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Correlating sea otter density and behavior to habitat attributes in Prince William Sound, Alaska: A model for prediction

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CORRELATING SEA OTTER DENSITY AND BEHAVIOR TO HABITAT ATTRIBUTES IN PRINCE WILLIAM SOUND, ALASKA: A MODEL FOR PREDICTION

BY

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

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University of New Hampshire, December, 2006

As a benthic foraging marine mammal, sea otters *(Enhydra lutris)* present a unique opportunity for conducting a quantitative assessment of behavior based on habitat use as well as developing a habitat based density model using GIS because of the sea otter's well defined habitat requirements. Several studies have documented sea otter behavior but none have calculated the probability of occurrence of a particular behavior based on habitat attributes. Previous predictive models of sea otter density have been constructed, however these models have excluded offshore habitat. Seven aerial surveys, that included offshore habitats, were conducted between 1995 and 2005 in western Prince William Sound to estimate distribution and abundance of sea otters (*Enhydra lutris*).

The location and densities of sea otters that resulted from these surveys were used to explore relationships between sea otters and habitat attributes, both nearshore and offshore. These relationships described in western Prince

William Sound were then used to construct habitat based models to predict sea otter carrying capacity and total abundance at different spatial scales. The data from the aerial surveys were also used to quantify the relationship between a sea otter's behavior and the habitat attributes associated with the location of the animals when the behavior occurred.

Stepwise logistic regression was used to describe relationships between behavior, diving or not diving (assumed resting) and habitat attributes. Three subsets of the data were examined; all animals, all single animals without pups and all single animals with pups. Bathymetry was consistently significant (alpha = 0.05) in determining the probability of a behavior being diving or not diving, regardless of size of group or reproductive status. Group size was the first variable to enter the stepwise regression analysis of all available sightings, regardless of reproductive status, with bathymetry as the second and final variable. Among single animals with pups bathymetry was the first variable and distance to shore was the second and final variable to enter the model. Bathymetry was the only significant variable in the analysis of single animals without pups.

The aerial survey data from western Prince William Sound, AK, was used to create a predictive density model based on five habitat attributes; bathymetry, distance to the closest shoreline, distance to the closest protected shoreline, distance to the closest tidewater glacier and distance to the closest anadromous stream. The mean predictive density estimate was 2.0316/ km^2 with a total corrected population estimate within the survey boundaries of 16,441, with a range of 14,468 to 18,803 (alpha = 0.05).

Special attention was given to northern Knight Island, an area heavily impacted by the Exxon Valdez oil spill in 1989. Predicted densities within that area were 1.5792/km^2 with an estimated abundance of 384. The actual mean abundance estimate at northern Knight Island between 1995 and 2005 was 68 with a range of 34 to 102 (alpha = 0.05), illustrating a discrepancy between predicted estimates and of actual survey abundance estimates.

The analysis and results presented in this work give insight into the density and distribution variation of sea otters in Prince William Sound as well as contribute to the understanding of the sea otter's use of its nearshore habitat.

GENERAL INTRODUCTION

Sea otters (*Enhydra lutris*) were once hunted to near extinction until the implementation of the Fur Seal Treaty in 1911, protecting the otters from further commercial harvest. At this time, little pre-decline abundance data were available. Eleven remnant populations persisted, most of these in the Aleutian Archipelago, and they increased in size to eventually repopulate most available habitat between Prince William Sound and the Kuril Islands in Russia. The Rat Island group in the Aleutian Archipelago displayed the earliest and most extensive sea otter population recovery in Alaska (Kenyon 1969). The population at Amchitka, the largest of the Rat Islands, was thought to be at carrying capacity by the mid 1960s and was likely providing immigrants to the other islands (Kenyon 1969). By the 1980s most of the Aleutian Islands were re-populated (Estes 1990) with an estimated 55,000 to 74,000 otters (Calkins and Schneider 1985). However, subsequent skiff based surveys conducted in the early 1990s, in the Rat and Andreanof Islands, showed a rapid population decline (Doroff et al 2003). This rapid decline was apparently due to predation from killer whales (*Orcinus orca*) (Estes et al. 1998). Subsequent aerial surveys throughout the Aleutian Islands and Alaska Peninsula have identified the geographic extent of the decline to include most of the entire southwest stock of sea otters extending 1500km from near Kodiak Island to Attu Island (US Fish and Wildlife Stock Assessment Report 2002) (Fig.1).

Three distinct stocks of sea otters have been defined in Alaska, the southeast stock, the southcentral stock and the southwestern stock (Gorbics and Bodkin 2001). The US Geological Survey has conducted annual aerial surveys of sea otters in Prince William Sound as well as along the Kenai Peninsula/Cook Inlet. The Kenai Peninsula/Cook Inlet area borders the Alaska Peninsula to the east. Prince William Sound and the Kenai Peninsula/Cook Inlet area comprise much of the southcentral stock and this population is bordered on the west by the southwestern stock (Fig. 1). While the data from the Prince William Sound surveys have resulted in population size estimates, little work has been done to relate variation in abundance and distribution with habitat attributes. The Prince William Sound population is considered stable (Bodkin et al. 2002) and will serve as the base to construct a habitat based population density model. This will aid in understanding variation in the distribution and density of sea otters within Prince William Sound and may be applied elsewhere where little pre-decline population data exists.

There are three main objectives to this study. The first one is to build a model correlating sea otter densities, distribution and behaviors to various habitat attributes within Prince William Sound. The various habitat characteristics that have been chosen are described further and could be applicable to other sea otter populations outside Prince William Sound. Understanding variation in sea otter distribution may aid managers in decision making processes such as habitat protection or resource allocation during a natural or anthropogenic occurrence. The second objective to this study is the creation of a methodology for calculating densities based on habitat attributes utilizing abundance data. This methodology may be applied to other areas where population density and distribution may be unknown, uncertain or, as in some Aleutian Islands, where abundance prior to the recent decline may be unknown, but where some survey data have been collected. The model may change based on area of study; however, the model development process will remain the same and possibly aid in the comparison of available habitat across the sea otter's range.

The third objective is to calculate sea otter densities in areas where little data are available or where recovery or re-colonization has not yet occurred. For Prince William Sound this is accomplished by applying the model to the entire Sound and calculating densities in the northern Knight Island region (a highly impacted area from the Exxon Valdez oil spill) to determine if the current abundance, as measured by aerial survey results, is below the estimate of what the habitat can support based on the habitat model results. This may be because sea otters are still being impacted by the Exxon Valdez oil spill.

General Study Area

Prince William Sound is located in southcentral Alaska, along the northern curve of the Gulf of Alaska (Fig. 2). It is completely surrounded by the Chugach National Forest, while the Kenai Fjords National Park is located to the southwest (Morris and Loughlin 1994). Bathymetry varies within the Sound, with depths averaging about 200m with a maximum depth of 750m in the northern region

(Weingartner 2005). The Sound is considered a central estuarine basin with several fjords (Muench and Heggie 1978, Irons et al. 1988). The Alaska Coastal Current flows into the Sound through Hinchinbrook entrance on the southeastern end and exits on the southwestern end through Montague Strait (Muench and Heggie 1978, Weingartner 2005) (Fig. 3). The study area for this work is defined by the area encompassed by the aerial survey. The 100m bathymetry contour, which generally parallels the shoreline and varies in distance offshore, as well as a minimum distance offshore, regardless of depth, define the survey area. Prince William Sound supports large numbers of various forms of wildlife from marine birds and mammals to a diverse intertidal and subtidal community as well as several commercially important fish species (Spies et al.1996). However, there are relatively few shallow water areas that would provide suitable habitat for a variety of shallow benthic foraging species like sea otters and some marine birds (Irons et al. 1984).

General Methods

Aerial Survey

Aerial surveys of Prince William Sound were completed in 2005. Details of the survey method are described thoroughly in, *An aerial survey method to estimate sea otter abundance* (Bodkin and Udevitz 1999). Briefly, the survey design consists of two basic components. The first component is the strip transect. These transects are stratified by depth into high (0-40m) and low (40-

100m) density transects. Strips are 400m in width and are spaced at least 1.2km apart. Transects run perpendicular from the shoreline to the 100m depth contour or 400m offshore, whichever is greater. Allocation of strip survey effort was proportional to anticipated sea otter density with 80% of the effort in the high strata (<40m depth) and 20% in the low strata (>40m depth) (Fig. 4). The second component to the survey method is the intensive search unit or ISU. Intensive search units are sampled by flying five 400m diameter concentric circles within the strip transects and are conducted systematically to account for animals not seen on the strip so a correction factor can be applied to the strip counts (Fig. 4). Correction factors vary little across survey years (Table 1). These correction factors are applied to the final unadjusted model estimates. The ISU observer also records a behavior for each animal seen in the circle. Behavior is described as either diving (D) or not diving (N) (assumed resting). Data collected include group size, behavior (diving or not diving) and location. A group is defined as 1 or more otters separated by less than 4 meters (Bodkin et al. 2002). Flights were conducted during daylight hours in the summer (June to August) and only while the sea state was calm (Beaufort <2) and the ceiling was > 500ft. Therefore, behavior during hours of darkness, other seasons of the year or in weather conditions less than suitable for surveying cannot be assumed from this analysis.

The aerial survey method described above was created in this manner to take into account the sea otter's diving habits, both duration and depth. At the time of its implementation, it was generally believed that otter foraging depths were concentrated inshore of the 40m depth contour (Lensink 1962, Kenyon

1969, Estes 1981, Riedman and Estes 1990) with occasional records of animals foraging in waters as deep as 54-100m (Kenyon 1969, Newby 1975). Bodkin et al. (2004) conducted a study implanting time depth recorders (TDR, Wildlife Computers, Redmond, WA) into otters in Southeast Alaska. The study (Bodkin et al. 2004) created a more accurate estimate of sea otter dive depths than has been possible before. Data analysis revealed two types of divers, unimodal and bimodal. Most foraging dives occurred in less than 25m for the unimodal divers, while the bimodal divers exhibited dive behavior that occurred in the less than 20m contour as well as in the 35-55m contour (Bodkin et al. 2004). Besides depth, duration of the dive is an important factor in detection as well. Sea otter dive duration averages 74 seconds but can last for upwards of 200 seconds in California sea otters *(Enhydra lutris nereis)*, a sub-species of the sea otter (Ralls et al. 1995). However, during the Southeast Alaska TDR study, the average foraging dive lasted 85 seconds (Bodkin et al. 2004) and in 2006 an otter in Prince William Sound had a recorded dive of over seven minutes (Bodkin unpub. data 2006). Intensive search Units (ISUs) are flown to account for animals not seen on the strip count, but that might be underwater at the time of ISU initiation. For the time it takes to complete the minimum of five circles (230 seconds), a majority of otters diving would have surfaced during the ISU to be counted.

There are three variations to the aerial surveys in terms of spatial coverage; all of Prince William Sound (Fig. 5), western Prince William Sound (Fig. 6) and replicate surveys which consist of northern Knight Island (spill effected) and northern Montague Island (reference) (Fig. 7). Replicate surveys are conducted a minimum of four times and a maximum of five times within one survey year. Replicate surveys are identical in design to the other surveys, but are required for small areas where sighting and ISU data are too sparse for precise or accurate estimates of abundance. These surveys vary in spatial coverage as well as sampling intensity. Therefore, the spacing of the transects depends entirely on which survey is being conducted. All variations of the aerial survey design are encompassed within the survey boundaries (Fig. 8).

Surveys conducted in only the western portion of Prince William Sound as well as the Montague portion of the replicate surveys are used to predict location and density of sea otters in the remaining areas of the Sound to validate the model as well as illustrate areas of concern where sea otter densities are lower than the model predicts. The data from the western Prince William Sound survey were chosen because there are seven years of aerial survey data from western Prince William Sound as opposed to only three years of data from the entire Sound survey. One set of data from the Montague replicate survey for each corresponding year of the western Prince William Sound survey were utilized as well because Montague is considered a reference area, unaffected by the 1989 Exxon Valdez oil spill, and has relatively high sea otter densities that may aid in the predictive capabilities of the model into other areas within Prince William Sound that have high sea otter densities.

Geographical Information Systems (GIS)

The overall design of this study is to use existing sea otter aerial survey abundance and location data to explore the relationships between sea otter densities and carefully chosen habitat attributes. These attributes include: bathymetry, distance to closest shoreline, distance to closest protected shoreline, distance to closest tidewater glacier, distance to closest human population center and distance to closest anadromous stream (Table 2). It is well documented that sea otters forage nearshore in rocky and soft sediment habitats and feed almost exclusively on benthic prey (Kenyon 1969, Estes 1981, Estes et al. 1981, Estes 1989, Reidman and Estes 1990, Bodkin et al. 2004). However, this observation doesn't fully explain the variation is sea otter distribution throughout Prince William Sound. A map of the sea otter distribution throughout Prince William Sound clearly illustrates this variation (Fig. 9). Habitat features other than bathymetry and benthic sediment composition have largely been overlooked as influences that explain sea otter distribution and density. Therefore, a new approach was implemented. Frequency of adults counted during the survey were plotted against each variable separately a priori to examine relationships between habitat characteristics and number of adult sea otters observed during a survey (Fig. 10-15). Based on the plots of adult sea otter frequency based on depth bins, there was a negative trend. As depth increases, sea otter frequency decreased (Fig. 10). There is a similar trend shown between frequency of adult sea otters and closest distances to both the shoreline as well as closest distance to protected shorelines (Fig. 11 and 12), where sea otter frequency tends to decrease as distances from shorelines increased. However, there was a slight increase in the frequency of adult sea otters from the <800m distance to the closest shoreline bin to the <1000m distance bin. This spike may have indicated shallow areas offshore. There was little discernable pattern from the a priori plot of adult sea otter frequency based on distance to the closest tidewater glacier (Fig. 13). However, because of the abundance of tidewater glaciers within Prince William Sound, tidewater glaciers were utilized in the model building process. The plot of adult sea otter frequency based on the closest distance to a human population center illustrated lower frequencies of sea otters close to the designated population centers, but increased steadily with a maximum frequency of adult sea otters at 5km from the closest human population center. Frequencies dropped rapidly as distance increased above 5km (Fig. 14). However, because this graph did not show a clear positive trend and there were marked decreases in animal frequencies as distances from population centers increased above 5km, there was an assumption that some other physical or biological factor dictated sea otter distribution based on closest distances to a human population center such as protection from inclement weather or marine productivity. The decrease in sea otter frequency as distance from anadromous streams increased (Fig. 15) may be due to the marine derived nutrients deposited in the nearshore by the decomposing carcasses of the fish, therefore potentially increasing benthic productivity. An alternative hypothesis for the negative trend between sea otter frequency and distance to the closest anadromous stream was that because of the high number of salmon streams evenly distributed within Prince William

Sound, distance to the closest anadromous stream could have correlated with distances to the closest shorelines. Without anadromous stream run size, power to detect the influence a large salmon run or a small run might have on the nearshore benthic community is decreased.

All survey results were digitized from paper maps using ArcGIS 9.1 (ESRI, Redlands, CA). Survey attribute tables consisted of several fields that include: transect number, number of adults in each group, number of pups in each group, total number observed on each transect (adults + pups), date, and notes. Geographically referenced shoreline data for Prince William Sound was obtained from US Geological Survey shoreline data and is the same shoreline layer used to create the aerial survey transects. Bathymetry data were obtained from the NOAA Geophysical Data Systems for Hydrographic Survey Data and is a categorical variable with 8 levels, 0-20m, 20-40m, 40-60m, 60-80m, 80-100m, 100-120m, 120-200m and >200m. Anadromous stream locations were obtained from the Alaska Department of Fish and Game Fish Distribution Database (FDD 2006). While several studies have taken place to understand the role of anadromous fish, particularly salmon, in the transport of marine derived nutrients (MDN) into terrestrial ecosystems (Ben-David et al. 1997, Ben-David et al. 1998, Bartz 2002, Stockner 2003) relatively little work has been done to examine the effect of salmon derived nutrients on the nearshore environment. Some populations of salmon, particularly chum (*Oncorhynchus keta)* and pink (*Oncorhynchus gorbuscha)* salmon, travel shorter distances to spawn and this allows the nutrients to be distributed to the estuaries and nearshore habitats

(Stockner and Ashley 2003). A majority of the salmon runs in Prince William Sound are composed of pink and chum salmon (Moffitt and Merizon 2006). Distance to the nearest salmon stream was calculated for each group of sea otters and used as a covariate of sea otter density from this dataset. Because the size of each salmon run is not available from this dataset, the lack of scale in salmon run size will likely reduce the power to detect the influence a large salmon run might have on the nearshore benthic community versus a small run, and whether these factors influence nearshore sea otter habitat use and productivity.

Data pertaining to human population centers in Prince William Sound were obtained from the Alaska State Geo-Spatial Data Clearinghouse (ASGDC 2005a) created by the Alaska Department of Community and Regional Affairs (ADCR 1998). The population centers within Prince William Sound are: Cordova, Valdez, Whittier, Tatitlek, Eyak and Chenega Bay (Fig. 3). There are ferry services between Valdez, Whittier and Cordova that lead to potential boat traffic increases near harbors. Whittier is the closest boat harbor access to Prince William Sound from Anchorage on the road system in Alaska. Due to the large human population in Anchorage as well as the number of tour operators and cruise ships that come and go from Whittier, the town contributes to increased boat traffic within Prince William Sound. The remaining communities of Eyak, Tatitlek and Chenega Bay are primarily Native Alaskan villages. The Alutiiq people populate the majority of Tatitlek and Chenega Bay. Eyak is populated mainly by the Eyak Athabascan people (Alaska Department of Commerce, Community and

Economic Development 2006). Harvest of sea otters for subsistence purposes is provided for under an exemption to the U.S. Marine Mammal Protection Act (MMPA) of 1972 (Public Law 92-522) and there are no limits to the harvest. While it is likely that subsistence hunting has not contributed to the decline of sea otters in other areas of its range (Burn 2005), it is possible that the location of human population centers effects variation in the sea otters density and distribution.

Protected areas are potentially important by providing sheltered waters from storms or inclement weather. Protected shorelines are defined by the protection offered from prevailing winds in this analysis. NOAA historical data was taken from buoy # 46061. The historical wind data from buoy #46061 was used to determine prevailing winds in the region based on average direction over 11 years (1995-2005) during the months of July and August. The average wind direction was calculated to be from the southeast. From this calculation, protected bays were determined by shifting the Prince William Sound coverage on the x- and y-axis to shadow protected shorelines from prevailing winds. These shorelines were then used to create a new raster layer for distance measurements from observed sea otter locations.

Location of tidal glaciers are potentially important because of the benthic community they support (Hoskin 1977, Carpenter 1983, Feder and Jewett 1987). Glacier location data were obtained from the Alaska State Geo-Spatial Data Clearinghouse (ASGDC 2005b) and created by the Alaska Department of Natural Resources (ADNR). Prince William Sound has over 40 glacial fjords, 20 of these glacial fjords are tidewater glaciers (Molnia 2001). These 20 tidewater glaciers reside in the western and northwestern region of the Sound (Fig. 3). Tidewater glaciers were digitized into ArcGIS. Distance measurements from observed sea otter locations to these habitats were calculated. Tidal glaciers exhibit slower sedimentation rates than turbid outwash glaciers because the terminus of a turbid outwash glacier does not reach the sea (Lethcoe 1987). The turbid outwash glaciers tend to be less productive because of increased sedimentation rates (Weslawski et al. 1995, Weslawski et al. 2000, Zajaczkowski and Legezynska 2001) as well as increased sediment loads (Carpenter 1983). High sedimentation rates from glacial streams inhibit mussels (Mytilus sp.), a prey item of sea otters, from settling as well as decreasing the distribution of other sessile organisms (Feder and Shaw 1986). There is evidence that tidewater glaciers contribute to increased biological production compared to turbid outwash glaciers (Hoskin 1977, Carpenter 1983, Feder and Jewett 1987). The increase in production and abundance of food sources may influence where sea otters are located (Irons et al. 1988). However, compared to areas with little sedimentation flux, biomass is considerably lower in the glacier fed water bodies, regardless of glacier type (Hoskin 1977).

For analysis, all data layers are converted into rasters. All distance rasters were created using ArcGIS Spatial Analyst Cost Distance tool (ESRI, Redlands, CA). By using the Cost Distance tool, the grid representing water was used as the “cost” grid, which allowed for the calculation of distances only across water bodies, excluding the land masses as possible routes of travel for sea otters.

Each raster represents a data layer such as sea otter locations, shoreline, bathymetry, etc. Each one of these rasters is made of an equal number and equal size of pixels or cells and all of these rasters are stacked or "snapped" to each other. When sample analysis is performed (Grid, ArcInfo Workstation, ESRI, Redlands, CA) all the values from each raster are exported for each cell. An example of a sea otter location and the various attribute data collected from the location is given in Figure 16.

A critical element in the model building process was the selection of the spatial scale of analysis. The pixel or cell size of 50m x 50m was chosen for two reasons. One was due to the transect lines. Only the area sampled by the observer while flying the transect lines are used in this analysis to build the model. It is difficult to represent a smooth transect line with cells, especially if the transect line does not necessarily run in the north-south direction or east-west direction. Because of this, if larger pixel sizes were created, the transects would become "stair-stepped" and not necessarily represent the surveyed area (Fig. 17). The second reason for the 50m x 50m pixel size was to optimize the availability of marine habitat. Prince William Sound has many small bays and inlets that would be classified as land, based on the land feature that might be present in the center of the cell (ESRI 2004) (Fig. 18), therefore these cells would be excluded from the analysis of sea otter habitat. Conversely, land masses that are not sea otter habitat could be classified as marine based in the same classification process mentioned above (Fig. 18). Opposing reasons for the 50m x 50m cell size are many as well. Certainly accuracy of the data, both the observed sea otter locations as well as the many habitat attributes used in the analysis, could come into question with this cell size. To overcome this issue of small cell size for the strip distribution and density analysis, a buffer was placed around each observed sea otter location. The buffer has a radius of 1.2km. The buffer size was chosen for two reasons.1) A sea otter study conducted in California showed an average daily movement of 0.13 – 1.15km per day (Kage 2004), inferring that a buffer size of 1.2km could encompass an individual's entire daily movements and 2) 1.2km is the minimum spacing between each high density transect. Therefore, each transect did not allow for the possibility of double counting an animal in the same day. The buffer surrounding each otter sighting was designated occupied sea otter habitat, however, only the attribute values associated with the 50m x 50m pixel that contained a sea otter were used in the analysis. If there were no additional observed sea otters in any pixels within this buffer, the values associated with each pixel were not used as zeros or non-locations in the analysis. Therefore, areas designated as zeros or non-locations for analysis, were a minimum of 1.2km from the nearest observed otter location (Fig. 19).

It was critical during analysis that occupied and unoccupied pixels were chosen in the same ratio as seen during the aerial survey. This was first discovered when individual survey years were used to calculate densities. Those results were compared to density calculations from the combined surveys. Large discrepancies existed between the single year survey density estimates and the combined survey density estimates. The discrepancy was due to the ratio of occupied to unoccupied pixels for the combined survey data. When the data had been combined, duplicate unoccupied pixels were discarded and occupied sea otter pixels values were summed. This methodology not only increased the ratio of occupied to unoccupied pixels that is actually observed during an aerial survey, but the resulting estimated density was also much higher. However, the desire was to utilize the combined survey data across years for analysis to generate one best model regardless of year instead of one best model per year. To overcome the issue of pixel ratios, duplicates were not discarded during analysis, nor were otter sightings summed if in the same pixel but observed during a different survey year. For example, a pixel may have been occupied in three of the seven surveys, unoccupied during two of the seven surveys and ignored because the pixel was within 1.2km of an occupied pixel for the remaining two surveys. The various habitat values for that pixel were entered into the analysis five times. The values were only entered five of seven times because of the ignored values when the pixels fell within the 1.2km survey buffer. Not only did this approach allow for an accurate occupied to unoccupied ratio, there was no overestimating abundance or density because of summing of data across years.

The same justifications as mentioned above were used to rationalize the small pixel size for behavior analysis.

Analysis

Methods for analysis vary for each of the following three chapters. Logistic regression and probability analysis was done in Chapter I. Poisson regression, Akaike's Information Criterion (AIC) and semivariograms were the methods chosen in Chapter II. Chapter III utilized data from Chapter II and compared density estimates from the best model explained in Chapter II for northern Knight Island to post-Exxon Valdez oil spill aerial survey estimates from the same area. Analysis methods are explained in detail in each chapter where they are first used.

CHAPTER I

CALCULATING BEHAVIOR PROBABLITIES OF SEA OTTERS IN RELATION TO HABITAT ATTRIBUTES

Introduction

As a benthic foraging marine mammal, sea otters *(Enhydra lutris)* are restricted in their use of habitat both by depth and distance to foraging depths. It is well documented that sea otters forage nearshore in rocky and soft sediment habitats and feed almost exclusively on benthic prey (Kenyon 1969, Estes 1981, Estes et al. 1981, Estes 1989, Reidman and Estes 1990, Bodkin et al. 2004). However, in some cases where shallow water extends several kilometers offshore, sea otters may be common (Kenyon 1969, Newby 1975, Reidman and Estes 1990).

Sea otters generally forage individually, but tend to rest in groups (Garshelis et al. 1984, Estes and Jameson 1988, Reidman and Estes 1990). Resting and foraging behaviors often occur in different locations (Shimek and Monk 1977, Reidman and Estes 1990). If kelp is present, resting areas are typically in the kelp beds, which are limited to relatively shallow habitats of <20m, to protect a resting otter from winds, rough water or currents (Kenyon 1969). Without the presence of kelp beds, the assumption was that an otter would desire finding an area that offers similar protection. Observed otters on Adak Island occupied nearshore, sheltered areas during inclement weather (Gelatt

1996). These resting groups are usually segregated by sex with male resting groups usually much larger than female resting groups (Reidman and Estes 1990), but with most habitats occupied by lower densities of females and territorial males (Ralls et al. 1985).

Because of their restricted diving capabilities and almost exclusive consumption of benthic prey (Kenyon 1969, Estes 1981, Estes et al. 1981, Estes 1989, Reidman and Estes 1990), sea otter foraging areas were thought to be largely within the 20m contour (Kenyon 1969, Reidman and Estes 1990) with relatively little foraging in depths >20m (Newby 1975). However, with recent Time Depth Recorder (TDR) (Wildlife Computers, Redmond WA) data from a study conducted in Southeast Alaska, a majority of foraging took place within the 30m contour. Foraging ranged to the 100m contour, with significant foraging of some individuals to the 60m contour (Bodkin et al. 2004).

The time of day the aerial survey was conducted should not have an effect on the ISU results. Time allocated to foraging seems to be positively related to sea otter density and negatively related to prey availability, allowing a conclusions that sea otter densities may respond to food availability. When prey are abundant, less time is required to meet energetic requirements from foraging and densities of sea otters may increase (Estes, et al. 1982, Garshelis 1983). Conversely, as densities increase prey may become more limiting, more foraging time may be necessary to meet energetic requirements and densities may decline. There are, however, several examples of daily activity patterns in sea otter activity. A study conducted on Amchitka Island, Alaska illustrated that the majority of feeding occurs during daylight hours with the exception of females with small pups (<10 weeks) presumably to avoid pup predation by bald eagles *(Haliaeetus leucocephalus)* (Gelatt 1996, Gelatt et al. 2002). Another study in Prince William Sound determined that sea otters forage in daylight during the early morning and early evening (Garshelis 1983). However, this work did not show similarities in activity budgets beyond the morning and evening foraging bouts between different study sites. More recent TDR work is currently being analyzed from Prince William Sound, AK where foraging behavior may be related to seasons, tides and time of day (Bodkin per. comm.) Because current available data shows that otters tend to forage and rest at various times of the day with little discernable time pattern (Kenyon 1969, Bodkin et al. 2004, Tinker 2004), behavior is largely independent of time of day at the population level. Therefore, the time of day the aerial survey was conducted should not have an effect on the ISU results.

Little or no work has been done to determine if foraging and resting occurs in different areas within Prince William Sound nor has any work been done to examine the differences in habitat characteristics of these resting and foraging areas if they exist. Also, no work has been done to quantify these differences in behavior based on habitat attributes or group size. This chapter quantifies these relations. The expectation had been that variation in habitat use that was determined to exist from this analysis might have aided in the explanation of the variation of sea otter distribution throughout Prince William Sound. However, as Chapter II will explain, this was not the case.

Methods

Aerial Surveys

The intensive search unit or ISU portion of the aerial survey was used in this analysis. Intensive search units are sampled by flying five 400m diameter concentric circles within the strip transects and are conducted systematically to account for animals not seen on the strip so a correction factor can be applied to the strip counts. The ISU observer also records a behavior for each animal seen in the circle. Behavior is described as either diving (D) or not diving (N) (assumed resting). Data collected include group size, behavior (diving or not diving) and location.

Analysis

The behavior information from each ISU is used to assess if there is a relation between habitat attributes associated with each otter location and the behavior of the animal at that location. Because of the relatively low sample size of ISUs (intensive search units) collected per survey year, all survey years were combined to examine the behavior recorded for each otter sighting during an ISU (n=1764) based on habitat characteristics. Stepwise logistic regression (SAS ASSIT, SAS Institute, Cary, NC) analysis was used to examine the behavior data. In stepwise logistic regression, the first variable selected is the variable most strongly associated with the response (SAS Institute, Cary, NC). The probability of diving was modeled and effects left in the model were significant at the 0.05 level. Analysis was performed on three aspects of the data; all the ISUs (n=1764), only the ISUs of single animals without pups (n=766), and all the ISUs of single animals with pups (n=567). The purpose of analyzing the data for all ISUs was to determine if there was a difference in habitat use based on behavior at the population level, regardless of reproductive status. In addition to the variables listed in the general methods section, the number of adults in a group was one of the independent variables for the analysis of the all the data, while D (diving) or N (not diving) was the categorical response variable. The number of adults was one of the variables because of the sea otters' documented tendency to rest in groups and forage independently (Garshelis et al. 1984, Estes and Jameson 1988, Reidman and Estes 1990). Analysis of the ISUs comprised of single animals with pups and single animals without pups were conducted to examine the potential differences in habitat use based on reproductive status. Time budget studies of sea otters on Adak Island have shown that females with young pups forage for 21% less time than single animals (Gelatt 1996) and more recent TDR data from Prince William Sound shows similar differences in foraging activity between females with small pups and single animals (ASC, USGS unpub. data), however, no studies have been conducted to examine habitat use differences between sea otters with pups and sea otters without pups. Data analysis from the intensive search units was used to calculate a percentage of animals partaking in one behavior or the other. The location of an animal during an ISU was used to determine habitat use differences based on the recorded behavior by the observer. ArcInfo Workstation Grid (ESRI, Redlands, CA) was used to sample all the variables based on the location of the ISU.

Results

All ISUs

A total of 1,759 observations were used from the survey years of 1994 to 2005. Of the 1,759 observations, 31% (549 of the total ISUs) were categorized as diving and 69% (1,210 of the total ISUs) were categorized as not diving. All effects were removed from the model except number of adults in a group and bathymetry because of failure to reach the 0.05 significance level. The number of adults in a group was selected for the model first and had a p-value < 0.001. Bathymetry was selected second with a p-value <0.0001 as well. From the probability analysis, a single animal has a 37% probability of exhibiting diving behavior while a group of otters > 1 only has a 15% probability of exhibiting diving behavior. Figure 20 illustrates the decline in diving behavior in the survey as group size increases. Figure 21 illustrates the decline in diving behavior as bathymetry increases.

Single Animals without pups ISUs

A total of 763 observations of single animals without pups were sampled from the total ISUs. Of these 763 observations, 47% (357 of the 763 observations) were categorized as diving and 56% (199 of the 357 observations) of those categorized as diving were within the 20m depth contour and 85% (303 of the 357 observations) were within the 40m depth contour. Fifty three percent (406 of the 763 observations) of the observations were categorized as not diving. All habitat attributes were removed from the model except bathymetry ($p<0.001$) because of failure to reach the 0.05 significance level. Figure 22 illustrates the decline in diving behavior as depth increases. In this figure, the probability of observing diving behavior as a function of depth is also modeled for single animals with pups to illustrate the potential differences in time allocated to foraging as a consequence of reproductive status.

Single Animals with pups ISUs

A total of 566 observations of single animals with pups were sampled from the total ISUs. Of these 566 observations, 24% (134 of the 566 observations) were categorized as diving and 63% (85 of the 134 observations) of those categorized as diving were within the 20m depth contour and 90% (120 of the 134 observations) within the 40m depth contour. Seventy six percent (432 of the 566 observations) of the observations were categorized as not diving. All effects were removed from the model except bathymetry ($p=0.0166$) and distance to the closest shoreline ($p=0.0172$) because of failure to reach the 0.05 significance level. Bathymetry was the first variable selected by stepwise regression and distance to the closest shoreline was the second. Figure 22 illustrates the decline in diving behavior as depth increases and figure 23 illustrates the decline in diving behavior as distance to the closest shoreline increases.

Discussion

While there have been numerous studies documenting and discussing sea otter behavior and the formation of resting groups and individual foraging, allowing for the assumption that group size is indicative of behavior (Garshelis et al. 1984, Estes and Jameson 1988, Reidman and Estes 1990), little work has been done to quantify the probability of diving based on group size. In this analysis, group size is a significant ($p<0.001$) indicator of diving behavior, with the probability of diving inversely related to group size and the probability of not diving positively related to group size.

Results from the single animal ISU (regardless of reproductive status) analysis show that the use of the varying bathymetric contours is fairly restricted during observed diving behavior with a majority of the diving occurring within the 0-20m depth contour and 85-90% of all diving behavior is done within the 40m depth contour, supportive of previous work (Kenyon 1969, Reidman and Estes 1990, Bodkin 2004). Single females with pups also showed a significant negative effect with the percent of diving behavior decreasing as distance to closest shoreline increasing. Females with young pups might require the shelter offered by land masses more frequently than single animals (Kenyon 1969). However, during aerial surveys, quite often sea otters recorded as pups are young and easily identified during ISUs because of their small size, inability to dive and are often being carried by the mother. However, small pups can be difficult to observe during strip counts, because of the same reasons they are easier to detect during ISUs. Identification of large pups is difficult for opposing reasons. Large pups may be of similar size to the mother, are no longer carried on the chest and may be diving independently of the mother. These large pups may be misclassified as independent animals (ASC, USGS unpub. data). Therefore, analysis of this type may be biased towards small pups and may not be applicable to larger pups that swim and dive independently of their mother.

While none of these conclusions are surprising based on prior studies of sea otter behavior, many of these prior studies were conducted with tagged animals for long term monitoring. In this study we were able to use survey abundance data of untagged animals to arrive at similar conclusions and quantify the results. In future applications, this type of behavioral data collection may aid managers in understanding a population's habitat use and allocation based on behavior before investing in long term tagging studies. For example, ISU analysis of diving behavior could indicate a food limited population based on the percent of observed sea otters that were recorded as diving. In this analysis, 47% of observed single animals without pups were diving. In other populations where prey availability may be decreased either because of high sea otter densities or some other factor effecting prey density, time allocated to foraging may be higher. The opposite can be said for low densities of sea otters or high prey availability. This theory could be tested by isolating high and low density areas of sea otters within this study area and examine differences in habitat use and behavior.

While individual tagging studies and TDR data give a detailed insight into an individual's activity budgets and use of habitat, these studies can be difficult to extrapolate into a larger population or study area. By implementing aerial surveys and conducting the analysis described in this chapter, populations or areas with anomalous results could be targeted for further intensive study, therefore efficiently allocating resources.

CHAPTER II

SEA OTTER DENSITY, ABUNDANCE AND DISTRIBUTION RELATED TO HABITAT ATTRIBUTES

Introduction

Sea otters were abundant in Prince William Sound prior to the Russian fur trade (Lensink 1962), but were hunted to near extinction across the sea otter's range. A remnant population persisted in Prince William Sound (Lensink 1962). As late as 1962, linear shoreline surveys of sea otter densities in Prince William Sound, based on the 10 and 50 fathom contours, were below the density values reported from the Aleutian Islands (Lensink 1962). Lensink flew several aerial surveys between 1959 and 1960. Calculated abundance estimates for Prince William Sound were 1,000 to 1,500 animals (Lensink 1962). In 1973 and 1974 coastline aerial surveys were conducted by the Alaska Department of Fish and Game. The survey design was based on the contour of the shoreline, about 200 yards off shore (Pitcher 1975). The summarized distribution data from these aerial surveys (Pitcher 1975) has been provided by Irons et al. 1988 (Fig.24). These surveys illustrated an expanding population with an estimated 5,000 sea otters (Alaska Department of Fish and Game 1973). By the mid-1970's, the population in Prince William Sound had re-colonized nearly all known available habitat (Johnson 1987). In 1984 and 1985, US Fish and Wildlife Service conducted coastline boat based surveys for sea otters throughout Prince William Sound to examine changes in abundance and distribution. Some offshore transects were completed as well but were conducted only when traveling in-between shoreline transects, with no explicit design. The corrected results for the 1984 and 1985 surveys of all of Prince William Sound were 4,509 animals with 259 animals at all of Knight Island and the population was more "evenly" distributed than in the previous 1973 survey (Irons et al. 1988) However, there were still questions as to the sources of observed variation in the distribution. Irons et al. (1988) hypothesized that the potential reasons for the variation in distribution were related to available habitat and the possibility that the Prince William Sound population was still re-colonizing the area in 1985 (Irons et al. 1988). Johnson (1987) estimated a carrying capacity of 8 otters per mi^2 (3.1/ km^2) resulting in a population estimate of 6,500 animals based on the 30 fathom contour. However, the density estimate used was derived from an area thought to be at carrying capacity and ignored areas in the northwest because of a lack of bathymetry data (Johnson 1987).

Currently, the entire Prince William Sound population is considered stable with the actual mean abundance estimate in Prince William Sound between 1995 and 2005 was 12,536 with a range of 11,289 to 13,783 (alpha = 0.05 significance level) (Table 4). The mean aerial survey calculated density was 1.7607/ km^2 (ASC, USGS unpub. data).

In western Prince William Sound, an area heavily affected by the Exxon Valdez Oil Spill in 1989, there are about 2500 sea otters. Between 1993 and

2000, the population has been increasing in numbers at a rate of about 5% or half the recovery rate of the sea otter re-colonization of Prince William Sound after exploitation ended in the early 1900's (Bodkin et al. 2002). The data from the Western Prince William Sound surveys was used to build a habitat based population density model to be applied to the entire Sound. This will aid in the understanding of the variation of the distribution and density of sea otters within Prince William Sound. Little work has been done to relate sea otter population density or variation with habitat attributes.

Methods

Aerial Surveys

A detailed description of the aerial survey method is given in the General methods section of this paper. The information from each otter sighting (number and location) was used to establish and quantify if there is a relation between habitat attributes and the densities of sea otters in western Prince William Sound. Only the survey data from the western portion of Prince William Sound (excluding northern Knight Island) was used to create the model. The model was then used to estimate abundance and calculate density for the entire Sound including northern Knight Island.

Analysis

Because count data were collected, Poisson regression was used to analyze the survey data and create a predictive density model for the entire Sound. However, Poisson regression assumes all count events are independent which may cause error by overestimating where densities are low and underestimating where densities are high (Jones et al. 2002). All western Prince William Sound surveys were sampled and analyzed using ArcInfo Workstation GRID (ESRI, Redlands, CA) and SAS PROC GEMOD (SAS Institute, Cary, NC). All occupied and unoccupied pixels within the survey boundaries were sampled, except those pixels that were within the buffer of an observed sea otter location (Fig. 19). Predicted densities as well as upper and lower confidence intervals (alpha=0.05) were calculated based on the best fitting model. The model was determined by calculating Akaike's Information Criterion (AIC) (Akaike 1973) and calculating the weight of evidence, $w_{i,}$ to quantify the strength of the model (Burnham and Anderson 2002) (Table 3).

Seven separate yearly surveys from Western Prince William Sound were analyzed for the variables listed in the General Methods Section (Table 1.) Year was also added as a continuous variable to determine if year improved the model. As a continuous variable, years are analyzed in chronological order. If year had been entered as a discrete variable, order would have been ignored. Because there was an assumption that the sea otter population was linearly increasing as time passed, chronological order of the data was important.

Once calculations were completed, predicted densities and their respective x, y location data were imported into ArcMap to create a map of the predicted high and low density areas within Prince William Sound. Digitized results of whole Sound surveys were superimposed on the predicted map of high and low density areas to examine any discrepancies in the predictive mapping.

Spatial autocorrelation, as defined by Ver Hoef et al., is “a random variable [that] may be correlated to itself when separated by some non-zero distance” (Ver Hoeff et al. 2001). In this type of analysis, unaccounted for spatial autocorrelation was present if residual deviations that are close (in distance) to each other are less variable than random deviations. Generally, proximate neighbors are more alike than distant neighbors (Barbujani 1988, Henebrey and Merchant 2002).The presence of unaccounted for spatial autocorrelation could lead to the inflated significance of variables within the model (Henebrey and Merchant 2002). To determine whether or not the model accounted for spatial autocorrelation, semivariograms were created in ArcMap Geostatistical Analyst (ESRI, Redlands, CA) based on the residuals calculated from known densities minus predicted densities on a per pixel basis. In this analysis, variograms or semivariograms illustrated how correlated the residuals from one pixel to another are given a certain distance (LeCorre et al. 1998, Thompson et al. 2005). These residuals were plotted by distance and direction on an x, y-grid to determine if a pattern existed. If no discernable pattern was present, then the model had taken into account spatial autocorrelation and any remaining error was random. If there was a pattern present, an assumption was made that the model is missing some variable and error was not just random (Cooper per. comm.). Spatial autocorrelation was looked for over 10 intervals (lags) of a distance up to 5,000m (lag size) from each pixel. These values were calculated from a general rule in geostatistics that the number of lags multiplied by the distance (lag size) should not exceed the value of the maximum distance of one pixel to another (LeCorre et al. 1998, Thompson et al. 2005). Partial sills were also calculated in this analysis. Partial sills are the sill (the value that the semivariogram model attains when it levels out on the y-axis) minus the nugget (parameter of a semivariogram model that represents independent error or measurement error) (ERSI, Redlands, CA). Partial sills close to 0 indicate that little or no spatial autocorrelation exists because there is no increase in the variability as distance or direction change. Partial sills were determined with anisotropy in several directions ranging from 0° to 180° at 45° increments. Only the interval from 0° to 180° was used because semivariogram values in one direction are equal in the opposite direction (Johnston et al. 2004). Partial sills were also calculated without anisotropy.

Results

The addition of year as a variable improved the model slightly; however, year was left out of the final model for several reasons. Uncorrected estimates from the best model by year showed almost no difference on a per year basis. Only when the estimates were corrected based on calculated correction factors from the aerial survey (Bodkin and Udevitz 1999) of that given year was there a marked difference in estimates by survey year (Figures 25 and 26). Because uncorrected counts were used to build the model, not corrected estimates, year was left out of the model. Also, because the model is being tested on a stable population (entire Prince William Sound) and is built based on an unstable population (western Prince William Sound), surveys across years were combined to eliminate the variation associated with yearly surveys and utilize seven years of data to reduce the result to one best model for prediction.

All other remaining variables (Table 2) were left in the model except distance to population centers. Distance to human population centers actually showed a negative correlation to sea otter density. As distances from towns and villages increased, sea otter density decreased, implying that there is some other biological or physical feature that contributes to the presence of both sea otters and human population centers such as protection from inclement weather or marine productivity. Bathymetry levels 7(120m-200m) and 8(>200m) were collapsed into 1 level, which improved the model slightly. This inferred that at these depths, there is little difference between the two bathymetry levels. Two interaction terms were part of the model; 1) The interaction between bathymetry and distances to land, and 2) the interaction between distances to land and distances to protected shoreline. Because bathymetry was a categorical variable with 7 levels, K, or the number of model variables =20 (Burnham and Anderson, 2002) (Table 2).

From the best model, densities on a per km^2 basis had an uncorrected mean of $2/km^2$ and a range of $0/km^2$ to $37/km^2$. Correction factors from all seven surveys were averaged and applied to the total predicted values (predicted, upper and lower confidence intervals) to attain a final corrected density estimate for all of Prince William Sound. The estimated total population size with upper and lower confidence intervals (alpha = 0.05) was calculated by summing the densities of all the pixels. Estimated corrected total population size is 16,441 (+2,363, -1,973). Figure 27 illustrated the predicted high and low density areas across Prince William Sound calculated from the best model. From visual inspection of the superimposed survey results onto the predicted map of high and low density areas, many of the high density areas corresponded with actual Prince William Sound survey results (Fig. 28). However, in Port Wells, an area in the northwest region of Prince William Sound, there were high densities of sea otters from actual survey results. The model predicted relatively low densities of sea otters.

Visual examination of the semivariogram created from the residuals showed almost no spatial autocorrelation present. With and without anisotropy, partial sills ranged from 0.00030497 to 0.00094578.

Discussion

Sea otter densities vary across their range. Studies conducted in the Commander Islands showed densities as high a $9.2/km^2$ (Bodkin et al 2000).

However, these high densities preceded a density-dependent population decline. Post-decline estimates were calculated to be 6.1/km^2 (Bodkin et al. 2000). Laidre et al. (2001) reported sea otter densities at about 5/km^2 along the California coast in rocky substrate habitats and densities varying from 4.55/km^2 within the 20m contour to 0.97/km^2 within the 40m contour along the Washington state coastline (Laidre et al. 2002). Reported densities of sea otters in Prince William Sound have been estimated to be as high as 8 otters per mi^2 (3.1/ km^2) within the 30 fathom contour (Johnson 1987). However, an aerial survey conducted in 1994 within Prince William Sound suggested an average sea otter density of 1.28/km^2 (Bodkin and Udevitz 1999). More recent estimates suggest the average density to be 1.7607/km^2 (ASC, USGS unpub. data). Density calculations from the best model created in this study were calculated to be 2.0316/km^2 within the entire Prince William Sound survey boundaries. However, density calculations from the best model included northern Knight Island, an area heavily impacted by the Exxon Valdez oil spill, and an area that is still not at pre-spill population levels (Chapter III). Total corrected population was estimated at 16,441 (+2,363, -1,973).

Although there was no spatial autocorrelation in the model, indicating that error in the model was random and not due to any missing variables, improvements to the variables used in the model could enhance it. Benthic prey abundance and densities are unknown throughout Prince William Sound. The location and abundance of available food resources for sea otters most likely would have been a variable that improved the model. Until prey data is available, using bathymetry as an indicator of prey availability is the only option. A more accurate and precise bathymetry layer may improve the model by indicating areas of shallow water habitat, not currently available in digital form. For example, in Port Wells, an area of high sea otter density according to the aerial survey results, the model predicted low densities of sea otters. One reason may be the lack of shallow water habitat available according to the bathymetry data used in the analysis. There are many shallow moraines that are not represented by the bathymetry layer used in the analysis. If a more accurate digital bathymetry layer was available, the model would likely predict higher densities within Port Wells.

Applying correction factors from the aerial surveys by year so the model would have been built on population estimates instead of population counts might have improved the model somewhat.

Another possibility for the discrepancy between the predicted densities and the observed densities is the correction factor itself. The correction factor is calculated from the ISU data to account for animals not seen during strip counts. As mentioned previously, the ISUs last for a maximum of 230 seconds. Sea otter dive duration averages 74 seconds but can last for upwards of 200 seconds in California sea otters *(Enhydra lutris nereis)*, a sub-species of the sea otter (Ralls et al. 1995). However, during the Southeast Alaska TDR study, the average foraging dive lasted 85 seconds (Bodkin et al. 2004). However, a TDR dive record retrieved from an animal in Prince William Sound in 2006 had a recorded dive of 422 seconds (Bodkin unpub. data 2006). If longer dives (>230s) are more prevalent within the sea otter population than originally thought, ISUs utilized in this analysis to calculate correction factors might not have observed animals still underwater. If that was the case, then correction factors used to calculate abundance would underestimate the overall abundance of sea otters within Prince William Sound.

Predictive models of sea otter density have been constructed in the past (Laidre et al. 2001, Laidre et al. 2002). However, these models were built using data collected from shoreline based aerial or shore based observer surveys and calculated density based on the linear habitat parallel to the shoreline, excluding any potential offshore habitat. These models also do not account for variables other than substrate (i.e. rocky, sandy, or mixed) such as protection or distances to potentially biologically productive areas. The surveys conducted in Prince William Sound are created perpendicular to the shoreline and are based on the 100m bathymetry contour as well as geological formations that may contribute to sea otter presence such as protected bays or inlets. Because of this, sea otter counts are collected in a way that allowed for the inclusion of potential offshore habitats. Therefore, density calculations are not restricted to the shoreline since this may underestimate sea otter populations within Prince William Sound. Thus, estimates of abundance may be less biased than those based solely on the nearshore habitat.

This model will be further refined and tested in areas outside of Prince William Sound where similar aerial surveys for sea otter abundance have been conducted such as the Kenai Peninsula, Kodiak Island and possibly Southeast

Alaska. Once completed, the model may be able to calculate sea otter densities in areas where little abundance information is available. Understanding the variation in sea otter distribution may aid managers in making decisions related to habitat protection or human or financial resource allocation during natural or anthropogenic events. It may also allow managers to adjust current sea otter density estimates as habitat use by sea otters changes over time due to predation or some other biological or physical occurrence.

CHAPTER III

PREDICTED SEA OTTER DENSITY WITHIN NORTHERN KNIGHT ISLAND, A HEAVILY IMPACTED AREA FROM THE EXXON VALDEZ OIL SPILL, COMPARED TO SURVEY ESTIMATES FROM THE SAME AREA

Introduction

An estimated 1000 to 2800 sea otters were killed by the *Exxon Valdez* oil spill of 1989 (Garrott et al. 1993) in Prince William Sound. However, mortality was not equivalent in all areas. Northern Knight Island, in western Prince William Sound is considered to be a highly impacted area of the spill where an estimated 90% of sea otters residing there died as a result of the spill (Bodkin and Weltz 1990, Bodkin and Udevitz 1999) At northern Knight Island, numbers have not increased to estimated pre-spill abundance (Dean et al. 2000). An uncorrected count of sea otters at northern Knight Island in 1973 was 105 with a corrected range of 210 to 420 (Pitcher 1975). In 1984 and 1985, the US Fish and Wildlife Service conducted coastline boat based surveys for sea otters throughout Prince William Sound to examine changes in abundance and distribution. A count of sea otters throughout all of Knight Island gave an estimate of 259 (Irons et al. 1988). Dean et al. (2000) estimated 165 animals were residing around northern Knight Island based on analysis of carcass recovery data (Dean et al. 2000). Corrected aerial survey results of the northern Knight Island portion of the aerial survey from 1995 to 2005 are 68 sea otters with a range of 34 to 102 (Bodkin unpub. data) and indicate a declining population since 1995 (Table 5).

Predicted densities were calculated for the northern Knight Island survey area and compared to actual aerial survey results to determine if a discrepancy between actual sea otter densities and predicted sea otter densities based on the model created for the entire Prince William Sound existed. From this model, it is quite evident that sea otters have not recovered to pre-spill densities.

Methods

After completing the analysis from Chapter II, the predicted densities from the northern Knight Island area were isolated. Based on the current model developed in Chapter II previously, the predicted number of individuals for Northern Knight Island was calculated and compared to the survey results of the past 10 years.

Results

The model created in Chapter II calculated densities for the entire Sound. Northern Knight Island results were isolated to determine the densities that were predicted in that area. Northern Knight Island densities on a per $50m^2$ pixel basis had an uncorrected mean of 0.003948 and a range of 0.00000 to 0.035719.

These numbers convert to a mean density of 1.5792/km^2 ranging from 0/ km^2 to 14.2876/ km^2. Correction factors from all seven aerial surveys utilized to create the model in Chapter II were averaged and applied to the total predicted values (predicted, upper and lower confidence intervals) to attain a final corrected density estimate for northern Knight Island. The total estimated population size with upper and lower confidence intervals (alpha = 0.05) was calculated by summing the densities of all the pixels. The estimated corrected total population size is 384 (+53, - 43). Once calculations were completed, predicted densities and their respective x, y location data were imported into ArcMap to create a map of the predicted high and low density areas within the northern Knight Island survey area of Prince William Sound (Fig. 29). The digitized results of the northern Knight Island survey results were superimposed on the predicted map of high and low density areas to examine any discrepancies in between the aerial survey results and the predicted high and low density areas from the model (Fig. 30).

Discussion

From visual inspection, many of the high density areas correspond with actual survey results. However, many areas predicted as high density habitat are lacking animals or are well below predicted densities. Uncorrected results from surveys conducted before the Exxon Valdez oil spill estimate the Knight Island population to be 105 animals with a corrected range of 210 to 420 (Pitcher 1975) Corrected aerial survey results of the northern Knight Island portion of the aerial survey from 1995 to 2005 are 68 sea otters with a range of 34 to 102 (Bodkin unpub. data) (Table 5).

Although our estimate of 383 animals is greater than pre-spill estimates, Prince William Sound was still re-colonizing many areas as late as 1985. Although the distribution of sea otter abundance appeared more "even" in 1985 than in the previous 1973 survey, Irons et al. (1988) hypothesized that the potential reasons for the variation in distribution was related to available habitat and the possibility that the Prince William Sound population was still re-colonizing the area (Irons et al. 1983, Irons et al. 1988). Dean et al. (2000) estimated 165 animals were residing around northern Knight Island based on analysis of carcass recovery data (Dean et al. 2000). Unfortunately, there were no surveys conducted any later than 1985 to further refine the pre-spill abundance and density of sea otters in northern Knight Island.

Another possible cause of the discrepancy is in the nature of the Poisson regression. Poisson regression assumes all count events are independent which may cause error by overestimating where densities are low and underestimating where densities are high (Jones et al. 2002). Overdispersion occurs when an incorrect assumption of independence is made (Burnham and Anderson 1998). A possible alternative form of analysis may be the zero-inflated Poisson model (Jones et al. 2002). This allows for the probability of the density of a pixel being predicted as zero to be larger than a normal Poisson regression would calculate (Long 1997).

By testing our model in other areas outside Prince William Sound, this model may be further refined. Once completed, this model may be able to more accurately calculate sea otter densities in areas where pre-decline abundance information is not available, or in the case of the Aleutian Islands, prior to the decline. Understanding the variation in sea otter distribution may aid managers in making decisions related to habitat protection or resource allocation during natural or anthropogenic events.

			Uncorrected				Corrected
Survey Year	Correction Factor	Predicted	Upper CI	Lower CI		Predicted	Upper CI
1995	1.36	11458.99	13097.4	10091.02		15584.226	17812.464
1996	1.27	11446.74	13083.46	10080.22		14537.36	16615.994
1997	1.34	11488.48	13128.76	10118.52		15394.563	17592.538
1998	2.04	11494.96	13137.24	10123.62		23449.718	26799.97
2000	1.54	11476.99	13117.16	10107.24		17674.565	20200.426
2004	1.59	11551.1	13205.96	10170.3		18366.249	20997.476
2005	1.31	11488.57	13134.66	10114.79		15050.027	17206.405
Average CF	1.492857143						
			Uncorrected				Corrected
		Predicted	Upper CI	Lower CI		Predicted	Upper CI
Single Model		11012.98	12595.55	9691.58		16440.806	18803.357

Table 1 Correction factors for survey years with calculated corrected and uncorrected estimates by year as well as single and uncorrected estimates that removed year effect from the model.

Variable	Description	Type
Bathymetry	Depth contours classifies into 8 categories - 1-20m, 20-40m, 40-60m, 60-80m, 80-100, 100-120m, 120-200m, >200m	Categorical
Distance to shorelines	Shortest distance to a land mass	Continuous
Distance to protected shorelines	Shortest distance to protected bays - based on prevailing winds;	Continuous
Distance to tidewater glaciers	Shortest distance to tidewater glaciers within Prince William Sound; n=20	Continuous
Distance to population centers	Shortest distance to populations centers within Prince William Sound; n=5	Continuous
Distance to anadromous streams	Shortest distance to anadromous streams - run size is not available; n=1337	Continuous

Table 2 Variables used in the model building process, description and type

Model	K (# of parameters)	Δ	exp(1/2Δ)	w	Log li
All variables (sw*land)(land*bath) - (town)	20.0000	0.0000	1	0.6098306	-1937
All variables (sw*land)(land*bath) - (town)	22.0000	0.8932	0.6397998	0.3901694	-1937
All variables (bath*land) - (town)	19.0000	200.7278	2.585E-44	1.577E-44	-1947
All variables (bath*land) - (town)	21.0000	202.0874	1.31E-44	7.989E-45	-1947
All variables (bath*land) - (town,sal)	18.0000	226.9182	5.313E-50	3.24E-50	-1949
All variables (bath*land) - (town,sal)	20.0000	228.3350	2.616E-50	1.595E-50	-1949
All variables (bath*sw) - (town)	21.0000	313.7808	7.301E-69	4.452E-69	-1953
All variables (bath*sw) - (town)	19.0000	322.3678	9.971E-71	6.08E-71	-1953
All variables (bath*sw) - (town-glac)	20.0000	441.1866	1.576E-96	9.611E-97	-1959
All variables (bath*sw) - (town-glac)	18.0000	448.9424	3.261E-98	1.989E-98	-1960
All variables (sw*land) - (town)	14.0000	491.2938	2.07E-107	1.27E-107	-1962
All variables (sw*land) - (town)	13.0000	495.4754	2.56E-108	1.56E-108	-1963
All variables (sw*land) - (town,sal)	13.0000	523.2600	2.37E-114	1.45E-114	-1964
All variables (sw*land) - (town,sal)	12.0000	526.7758	4.09E-115	2.5E-115	-1964
All variables - (town)	13.0000	552.1684	1.25E-120	7.64E-121	-1965
All variables - (town)	12.0000	555.2822	2.64E-121	1.61E-121	-1966
All variables - (town - glac)	12.0000	653.6876	1.13E-142	6.9E-143	-1971
All variables - (town - glac)	11.0000	658.1506	1.21E-143	7.41E-144	-1971
All variables (bath*land) - (town-sw)	18.0000	718.5540	9.29E-157	5.66E-157	-1973
All variables (bath*land) - (town-sw)	20.0000	719.1434	6.92E-157	4.22E-157	-1973
All variables - (town-sw)	12.0000	1078.5026	6.4E-235	3.9E-235	-1992
All variables - (town-sw)	11.0000	1082.4002	9.12E-236	5.56E-236	-1992
PWS_bath_50m	9.0000	1410.8538	4.34E-307	2.64E-307	-2009
PWS_bath_50m	8.0000	1416.0268	3.26E-308	0	-2009
sw_dist_int	2.0000	2903.4812	0	0	-2084
land_dist_int	2.0000	3571.5042	0	0	-2118
sal_dist_int	2.0000	3763.1418	0	0	-2127
glac_dist_int	2.0000	3967.3214	0	0	-2137
town_dist_int	2.0000	3992.9738	0	0	-2139

Table 3 Weights and AIC scores of various density models

		PWS	
Date	**Population size**	**$x_i-\bar{x}$**	**$(x_i-\bar{x})^2$**
Jul-99	13,234	698	487,204
Jul-02	12,385	-151	22,801
Jul-03	11,989	-547	299,209
total	37,608	0	809,214
average	12,536		
variance	404607		
SE	636		
CI's	11289	13783	

Table 4 Average population estimates from the aerial surveys of Prince William Sound with calculated upper and lower confidence intervals (alpha = 0.05).

	Northern Knight		
Date	Population Est.	$x_i-\bar{x}$	$(x_i-\bar{x})^2$
Jul-95	89	21	441
Jul-96	65	-3	9
Jul-97	76	8	64
Jul-98	76	8	64
Jul-00	79	11	121
Jul-04	54	-14	196
Jul-05	37	-31	961
total	476	0	1856
average	68		
variance	309		
SE	18		
CI's	34	102	

Table 5 Average population estimates from the aerial surveys of northern Knight Island with calculated upper and lower confidence intervals (alpha = 0.05).

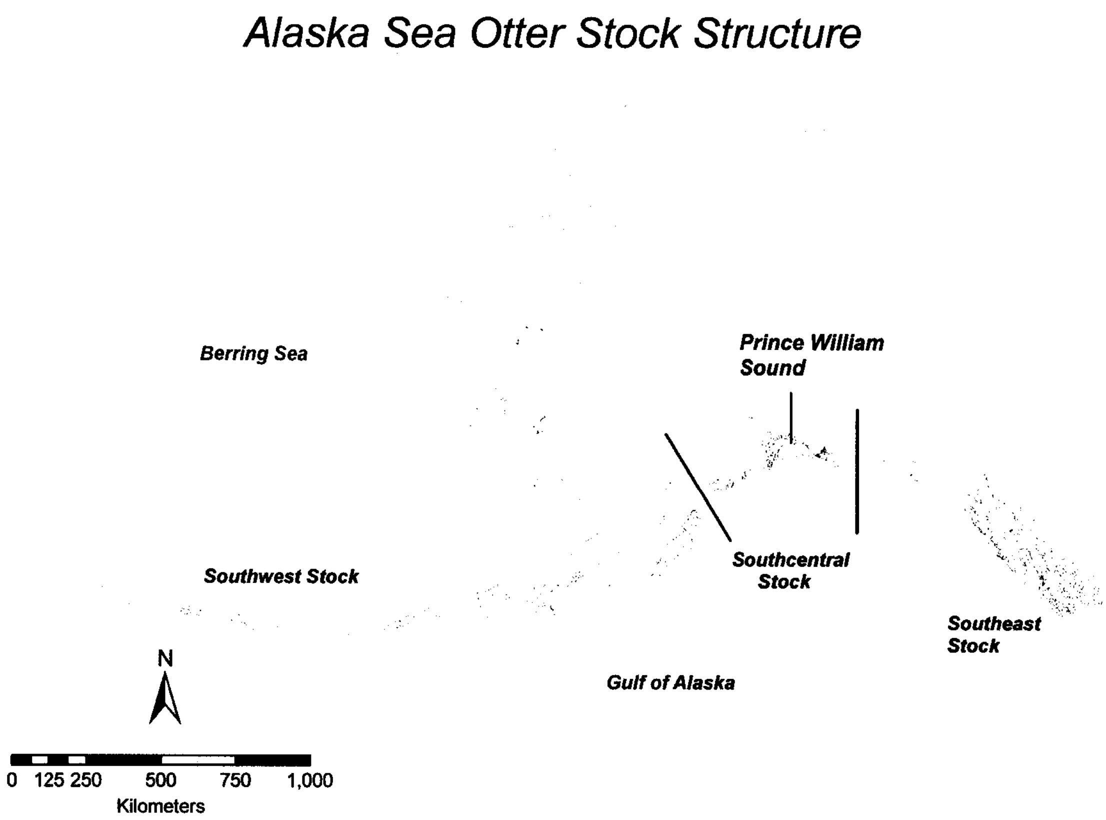

Figure 1 Alaska sea otter stock structure

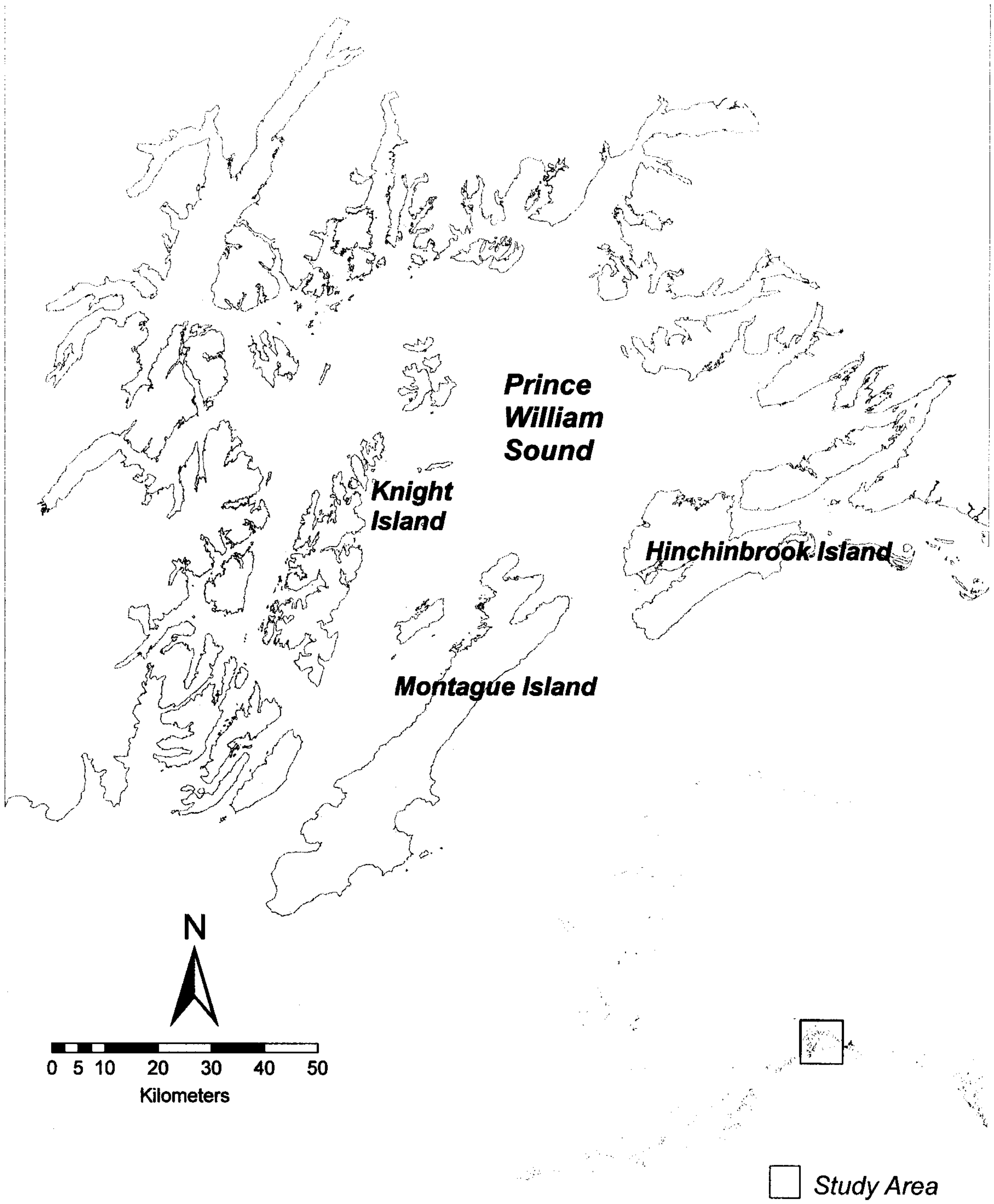

Figure 2 Study Area: Prince William Sound

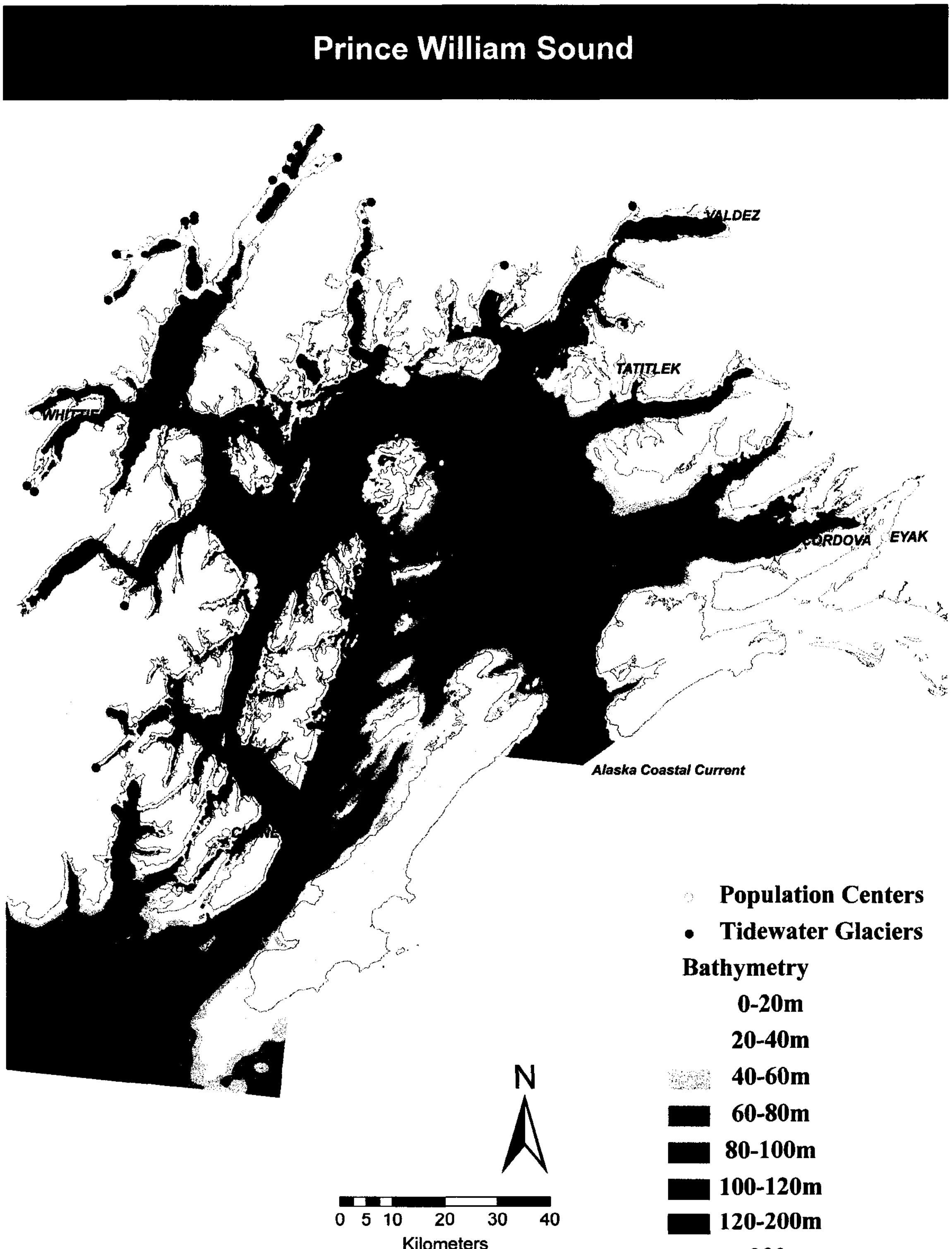

Figure 3 Physical features and population centers of Prince William Sound

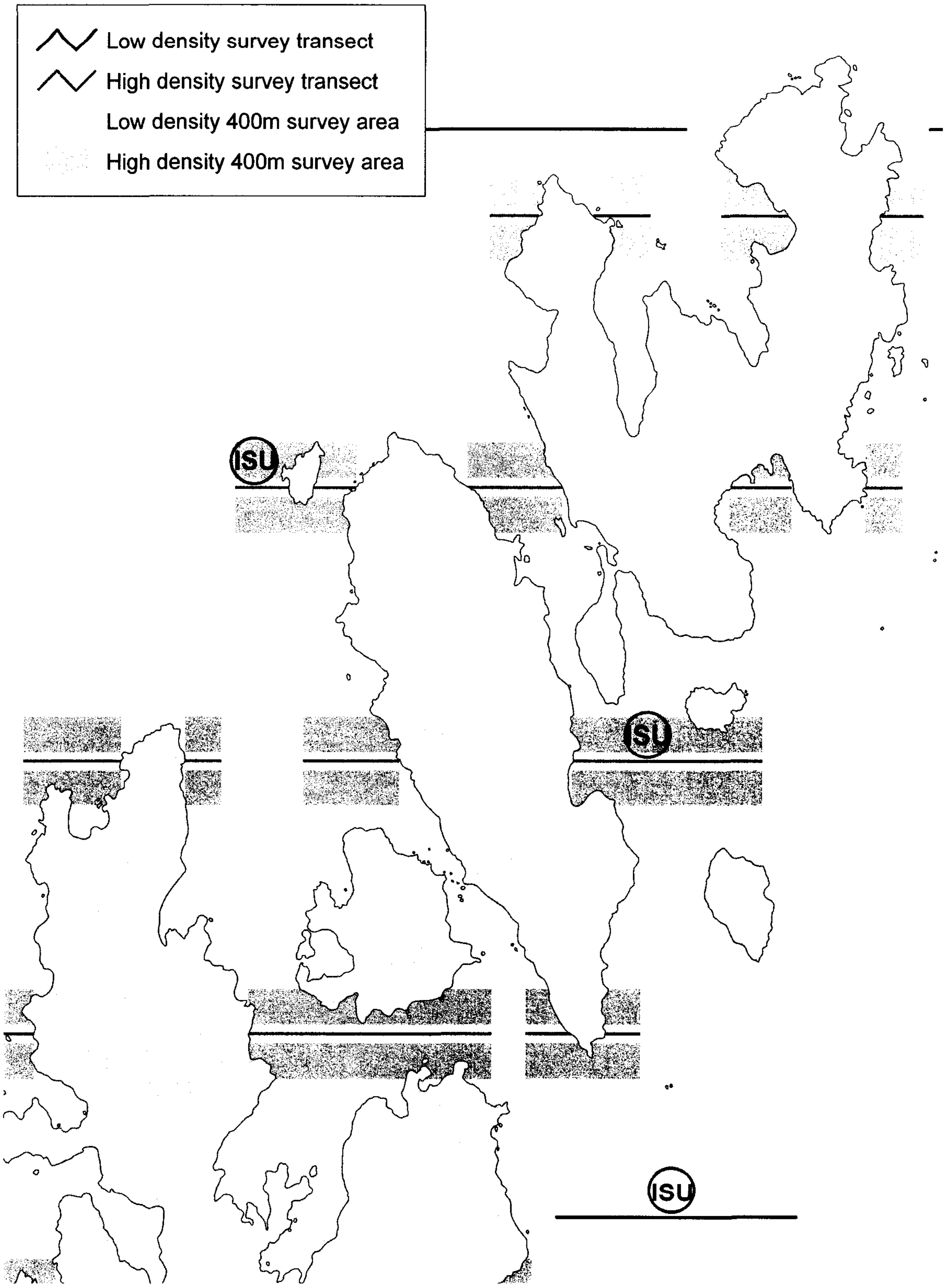

Figure 4 Aerial survey design

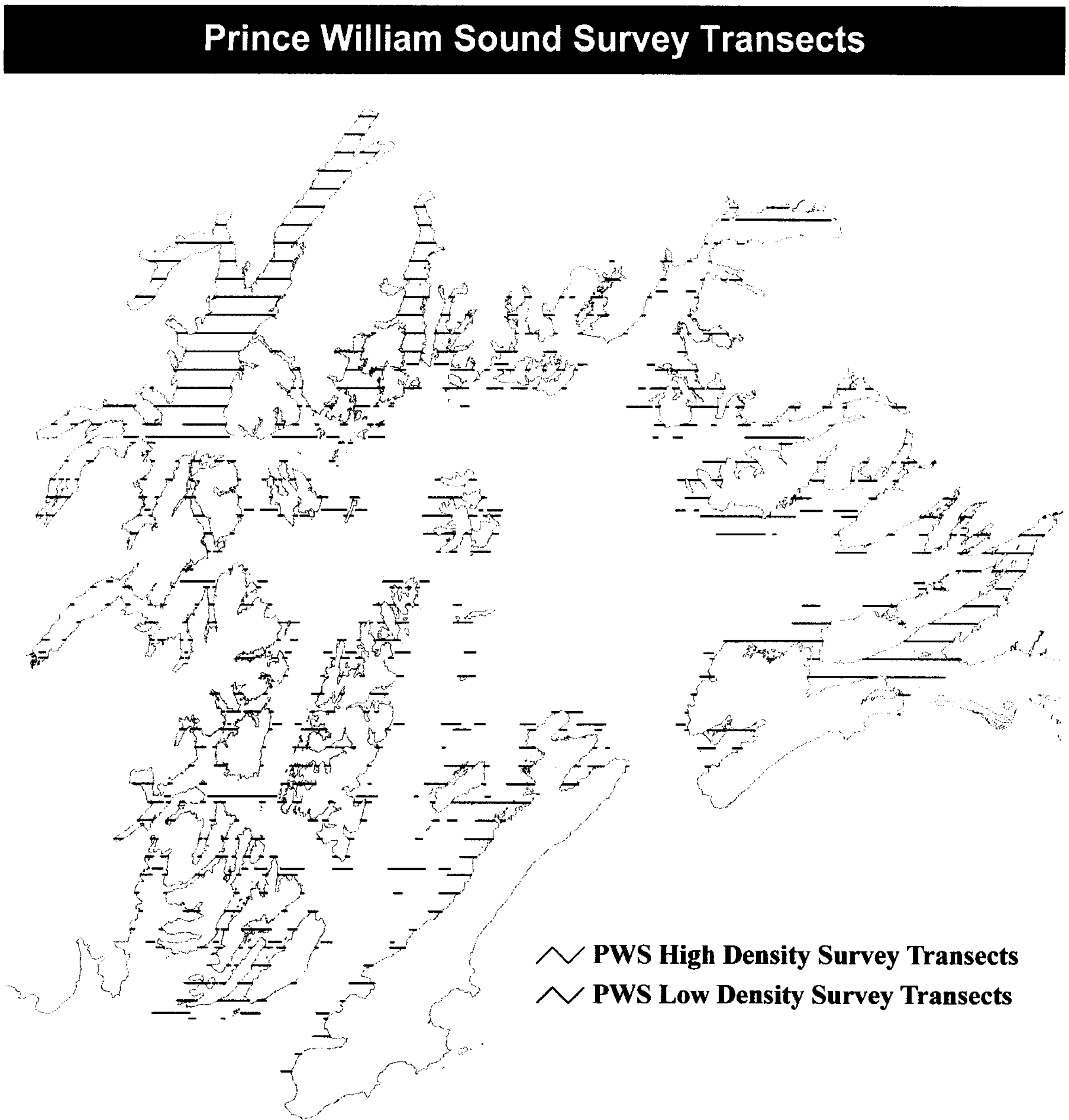

Figure 5 Prince William Sound aerial survey transects

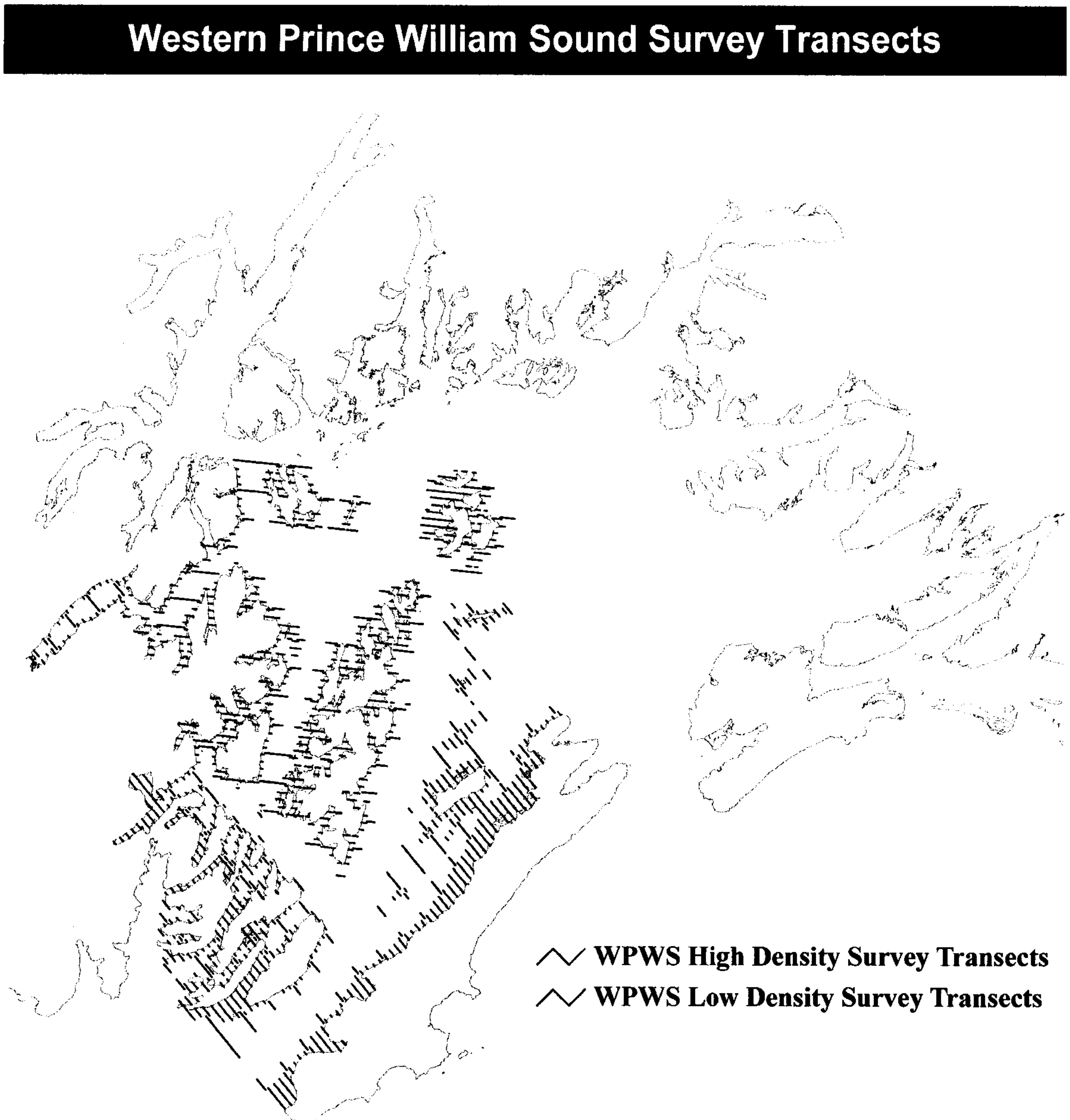

Figure 6 Western Prince William Sound aerial survey transects

Prince William Sound Replicate Survey Transects

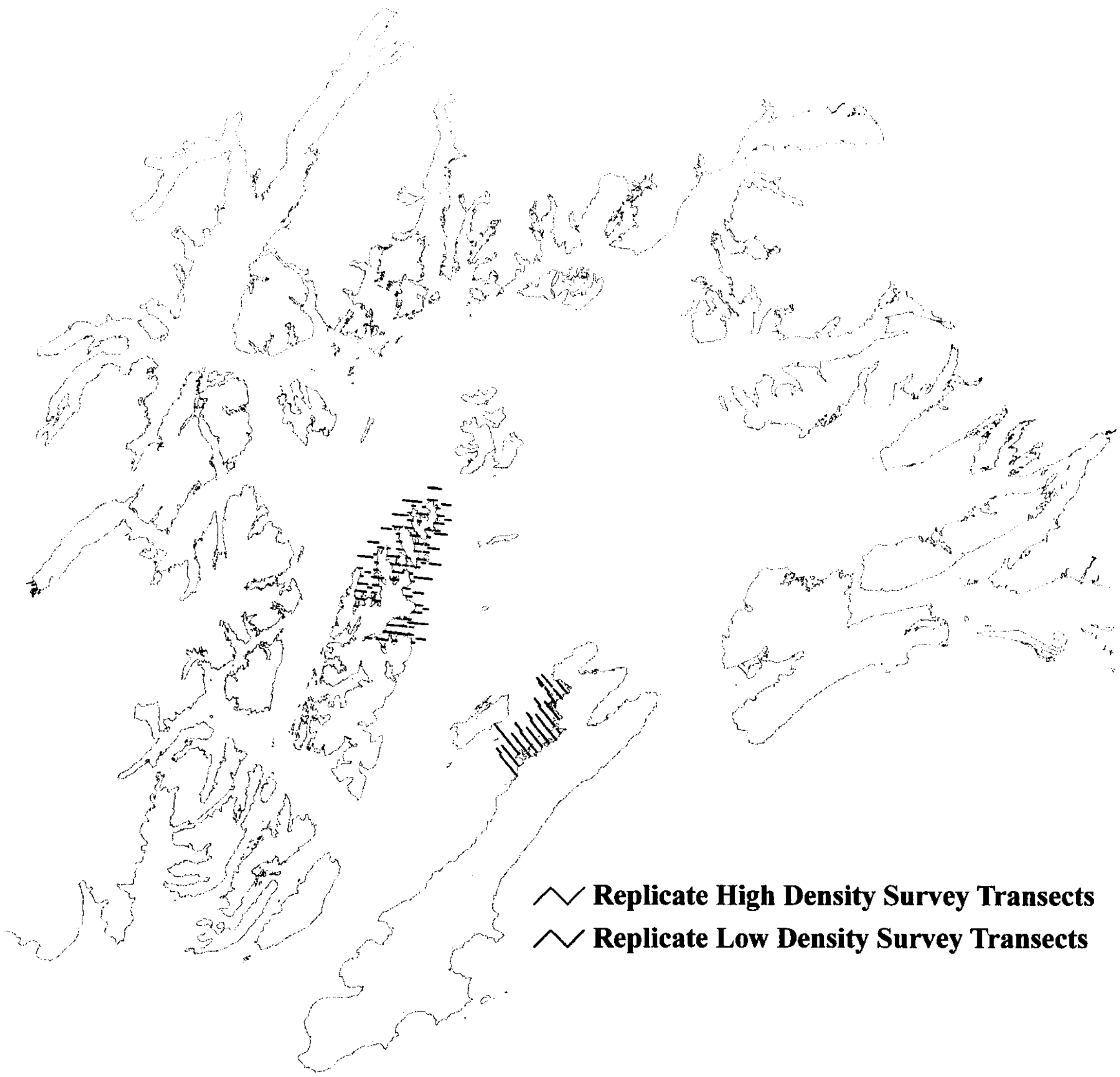

Figure 7 Prince William Sound replicate aerial survey transects

Aerial Survey Boundary for Prince William Sound

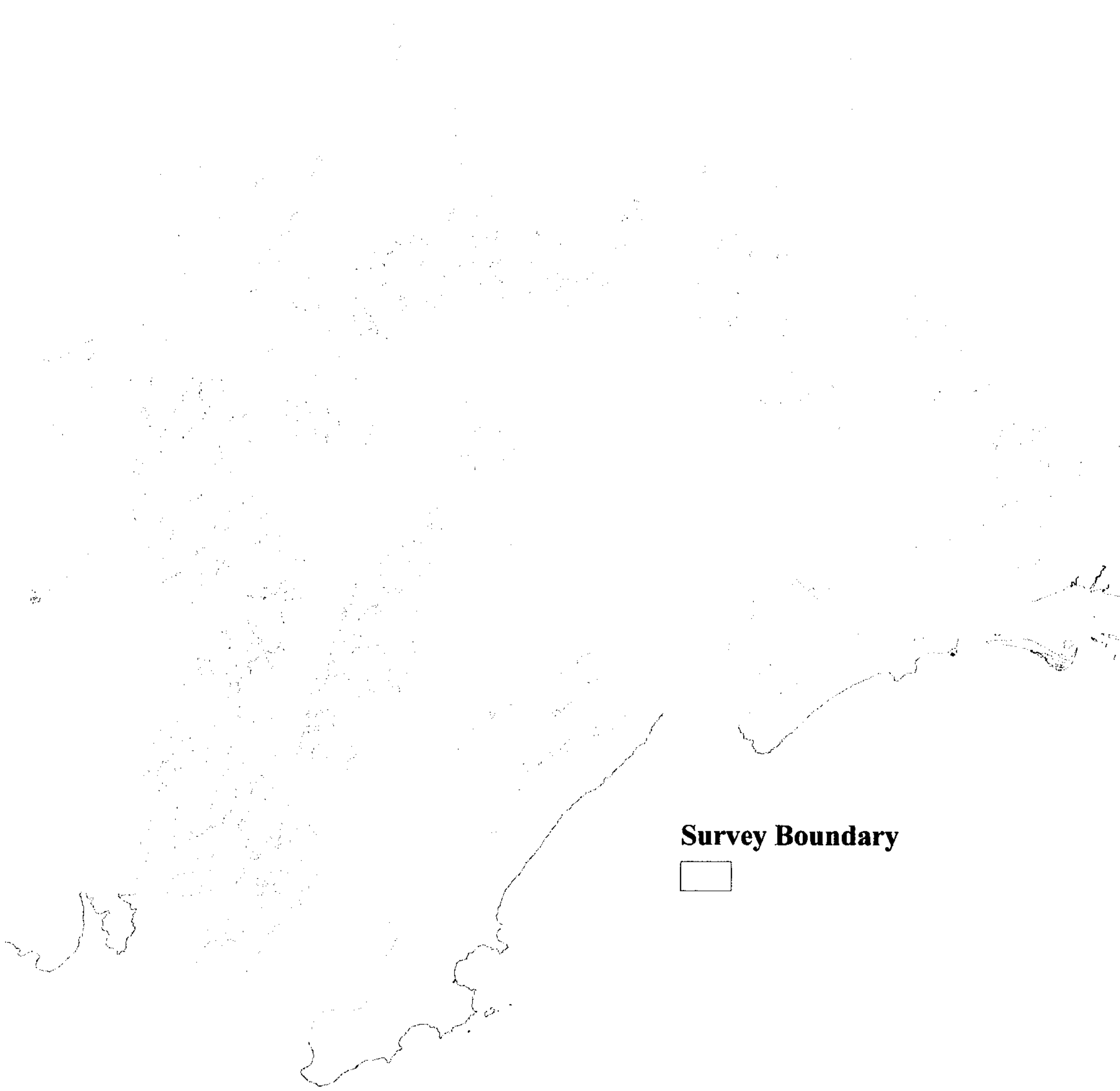

Figure 8 Prince William Sound aerial survey boundary

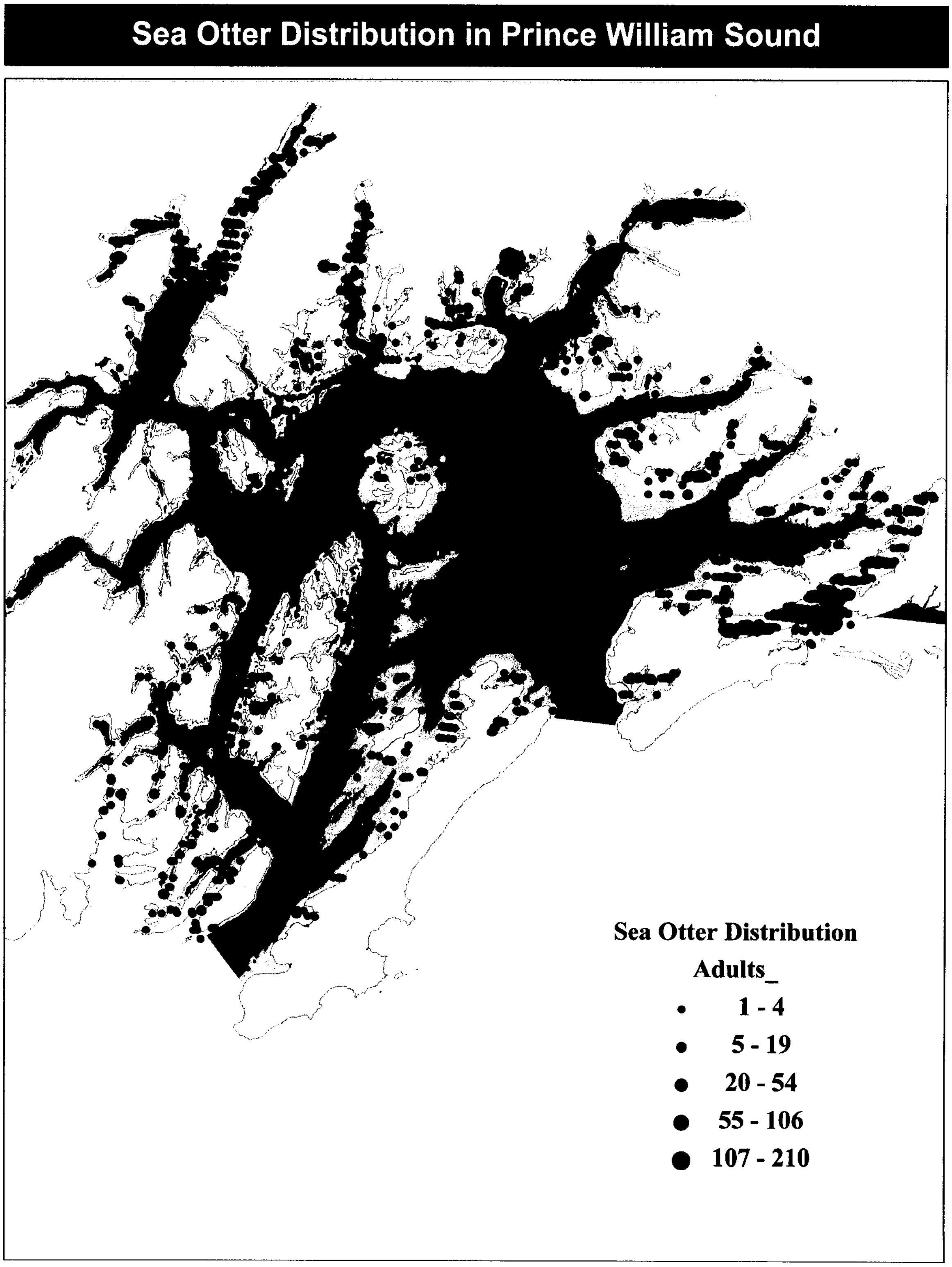

Figure 9 Sea otter distribution in Prince William Sound – illustration of the 3 years of survey data available for the entire Sound.

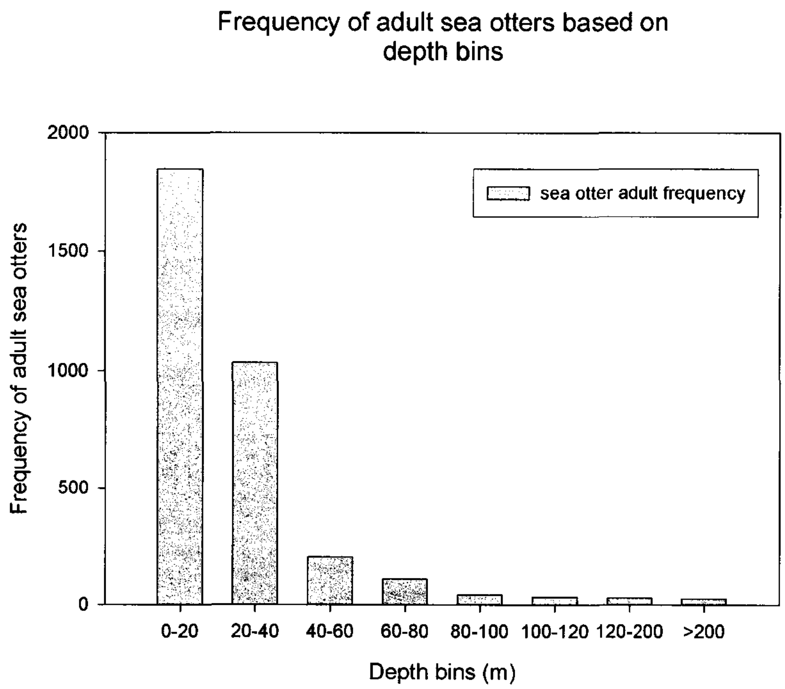

Figure 10 Frequency of adults observed during aerial surveys based on depth bins (in meters).

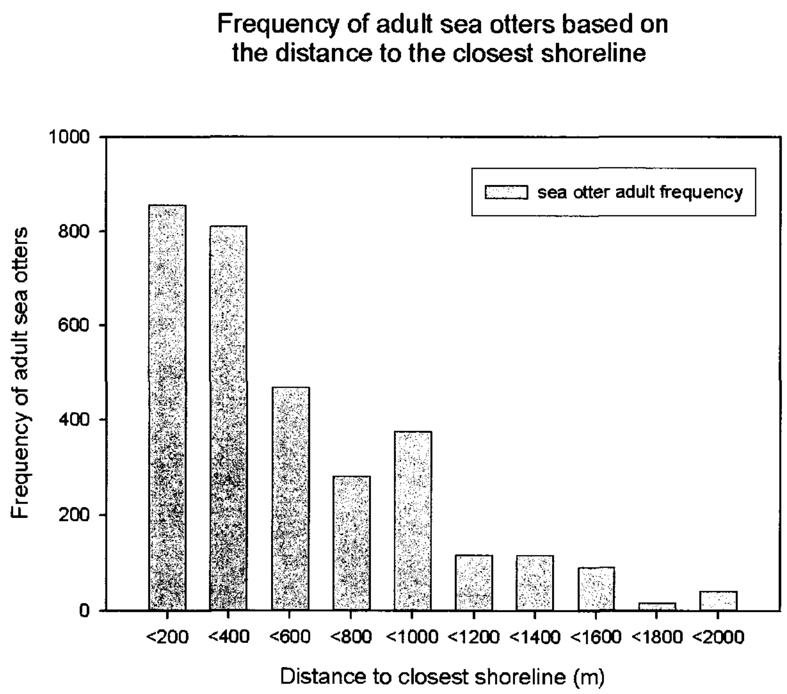

Figure 11 Frequency of adults observed during aerial surveys based on distances to the closest shoreline (in meters). Increase of adult sea otters in the >1000 bin may indicate shallow areas offshore.

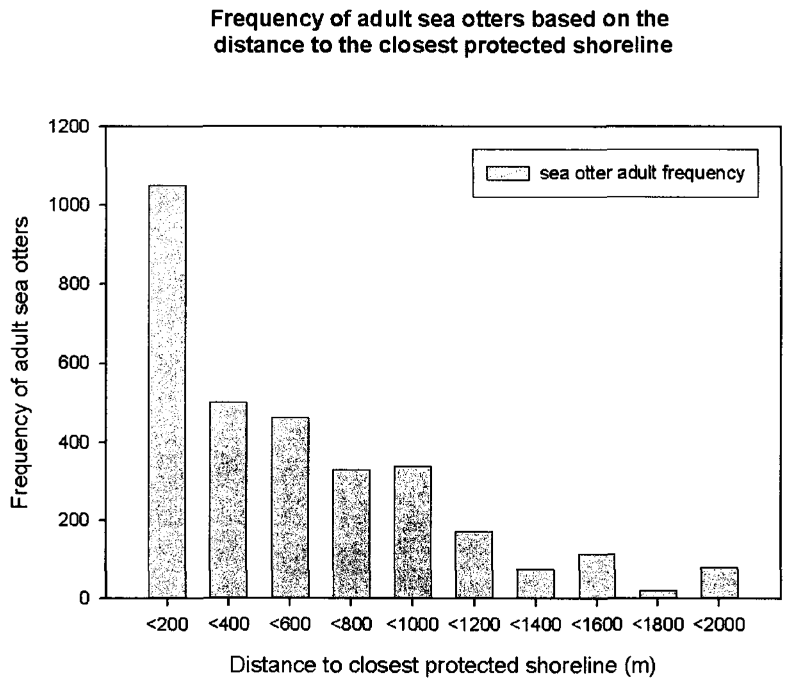

Figure 12 Frequency of adults observed during aerial surveys based on distances to the closest protected shoreline (in meters).

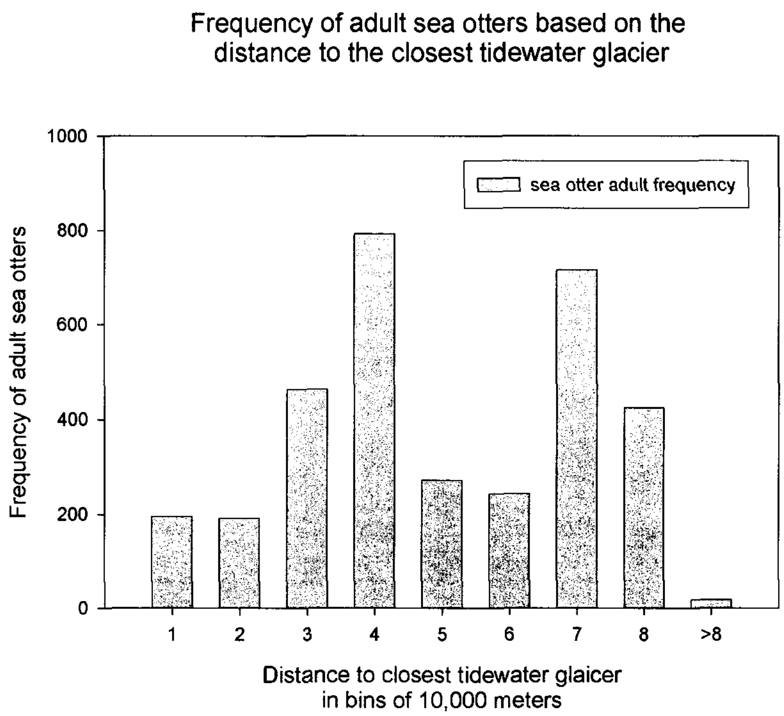

Figure 13 Frequency of adults observed during aerial surveys based on the closest distance to a tidewater glacier. Spikes in sea otter frequency may indicate shallow water moraines.

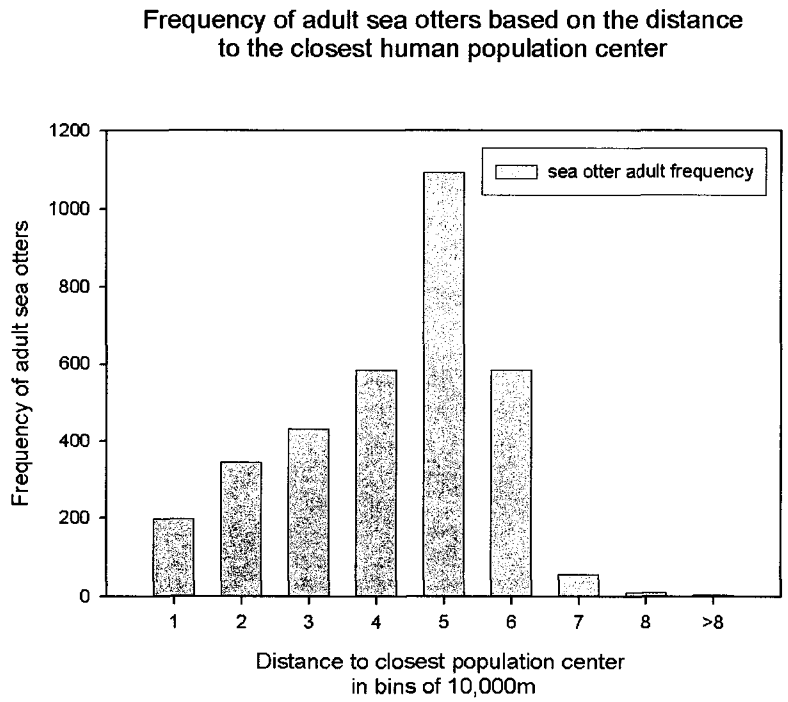

Figure 14 Frequency of adults observed during aerial surveys based on the closest distance to a human population center. Although this graph indicates an increase in adult sea otters as distance from population centers increases, the final model indicated a significant negative relation between sea otter density and distance from population centers. The final model indicates that there is some other biological or physical feature that influences sea otter presence and population center presence in the same areas such as protection from inclement weather or productivity.

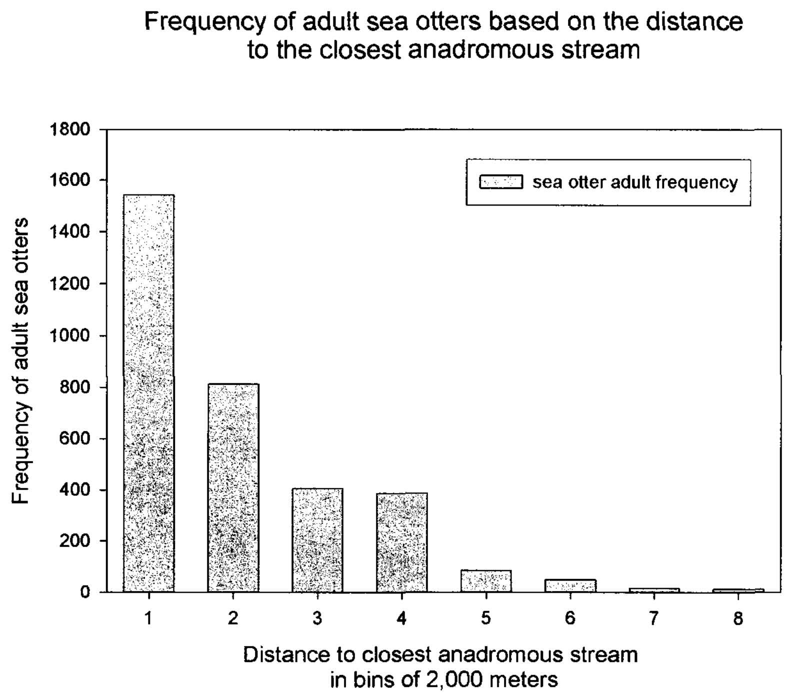

Figure 15 Frequency of adults observed during aerial surveys based on the closest distance to an anadromous stream. The decrease in sea otter abundance as distance from anadromous streams increased may be due to the marine derived nutrients deposited by the decomposing carcasses of the fish in the nearshoreor because of the high number of salmon streams within Prince William Sound. High numbers of salmon streams evenly distributed within Prince William Sound could have correlated with distances to closest shorelines.

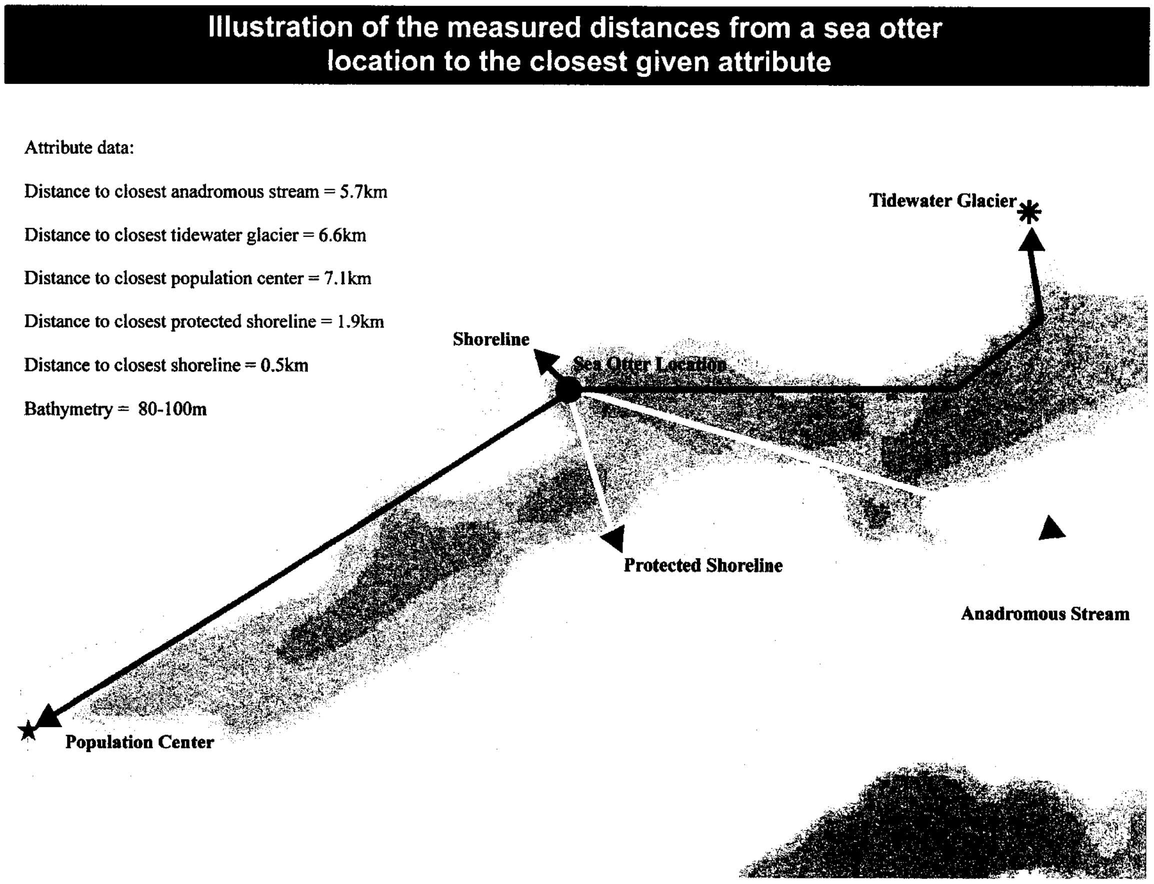

Figure 16 Example of a sea otter location and the various attribute data collected from the location

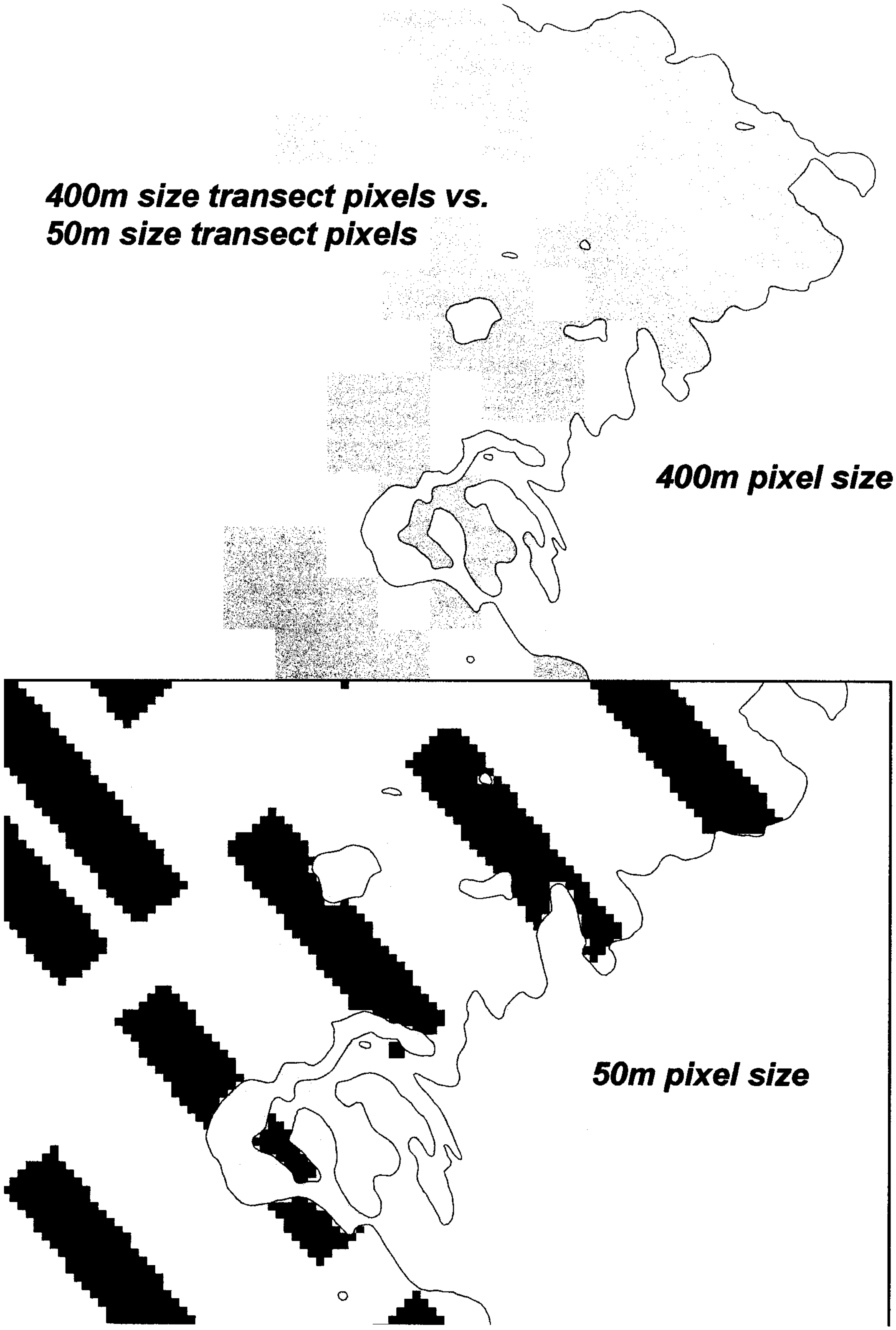

Figure 17 Example of the "stair stepped" phenomenon if pixel size is too large (ESRI)

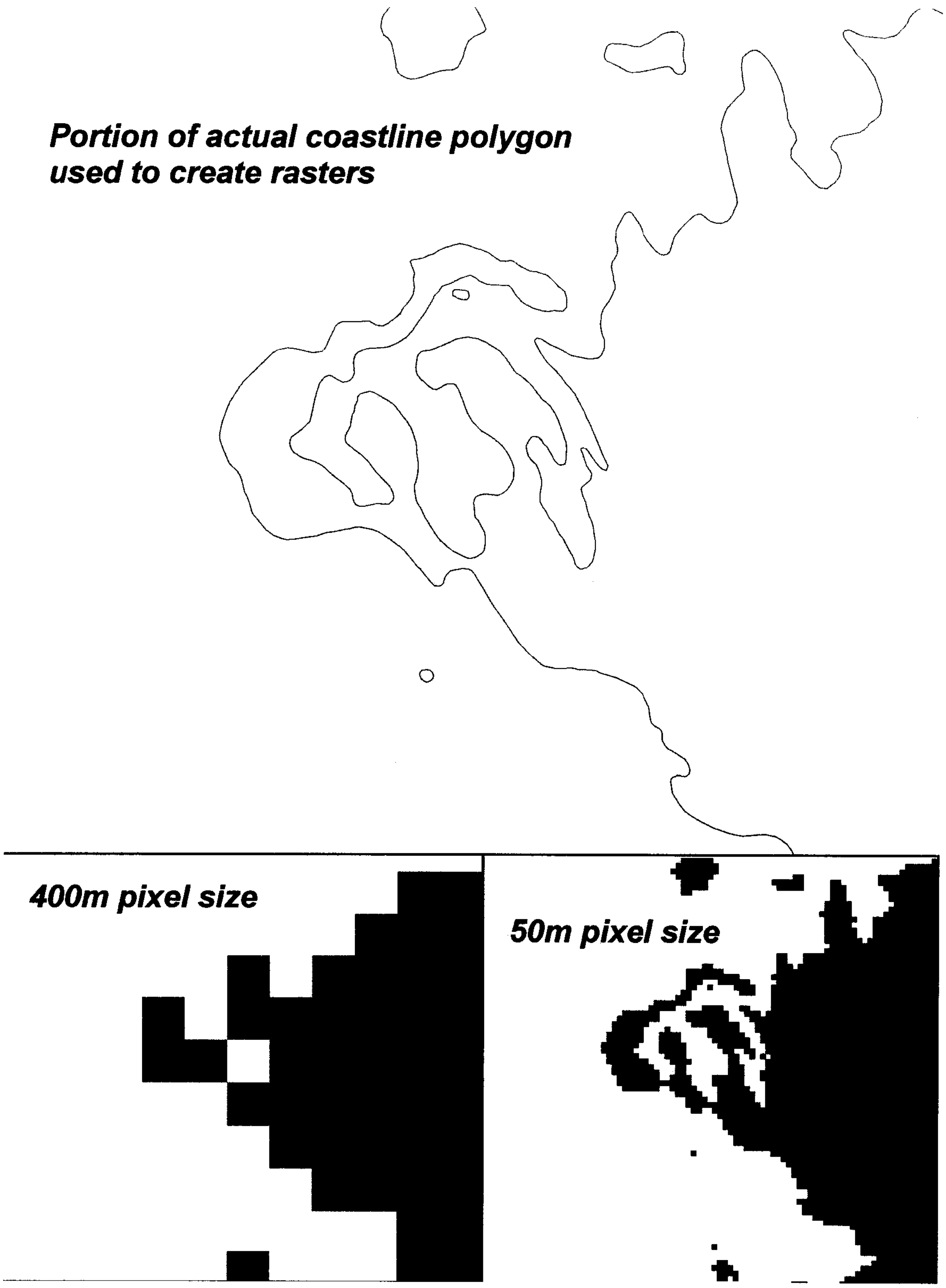

Figure 18 Example of the potential loss of sea otter habitat because of classification error due to large pixel size (ESRI)

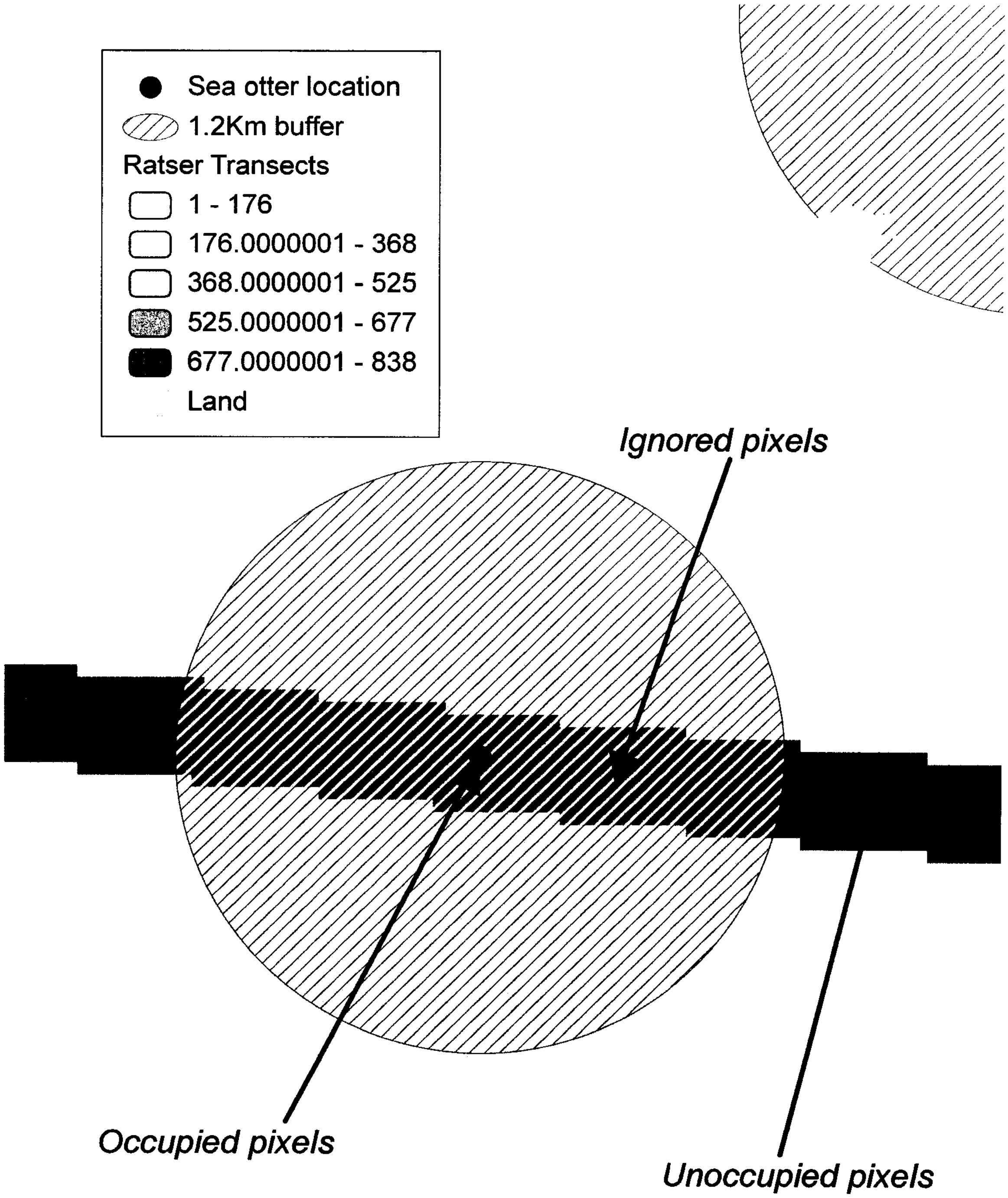

Figure 19 Example of 1.2km buffer around an otter sighting. Transect pixels outside of the hashed area are considered unoccupied, while pixels inside the hashed area are not classified as unoccupied and are not used in the analysis. Raster transect values listed in the map key represent the assigned transect number.

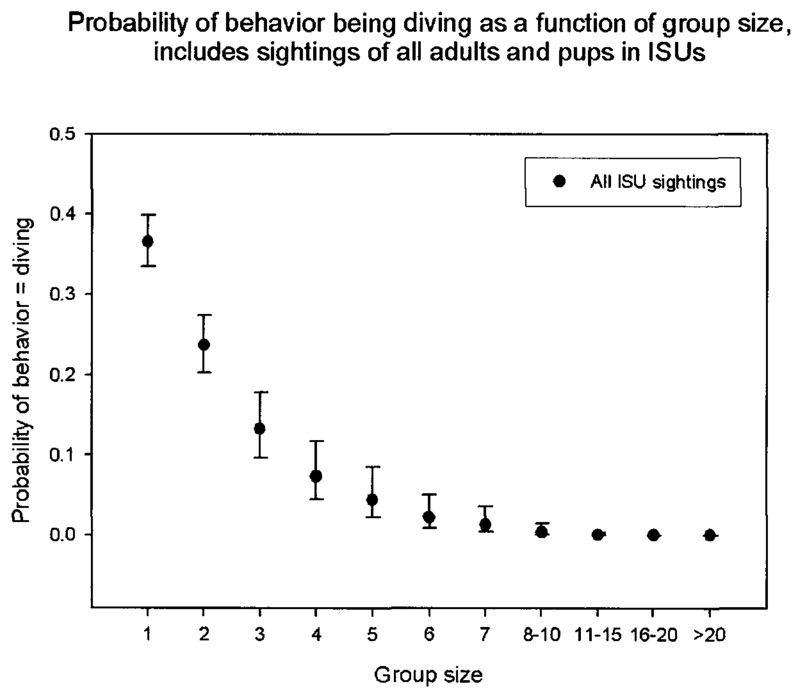

Figure 20 Probability of behavior being diving as a function of group size. Bars indicate 95% confidence interval (alpha = 0.05).

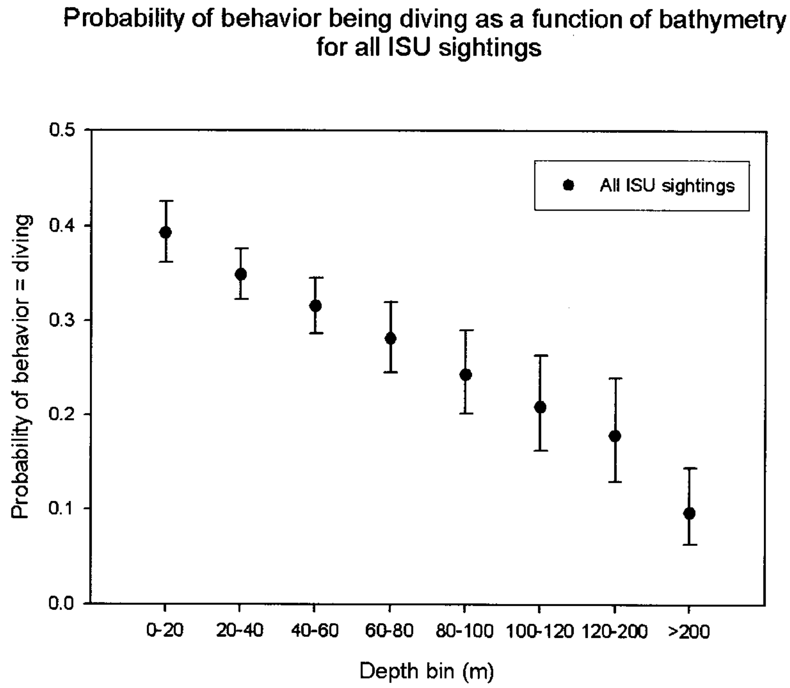

Figure 21 Probability of behavior being diving as a function of depth. Bars indicate 95% confidence interval (alpha = 0.05)

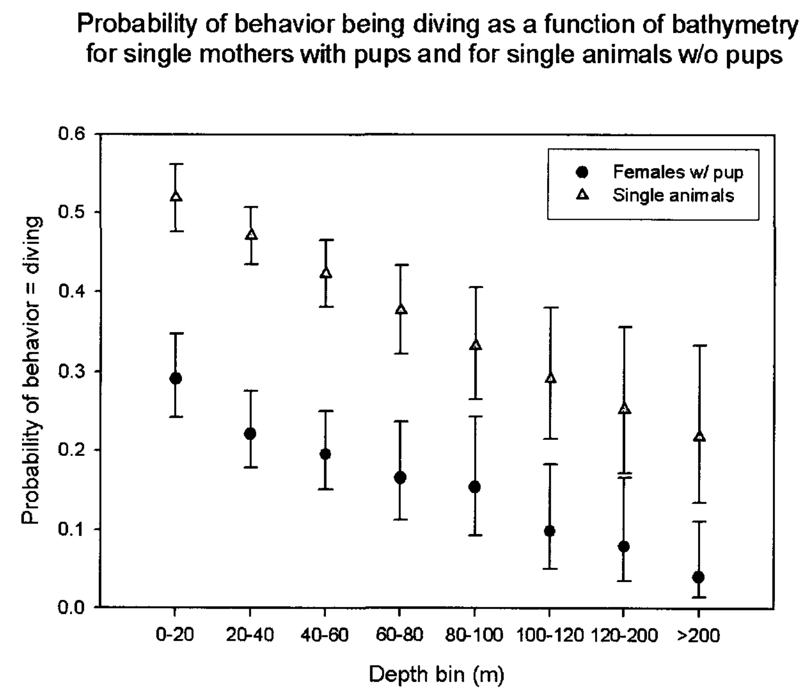

Figure 22 Probability of behavior being diving as a function of depth for both single animals with pups and single animals without pups. Bars indicate 95% confidence interval (alpha = 0.05)

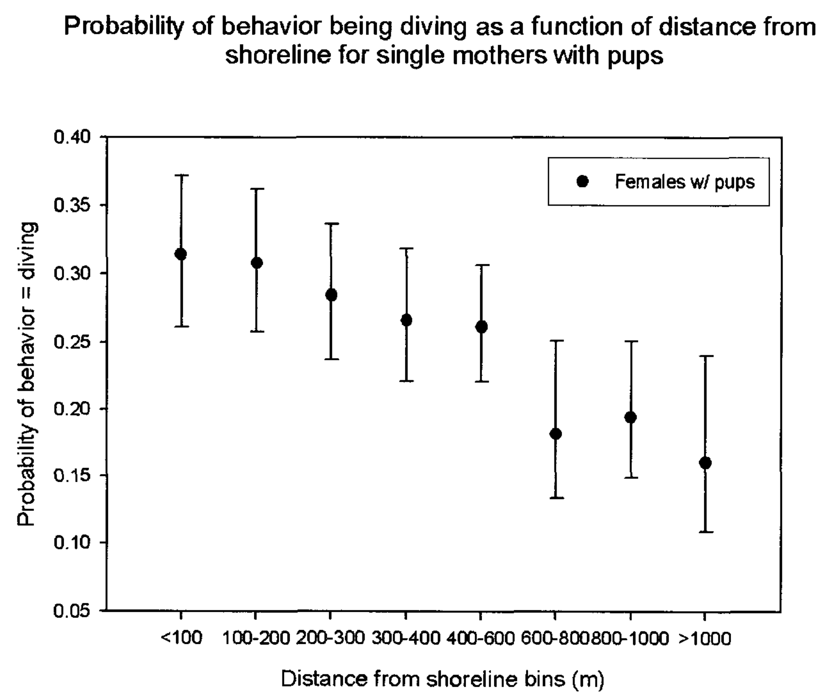

Figure 23 Probability of behavior being diving as a function of distance from shoreline for single animals with pups. Bars indicate 95% confidence interval (alpha = 0.05)

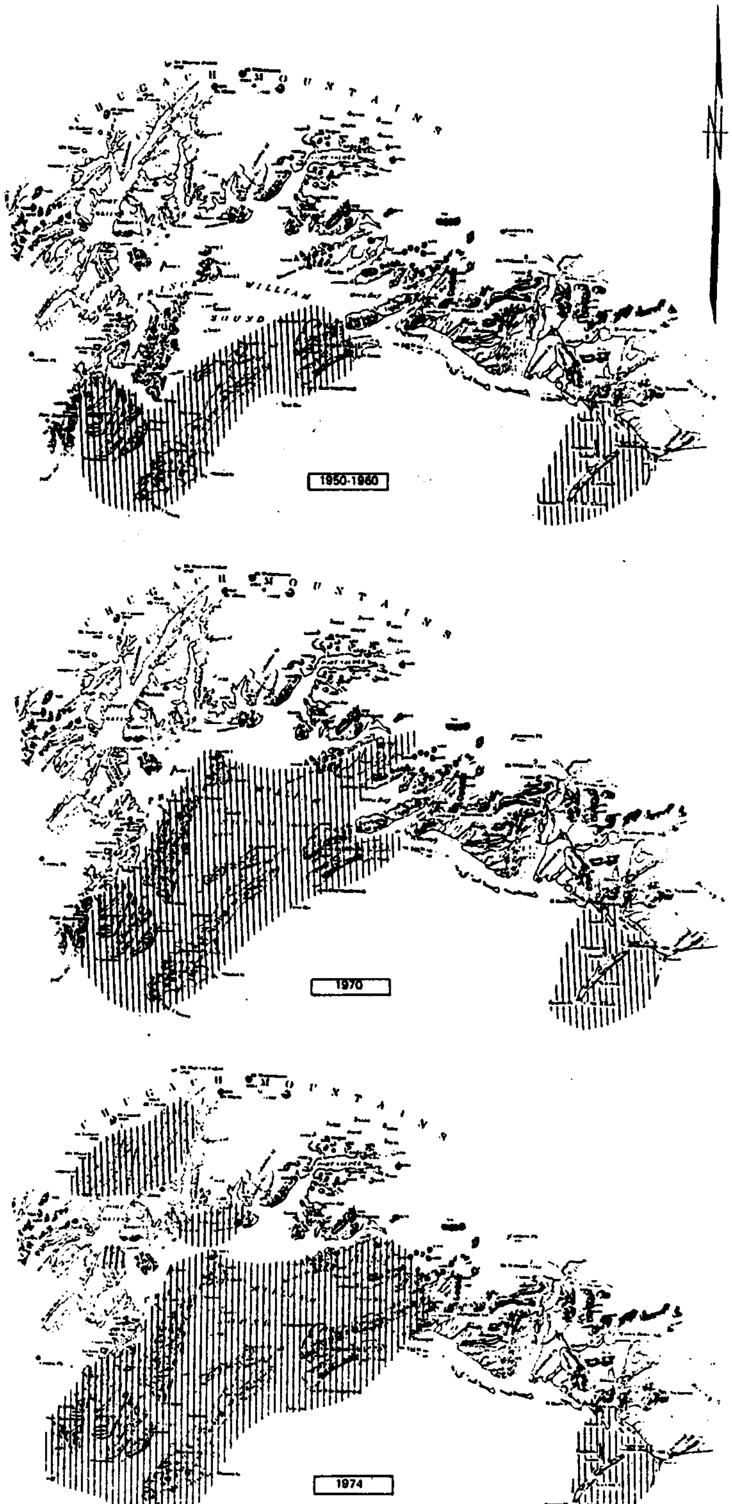

Figure 1. Distribution of sea otters in Prince William Sound, Alaska, from 1950 to 1974. Shaded areas indicate presence of sea otters (Reproduced from Pitcher 1975).

Figure 24 Irons et al. 1988, sea otter distribution and re-colonization. Top image: 1950-1960; middle image: 1997; bottom image: 1974

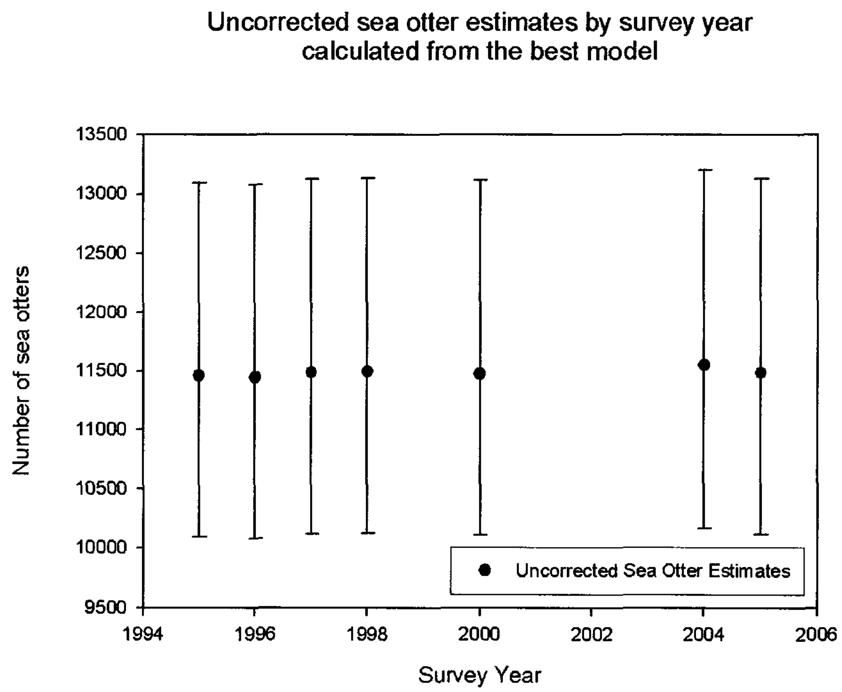

Figure 25 Uncorrected model prediction of sea otter abundance by year. Error bars indicate 95% confidence intervals.

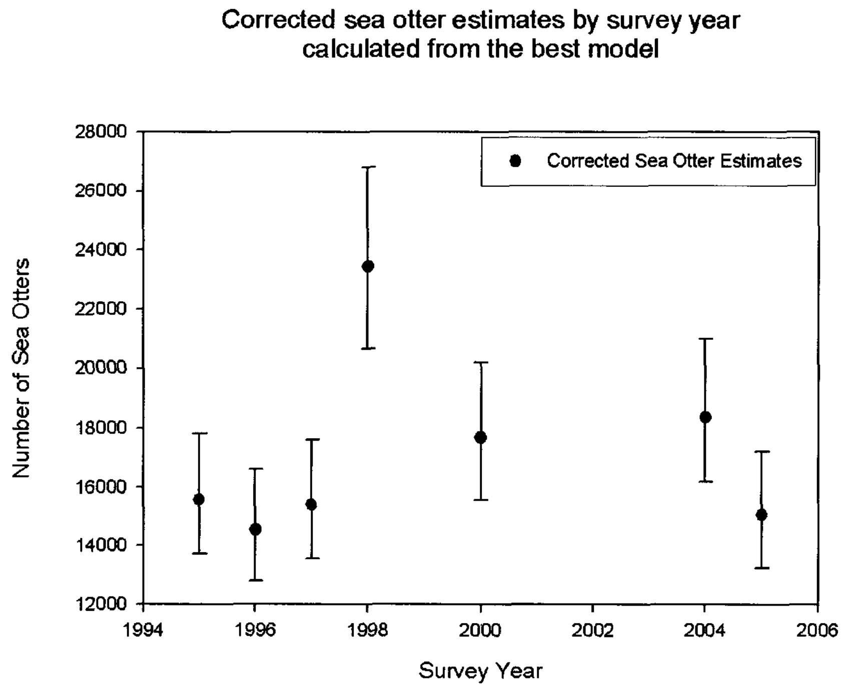

Figure 26 Corrected model prediction of sea otter abundance by year. Error bars indicate 95% confidence intervals.

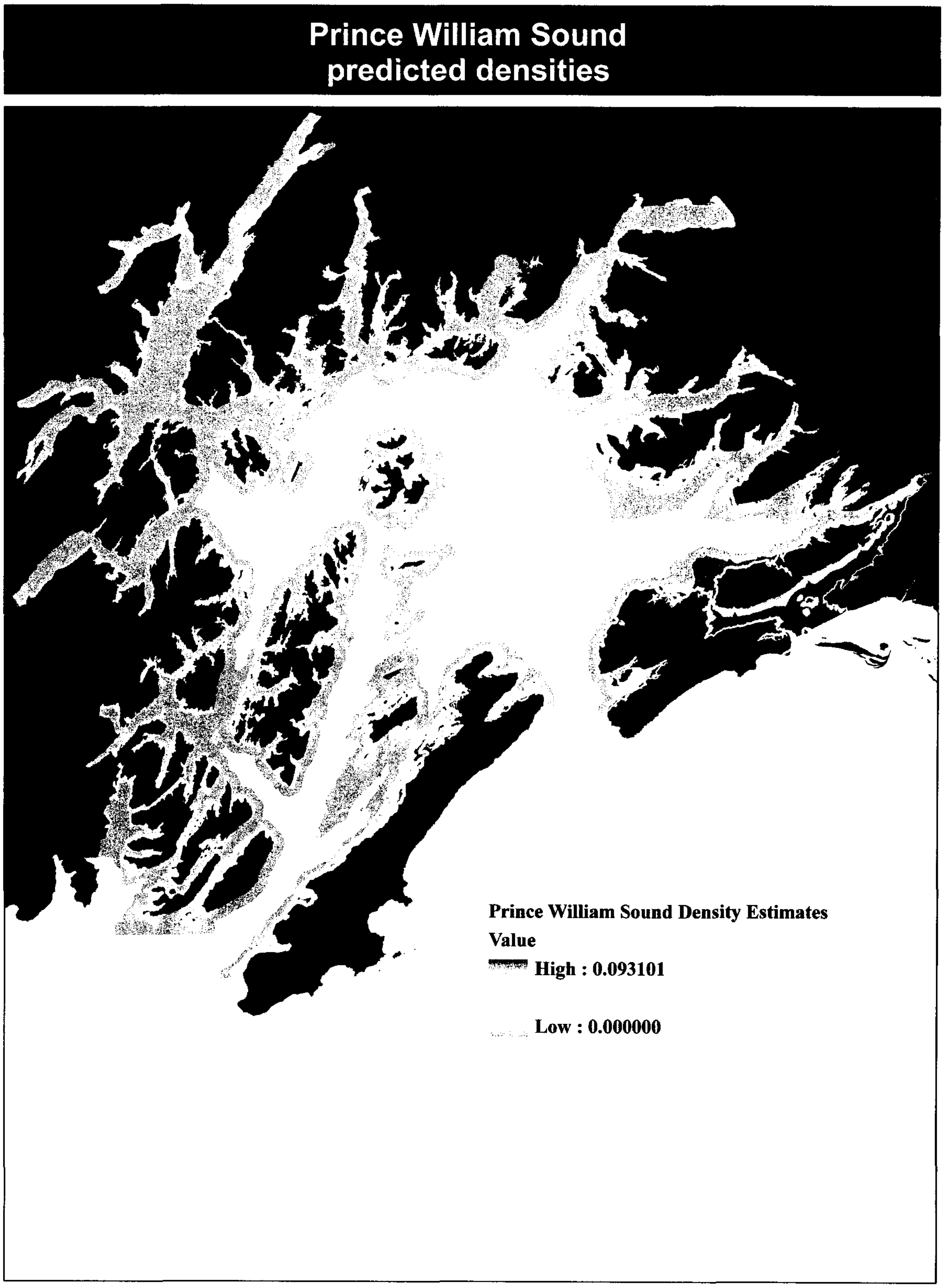

Figure 27 Colored relief of high and low predicted sea otter density. Areas of blue are high density while areas of red are low density.

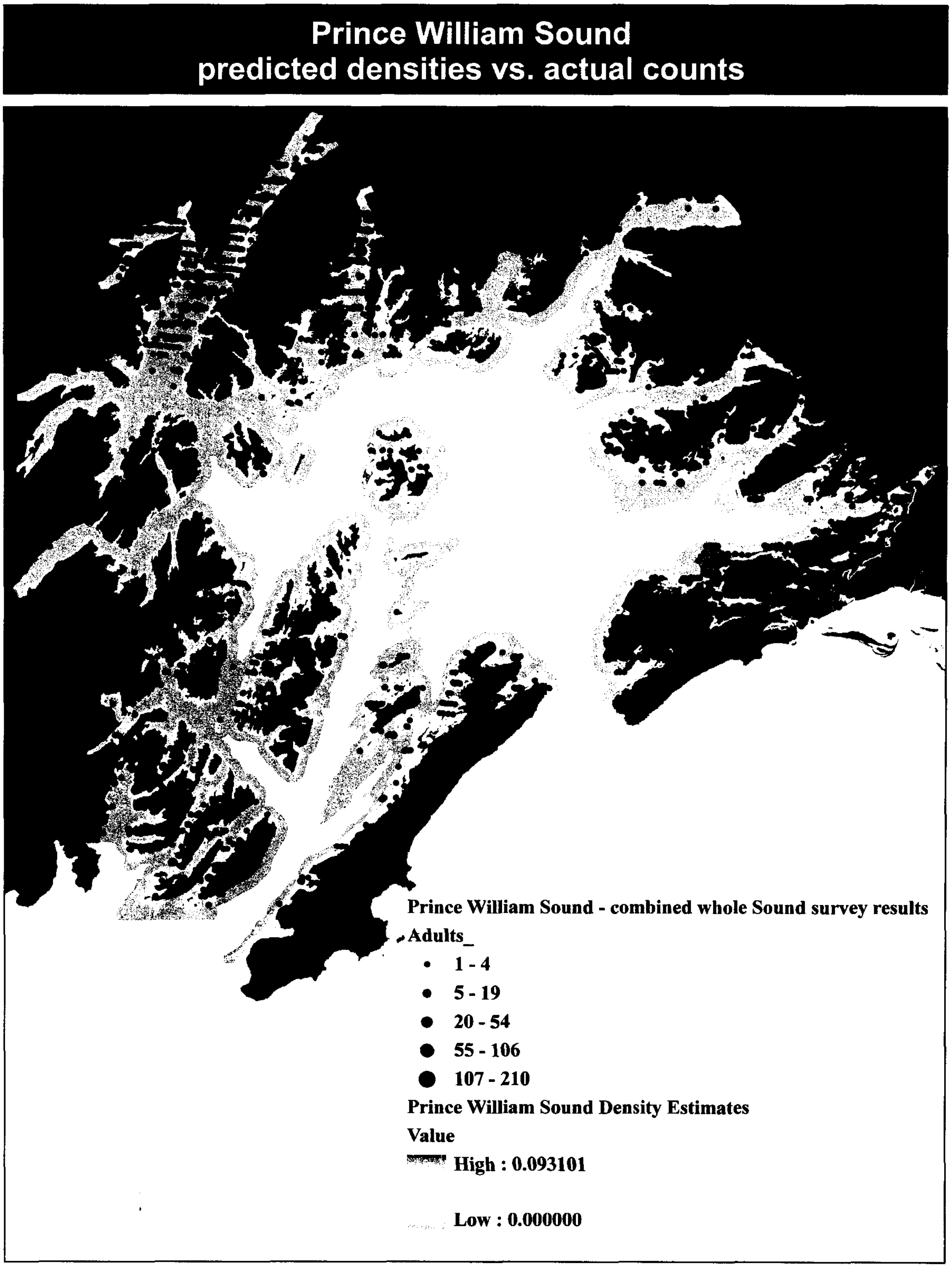

Figure 28 Colored relief of high and low predicted sea otter density. Areas of blue are high density while areas of red are low density. Black dots indicate previous aerial survey counts of all three years of available survey data.

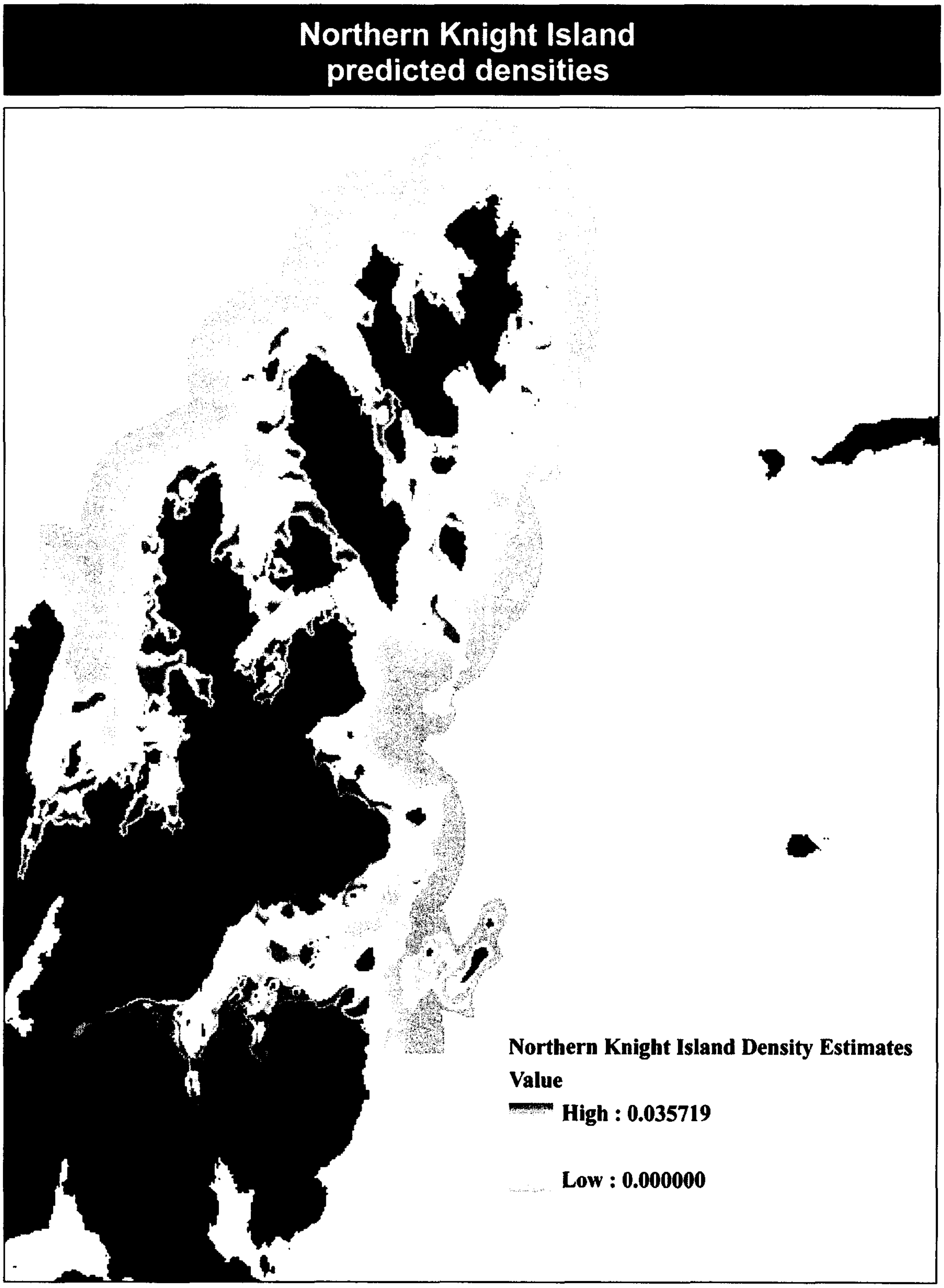

Figure 29 Colored relief of high and low predicted sea otter density. Areas of blue are high density while areas of red are low density.

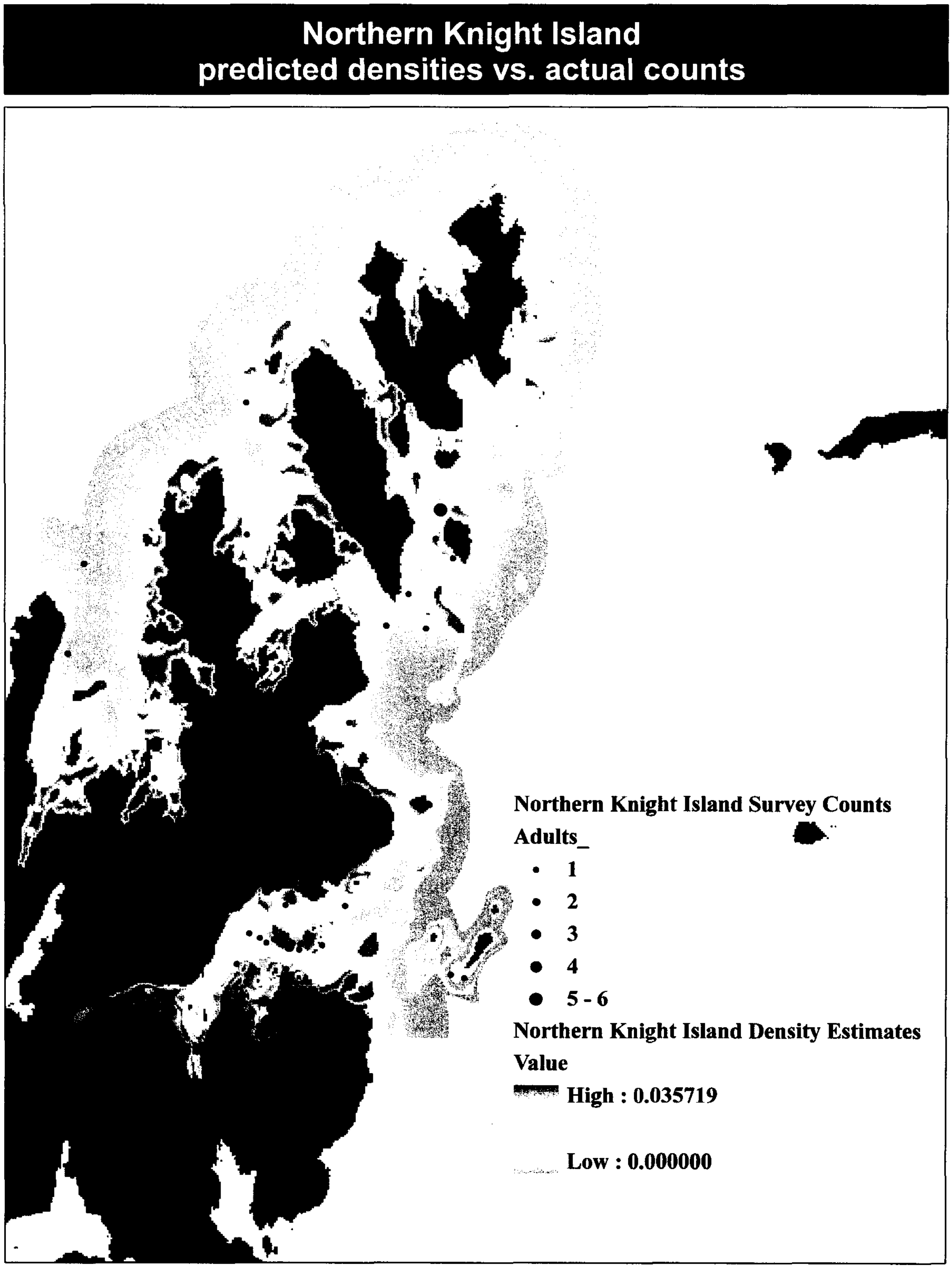

Figure 30 Colored relief of high and low predicted sea otter density. Areas of blue are high density while areas of red are low density. Black dots indicate previous aerial survey counts of all seven years of available survey data.

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