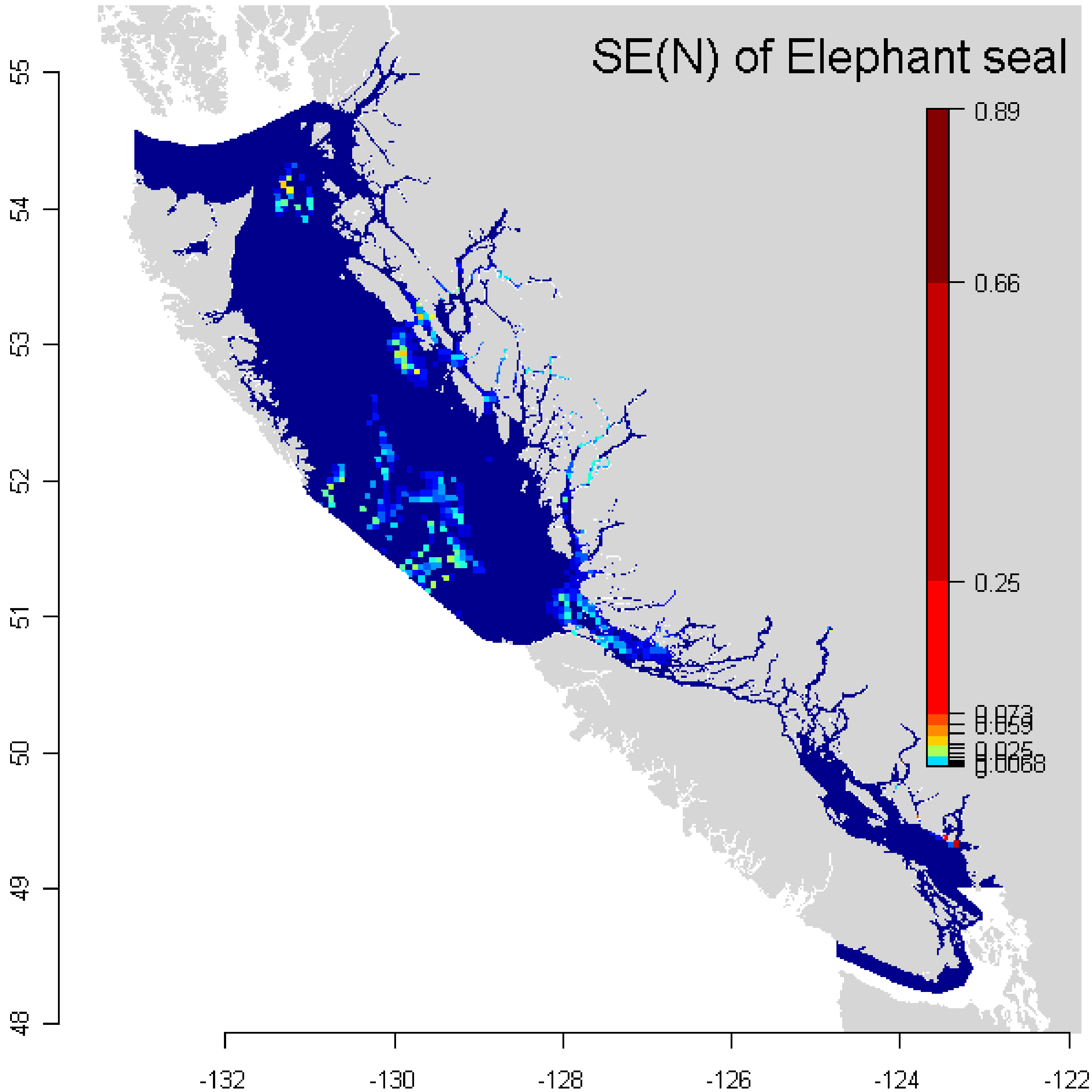

Figure 74. Standard error for density surface model of Steller sea lion.

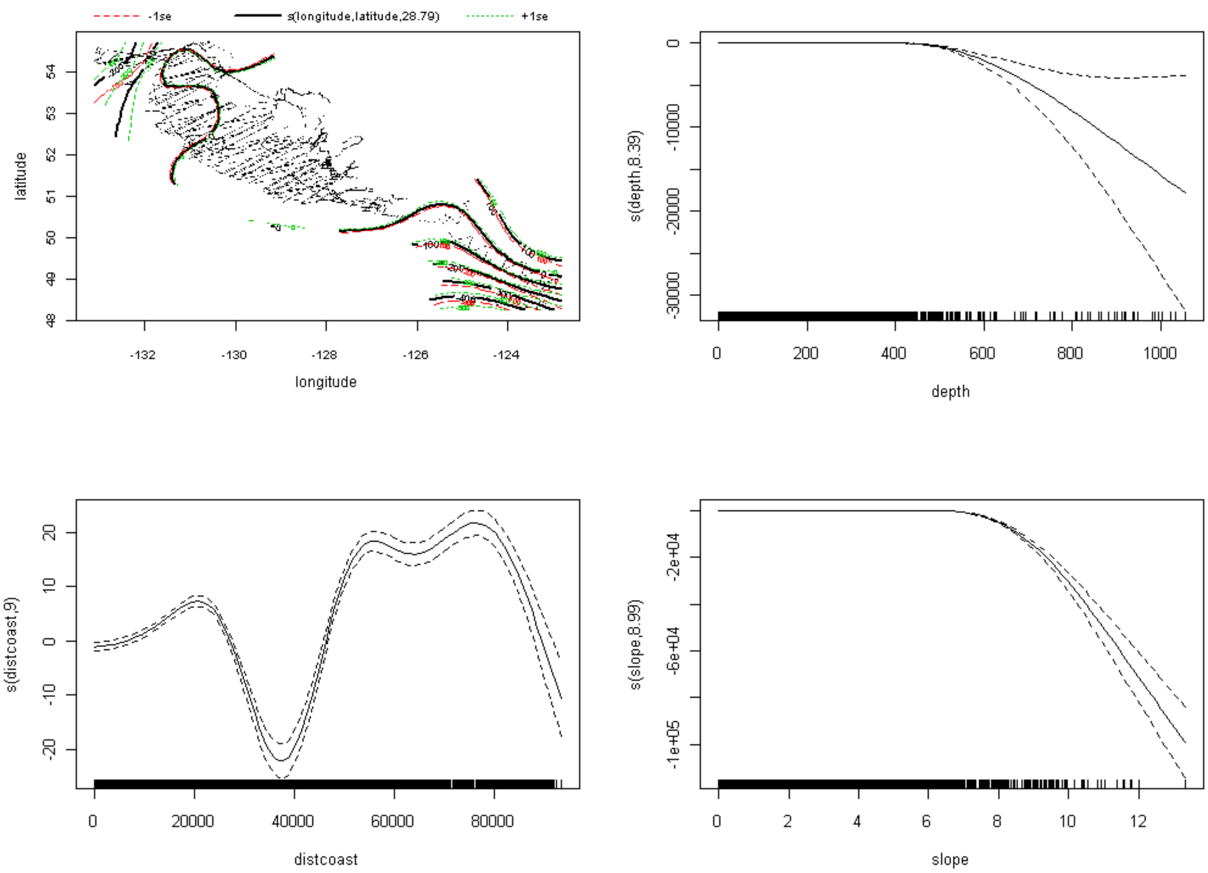

Figure 75. GAM terms plot for density surface model of elephant seal.