Fisheries and Oceans Canada / Pêches et Océans Canada
Science / Sciences

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Research Document 2008/063 / **Document de recherche 2008/063**

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Abundance of Steller sea lions (*Eumetopias jubatus*) in British Columbia

Abondance de l'otarie de Steller (*Eumetopias jubatus*) en Colombie-Britannique

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ISSN 1499-3848 (Printed / Imprimé)

ABSTRACT

Recent trends in the abundance of Steller sea lions (*Eumetopias jubatus*) in British Columbia were assessed based on a series of 10 province-wide aerial surveys conducted during the breeding season (27-June to 06-July) between 1971 and 2006. Numbers of non-pups (juveniles and adults) increased at an average rate of 3.5% per annum and pup production at a rate of 3.9% per annum during the study period, resulting in a three-fold increase in abundance since the species was protected in 1970. In both cases, numbers on B.C. rookeries appeared to be relatively stable during the 1970s and early 80s, with most of the increases occurring since the mid-1980s. Steller sea lions recently resumed breeding on the Sea Otter Group, a rookery that had been eradicated by predator control programs in the 1920s and 1930s, and the number of year-round haulout sites has almost doubled from about 12 to 23. During the most recent province-wide survey in 2006, we counted 4,118 pups and 15,700 non-pups (7,171 on rookeries and 8,529 on haulouts). Applying correction factors of 1.05 to account for pups obscured in oblique 35mm photographs (Olesiuk et al. 2007) and 1.1 for pups not included in censuses (Trites and Larkin 1998; Pitcher et al. 2007), total pup production in B.C. was estimated to be about 4,800 pups. Based on life tables for a stable population (Calkins and Pitcher 1982; Trites and Larkin 1996) and taking into account the uncertainty in how demographics differ for an increasing population, it was estimated that total abundance could range from 4.0 to 5.8 times the number of pups born. Its thus estimated that at least 20,000 and perhaps as many as 28,000 Steller sea lion currently inhabit coastal waters of B.C. Despite the recent increases, the proportion of the population occupying breeding rookeries appears to have remained relatively constant at about 61% (range 51-67%) over the last 35 years, suggesting that numbers on rookeries provides an index of total abundance. A review of historic counts at rookeries (Bigg 1985) indicated that control programs and commercial harvests conducted in B.C. during 1912-1967 eradicated one breeding area and reduced numbers on the remaining rookeries to about 25-30% of peak levels observed in the early 1900s. Abundance of Steller sea lions in SE Alaska has also increased in recent years, where 5 new rookeries have become established, including what is now the largest Steller sea lion breeding site at Forrester Island just a few kilometres north of the B.C.-Alaska border (Calkins et al. 1999; Pitcher et al. 2007). These recent increases likely represent the recovery of populations from control programs and harvests, but abundance in this region now appears to have surpassed peak historic levels by a factor of two.

RÉSUMÉ

Les tendances récentes affichées par l'abondance de l'otarie de Steller (Eumetopias jubatus) en Colombie-Britannique ont été évaluées à partir d'une série de dix relevés aériens menés à l'échelle de la province pendant la saison de reproduction (du 27 juin au 6 juillet) entre 1971 et 2006. Le nombre de juvéniles et d'adultes s'est accru à un taux moyen de 3,5 % par année, tandis que la production de petits a augmenté à un taux de 3,9 % par année pendant la période d'étude, ce qui représente une multiplication par trois de l'abondance depuis que l'espèce a été protégée en 1970. Dans les deux cas, le nombre d'individus aux roqueries de la C.-B. a semblé demeurer relativement stable pendant les années 1970 et le début des années 1980, la majeure partie des augmentations ayant débuté au milieu des années 1980. Les otaries de Steller ont récemment recommencé à se reproduire dans le groupe Sea Otter, une roquerie qui avait été éliminée par les programmes d'abattage des prédateurs menés dans les années 1920 et 1930, et le nombre d'échoueries occupées à l'année longue a presque doublé, passant d'environ 12 à 23. Au cours du dernier relevé mené à l'échelle de la province, en 2006, nous avons dénombré 4 118 petits et 15 700 juvéniles et adultes (7 171 individus aux roqueries et 8 529 individus aux échoueries). En appliquant des facteurs de correction de 1,05 pour tenir compte des petits non détectés dans les photographies 35 mm prises à angle oblique (Olesiuk et al., 2007) et de 1,1 pour les petits non inclus dans les recensements (Trites et Larkin, 1998; Pitcher et al., 2007), nous avons estimé que la production totale de petits en C.-B. se situait à environ 4 800 individus. Selon les tables de survie pour une population stable (Calkins et Pitcher, 1982; Trites et Larkin, 1996) et en tenant compte de l'incertitude concernant la variation des caractéristiques démographiques d'une population en croissance, nous avons estimé que l'abondance totale pouvait se situer entre 4,0 et 5,8 fois le nombre de nouveau-nés. Nous avons ainsi estimé qu'au moins 20 000 otaries de Steller et peut-être jusqu'à 28 000 d'entre elles vivaient actuellement dans les eaux côtières de la C.-B. Malgré les augmentations récentes, la proportion de la population occupant les roqueries semble être demeurée relativement constante, à environ 61 % (fourchette de 51 à 67 %) au cours des 35 dernières années, ce qui laisse sous-entendre que le nombre d'individus observés sur les roqueries constituait un indice de l'abondance totale. Un examen des dénombrements historiques effectués aux roqueries (Bigg, 1985) révèle que les programmes d'abattage et la chasse commerciale menés en C.-B. de 1912 à 1967 ont entraîné la disparition d'une aire de reproduction et ont réduit le nombre d'individus observés aux roqueries restantes à environ 25 à 30 % des niveaux maximaux observés au début des années 1900. L'abondance de l'otarie de Steller dans le sud-est de l'Alaska s'est également accrue ces dernières années, cinq nouvelles roqueries ayant été établies par les otaries, y compris ce qui constitue maintenant le plus important site de reproduction de l'otarie de Steller, à l'île Forrester, à quelques kilomètres au nord de la frontière entre la C.-B. et l'Alaska (Calkins et al., 1999; Pitcher et al., 2007). Ces augmentations récentes sont vraisemblablement le signe du rétablissement des populations touchées par les programmes d'abattage et la chasse, mais l'abondance dans cette région semble maintenant avoir dépassé les niveaux maximaux historiques par un facteur de deux.

INTRODUCTION

Steller sea lions (*Eumetopias jubatus*) breed along the North Pacific Rim from the Kuril Islands and Kamchatka Peninsula, west through the Aleutian and Pribilof Islands into the Gulf of Alaska, and south along the continental shelf as far as central California. Three stocks are recognized based on genetic differences (Bickham et al. 1996; Baker et al. 2005) and phylogeographic patterns (Loughlin 1997). The Asian stock breeds in Russia west of the Commander Islands and the Western stock breeds in the Commander and Aleutian Islands and Gulf of Alaska west of Cape Suckling (144°W). The Eastern stock breeds in Southeast Alaska, British Columbia, Oregon and north-central California. The Western stock, having declined by about 80% since the 1970s (Merrick et al. 1987; Loughlin et al. 1992; Trites and Larkin 1996; Loughlin 1998, NMFS 2007), was listed as *endangered* under the U.S. *Endangered Species Act,* has been the focus of much research in recent years (NMFS 2007). In contrast, the eastern stock appears to be stable or increasing over much of its range (Calkins et al. 1999; Brown and Riemer 1992; Olesiuk 2003; Pitcher et al. 2007), but it was nevertheless listed as *threatened* in the U.S. due to concerns the declines – which were first observed in eastern Aleutian Islands and spread to the Gulf of Alaska (Braham et al. 1980) – may continue spreading to the Eastern stock, and because there was some uncertainty at the time regarding the genetic division of stocks. In Canada, *COSEWIC* originally concluded in 1987 that the species was *not at risk* (Bigg 1988), but more recently in 2003 recommended the species be designated as *special concern* under the new *Species at Risk Act.* The re-designation was recommended primarily on the basis of the unexplained declines that had occurred in western Alaska, the species is sensitive to disturbances while on land, and there were a limited number of breeding sites (3) in Canadian waters. Given the widespread distribution of Steller sea lions, which spans several state and federal jurisdictions, it is important that survey procedures be co-ordinated and calibrated amongst the various department and agencies responsible for monitoring populations in different parts of their range.

In British Columbia, Steller sea lion populations have been monitored since the early 1970s by conducting province-wide aerial surveys during the breeding season. Bigg (1984, 1985) compiled and examined historic records of kills and counts, and analyzed trends from aerial survey data up to 1982. He concluded that the control programs and commercial harvests had reduced populations to about one-quarter to one-third of historic levels, but that populations had remained stable since being protected in 1970. He estimated that counts on rookeries during 1971-82 averaged about 3,800 (including pups), with an additional 1,900 animals on year-round haulouts, compared with 11,000-14,000 on rookeries when the first studies in B.C. were initiated in 1913. Olesiuk et al. (1993) analyzed survey data for the Queen Charlotte Islands up to 1992 and reported that both pup production and total abundance was slowly increasing at rates of 2.4% and 2.1% respectively. Olesiuk (2003) extended that analysis to the entire B.C. coast up to 2002 and concluded that both pup production and total abundance was increasing at a rate of 3.2% per annum, and noted that in both cases the rate of increase appeared to have accelerated since the 1980s. Pitcher et al. (2007) compiled information for the entire Eastern stock up to 2002 and concluded that Steller sea lions were increasing at similar rates of 2.5% per annum in Oregon and 3.2% per annum in SE Alaska.

In this Research Document, I present the results of the most recent province-wide Steller sea lion survey conducted in B.C. in 2006. The survey results are incorporated into analyses of trends in pup production, total number of animals on rookeries, and total number of animals at haulout sites and rookeries. Historic and recent information on abundance in neighbouring waters, particularly the large rookery – now the world's largest – at Forrester Island in SE Alaska are also reviewed. Using estimates of pup production and life table statistics (Calkins and Pitcher 1982; Trites and Larkin 1996), with adjustments for an expanding population (Pitcher et al. 2007; Olesiuk, unpublished), I also estimate actual abundance in coastal waters of B.C. (including animals at sea or hauled out on sites missed during surveys).

METHODS

Site Classification:

Following Bigg (1985), Steller sea lion haulout sites were classified into 3 distinct categories: 1) *breeding rookeries*; 2) *year-round haulout sites*; and 3) *winter haulout sites* (Table 1; Figure 1). As did Bigg (1985), I used all available information on the presence of sea lions at particular locations for classifying sites. This included not only the systematic survey data presented in this report, but also observations made during harbour seal surveys and other field studies, as well as anecdotal records and unpublished sightings from other researchers, lighthouse keepers, mariners, parks staff, fishing lodges, naturalists, and the general public.

Rookeries are located the farthest from land masses, and generally are the most exposed to oceanic swells. The vast majority (>99%) of births and most breeding activity occurs at rookeries. In their assessment of the Eastern Steller sea lion population, Pitcher et al. (2007) defined rookeries as locations where 50 or more births had occurred, and I adopted the same definition in this assessment. Some animals occupy rookeries throughout the year, but there is a distinct seasonal peak in abundance during the June-August breeding season (Bigg 1985). Social structure is well developed on the rookeries themselves (Edie 1977), but there are usually aggregations of non-breeding animals on their periphery or nearby islands. In many cases, there may be multiple rookery sites located on the same or neighbouring islands separated by up to several 10's of kilometres, and I collectively refer to these larger groupings as *breeding areas*.

Some non-breeding haulout sites appear to be used continuously throughout the year. These sites tend to be situated along the outer coast and exposed to ocean swells, but unlike rookeries are often close to land masses. A few births may occur and matings have been reported by some researchers (Trites, unpubl. data) but not others (Harestad and Fisher 1975). Animals are present in all months, with no marked seasonal variation in abundance (Bigg 1985). The presence of animals on a regular basis, and in particular during the June-July breeding season, is characteristic of these sites, referred to as *year-round haulout sites*.

Steller sea lions also use many additional sites on a seasonal basis. These can be located in exposed locations, as well as in sheltered inlets and channels and sometimes even up rivers. Sites in exposed locations are generally not directly exposed to ocean swells, but rather are sheltered to some extent by the surrounding topography, such as in a bay or on a leeward side of an island. The main period of occupancy is during winter months, but animals can also be present sporadically during May-August. Occupancy can be continuous or intermittent during winter months. The absence of animals, or the presence of only a few animals, or the intermittent use during June-July is characteristic of these sites, referred to as *winter haulout sites*.

Bigg (1985) recognized a fourth type of site referred to as winter rafting areas, where animals rest in groups in the water adjacent to land. These tend to occur more in inshore waters during the winter and were therefore not often observed during breeding season surveys, although we occasionally observed animals rafting adjacent to haulout sites, especially low-lying sites when there was a large oceanic swell. Counts of any rafting or swimming animals observed during the surveys were added to the nearest haulout site.

Survey Procedures:

Aerial surveys were conducted by 1-3 observers from a *DeHavilland* Beaver floatplane flown at an altitude of 150-200 meters and airspeed of 125 $km \cdot hr^{-1}$. In order to insure consistency, the author participated in all surveys since 1982, and all surveys up to 1987 were conducted by the late Dr. Michael Bigg. We flew the 1982 and 1987 surveys together. During the surveys, we flew to all rookeries and attempted to check all known year-round haulouts and as many winter haulout sites as possible. Between known haulout sites we usually scanned the shoreline and opportunistically checked offshore potential hauling areas for new sites, but coverage of the coastline was by no means complete[1].

Visual counts were made of swimming animals and small groups (<5-10) of animals on land. Larger groups were generally photographed with a hand-held 35mm single lens reflex camera equipped with a motor drive and 135-210mm zoom telephoto lens. When light conditions permitted, we preferred to use *Kodachrome* 200 ISO slide film, which in our experience provided greater resolution and warmer contrast, which was especially important for discerning pups on dark substrates. When there was insufficient light to maintain a shutter speed of 1/500th second or less, we switched to *Extrachrome* (or in more recent years *Provia-F*) 400 ISO slide film, which when necessary was exposed and subsequently push-processed at 800 ISO. Light-meter readings were taken from the ocean surface away from land to prevent anomalous readings caused by the reflection off breaking surf. We tended to slightly over-expose film at rookeries which enhanced the visibility of the dark pups, especially on darker substrates. Non-pups were very easy to photograph, but for pups we made a special effort to insure 35mm slides were taken from acute angles and with sufficient magnification for counting pups (see Olesiuk et al. 2007).

The 35mm slides were counted by projecting the image onto white paper using a Prado Leitz projector, which provided superior optics. We began by viewing all passes and selecting the highest quality images. Groups of animals were usually counted from the same pass, so we could use both individual animals and physical features to delineate boundaries between overlapping slides. We generally tried to make counts from the centre of overlapping frames, where optical distortion was minimal. Non-pups were generally very easy to discern and the counts can be considered as representing essentially the exact number present, but more subjectivity was required in identifying pups. Pups were distinguished on the basis of colour and small size and marked on the paper with felt pen, and the marks tallied once the count was complete. We adopted a "balance-of-probability" approach, rather than counting only images that could positively be identified as pups (which would lead to an underestimate) or all images that could possibly have been pups (which would lead to an overestimate). We began by quickly going over the slide and marking those that were very clearly pups, and then carefully deliberating over each of those where there was some uncertainty.

In 2006, we began a transition from using film to digital photography. Photographs at rookeries and haulout sites were taken with a 10.2 mega-pixel *Nikon D200* single-lens reflex camera equipped with a *Nikon* AF-S 80-200mm f2.8 lens. Images and were taken in RAW format, which offered 12 bits of depth for each of the 3 colour channels (compared with 8 bits for compressed JPEG and TIFF files). Digital images and PhotoShop psd files were managed (and if necessary adjusted) in an *Aperture* library in *OS X* on a *MacPro* computer with a 24" *Dell UltraSharp* LCD monitor featuring high-colour range (i.e. 92% colour gamut compared with 72% on standard LCD monitors). Images were counted in PhotoShop CS2 using the Reindeer

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[1]Although shoreline coverage was incomplete during Steller sea lion surveys, the entire shoreline and all possible haulout locations were searched during harbour seal surveys, which were also conducted during the June-August Steller sea lion breeding season, and any new sea lion haulout sites would have been noted.

Graphics Image Processing Tool Kit. Following the methodology developed by Withrow (National Marine Mammal Laboratory, Seattle, WA, pers. comm.) and adapted by Olesiuk (2006) for harbour seal surveys, separate layers were created for pup counts, non-pup counts, and demarcation lines and notes. Animals were marked on each layer with colour-coded symbols using the brush tool, and tallied using the Reindeer Graphics Count filter. To insure consistency of the survey time series, especially for pup counts, in 2006 all rookeries were photographed with both film and digital cameras within a few minutes of each other. Comparison of the counts from the digital images and photographic slides indicated close agreement, and the counts were statistically indistinguishable from one another (Olesiuk et al. 2007).

To facilitate comparison with previous surveys, censuses were conducted under standardized conditions when maximum numbers of animals would be expected to be hauled out. The two most important factors were date and time-of-day. Date was especially important for counts of breeding animals and pups on rookeries which, as will be shown, provide the best index and estimates of the total abundance. Throughout their range, Steller sea lions pup from mid-May to early July (Bigg 1985, Pitcher, unpublished manuscript). On average, less than 10% of births have occurred prior to the end of May, and about 95% have occurred by the end of June (Bigg 1985). Moreover, pups are poor swimmers at birth and are confined to rookeries for about the first month of life (Sandegren 1970). Censuses were therefore conducted in late June or early July (range June 27th to July 6th), by which time most pups had been born, but before they begin to disperse from rookeries. Based on the pupping data given in Eddie (1977), it was estimated that pupping would have been 98-100% complete at the time of the surveys. Since females give birth within a few days of their arrival on rookeries (Edie 1977), their peak numbers coincides with that of pups. Non-pups typically leave for foraging trips in the evening and return in the morning, so we attempted to make counts between 10:00 and 18:00 local time when peak numbers were hauled out (Withrow 1982).

Trend Analysis:

Since it was not always possible to survey all sites due to weather or other logistical constraints, some minor adjustments were made to account for animals that may have been present on sites that were missed. The number of animals missed was estimated by linearly interpolating between the preceding and proceeding counts for that site, or where necessary extrapolated from the first or last count for that site. This differed from the correction method used by Bigg (1985), who applied the mean count for the site over all years it had been surveyed (1971-82). I considered the interpolations to be more appropriate because the population was increasing during the latter part of the study period, whereas Bigg (1985) considered to have been stable. In any event, the corrections had little effect on the overall results, because with the exception of 1973 count, surveys were nearly complete (mean=99.1%; range 94-100%). The 1973 survey was attempted during a period of persistent fog, and 9 sites that were projected to account for 20% of all non-pups were missed. Nevertheless, I opted to include this incomplete survey, since the adjusted count was very similar to others conducted in the 1970s, so it had little influence on the overall trends.

Temporal trends in abundance were assessed by regressing logarithmically transformed counts on time:

[1] $\ln N_t = \ln N_o + rt$

such that:

$$[2]\ N_t = N_o e^{rt}$$

where N_o and N_t denote the numbers of animals estimated to have been present in 1971 and 2006 respectively, and r the exponential or intrinsic rate of increase (Caughley and Birch 1971). The mean finite annual rate of increase, expressed as a percentage, was subsequently calculated as $100(1\text{-}e^r)$.

I also tested for possible changes in population growth rates by fitting piecewise linear regressions to the logarithmically transformed counts on time:

$$[3]\ \ln N_t = \ln N_o + r_1 t + r_2(t\text{-}x)Y_t$$

where x represents the year the rate changed, r_1 and r_2 the intrinsic rates of increases before and after change, and Y_t a dummy variable assigned a value of 0 for all years before and a value of 1 for all years after the rate changed. Whether or not the rate changed and the year in which it changed was determined iteratively by fitting all possible regressions and comparing their fit. The fit of regression models was evaluated based on its *R*-square values adjusted to account for the loss of a degree of freedom with each parameter incorporated into the model (SAS Institute 1998).

Abundance Estimation:

The number of animals counted during surveys represent minimum abundance. Some animals were dispersed at sea and hence missed during surveys. The surveys were flown from site to site with only superficial searching between them, so its possible undocumented sites were missed. However, the vast majority of pups are born on rookeries, and all rookeries were surveyed. Moreover, surveys timed to coincide with time by which pupping is nearly completed and the oldest pups were still too young to disperse, so the pup counts provide a nearly complete record of pup production. We applied a correction of 1.05 to account for pups present on rookeries but hidden in oblique 35-mm images (Olesiuk et al. 2007). Following Trites and Larkin (1996) and Pitcher et al. (2007), we also applied an arbitrary correction of 1.10 to account for the fact some foetuses may have been aborted just prior to the breeding season (pup multipliers were based on late-term pregnancy rates), some pups may have died and/or been washed off the rookery prior to surveys, and some pups may have been born following the survey or at unsurveyed sites.

Total abundance can be extrapolated from pup production based on the ratio of total number of animals to pups in the population as determined from life tables. The only life table for Steller sea lions was based on samples collected in the Gulf of Alaska during 1975-78 (Calkins and Pitcher 1982), which was believed to represent a period of relative stability just prior to the sharp declines that occurred in that area during the 1980s (Loughlin et al. 1992; Trites and Larkin 1996; NMFS 2007). The life table for the stable population indicated a pup to non-pup ratio of about 4.5-4.6:1 (Calkins and Pitcher 1982; Trites and Larkin 1996), but the ratio can vary depending on population status. Life tables are not available for the increasing Eastern stock of Steller sea lions, and its unclear how the life history parameters differ from those of a stable population. I thus explored the potential range of pup multipliers in a growing population using simulations. Sensitivity analyses were conducted by adjusting each one of the four key vital rates (juvenile survival, adult survival, age at first birth, and fecundity rate) independently, and then using matrix projection models to determine how the multiplier changed as a function of the population growth rate. In reality, the vital rates presumably change in unison (e.g. Eberhardt 1985), so the actual multiplier likely falls somewhere within the range indicated by the simulations that adjusted each vital rate separately.

RESULTS

Distribution:

In British Columbia, Steller sea lions pup on 3 main breeding areas: 1) the chain of islands that extend off the north-western tip of Vancouver Island known as the Scott Islands; 2) off Cape St. James at the southern end of the Queen Charlotte; and 3) off the central-northern mainland coast on North Danger Rocks (Figure 1). On the Scott Islands, rookeries are situated on Maggot Island (Figure 2a), on the rocks to the east of Sartine Island (Figure 2b), and at several sites distributed around Triangle Island (Figure 2c). At Cape St. James, rookeries are situated at two sites on the Keourard Islands (Figure 2d). The rookery at North Danger Rocks is comprised of a cluster of rocks, with most pups born on the two larger ones lying to the southwest (Figure 2e). Non-breeding animals, mainly bachelor males and barren females, are distributed near the periphery of rookeries and on adjacent islands. Although several pups were occasionally observed at haulout sites, it was unclear whether they had been born there or had moved there, and they were added to the count for the nearest rookery.

Historically, a fourth breeding site was situated at Virgin and Pearl Rocks in the Sea Otter Group off the central coast. Although a few pups (<5) had been observed in previous surveys, it was utilized mainly by non-breeding animals as a year-round haulout site. However, increasing number of pups have been observed during brand resight and scat collecting trips in recent years (Olesiuk, unpublished. data), and in the 2006 survey a total of 51 pups were counted on Virgin Rocks, and another 4 on Pearl Rocks. These counts surpass the threshold adopted for rookery status (Pitcher et al. 2007), and the site was thus re-designated as a rookery. Steller sea lions are thus currently breeding at all known historic rookeries in Canadian waters. Breeding areas appear to be especially stable, with no new rookeries having been discovered in B.C. during the study period. Indeed, the four breeding areas in use today were all known to have existed when the first sea lion studies in B.C. were published in 1913.

The 4 breeding areas accounted for essentially all (>99%) the pup production in B.C. (Table 2). Although the distribution of pup production has remained fairly stable among the breeding areas, some minor shifts have occurred, especially in recent years (Figure 3). The Scott Islands accounted for 55-65% (mean 61%) of total pup production during the 1970s, 80s and early 90s, but that figure has recently risen to 74% by 2002 and 71% by 2006. In contrast, Cape St. James accounted for 27-36% (mean 32%) of pup production up to the mid 1990s, but that figure has recently declined, to 19% by 2002 and 18% by 2006. North Danger Rocks has accounted for about 5-10% (mean 7%) of total pup production throughout the study period. In 2006, the rookery re-established on the Sea Otter Group accounted for 1.3% of total pup production (Figure 3).

The location of rookeries at Cape St. James and North Danger Rocks has not changed much during the study period, although it now appears that pupping is more widely distributed at the latter site and the rookery on the southern Keourard Islands has shifted slightly. On the Scott Islands, however, there has been a dramatic shift in distribution away from Maggot and Sartine Islands onto Triangle Island. The proportion of Scott Island pups born on Triangle Island increased from about 22-33% (mean 27%) in the early 1970s to 77-83% (mean 80%) in the 1990s and to 92% in 2006 (Figure 4). There has also been a pronounced re-distribution of where animals breed on Triangle Island (Figure 2c). During the 1970s and 80s, the major rookeries were situated on the rocks lying off the north and northeast tip of the island. During the 1980s, animals began pupping in increasing numbers on rocky ledges off the southeast tip of the island, and by the early 1990s breeding animals began to extend onto the pebble beaches and flat rock ledges that run along the southeast coast. The latter two sites now account for nearly 90% of total pup production on Triangle Island. Based on his examination of historic records, Bigg (1985) concluded there was little evidence of movements between the major breeding areas, but noted that pronounced shifts in distribution among the rookeries within each breeding area apparently occurred. Some of these shifts appeared to be related to disturbances during control programs and harvests. Due to the shifts in distribution and movements within major breeding areas, I combined counts for all rookeries within each (i.e. data were aggregated by major breeding area, rather than individual rookery or island) for analyzing population trends.

Bigg (1985) noted that the utilization of year-round and winter haulouts sometimes changed over time, and I made a number of additional revisions to his designations (Table 2). Based on a examination of various sighting records from 1892-1982, and the systematic aerial surveys conducted up to 1982, Bigg (1985) recognized a total of 15 year-round haulouts. However, he noted that one (Isnor Rocks) had been abandoned since the mid-1960s, and two others (Solander Island and Langara Rocks) had been used only as winter haul-out sites since the mid-1960s, so only 12 year-round sites were being utilized during the 1970s. All 12 of those year-round haulouts are still currently in use (but Virgin Rocks has been re-classified as a rookery), and new year-round haulout sites are now used. Bigg (1985) considered Carmanah Point to be a winter haulout, but I re-classified it as a year-round haulout site because animals have been seen there on every survey since 1977, and according to local lighthouse keepers the site is almost always occupied during summer months (J. Etzkorn, Carmanah Lighthouse, B.C., pers. comm.). I suspect animals might have been missed during some of the earlier surveys, since the site is relatively low lying and animals are often forced off and raft nearby when there is a large oceanic swell. Interestingly, the two year-round sites that Bigg (1985) considered to have changed to winter sites in the mid-1960s (Solander and Langara Islands), have been used on a regular basis in during June-July in recent years, and I reclassified them as year-round haulouts. Two other haulout sites Bigg (1985) considered to be winter haulouts – Ashby Point in Queen Charlotte Strait and Mara Rocks in Barkley Sound – have also been used on a fairly regular basis during June-July, and have been reclassified as year-round haulouts. Another winter site at Cherney Island off the northern mainland coast has been occupied by large numbers of animals in the last few years, and animals (mainly bulls) regularly occur during the breeding season at what was formerly regarded as a winter haulout at Rose Spit off the Queen Charlotte Islands (Olesiuk, unpubl. data). Substantial numbers of animals were seen at Isnor Rock off the northern mainland coast during the 2002 and 2006 sea lion surveys, after having been occupied no more than one or two animals for several decades. In addition to the aforementioned changes in haulout use, four entirely new year-round haulouts have been established: animals were first seen at Anthony Island in the Queen Charlotte Islands in 1987, at Warrior Rocks off the northern mainland coast in 1992, and both sites have been inhabited by significant numbers of animals in subsequent surveys. Animals were also first seen at Garcin Rocks in the Queen Charlotte Islands during the 2002 survey and were also present during the 2006 survey. No animals were present when a detailed reconnaissance was conducted during harbour seal surveys in 1992 (Olesiuk et al. 1993), but the site had apparently been used on a fairly regular basis since the late 1990s (Ray Breneman, Canada Parks Service, Queen Charlotte City, B.C., pers. comm.), and may have been overlooked during the 1998 Steller sea lion survey. Animals were first observed on Perez Rocks off the west coast of Vancouver Island in 2004, and large numbers were present in 2006 and during harbour seal surveys in 2007 (Olesiuk, unpubl. data). The number of year-round sites being utilized has thus approximately doubled from 12 to 23 over the last 3 decades.

During surveys, an average of 61% (range 51-67%) of the total count (including pups) has occurred on rookeries. Despite the population increases in recent years (see Recent Trends), this proportion seems to have remained fairly constant over the study period (Figure 5). The number of animals on winter sites that were occupied during the breeding season seems to have increased slightly in recent years, from negligible levels to about 1-4% of total non-pup count in recent years. The latter increase would have been much greater (8-19%), but several sites that had been considered winter haulouts early in the study period have since been re-classified as year-round haulouts (Table 2).

Recent Trends:

Numbers of Steller sea lions increased during 1971-2006. Total abundance of non-pups (at both rookeries and haulout sites) increased at a mean rate of 3.5% per annum (SE=0.37% r^2=0.9135; $F_{(1,9)}$=84.5 P<0.0001) (Figure 6a). Non-pup numbers increased at an average rate of 2.8% on rookeries (SE=0.34% r^2=0.8930; $F_{(1,9)}$=66.8; P<0.0001), and 4.3% at haulout sites (SE=0.59% r^2=0.8640; $F_{(1,8)}$=50.8; P<0.0001), but the rates were not significantly different. Much of the increase appears to have occurred recently, with a piecewise regression model indicating non-pup numbers were stable during 1971-82 ($F_{(1,3)}$=0.13; P=0.7509), but subsequently increased at a rate of about 5.0% (SE=0.17% r^2=0.9937; $F_{(1,6)}$=785.4; P<0.0001). Overall, the number of non-pups counted during surveys more than tripled from an average of 4,718 (SE=249) during 1971-82 to 15,700 by 2006.

Pup production on B.C. rookeries has also increased. During 1971-2006, pup counts increased at a mean rate of 3.9% per annum (SE=0.73%; r^2=0.7737; $F_{(1,8)}$=27.3; P=0.0008) (Figure 5b), which was similar and not significantly different from the rate of increase for non-pups. Similar to the pattern found for non-pups, a piecewise regression model indicated that non-pup numbers were stable during 1971-87 ($F_{(1,4)}$=2.52; P=0.211), but subsequently increased at a rate of about 7.9% (SE=1.17% r^2=0.9139; $F_{(1,4)}$=42.5; P=0.0029). Pup counts exhibited particularly large increases during the last three surveys – the 1998 count was 40% greater than any previous count, the 2002 count was 60% greater than the 1998 count, and the 2006 count was 26% greater than the 2002 count. As a result of the recent increases, pup production has more than quadrupled since the 1970s.

Historic Trends:

The earliest studies of Steller sea lions in British Columbia focused mainly on rookeries, and the few counts available for haulouts are too incomplete to assess trends in total abundance. Furthermore, pups were not always distinguished from non-pups, and in many cases only the total number of animals on rookeries was reported. Nevertheless, the aerial surveys conducted during 1971-2006 indicated a consistent linear relationship between total number of animals on rookeries and province-wide abundance, suggesting that rookery counts serve as a good index of total abundance (Figure 5). The early censuses were made by boat or from vantage points on land, and often on sub-optimal dates or following disturbances, so data and field notes have to be carefully examined when interpreting the historic data. Bigg (1985) presented what is probably the most thorough and objective interpretation possible for the historic data, which I have revised only slightly.

Despite the limitations and potential biases in the historic data, it is clear that abundance of Steller sea lions declined during the first part of the 20th century (Figure 8). When the first systematic studies were conducted in 1913-16, there were at least 11,000 animals on rookeries in B.C. and, when allowance is made for disturbances and the sub-optimal timing of counts, there were more likely about 14,000 animals (pups and non-pups) present (Newcombe and Newcombe 1914; Newcombe et al. 1918; Bigg 1985). Numbers on rookeries had been reduced to about 9,500 by the mid-1950s, mainly because of the eradication of the rookeries on the Sea Otter Group (Virgin and Pearl and possibly Watch Rocks). During 1923-1939, fishery officers visited these rookeries annually toward the end of the pupping season, shooting non-pups with machine guns as they approached with boats, before landing and killing the pups which were generally too young to escape into the water (Figure 7). Thus most of the cohort was killed, and by eliminating any new recruitment to this stock the number on animals returning to breed on Virgin and Pearl Rocks declined exponentially. By the late 1930s numbers had been reduced to a few hundred animals, and since the 1950s the site has been used mainly as haulout with only a few sporadic births. However, increasing numbers of pups have been observed at the site in recent years, and based on the pup count during the 2006 survey the site was re-classified as a rookery.

While not as intensive, control programs were also conducted at the Scott Islands in 1913-1916 and again in late-1930s (Figure 7). Both the Scott Islands and Cape St. James were subject to intense commercial harvesting in the 1960s, which seemed to have had an appreciable impact on abundance. Bigg (1985) estimated that total abundance and pup production on Cape St. James declined by about 70% during the 1960s. During the same period, abundance and pup production on North Danger Rocks was also estimated to have declined by about 70%, and on the Scott Islands it appeared to have declined by about 40%. Thus, by the time the species was protected in 1970, total abundance on rookeries had been reduced to roughly 25-33% of the peak known levels that had been observed just after the turn of the 20th century, which was prior to any large-scale kills.

With the recent increases in abundance on B.C. rookeries, Steller sea lions have largely recovered from the predator control kills and commercial harvests. The numbers of animals counted on B.C. rookeries during the 2006 survey (4,095 pups and 7,171 non-pups) represents 80-100% of the peak historic levels present prior to major kills (Figure 8). In addition to the increases that have occurred in B.C., Steller sea lion populations have also expanded in neighbouring waters in SE Alaska. Steller sea lions were not known to breed in SE Alaska in the early 1900s (Calkins et al. 1999; Pitcher et al. 2007), but established a rookery on Forrester Island just north of the B.C. border as populations were being controlled in B.C. There is little historic information for Forrester Island, but one count made in the summer of 1929 indicated that less than 100 animals were present (Rowley 1929), and another count in August of 1945 indicated only about 350 animals (Imler and Sarber 1947). Forrester Island expanded rapidly while control programs were underway in B.C., with 800 pups counted when the first aerial survey was flown in 1961. Forrester Island is now the largest Steller rookery in the world, with 4,429 pups counted in the most recent survey in 2005 (Pitcher, pers. comm.). Given the close proximity (~50 kilometres) and size of this breeding aggregation, its difficult to assess overall trends without considering the influence of sites in SE Alaska.

The combined abundance of Steller sea lions on B.C. rookeries and Forrester Island observed during the most recent surveys (2005 in SE Alaska and 2006 in B.C.) appears to have surpassed peak historic levels in this region (Figure 8). In addition, several new rookeries have been established in SE Alaska during the last 3 decades, including Hazy Island in about 1985, White Sisters Island in about 1992 (Calkins et al. 1999). Baili Rocks and Graves Rocks have also attained rookery status in the last few years (Pitcher et al. 2007). Overall, breeding populations on rookeries in B.C. and SE Alaska have sustained annual growth rates of about 2.7% since the early 1960s, with pup production having increased at 3.0% per annum over the same period (Figure 9). During the 1960s and 1970s, the increases occurred almost entirely at Forrester Island (numbers were in fact reduced in B.C. during the 1960s). However, beginning in the 1980s, growth began to slow at Forrester Island but increase at B.C. rookeries, and the new rookeries began to be established in SE Alaska. Since the early 1990's, rookeries in B.C. have accounted for 63% of the overall increase in pup production in this region, Forrester Island for 11% of the increase in pup production, and the new rookeries in SE Alaska for 26% of the total increase in pup production. Total pup production has quadrupled over the last 45 years, and total abundance on rookeries in B.C. and SE Alaska has now surpassed the peak historic levels that occurred in the early 1900s by a factor of two. Nevertheless, there is no evidence of the growth rates slowing; pup production has increased at 4.5% since the mid-1970s compared with an average of 3.0% since the early 1960s. The reduced rate of increase during the early part of the time-series may be due to the fact that predator-control kills and commercial harvests were still occurring on B.C. rookeries during the 1960s.

Absolute Abundance:

Total abundance (including animals missed during surveys) can be estimated from pup production by applying multipliers based on the ratio of pups to non-pups as indicated by life tables. Pup production on B.C. rookeries, when adjusted for pups missed in oblique 35mm photos and for pups missed during censuses was estimated to be about 4,800 pups. As noted previously, published life tables are for a stable population in the Gulf of Alaska, and not directly applicable to the increasing populations in B.C. and SE Alaska. Simulations indicate the multiplier (ratio of non-pups to pups) would increase if the population growth in local waters was due to improved juvenile survival, or the multiplier would decrease if the population growth was attributable to increased fecundity or earlier maturation. The multiplier is relatively insensitive to changes in adult survival. The multiplier thus potentially ranges from 4.0 to 5.8 for a population increasing at 4.5% (the average rate observed in B.C. and SE Alaska over the last 3 decades) (Figure 10). Thus, it would require a population of 19,000 to 28,000 to support the level of pup production observed on B.C. rookeries in 2006. During the 2006 survey, a total of 19,818 animals were actually counted on the B.C. coast and some animals were undoubtedly missed, suggesting the actual abundance may be nearer the higher end of our extrapolated estimate.

Given the close proximity of the Forrester Island rookery to the border, it may be more meaningful to consider abundance for B.C. and SE Alaska combined. Based on total pup production during the most recent surveys (2005 in SE Alaska and 2006 in B.C.), total combined pup production is estimated at about 10,800 pups. Based on the calculations outlined above, this represents a total population size of 43,000-63,000. During the 2005/2006 surveys, 44% of pup production in this region occurred on Canadian rookeries. During the most recent complete survey of non-breeding haulout sites in 2002, 43% of non-pups in this region were counted in Canadian waters. Based on this distribution, its estimated that 19,000 to 27,000 Steller sea lions inhabit Canadian waters, which was very similar to the number estimated to support the pup production observed on local rookeries. Thus, although Steller sea lions are trans-boundary species, it appears there is roughly equivalent exchange between animals breeding in B.C. and those breeding in SE Alaska and other areas.

DISCUSSION

Steller sea lion populations in British Columbia have been growing in recent years. The growth is evident from significant increases in pup production, in numbers of non-pups observed on rookeries, and in total number of animals observed on rookeries and haulout sites. In contrast, when Bigg (1985) analyzed survey data up to 1982, he concluded that populations were stable and had not exhibited any significant recovery since being protected in 1970. This discrepancy can be explained by the fact that most if not all of the increase in non-pup numbers has occurred since the 1982 survey, and the growth in pup production has occurred since the 1987 survey. The shorter time series may also have provided less power for detecting population trends, particularly for such modest rates of increase, but even in retrospect there appears to be little evidence of growth of populations in B.C. prior to 1982.

Our time-series of surveys were all conducted within a fairly narrow window (27-June to 06-July), so it is unlikely that the trend estimates have been confounded by seasonal patterns in attendance at haulouts or rookeries. Most of the counts were within the prescribed window of 10:00 to 18:00 when peak numbers would be expected to be on land (Withrow 1982), so it is also unlikely trends were confounded by diurnal patterns in attendance. Surveys were conducted without regard to tide, which has been reported to have significant effects at some sites (Kastelein and Weltz 1990), but not at others (Withrow 1982). Most of the year-round haulouts were comprised of relatively large islets with substrate available at all tides, and I suspect that swell height would have been a more important factor. Calkins et al. (1999) applied co-variate analysis to account for date, tide and time effects, but none of the resulting trends estimates differed significantly as a result of the adjustments. Moreover, I am somewhat sceptical of the predictive value of their co-variates, since they often fitted multivariate quadratic equations to as few as 6 observations, and their time-series spanned only 7 years. The issue of co-variate effects warrants further examination with more extensive datasets, but their effects are likely small relative to the sustained growth observed over the last 45 years.

Increases in Steller sea lion abundance have also been reported in the waters neighbouring B.C. The species was not known to breed in SE Alaska until the rookery at Forrester Island became established, probably sometime near the middle of the 20th century (Bigg 1985). It was initially a minor site with about 50-100 animals during the 1920s with no mention of pupping (Rowley 1929), and perhaps 350 animals during the mid-1940s (Imler and Sarber 1947). However, it grew rapidly during the 1950s, 60s and 70s, and now represents the single largest breeding aggregation of Steller sea lions. Growth at Forrester Island slowed by the early 1980s, but other new rookeries were established in SE Alaska at Hazy Islands in the 1980s and White Sisters Island in the 1990s, and by 1997 total pup production in SE Alaska was estimated at 4,160 (Calkins et al. 1999). Calkins et al. (1999) estimated pup production increased at a rate of 5.9% during 1979-97, but appears to have slowed to 1.7% during 1989-97. That slowing appears to coincide with the sharp increase in pup production that began to occur on B.C. rookeries in the mid-1980s. Also, two additional rookeries appear to have been established at Graves Rocks and Biali Rocks during the 1990s, with total pup production in SE Alaska increasing to 5,510 by 2005 (Pitcher et al. 2007). Overall, combined pup production in B.C. and SE Alaska, which are difficult to separate due to the large rookery near the border, seems to have sustained steady growth of about 3.0% since the early 1960s. Forrester Island accounted for most of the growth up to the late 1970s, but since the early 1980s most subsequent growth has occurred at the new rookeries established in SE Alaska and at the existing rookeries in B.C. Steller sea lions do not breed off Washington, but numbers in Oregon also appear to be have increased at a mean rate of 2.5% during 1977-2002, and abundance has more than doubled (Brown and Reimer 1992; Pitcher et al. 2007).

The recent increases almost certainly represent, at least in part, the recovery of populations that had been depleted during the 1913-1967 by control programs and commercial harvests. By the time the species was protected in 1970, abundance in B.C. was estimated to have been reduced to 25-33% of levels that existed at the turn of the century, which was prior to any large scale control programs or harvests (Bigg 1985). Since 1970, abundance in B.C. has more than tripled, and the breeding population is currently estimated to be at 80-100% of peak historic levels. While large-scale killing was underway in B.C., only one small kill (187 animals at Forrester Island in 1960; Bigg 1984) was known to have been made in SE Alaska, which may explain why sea lions flourished in that region. With the colonization of new rookeries in SE Alaska, and the recent growth at B.C. rookeries, it now appears the species has fully recovered, with breeding populations currently twice as large as the peak known historic levels present in the early 1900s. Assuming that carrying capacity has not changed, one might expect density-dependent mechanisms to become more prevalent, and populations to stabilize (or exhibit long-term oscillations). Its also possible that Steller sea lion populations in B.C. and SE Alaska had already been depleted by subsistence harvesting when the first surveys were conducted in the early 1900s (Newcombe et al. 1918; Wailes and Newcombe 1929; Alaska Department of Fish and Game 1973; Bigg 1985). Native populations and their reliance on sea lion products both declined during the 1800s (Duff 1977). The first Steller sea lion studies in B.C. were prompted by increasing complaints about growing sea lion numbers and their impacts on salmon fisheries, but its unclear to what extent the complaints were attributable to the growth of sea lions populations or expansion of salmon fishing operations.

The only published life tables for Steller sea lions are for a stable population, and its unknown which or how vital rates differ to account for the increasing trends observed in B.C. and SE Alaska. Simulations indicate that the multiplier (ratio of total animals to pups) could vary from as low as 4.0 if the population growth were attributable to increased fecundity or earlier maturation, to as high as 5.8 if the growth were due to increased juvenile survival, and intermediate if due to improvement in adult survival. In reality, several if not all of these life history parameters probably vary concurrently, so the actual multiplier probably lies somewhere within this potential range. The total number of animals observed in B.C. during the most recent survey in 2006 and in the entire Eastern stock during the last range-wide survey in 2002 was actually slightly above (105-113%) the population estimates corresponding to the lowest pup multiplier. Its therefore unlikely that the population growth was due to just increased fecundity or earlier maturation. Based on the number of non-pups observed in surveys in the Gulf of Alaska compared with the number expected based on life tables, Loughlin et al. (1992) suggested that 75% of non-pups were represented in the survey counts. If this were the case on our surveys, the total B.C. population would be about 26,000, near the high end of our range based on pup multipliers. The high ratio of non-pups to pups observed in B.C. (3.6 non-pups per pup in the most recent survey in 2006) and in fact the entire Eastern stock (3.5 non-pups per pup in the most recent range-wide survey in 2002; Pitcher et al. 2007) compared with 2.6 non-pups per pup in surveys in the Gulf of Alaska (Loughlin et al. 1992) also suggests the difference in status of the two populations are more likely due to differences in juvenile survival rather than fecundity or age at maturity. In support, York et al. (1994) examined changes in age structure in samples taken before and following the decline in the Gulf of Alaska, and concluded that juvenile survival was the main driving factor, with adult survival also playing a role (Holmes and York 2003). In the absence of more specific information on how the demographics differ between stable and increasing populations, it can be concluded that at least 20,000 and perhaps as many as 28,000 Steller sea lions currently inhabit Canadian waters.

The recent increases in eastern stock contrast sharply with the western stock, where numbers have declined by about 80% since the 1970s, and are now considered as *endangered* (Merrick et al. 1987; Loughlin et al. 1992; Trites and Larkin 1996; Loughlin 1998, NMFS 2007). The reasons for the decline in the Gulf of Alaska and Bering Sea are poorly understood, although a leading hypothesis is nutritional stress caused by ecological changes that resulted in reduced availability or diversity of prey (Alverson 1992; Alaska Sea Grant 1993; DeMaster and Atkinson 2002; Trites and Donnelly 2003). While these ecological processes are still poorly understood, Steller sea lions are likely limited by bottom-up forcing and could serve as an indicator of the state of marine food chains. With the recovery of Steller sea lions, which are now at twice the peak known historic levels that occurred a century ago, their prey requirements and consumption and potential interactions and impacts on other fishery resources warrants further assessment.

ACKNOWLEDGEMENTS

I am particularly grateful to the late Michael Bigg for introducing me to sea lion surveys and laying the foundation for this assessment, and to Graeme Ellis, John Ford, Chrys-Ellen Neville, and Debbie Lacroix for assisting with aerial surveys, to Don Calkins, Tom Gellatt and Ken Pitcher for making the necessary arrangements that allowed us to extend our surveys to Forrester Island and providing access to Alaskan data, to Wayne Perryman, Morgan Lynn, and Charlie Stinchcomb for undertaking the medium-format pup surveys, to John Sease for sharing ideas on analysis of sea lion counts, and to the many pilots who got us to these remote sites safely, in particular the late Tom Blaesing, Jim Kirk, Louis Rouleau, and Dave Weintraub.

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Table 1. Names and locations of Steller sea lion haulout sites in British Columbia. Only sites where animal were observed during breeding season surveys in 1971-2002 are listed; see Bigg (1984, 1985) for more extensive data on abandoned and winter haulout sites.

Site Name	Latitude		Longitude	
	Degrees	Minutes	Degrees	Minutes
CARMANAH PT	48	36.92	124	45.68
MARA ROCK	48	52.53	125	28.66
LONG BEACH	49	2.33	125	43.13
RAPHAEL PT	49	18.50	126	13.67
BARRIER ISLS	50	0.74	127	31.57
O'LEARY ITS	50	6.14	127	38.77
SOLANDER ISL	50	6.71	127	56.41
CAPE SCOTT	50	47.10	128	25.19
MAGGOT ISL	50	48.11	128	46.76
BERESFORD ISL	50	47.52	128	46.18
SARTINE ISL	50	49.19	128	54.18
TRIANGLE ISL	50	52.28	129	4.64
ASHBY POINT	50	56.53	127	55.24
VIRGIN ROCKS	51	16.81	128	12.24
PEARL ROCKS	51	21.79	128	0.11
GOSLING ROCKS	51	52.14	128	27.59
MCINNES ISL	52	15.75	128	43.25
STEELE ROCK	52	27.83	129	22.25
ASHDOWN ISL	53	2.95	129	12.68
NORTH DANGER RKS	53	15.34	130	20.50
BONILLA ISL	53	27.97	130	36.77
REEF ISL	52	52.30	131	29.06
CAPE ST. JAMES	51	54.67	130	58.91
S TASU HD	52	42.17	132	4.50
MORESBY ITS	52	58.36	132	21.55
CONE HD	53	22.57	132	43.27
JOSEPH ROCKS	53	48.83	133	8.00
LANGARA ISL	54	15.63	133	0.88
ANTHONY ISL	52	6.20	131	14.40
WARRIOR ROCKS	54	3.88	130	51.12

Table 2. Number of non-pup Steller sea lions counted during province-wide breeding season surveys during 1971-2006. Sites were classified as R-rookeries, Y-year-round haulouts, and W-winter haulouts, although in some cases site use appeared to change over the course of the study period. NS denotes the site was not surveyed and animals likely missed, and (NS) denotes the site was not surveyed but it was not expected that animals were missed based on the preceding and proceeding surveys. ?- denotes the site was not known to exist, and could have been overlooked. The estimated number of animals missed (and the number of missed sites) is given near the bottom of the table.

Site Name	Type	28 June to 30 June 1971	29 June to 03 July 1973	27June to 30 June 1977	28 June to 01 July 1982	29 June to 03 July 1987	28 June to 03 July 1992	28 June to 01 July 1994	29June to 04 July 1998	02 July to 06 July 2002	01 July to 03 July 2006
CARMANAH PT	Y	0	NS	181	170	146	103	150	255	237	247
PACHENA PT	W	0	0	0	0	0	0	0	0	0	44
WOUWER ISL	W	0	0	0	0	0	0	0	0	31	4
MARA ROCK	W/Y	0	(NS)	0	3	0	0	41	87	296	264
LONG BEACH	Y	394	265	10	262	231	344	298	535	714	388
PEREZ RKS	Y									?	353
RAPHAEL PT	W	0	0	0	0	0	0	58	0	0	0
FERRER PT	W	0	0	0	0	0	0	0	0	0	16
BARRIER ISLS	Y	NS	NS	105	153	149	274	290	843	585	542
O'LEARY ITS	Y	331	NS	200	85	60	81	14	74	2	141
SOLANDER ISL	W/Y	0	3	1	0	0	51	419	179	187	876
CAPE SCOTT	W	0	(NS)	1	0	1	42	68	0	0	0
MAGGOT ISL	R	418	416	627	442	550	511	371	245	456	603
BERESFORD ISL	R	71	6	24	100	124	164	119	5	147	
SARTINE ISL	R	628	616	879	806	600	575	343	262	268	379
TRIANGLE ISL	R	550	375	570	376	1057	1603	1626	2540	2995	3576
ASHBY POINT	W/Y	NS	82	4	1	210	3	226	225	519	786
BUCKLE GROUP	W								(NS)	47	2
VIRGIN ROCKS	Y	317	205	62	190	229	157	131	168	419	516
PEARL ROCKS	Y	100	81	276	23	128	126	98	199	467	449
GOSLING ROCKS	Y	106	NS	37	179	135	72	192	133	160	257
MCINNES ISL	Y	196	NS	45	0	0	109	241	163	25	60
STEELE ROCK	Y	NS	NS	85	150	7	35	137	227	101	92
ASHDOWN ISL	W	(NS)	(NS)	0	NS	NS	25	NS	0	(NS)	(NS)
ISNOR	Y								0	72	29
N DANGER RKS	R	148	347	230	288	339	301	309	583	592	1003
BONILLA ISL	Y	29	158	333	219	19	265	272	303	215	375
CHERNEY ISL	W/Y								0	19	498
ROSE SPIT	Y								0	?	30
REEF ISL	Y	207	105	88	36	482	489	538	216	370	253
SKEDANS	W	0	(NS)	0	45	0	0	0	0	0	0
CAPE ST. JAMES	R	631	549	782	698	1021	867	797	763	982	1094
S TASU HD	Y	76	NS	278	117	263	80	196	285	151	47
MORESBY ITS	W	(NS)	(NS)	(NS)	(NS)	0	3	115	65	2	1
CONE HD	W	(NS)	(NS)	(NS)	(NS)	0	70	21	1	131	27
JOSEPH ROCKS	Y	408	NS	399	366	309	327	397	601	696	770
LANGARA ISL	W/Y	6	NS	0	3	3	NS	0	217	3	484
ANTHONY ISL	Y	?	?	?	?	44	279	617	359	313	513
WARRIOR ROCKS	Y	?	?	?	?	?	416	2	282	588	692
GARCIN ROCKS	Y								?	329	261
Miscellaneous	-	1		2	1	2	4	5	3	3	28
Number Counted	-	4617	3208	5219	4713	6109	7376	8091	9818	12121	15700
Missed (sites)	-	272(3)	831(9)	0(0)	13(1)	13(1)	2(1)	13(1)	0(0)	0(0)	0(0)
Total Estimated	**-**	**4889**	**4039**	**5219**	**4726**	**6122**	**7378**	**8104**	**9818**	**12121**	**15700**

Table 3. Number of Steller sea lion pups counted during province-wide breeding season surveys during 1971-2006.

Site Name	28 June to 30 June 1971	29 June to 03 July 1973	27June to 30 June 1977	28 June to 01 July 1982	29 June to 03 July 1987	28 June to 03 July 1992	28 June to 01 July 1994	29 June to 04 July 1998	02 July to 06 July 2002	01 July to 03 July 2006
MAGGOT ISL	174	188	147	171	180	107	76	72	77	62
SARTINE ISL	163	273	309	409	176	253	62	148	146	178
TRIANGLE ISL	181	189	140	185	305	476	630	1211	2199	2674
VIRGIN RKS	0	0	0	0	2	0	0	0	1	55
N DANGER RKS	86	93	64	74	54	148	84	144	219	403
CAPE ST. JAMES	337	272	303	404	367	484	333	484	635	723
Miscellaneous	0	0	0	2	0	0	1	4	4	23
B.C. Total	**941**	**1015**	**963**	**1245**	**1084**	**1468**	**1186**	**2073**	**3281**	**4118**
FORRESTER ISL	NS	2371	NS	2120	2073	NS	2073	2364	NS	NS

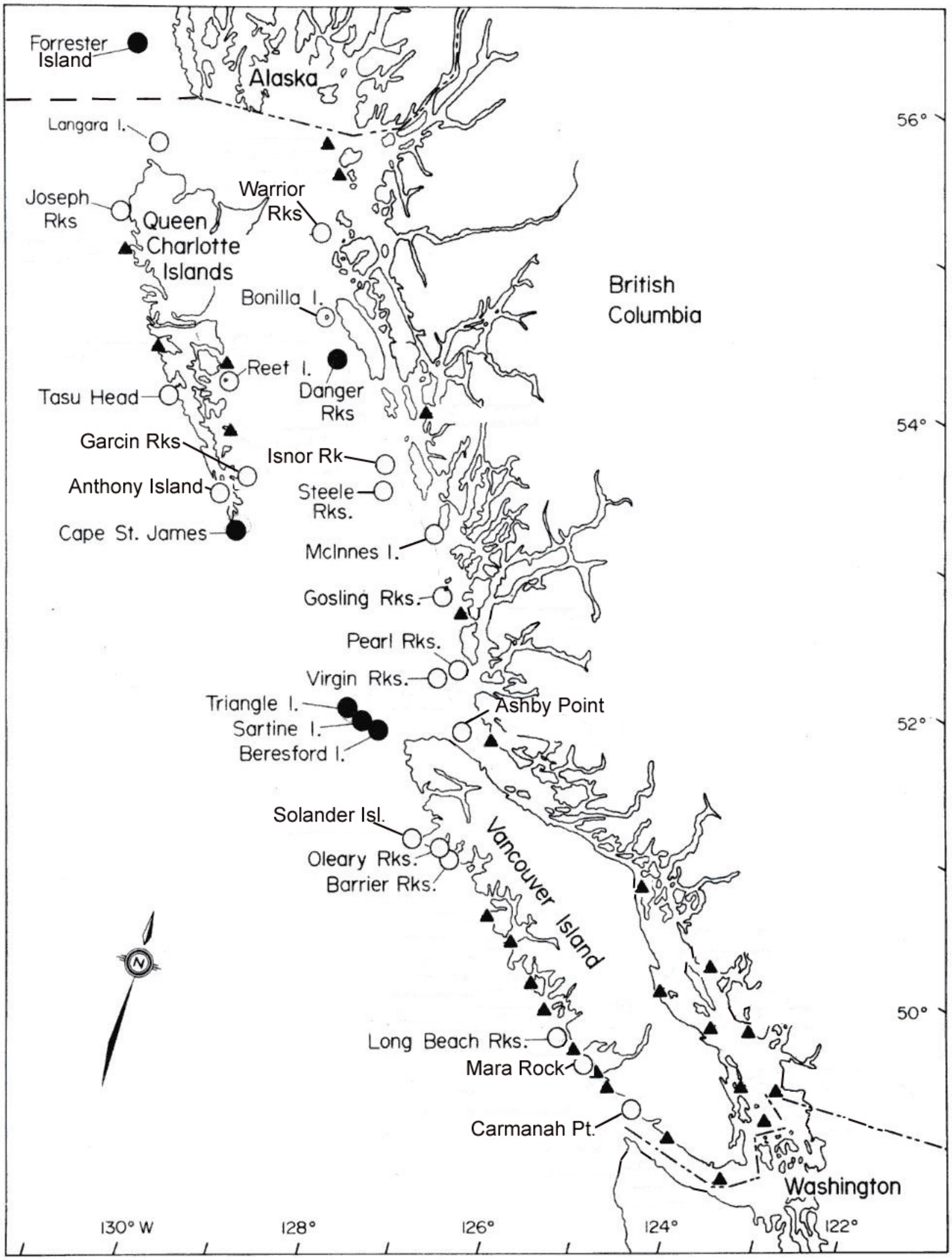

Figure 1. Map showing location of Steller sea lion breeding rookeries (●), year-round haulout sites (○), and major winter haulout sites (▲) in British Columbia and at Forrester Island, Alaska (modified from Bigg 1985).

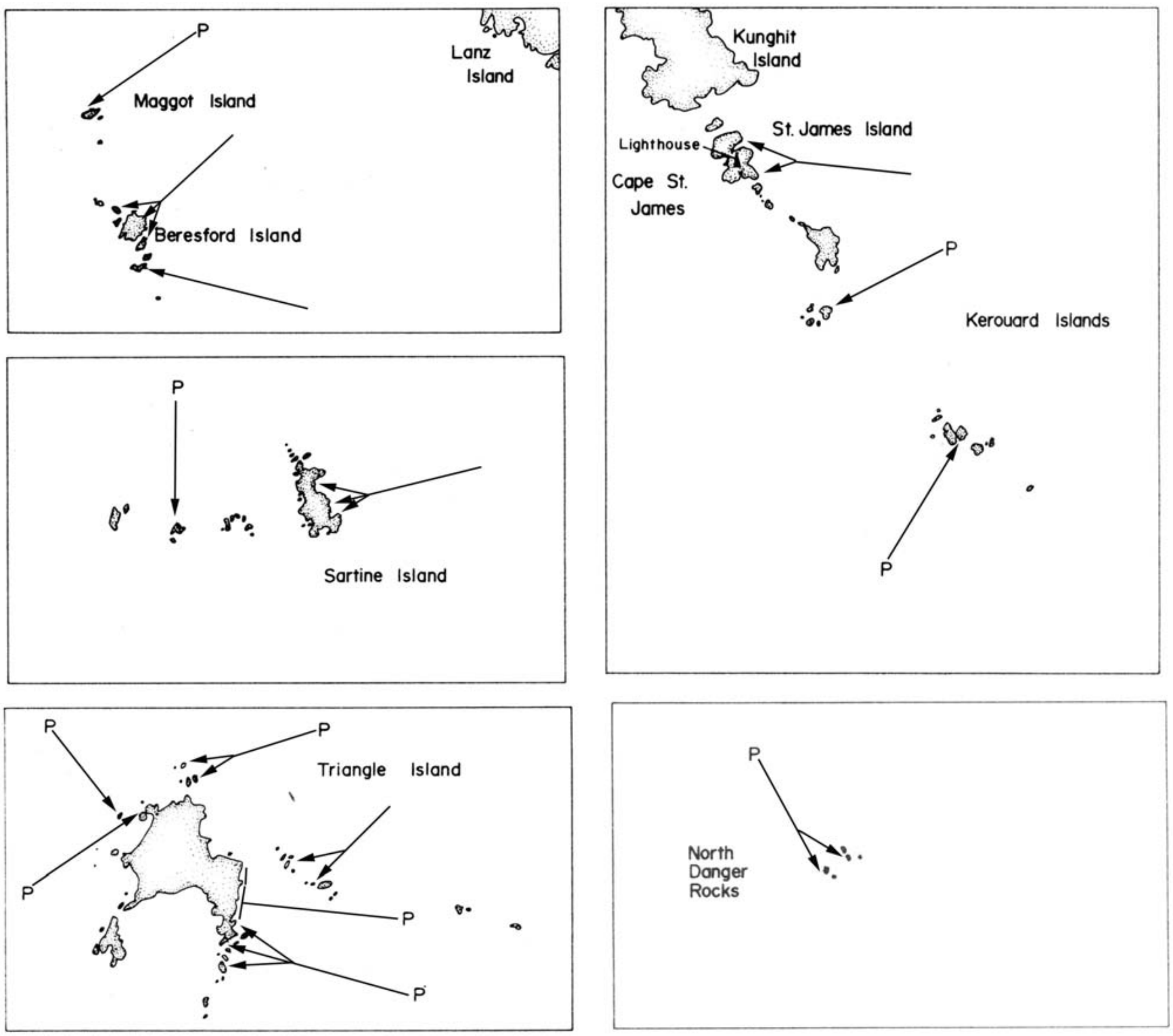

Figure 2. Detailed maps showing locations of rookeries. Arrows show location of aggregations of animals, and those denoted with P’s denote major pupping areas (modified from Bigg 1984).

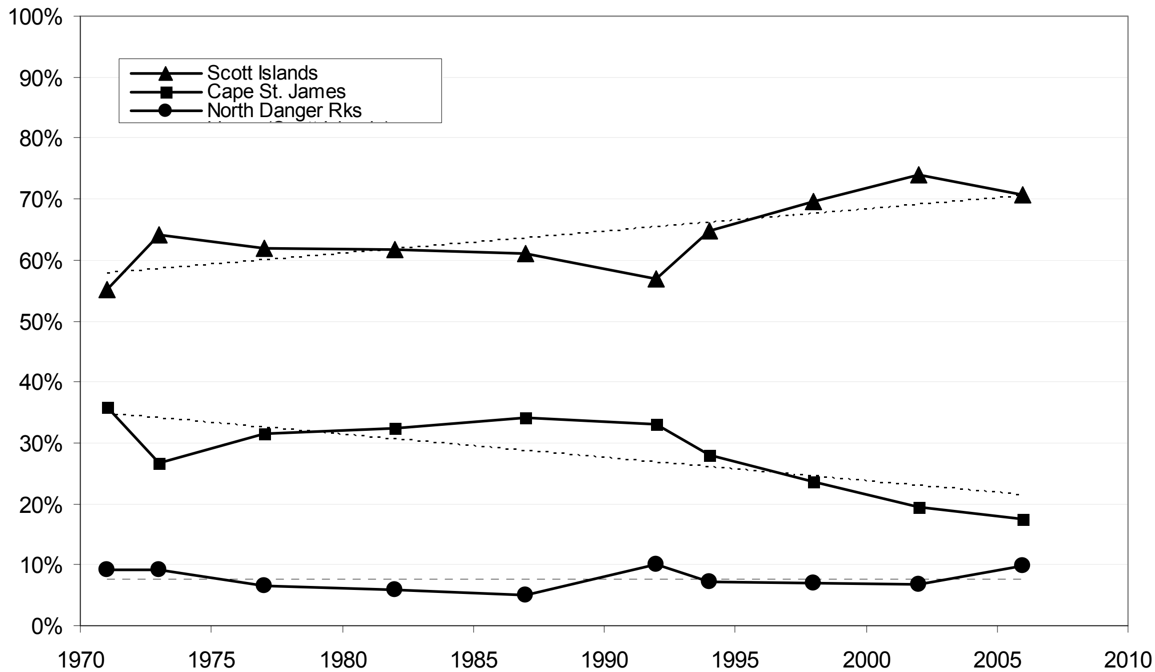

Figure 3. Temporal changes in relative distribution of pup production among major breeding areas in B.C. during 1971-2006. Dashed lines denote regressions showing long-term average rates of change over time. Regressions indicate a significant decreasing trend for Cape St. James ($F_{(1,9)}$=10.2; P=0.0129), and a significant increasing trend for the Scott Islands ($F_{(1,9)}$=9.3; P=0.0157), but no significant trend for North Danger Rocks ($F_{(1,9)}$=0.001; P=0.9922).

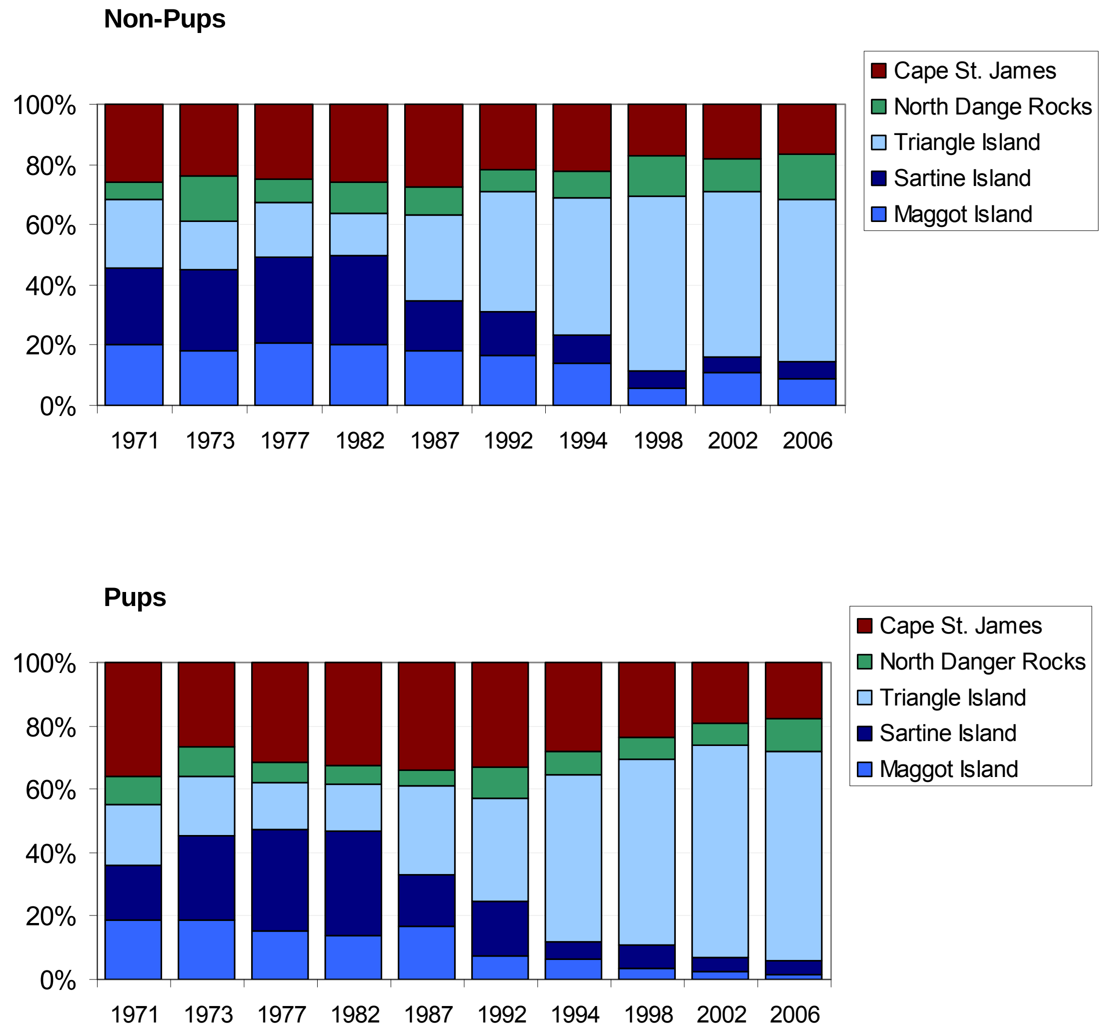

Figure 4. Changes in relative distribution of non-pups (top panel) and pups (bottom panel) among rookeries in B.C. observed during province-wide aerial surveys during 1971-2006. Rookeries at Cape St. James are shown in red, rookeries on North Danger Rocks in green, and rookeries on the Scott Islands in various shades of blue.

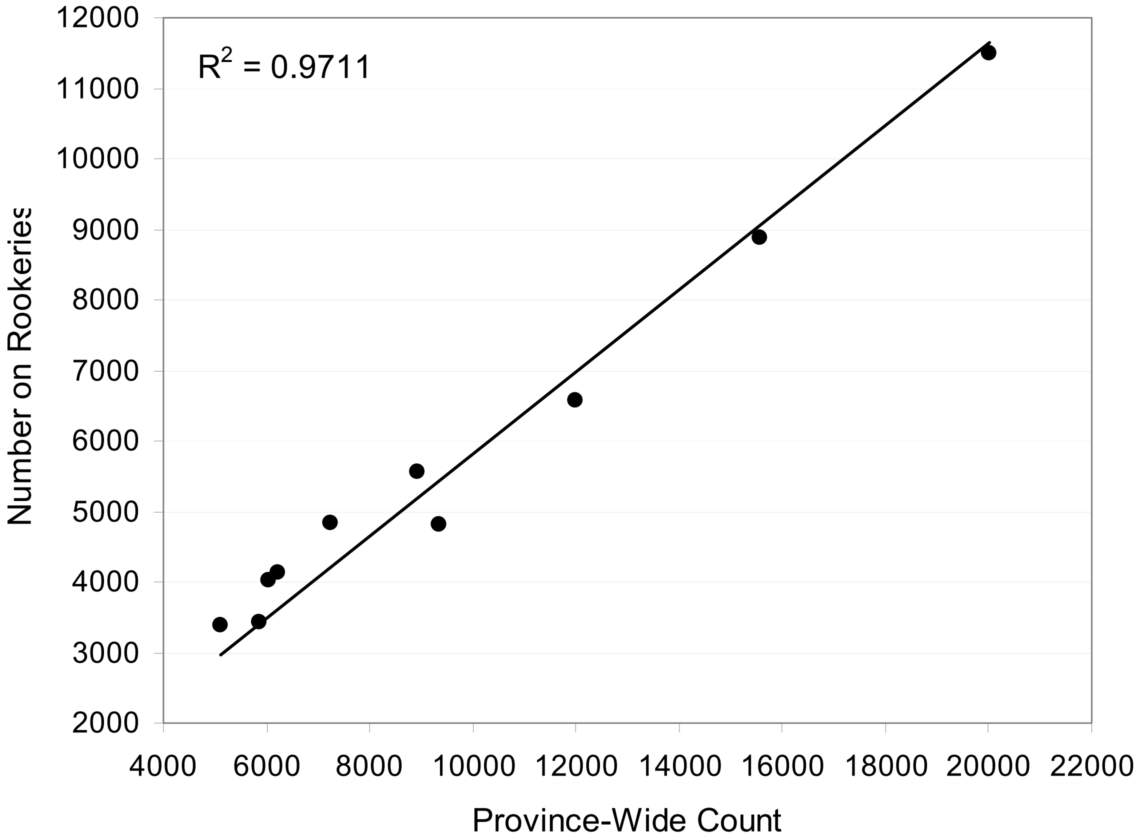

Figure 5. Proportion of the total province-wide count (including pups) that occurred on rookeries during 1971-2006. Solid line represents a least squares regression. The slope was significantly different from zero ($F_{(1,9)}$=578.1; P<0.0001), but the intercept was not quite significantly different than zero ($F_{(1,9)}$=4.3; P=0.0715), so the regression was forced through the origin. The resulting slope indicated that 58% (SE=1.3%) of pups occurred on rookeries.

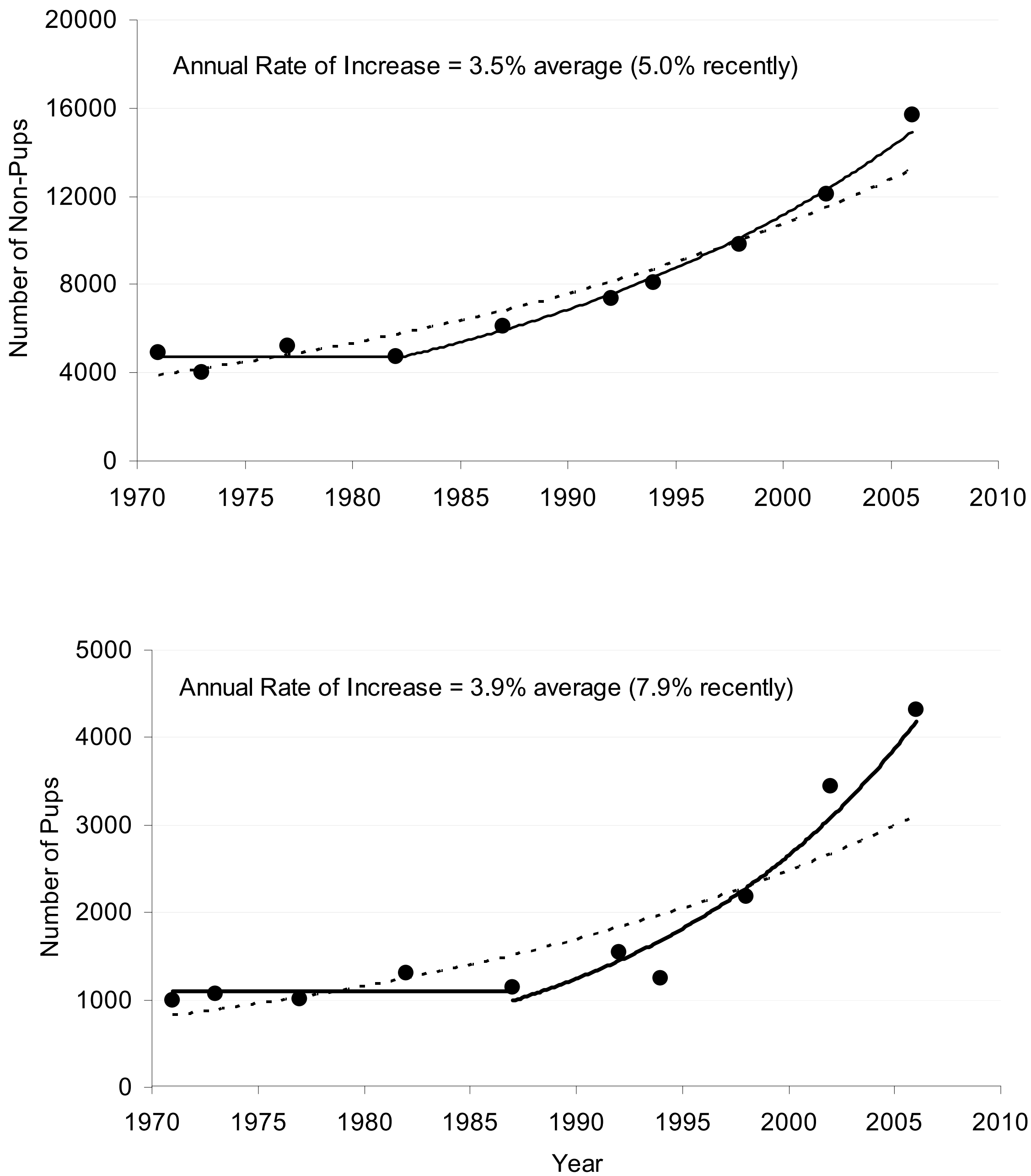

Figure 6. Recent trends in the number of non-pups (top panel) and pups (bottom panel) in B.C. based on province-wide aerial surveys conducted during 1971-2006. Pups counts were obtained from 35mm slides or images and have been multiplied by a factor of 1.05 to account for animals obscured in oblique photographs (Olesiuk et al. 2007). Dashed lines indicate simple log-linear regressions, but in both cases piecewise log-linear regression models that allowed a change in the rate provided a better fit (see text).

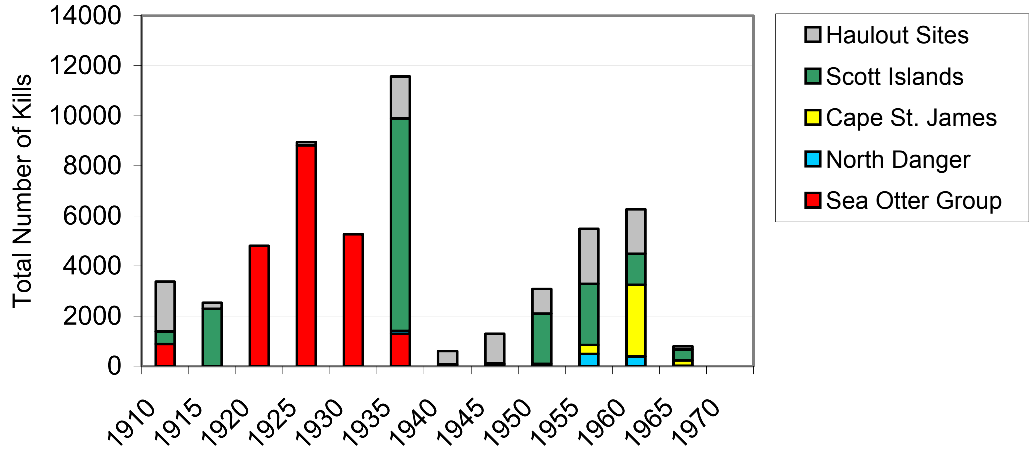

Figure 7. Total numbers of Steller sea lions (pups and non-pups) killed during control programs and harvests in B.C. during 1912-68. Bar heights represent the total number killed during each 5-year period, and colours denote the distribution of kills among major breeding areas and haulout sites. Based on data from Bigg (1984).

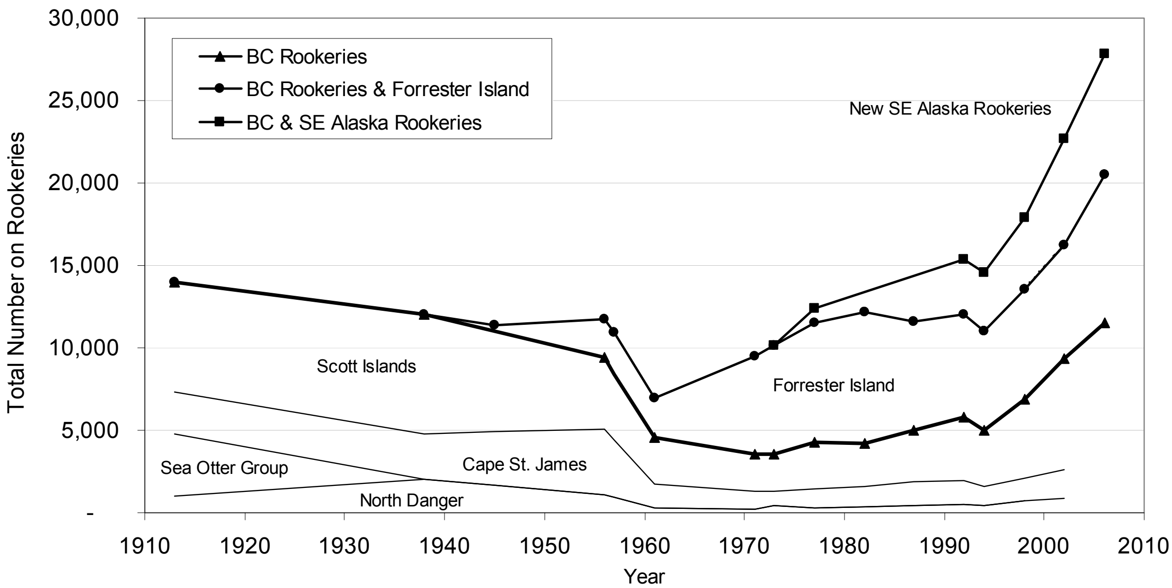

Figure 8. Historic trends in total numbers of Steller sea lions (pups and non-pups) on breeding rookeries in B.C. (lower thick line) and on Forrester Island, Alaska (middle thick line), and total for SE Alaska (upper thick line). The thin lines show distribution among main breeding areas in B.C. Pup counts made from 35mm slides were inflated by a factor of 1.05 for rookeries in B.C., and by a factor of 1.25 for Forrester Island, to account for pups that were obscured in the oblique photographs (see Olesiuk et al. 2007)(modified from Bigg 1985).

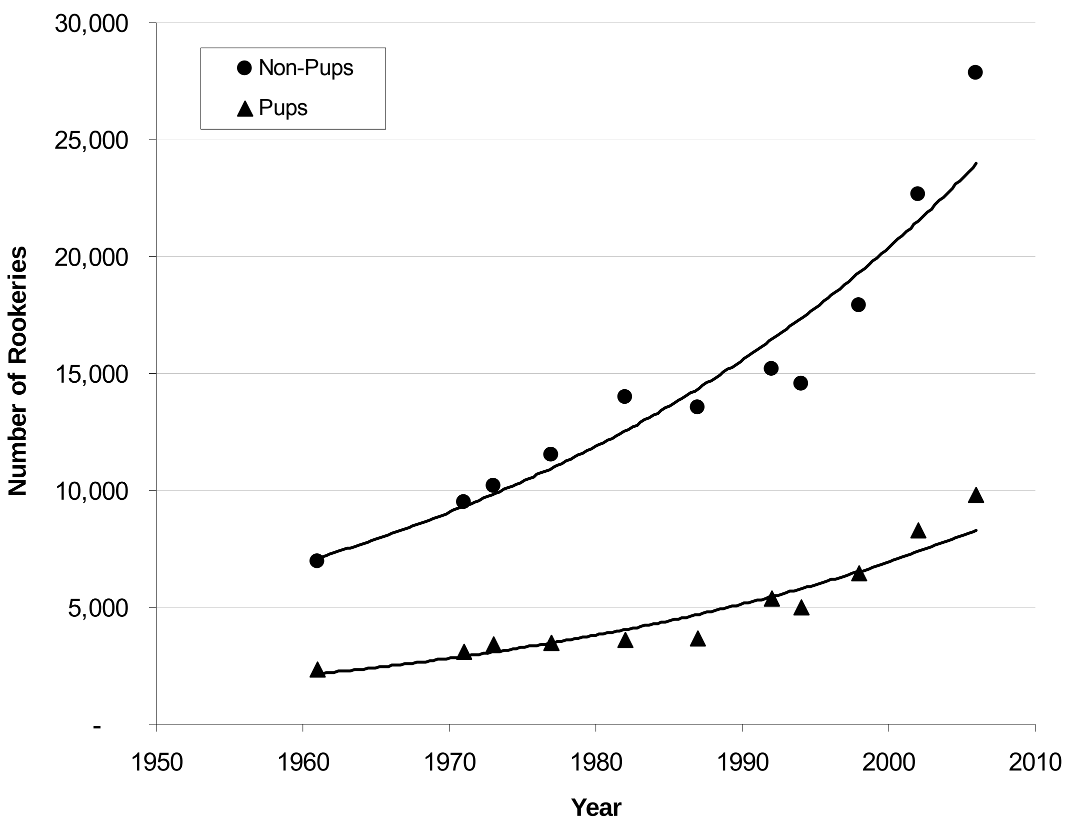

Figure 9. Overall increase in total number of animals (pups and non-pups) and pups on rookeries in B.C. and SE Alaska during 1961-2006. Because surveys were not always conducted in the exact same years in SE Alaska and B.C., estimates coinciding with B.C. survey years were linearly interpolated between the Alaskan surveys. Solid lines represent log-linear regressions, which indicated total numbers increasing at an average rate of 2.7% per annum (r^2=0.994; $F_{(1,10)}$=152.0; P<0.0001) and pup production increasing at an average rate of 3.0% per annum (r^2=0.9204; $F_{(1,10)}$=104.1; P<0.0001), but in both cases the rate of increase appears to have accelerated in recent years.

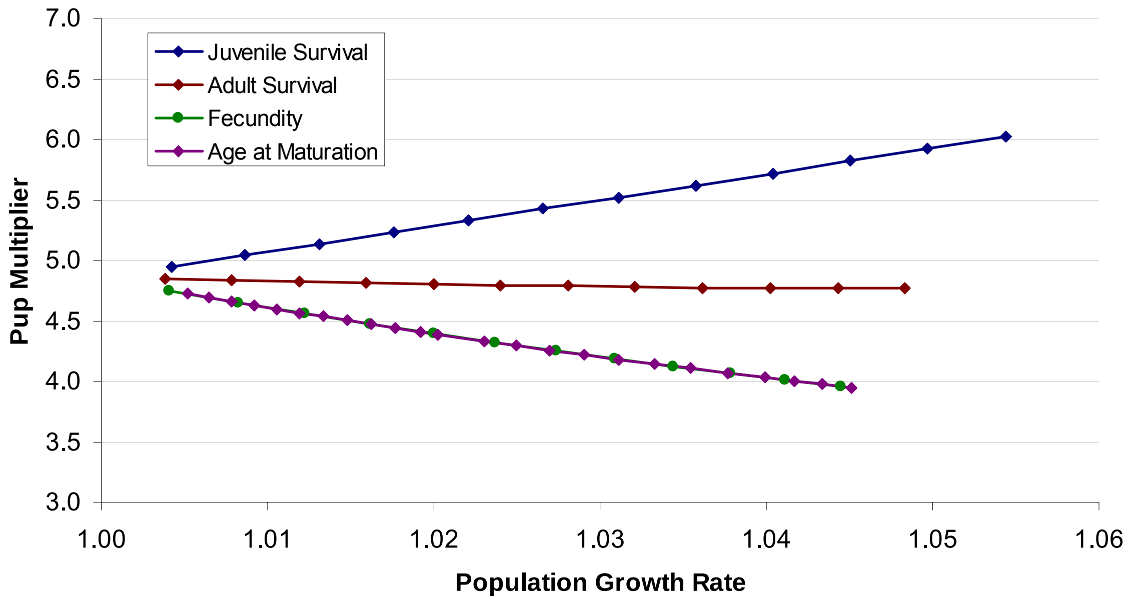

Figure 10. Sensitivity of the pup multiplier (ratio of total abundance to number of pups born) as a function of the population growth rate corresponding to changes in four key vital rates (juvenile survival, adult survival, fecundity and age at maturation). The baseline model for a stable population was based on the life tables given in Calkins and Pitcher (1982). The simulations indicate the pup multiplier for a population increasing at 3.5% per annum could range from 4.0 to 5.8 depending on which vital rate is changed to induce population growth.