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STATUS OF THE CALIFORNIA SEA LION

POPULATION ON THE U.S. WEST COAST

By

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ON THE U. S. WEST COAST

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INTRODUCTION

The California sea lion (<u>Zalophus californianus</u>) is one of several species of pinnipeds which inhabit the Pacific coast of the United States. Management and conservation of sea lions are subject to the guidelines and regulations of the Marine Mammal Protection Act of 1972 (MMPA) and subsequent amendments to the Act. Management under the MMPA is premised on determinations about whether the population is in a condition that can be considered optimum (for the animals and the ecosystem) and whether a proposed level of take is detrimental. Consequently, the status of the population must be assessed periodically. This document is a synthesis of the currently available information on the status of the California sea lion population of the U. S. west coast.

History of Exploitation and Management

Information about the exploitation (Cass 1985, Helling 1984, Stewart <u>et al.</u> in press) and historical distribution (Seagars <u>et al.</u> 1985, Stewart <u>et al.</u> in press) of sea lions has been compiled from the literature or through personal communications. The data are not sufficient to allow estimation of historical or pre-exploitation population sizes. Only a brief summary of the historical exploitation and management of sea lions is presented here.

Extensive commercial harvests of sea lions took place in the middle 1800's (Scammon 1874), probably causing a large decline in numbers (Bonnot 1928a). From that time, until 1972, sea lions were subject to varying levels of harvest for purposes which included collection of "trimmings" (Bonnot 1928a), scientific and display collections (Bonnot 1928b, Howorth, in press), use in pet food (Abbott 1939, Cass 1985), and reduction of fishery depredation (e.g., Jones 1981). Most of these uses, except perhaps the latter, were not responsible for substantial kills of sea lions after about 1940 (Cass 1985). Most "sea lions" which were killed for bounty in Oregon during the 1920's (Scheffer 1928) probably were Steller sea lions (<u>Eumetopias jubatus</u>), but <u>Z. californianus</u> may have been included.

California sea lions were afforded legal protection several times prior to the MMPA (Bonnot 1928a, Jones 1981). However, several accounts (Bonnot 1928a, Cass 1985, Helling 1984) indicate that this protection was effective only in preventing sanctioned kills on some state and federal lands. Private hunters continued to kill sea lions for hides, trimmings, pet food, and to reduce fishery depredation, largely unchecked by management agencies. When the U. S. Marine Mammal Protection Act was passed in 1972, the population was probably at a level which was reduced from the pre-exploitation level, but the relative extent of the reduction is unknown (Stewart <u>et al.</u> (in press) discuss possible prehistoric exploitation).

Basis for Management (the Marine Mammal Protection Act)

The MMPA recognizes marine mammals as components of the marine ecosystem. Noting also that some species or stocks of marine mammals have already been endangered or exterminated by man's activities, the MMPA requires maintenance of stocks above levels at which they would lose their function in the ecosystem. Although ecosystem function forms the basis of the primary objective of the MMPA, marine mammal management in practice is directed toward maintaining optimum sustainable population sizes, a second major objective of the Act.

The working definition of "optimum sustainable population" (OSP) is a range of population size between the environmental equilibrium (carrying capacity, or K) and the level from which maximum productivity would result. The lower limit of the range of OSP is the maximum net productivity level (MNPL). Implicit in the notion of OSP is the recognition of the instability of population levels below MNPL; that stocks which are reduced to levels below MNPL may decline precipitously, even when subject to apparently moderate perturbations in the environment or the harvest (Beddington and May 1977, Clark 1976). On the other hand, populations at levels above MNPL should be more resilient to variable environments and harvests. Population dynamics theory predicts that populations above MNPL will tend to equilibrate with a harvest, provided that the harvest is not at a rate greater than the maximum at which the population is capable of growing (Beddington and May 1977).

The net production of the population at the MNPL is the theoretical upper limit to a rate of harvest or incidental take which could be consistent with maintenance of a population within the range of OSP. A level of take, spread proportionally over all age and sex classes, which would cause a population to equilibrate between MNPL and the carrying capacity (a "replacement yield" take), is necessarily lower than the rate of maximum net production (MNP).

The present assessment of the California sea lion population uses abundance data and information about incidental mortality, to estimate the current size of the population, to estimate the current replacement yield, and to estimate the status of the population relative to the OSP.

METHODS OF STOCK ASSESSMENT

OSP Determination

Two basic approaches have been used for determining the status of marine mammal populations relative to the OSP. One approach requires direct estimation of the the population sizes which define the range that is considered to be the OSP (i.e.,

determining the values of MNPL and K). This method has been used for dolphins in the eastern tropical Pacific tuna fishery (Smith 1983) and for harbor porpoise on the U. S. west coast (Barlow 1987). In both of those assessments, the method used to estimate K was a back-projection from current abundance levels to the pre-exploitation level (K), using estimates of the history of the take (Smith and Polachek 1979). Using this method, the OSP determination is relative to the maximum population level that the environment could support at the beginning of the exploitation period (K is an estimate of "historic K"). This method is not applicable to California sea lions because the pre-exploitation abundance is unknown and cannot be estimated due to the lack of information about the magnitudes of the historical harvest.

The other approach to stock assessment depends on inference from some index (e.g., growth rates or other vital rates, physical growth rate or condition, parameters such as age at sexual maturity or age at first reproduction) which varies in a predictable fashion with population status (Eberhardt and Siniff 1977, Fowler 1987). Then the value of the index is used to determine qualitatively whether the population is depleted or in the range of OSP, without actually determining the population sizes which define the range. This approach makes an OSP determination that is relative to the current condition of the environment. Dynamic response analysis (Goodman 1987), which was previously applied to California sea lions (DeMaster et al. 1982) and to elephant seals (Boveng et al. 1987), is a form of this method which uses density dependent changes in the population growth rate (indexed by changes in pup counts) to infer the qualitative status relative to MNPL. The dynamic response analysis which I use in the present assessment extends the method applied in DeMaster et al. (1982) by incorporating a new technique for choosing the number of censuses to use (Boveng et al. 1987) and by including pup counts from colonies other than San Miguel Island.

Estimation of Rates of Increase and Replacement Yields

Rate of change in population size can be estimated directly from trajectories of a population index, or census, through time; or it can be estimated indirectly from estimates of survival and reproduction rates. The former method is an estimate of the actual population growth rate which occurred on the chosen time interval. In the absence of information about actual trends in population size, the latter method can be useful for determining the growth rates which would be expected on the basis of observed or assumed parameters of the species' life history. These values may then be compared to estimated or proposed rates of take to predict the degree of impact on the stock. In the present case, trends in pup counts are assumed to be at least as reliable as sea lion life history data. Therefore, the sections entitled

POPULATION GROWTH RATES and REPLACEMENT YIELD are based on analysis of trends in pup counts.

STOCK IDENTITY

The California sea lion, _Zalophus californianus_, comprises three subspecies: _Z. c. japonicus_ which bred on the coast of Japan and is probably extinct; _Z. c. wollebaeki_ which breeds on the Galapagos Islands; and _Z. c. californianus_ which breeds on islands of Mexico and California. This assessment is concerned only with the latter subspecies, hereafter referred to simply as the California sea lion.

The MMPA defines a "population stock" or "stock" as a group of the same species or smaller taxon in a common spatial arrangement, that interbreed when mature. Thus, the distinction between a population and a stock is not clear in the MMPA. In this assessment, I use the term "population" loosely, with no particular implication of demographic discreteness. I use the term "stock" more specifically, as defined in the Act. Evidence for identification of a California sea lion stock is presented below.

The breeding range of California sea lions extends south at least to Isla Margarita on the Pacific coast of Baja California and includes locations throughout the Gulf of California (Mate 1977, Aurioles et al. 1983, Le Boeuf et al. 1983). Births occur at only a few isolated locations north of San Miguel Island, California (Braham 1974, Pierotti et al. 1977, Keith et al. 1984, Huber et al. 1986). The range during the non-breeding season (particularly for males) extends north, at least to Vancouver Island, Canada (Hancock 1970, Mate 1975, Bigg 1985).

Because nearly all breeding by this species in U. S. waters occurs on four islands (San Miguel, San Nicolas, San Clemente and Santa Barbara), which are in relatively close proximity, the likelihood of genetic or demographic isolation of breeding colonies may be small. However, despite the application of thousands of tags to sea lion pups (e.g., Heath and Francis 1983, Heath and Francis 1984, Stewart and Yochem 1986, Oliver and Lowry 1987, Lowry et al. 1987), no detailed analyses of resights and movements have been reported. Some of the breeding sites used currently may have been exploited to the point of desertion (Scammon 1874, Bonnot 1928a, Bonnot 1928b, Bonnot 1931) and presumably have been re-established from nearby islands, but the evidence is largely anecdotal (Seagars et al. 1985, Stewart et al. in press). Thus, the extent of interbreeding between island colonies cannot presently be assessed.

Data are also not available for determining the extent to which exchange occurs between U. S. and Mexican sea lion rookeries. Furthermore, data on abundance of sea lions in Mexico

(Orr et al. 1970, Mate 1977, Le Boeuf et al. 1983) are not sufficient to conduct a population assessment for those animals.

Because of the close proximity of the four major U. S. breeding colonies and because no data are available to estimate rates of exchange between colonies, I assume for the purposes of this assessment that the entire U. S. population of California sea lions is a single stock as defined in the MMPA. The California sea lions breeding on Mexican coasts may be part of the same stock but data are not available to allow a reliable assessment of those animals.

ESTIMATION OF CURRENT POPULATION SIZE

There is no time in the life history of California sea lions at which the entire population is represented on shore. The one age class which can be found ashore in nearly its entirety is the cohort of pups which are born between late May and early July (Stewart and Yochem 1986). However, even a count during the peak abundance of live pups does not fully represent the number of pups born, because some pups will not survive from birth to census. I used pup count and mortality data from references to estimate the number of births and then scaled births to an estimate of total population size.

Figure 1 shows counts of California sea lion pups at the four major rookeries in the Channel Islands. The counts shown for San Miguel Island were reported in DeMaster et al. (1982), Delong and Antonelis[1], Stewart et al. (in press), and Gearin and Antonelis[1]. The San Nicolas counts were obtained from Odell (1971), Odell (1972), Bonnell et al. (1980), Heath and Francis (1984), Stewart and Yochem (1984, 1986) and Stewart et al. (in press). The Santa Barbara Island pup counts are from Odell (1971), Bonnell et al. (1980), Heath and Francis (1983) and Lowry et al. (1987). The pup counts at San Clemente Island are from Odell (1971), Bonnell et al. (1980) and Oliver[2].

Figure 1 also shows the total count for years during which counts were obtained at all four rookeries (solid squares). The open squares in the trajectory of total counts represent estimates for years in which a count was made at San Miguel Island but was missing from one or more of the smaller colonies. The estimate for each missing count was obtained by linear interpolation (by

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[1]Unpublished data. National Marine Fisheries Service, National Marine Mammal Laboratory, Northwest and Alaska Fisheries Center, Seattle, Washington.

[2]Unpublished data. National Marine Fisheries Service, Southwest Fisheries Center, La Jolla, California.

island, between adjacent years). The interpolated values were then added to the San Miguel counts to estimate the total counts. The total count for 1972 was not estimated because of the large number of premature births observed at San Miguel (DeMaster et al. 1982) and counts from the other rookeries were not available for that year. One count from all four rookeries, made in 1964 (Odell 1971), is not shown in Figure 1 but its position is indicated by the dotted line connecting it to the first recent census (1971). The 1986 total count of live pups at these four rookeries was 17,852.

Several studies (e.g., Brownell and Le Boeuf 1971, Bonnell et al. 1980, DeMaster et al. 1982, Heath and Francis 1983, 1984, Stewart and Yochem 1986) have estimated mortality of pups during portions, or all, of the pupping and breeding season. At San Miguel Island, between 1975 and 1981, dead pups counted during the census averaged 2.2 percent of the total of live and dead pups (DeMaster et al. 1982). During the same period, dead pups plus known premature births averaged 5.8 percent of the total of dead pups, premature births and live pups (DeMaster et al. 1982). Those values, based on single counts of dead pups at the times of the live pup counts, are known to be underestimates because carcasses are lost to decomposition, scavengers, tides, and drifting sand (DeMaster et al. 1982, Heath and Francis 1983, 1984, Stewart and Yochem 1984, 1986).

Stewart and Yochem (1986) made daily counts of dead pups at a site on San Nicolas Island in 1984 and found that 10 percent of the pups born died before 7 July. Heath and Francis (1983, 1984) used daily counts at two sites (and biweekly counts at remaining sites) on San Nicolas Island to estimate that about 11 percent of all pups born died during the breeding season (approximately 17 May-26 July) in 1982 and 18 percent died in 1983. These estimates, based on daily counts of dead pups, reduce the error from carcass loss, but are limited to a few study sites because of the disturbance which would be caused by daily surveys of large rookeries.

The studies cited above indicate that pup mortality prior to the census (at peak live pup abundance) varies between sites and between years. It seems unlikely that mortality prior to the peak live pup count is less than about 5 percent of total births and it may be 20 percent or more. I have assumed that 15 percent is typical of the population, but I present a range of population size estimates based on the entire range of 5 to 20 percent pre-census mortality. The estimated number of births corresponding to 15 percent mortality prior to the 1986 count of 17,852 live pups is 21,000. The range of 5 to 20 percent mortality would correspond to a range of 18,800 to 22,300 births.

I estimated total population size from the estimated number of births by computing the predicted proportion of newborn pups in a population with a stable age structure, a hypothetical schedule of survivorship, and an annual factor of increase of 1.0635 (see POPULATION GROWTH RATES). The assumed survivorship schedules for male and female sea lions (Table 1) were obtained by scaling[3] the relatively well-characterized survivorship schedule (Smith and Polacheck 1981) of the female northern fur seal (<u>Callorhinus ursinus</u>), by hypothetical values of longevity for California sea lions. Longevity, defined here as the 99th percentile of the age distribution[3], was assumed to be 22 years for females and 17 years for males. The predicted age structures are shown in Table 1. The proportion of newborn (c_1 in Table 1) is .261 for males and .223 for females. Assuming a sex ratio of 1:1 at birth, a population with that age structure and 21,000 births would contain 87,000 total individuals.

To investigate the sensitivity of the population estimate to the assumptions about pre-census mortality and male and female longevity, I generated population estimates based on ranges of values for those three parameters (Fig. 2). Extreme combinations of the parameters (e.g., 5 percent mortality, 15 years longevity for both males and females) yield a range of population estimates from about 67,000 to about 107,000 individuals. Extreme values for all three parameters seem unlikely to occur, so the probable range for population size may actually be narrower. The parameter combination and resulting population estimate (87,000) which I have assumed to be most likely is also shown in Figure 2(c).

POPULATION GROWTH RATES

Because more counts were available for total pups on San Miguel and San Nicolas Islands than for the total of all four rookeries, and because San Clemente and Santa Barbara Islands contribute little to the overall abundance (Fig. 1), only San Miguel and San Nicolas were used for analysis of growth rates. Figure 3 shows the natural logarithm of combined pup counts from San Miguel and San Nicolas Islands, versus year. The average rate of growth in those counts was 3.4 percent annually between 1971 and 1986 (slope=.033, s.e.=.0098, P<.01, R^2=.56). However, Figure 3 suggests that the series of pup counts between 1983 and 1986 (post El Niño) has a different slope than the counts prior to 1983. The estimated growth rate between 1971 and 1982 is 6.4 percent annually (slope=.0616, s.e.=.0062, P<.001, R^2=.95). The growth rate estimated from the (small sample of) years 1983 through 1986 is 11.9 percent annually (slope=.112, s.e.=.017,

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[3]Barlow, J. and P. Boveng. Modeling mortality for marine mammal populations. Unpublished manuscript. National Marine Fisheries Service, Southwest Fisheries Center, La Jolla, California.

$P<.025$, $R^2=.95$). The slopes of the latter two regressions are significantly different ($P<.05$).

The statistics given above indicate that the years since El Niño should not be included in an estimate of the current growth rate. The effects of El Niño on sea lion survival and reproduction may have been such that births increased at a much higher rate than total population size after the perturbation. In 1986, the pup count returned to nearly the 1982 level (Fig. 1). If the proportion of pups in the population has also returned to the 1982 level, the current growth rate would probably not have undergone much density dependent change from the growth rate in 1982. I have assumed that the growth rate estimated from the years prior to 1983 (6.4%) is closer to the current growth rate than that estimated from the entire trajectory or from only the 4 most recent counts. However, to the extent that pup counts are not representative of total population size, the relationship between current growth rate and that estimated prior to 1983 cannot be determined.

Berkson and DeMaster (1985) explored the theoretical reliability of pup counts for estimating discrete rates of change in population size. They found that the rate of change in pup counts may be a biased estimator of total population change and that the direction of the bias depends on the relative timing of density dependent pup survival and the census. Furthermore, they suggested that post-census pup survival may be density dependent in California sea lions and if so, that the rate of change in pup counts would be an overestimate of the total population rate of change. Therefore, 6.4 percent may be an overestimate of the total population growth rate, but the magnitude of the bias is unknown.

INCIDENTAL TAKE

California sea lions have been subject to taking for a variety of purposes in this century (see INTRODUCTION, History of Exploitation and Management). There is, however, little quantitative information available to assess the impact of any or all of those types of takes. Since 1972 and the passage of the MMPA, California sea lion take in U. S. waters has presumably been limited mostly to incidental take in commercial fisheries and an unknown level of killing to reduce gear loss and fishery depredation. The remainder of this section will focus primarily on recent levels of incidental take in commercial fisheries.

Quantitative studies of sea lion mortality in commercial fishing operations were not undertaken until 1979 (Miller et al. 1983). In that study of marine fisheries in California, an estimated 1494 (no standard error given) California sea lions were taken in 1980. Of those, nearly 1200 were estimated taken in ocean gill netting operations, with relatively small numbers

killed in other types of fisheries (e.g., trolling, purse seining and trawl fisheries). Furthermore, the drift gillnet fishery for sharks (and later for swordfish) was identified as the fishery in which most of the sea lion mortality occurred. Concern over the levels of sea lion take in drift gill nets reported by Miller et al. (1983) prompted additional studies of that fishery (Diamond et al. 1986a, 1986b). Those studies, which utilized a voluntary observer program, resulted in estimated sea lion mortality of about 900 (standard error of 380 or 469, depending on computational method used) for the 1983-84 fishing season. In the following year, about 225 (standard error of about 225) sea lions were estimated killed in the drift gillnet fishery. The large standard errors associated with these estimates reflect the difficulties of observing sufficiently large samples of the total fishing effort.

Provisional estimates (subject to change when more detailed stratification methods are used), based on a bootstrap resampling of the observed distribution of kills per net-pull (but not stratified by depths, season, or soak times), suggest that about 3000 (s.e.=774) sea lions were killed in California gill nets during the 1983-84 fishing year[4,5]. The portion of that estimate which represents the drift gillnet fishery was about 1000 animals, comparable to the estimate of 900 (Diamond et al. 1986a) obtained by more detailed stratification. The estimated mortality in all California gill net fisheries during the 1985-86 fishing season[4,5] was 2229 (s.e.=335)

An unknown number of California sea lions are killed by fishermen in Washington, Oregon, and British Columbia to protect gear and catches. Due to the age and sex composition of the sea lions which migrate north from the Channel Islands, these kills would be predominately adult and subadult males (Mate 1975, Bigg 1985).

OSP DETERMINATION

Dynamic Response Analysis

DeMaster et al. (1982) applied the dynamic response method to the trajectories of pup counts obtained between 1971 and 1981 at San Miguel Island. They found that the curve formed by fitting a second degree polynomial to the pup counts was concave downward,

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[4]Personal communication. Doyle Hanan, California Department of Fish and Game.

[5]Personal communication. Sandra Diamond, California Department of Fish and Game.

indicating a population above its MNPL. However, in that early version of the dynamic response method, there was no provision for considering the statistical significance of the fitted parameters and no method for determining the optimum number of counts to include in the analysis. Boveng et al. (1987), in an application to northern elephant seals, presented a method which addresses both of those problems. That method is described briefly here before applying it to the total sea lion pup counts from Figure 1.

When the mechanism of density dependence, the accuracy of the censuses and the amount of environmental variation are unknown (see Gerrodette 1987), a decision rule is needed to arrive at a compromise between using many censuses (the assessment will not be current) and using only a few recent censuses (the assessment may be influenced by noise in the data). In the technique presented in Boveng et al. (1987), curvature produced by noise in the trajectory is "filtered" out by increasing the number of censuses until the results are consistent with the expected results from a trajectory which has curvature due only to density dependence. This is accomplished by fitting second degree polynomials to intervals of the pup count trajectory. For a given interval size (number of censuses), repeatedly shifting the location of the interval by one pup count and computing a new regression produces a series of second order regression coefficients. The shape of the curve generated by plotting each coefficient against the mean date of the cenuses in the interval is examined for inconsistency with the shape predicted from density dependence considerations. The predicted shape is shown in Figure 4, which was generated by applying the method to the well known logistic growth curve. Census intervals which are too short will reveal curvature due to noise rather than density dependent changes in growth rate. The resulting plots of second order coefficients will cross the zero line more than once, or in the wrong direction.

The form of the dynamic response method applied here, namely that of analyzing the shape of the trajectory of pup counts, requires the assumption that fluctuations in the environment during the time interval of interest are limited to random, relatively small scale deviations from a constant mean (Goodman 1987). This is because large rare environmental perturbations will produce a trajectory of abundance with serial correlation in the deviations from the expected trajectory. Figure 1 reveals that this assumption cannot be met for intervals of censuses which span the decline in pup counts between 1982 and 1983. Therefore, I applied the dynamic response technique only to counts obtained in 1982 and earlier. The extension from status prior to 1982 to current (1986) status is made after presentation of the dynamic response results.

Another assumption underlying a dynamic response analysis of the shape of a growth curve is that there is no significant incidental mortality. The section entitled INCIDENTAL TAKE

described estimates of up to 3000 sea lions killed per year. Currently there are not enough reliable mortality estimates available to adjust estimates of per-capita production as suggested by Goodman (1987). However, unless incidental mortality rates decreased during the time interval of interest, the effect of the take would be to make the apparent status higher (relative to the MNPL) than the true status. In fact, evidence from halibut landings (Barlow 1987) suggests that gill net effort and by extension, sea lion mortality, was increasing in the 1970's. Therefore, the dynamic response analysis presented below ignores the incidental take, but a caveat -- that because of the take the true population status would probably be lower relative to the MNPL than the apparent result -- is applied to the conclusion.

Figure 5 shows the results of applying the moving interval method to the curve of total pup counts through 1982. The quadratic regressions computed on intervals of 5 or 6 censuses produce second order coefficients which alternate between positive and negative values, thus failing the requirement that the curve of second order coefficients cross zero only once, from positive to negative. Increasing the number of censuses to 7 results in a curve which might be judged consistent with the left tails of the curves in Figure 4. Note, however, that the approximate 95% confidence limits about the estimated regression coefficients cross the zero line for every interval. Relaxing the confidence limits to 90% would result in a curve not inconsistent with Figure 4. Including 8 cenuses in each regression interval results in a curve with only 3 points, but one which is not inconsistent with the shapes in Figure 4 (beginning at zero and increasing initially to positive values toward the right). No census interval size results in curves with significantly negative coefficients that are consistent in shape with the expected form.

The second order regression coefficient from the regression on the 8 most recent total pup counts (1975-1982) is 156.4 with an estimated standard error of 62.9. This result is taken to indicate a population below its MNPL (Goodman 1987, Gerrodette 1987, Boveng et al. 1987). This status determination should be considered a measure of the "average" status on the census interval rather than the status at the end of the interval (1982). The mean census date for the interval is about mid-1978. The extension to more recent population status is made below, based on the assumption that population status in 1982 was equivalent to the average estimated for the interval 1975-1982 (below MNPL).

Because 1986 pup counts had not fully recovered to 1982 levels, it seems unlikely that total population size in 1986 was larger than population size in 1982. Therefore, 1986 population status, relative to the MNPL, is not likely to have been higher than in 1982, unless El Niño caused a long-term contraction in the carrying capacity and MNPL. Such an environmental contraction

would be expected to cause the population growth rate to slow, imparting downward concavity to the ensuing census trajectory. The opposite appears to be the case (Fig. 1), but the sample size may be too small to draw conclusions about an increasing population growth rate from pup counts since 1983. In any case, several physical and biological parameters returned to approximately normal values within 2 years following the recent El Niño (Fiedler et al. 1986) and previous similar events (Smith 1985), suggesting that such events should be considered temporary perturbations to the carrying capacity (and MNPL) for sea lions.

Because the levels of incidental mortality during the years considered by the dynamic response analysis may not have been negligible, analyzing the shape of the growth curve may not result in the correct estimate of status (Goodman 1987). However, when the apparent status indicated by the dynamic response analysis is below MNPL, the existence of constant or increasing incidental mortality would imply that the actual population status is even lower, relative to the MNPL.

Estimating that the California sea lion population was, on average, below its MNPL between 1975 and 1982 is apparently at odds with the conclusion by DeMaster et al. (1982) that the San Miguel Island colony was above its MNPL (they used censuses between 1971 and 1981). However, the San Miguel colony represents only about 60% of the total pup count (Fig. 1). Furthermore, the preliminary version of dynamic response analysis used in DeMaster et al. (1982) did not incorporate a method for optimizing the census interval size or a significance test on the regression coefficients. Had those techniques been used, analysis of the San Miguel pup trajectory would have suggested a population near its MNPL, but not significantly above or below.

Other Information Bearing on Status

Time series of data on vital rates or physiological indices of condition which might indicate population status (Eberhardt and Siniff 1977, Fowler 1987) are not presently available for comparison with the results from the dynamic response analysis of pup counts. However, two studies related to historic range and exploitation are considered below.

Seagars et al. (1985) found that nearly all rookeries identified historically are currently used for pupping and breeding. Stewart et al. (in press) reviewed archeological studies which showed that frequency of pinniped remains found in Channel Island native American kitchen middens decreased in the more recent layers, leading them to speculate that over-exploitation may have occurred prior to European contact and that California sea lions may currently be more abundant than they ever were historically.

Evidence that the population is currently as abundant as, or perhaps more abundant than, its historic carrying capacity is not necessarily inconsistent with the results from the dynamic response analysis which indicated that the population may be below its current MNPL. The MMPA does not distinguish between current and historic carrying capacities or MNPLs.

REPLACEMENT YIELD

As discussed in the section on POPULATION GROWTH RATES, the rate of increase in counts of pups on U. S. rookeries between 1971 and 1982 was 6.4 percent per year. Because the 1986 pup count indicated near recovery to 1982 levels and because the increase in pup counts since the El Niño in 1983 is probably not a reliable estimate of current population growth rate, 6.4 percent is the best available estimate for the current rate of increase. Under the assumption that the rate of increase in pup counts is representative of the population growth rate, replacement yield is estimated as the product of the growth rate and the stock size. Thus, a current growth rate of 6.4 percent and a stock size of about 87,000 would result in an estimated replacement yield of about 5,600 animals. In other words, the estimate suggests that the current population could sustain a take of about 5,600 animals (but the take would have to be distributed in proportion to the age and sex structure).

It should be noted that because this population may be below its MNPL (see OSP DETERMINATION), the estimated replacement yield is strictly a theoretical construct. The population dynamics theory from which the concept of OSP was derived predicts that although there is an equilibrium yield for a population size below MNPL, the equilibrium is not stable and is likely to be disrupted by perturbations in the environment or changes in incidental mortality rates (Clark 1976, Beddington and May 1977).

Barlow (1987) suggested expressing replacement yields and maximum allowable take estimates in terms of percent of stock size, because numerical quotas can have a destabilizing effect. Expressed on that basis the current replacement yield estimate is simply equal to the estimate of the growth rate, 6.4 percent per year.

SUMMARY: STATUS OF THE STOCK

Although the extent to which interbreeding occurs between California sea lion rookeries could not be ascertained, the sea lions breeding on U. S. rookeries were assumed to compose a single stock for the purposes of this assessment. Data from references indicate that there were about 17,800 California sea lion pups counted on U. S. rookeries in 1986. That number of pups was estimated to represent about 21,000 births. On the basis of a hypothetical survivorship schedule and the observed rate of

increase in pup counts, that number of births would represent a total U. S. population (stock) size of about 87,000 animals. The current growth rate was assumed to be equal to that which was observed just prior to the 1983 El Niño event, 6.4 percent per year.

The trajectory of total pup counts from San Miguel, San Nicolas, San Clemente and Santa Barbara Islands was analyzed using the dynamic response technique. That analysis depends primarily on assumptions that no major deviations or systematic changes occur in the environment (carrying capacity) or in the proportion of pups in the population. Thus, the analysis is restricted to years prior to the occurrence of El Niño in 1983. To the extent that the assumptions hold, the average dynamics of the stock between 1975 and 1982 were like those of a stock below its MNPL (and therefore, below OSP). To infer current (1986) population status requires the additional assumption that average status between 1975 and 1982 was valid through 1982. Then the observation that 1986 pup counts had not entirely recovered to 1982 levels (especially at San Nicolas Island (Stewart et al., in press)), would imply that 1986 population status was not substantially higher, relative to the MNPL, than in 1982. The possibility that incidental mortality in fisheries was increasing during the time interval analyzed by the dynamic response method, implies that the actual stock status on that interval would be lower relative to the MNPL than indicated by the analysis.

Studies by Seagars et al. (1985) and Stewart et al. (in press) indicate that the current abundance of California sea lions may be higher than any historical level. That evidence, considered with the results of the dynamic response analysis, demonstrates the need to distinguish between status relative to current population parameters and parameters which may have been in effect previously. This distinction may become crucial in the process of determining status under the MMPA.

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Table 1. The stable age distribution[a] (C_x) for a population of California sea lions with assumed survivorship schedules (l_x) and an annual factor of increase, $\lambda = 1.0635$.

Age (x)	females l_x	$\lambda^{-x}l_x$	C_x	males l_x	$\lambda^{-x}l_x$	C_x
1	1.000	.940	.223	1.000	.940	.261
2	.702	.621	.148	.643	.568	.158
3	.534	.444	.105	.468	.389	.108
4	.431	.335	.080	.371	.290	.080
5	.365	.268	.064	.314	.231	.064
6	.320	.221	.053	.277	.191	.053
7	.289	.188	.045	.252	.164	.045
8	.266	.162	.039	.234	.143	.040
9	.248	.143	.034	.220	.126	.035
10	.235	.127	.030	.207	.112	.031
11	.223	.113	.027	.195	.099	.028
12	.213	.102	.024	.183	.087	.024
13	.204	.092	.022	.167	.075	.021
14	.195	.082	.020	.148	.063	.017
15	.185	.073	.017	.125	.050	.014
16	.174	.065	.015	.097	.036	.010
17	.161	.056	.013	.067	.024	.007
18	.145	.048	.011	.038	.013	.004
19	.127	.040	.009	.017	.005	.001
20	.107	.031	.007			
21	.084	.023	.005			
22	.060	.016	.004			
23	.038	.009	.002			
24	.021	.005	.001			
TOTALS		4.209	1.000		3.608	1.000

[a]The stable age distribution, $C_x = \dfrac{\lambda^{-x}l_x}{\sum_{y=1}^{\infty} \lambda^{-y}l_y}$
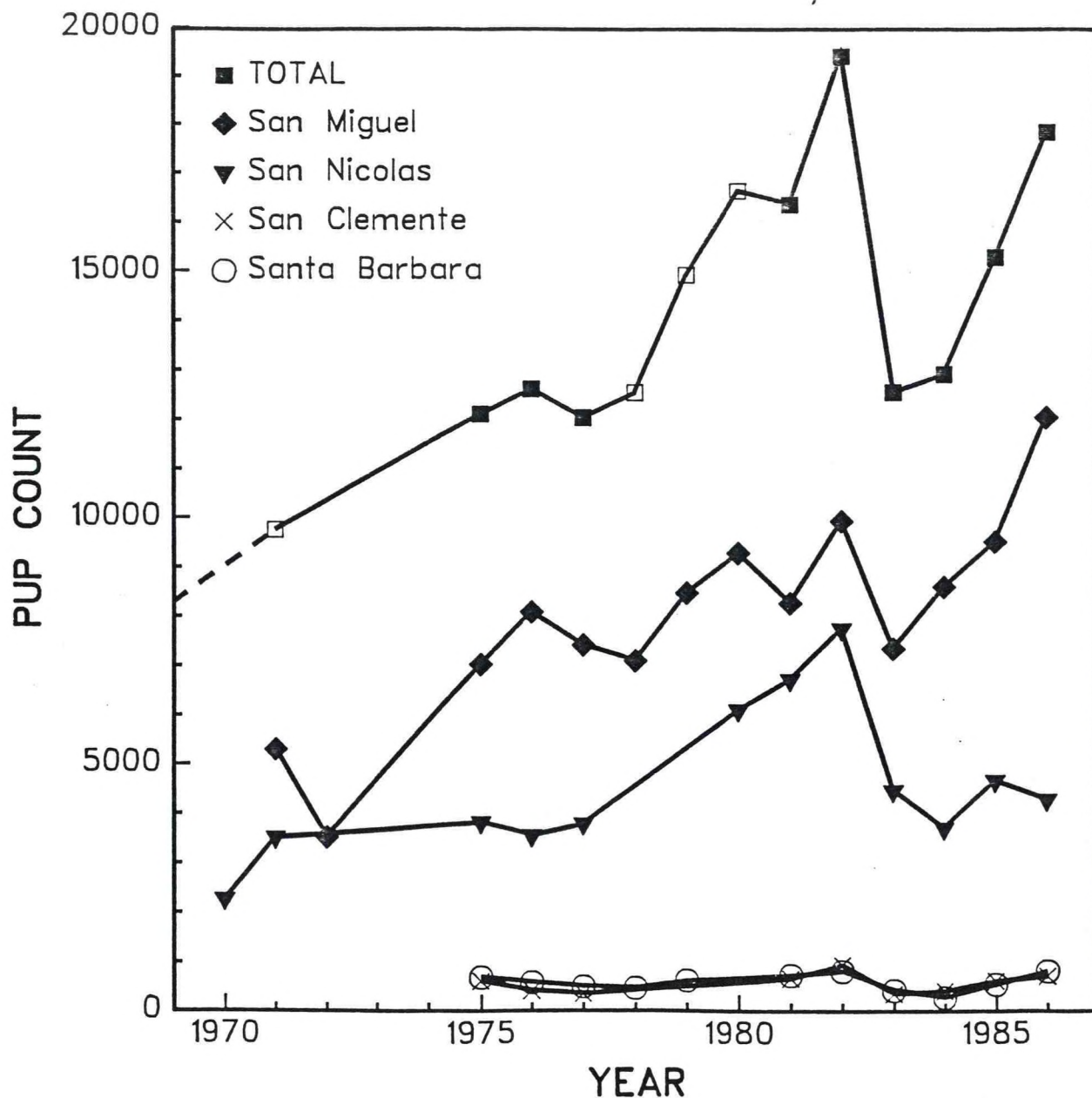
Figure 1. California sea lion pup counts at San Miguel, San Nicolas, San Clemente, and Santa Barbara Islands. The total count is shown for years in which counts were available from all 4 rookeries (solid squares). Estimated totals (open squares) were obtained by adding the San Miguel count to estimates for other islands obtained by interpolation (within island, between years). Sources of the data are cited in the text.
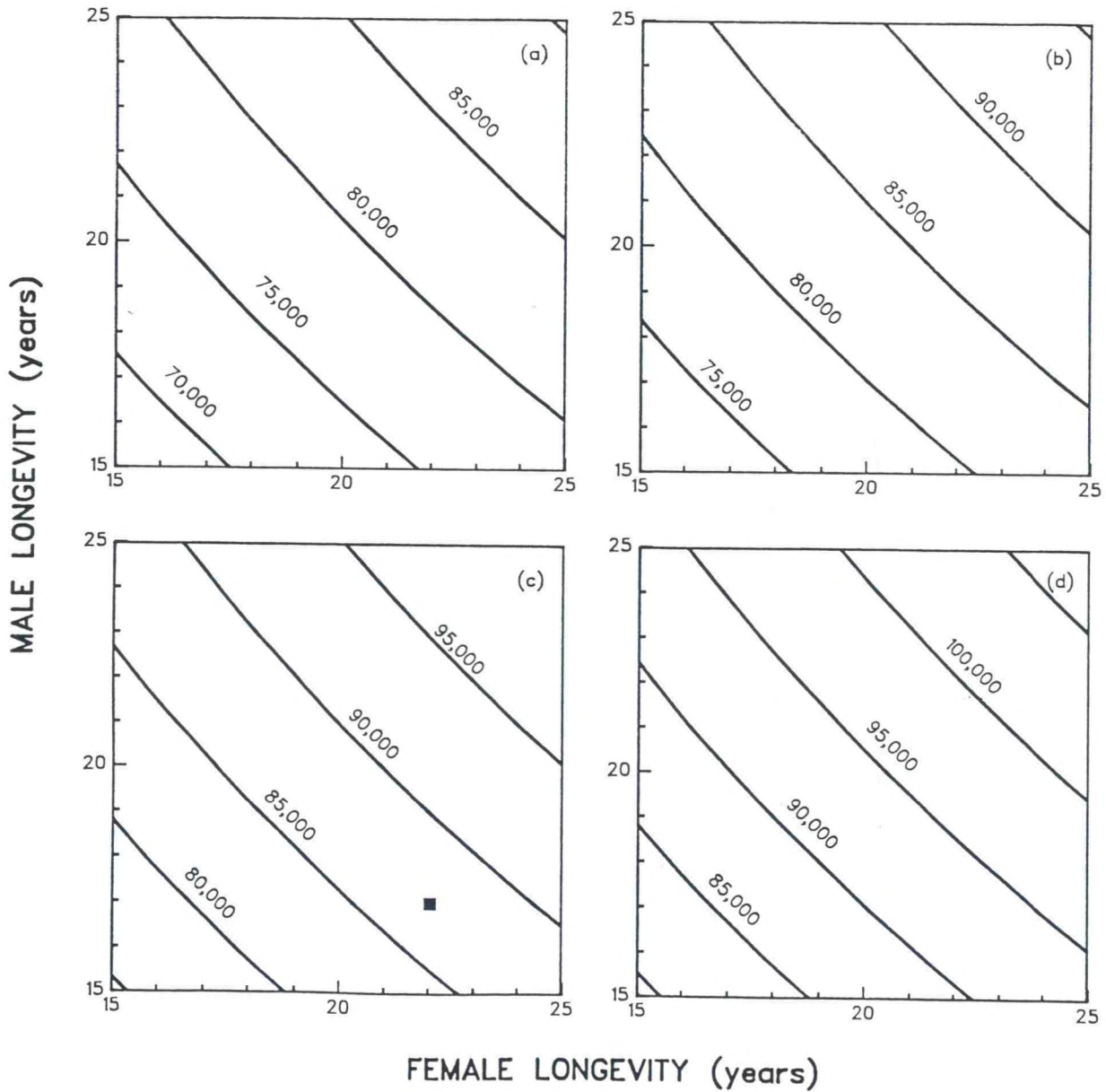
Figure 2. Contours of population sizes resulting from assumptions about male and female longevity (the 99th percentile of the age distribution) for: (a) 5 percent pre-census mortality, (b) 10 percent pre-census mortality, (c) 15 percent pre-census mortality, and (d) 20 percent pre-census mortality. The solid square in (c) represents the estimate assumed to be most likely (87,000).
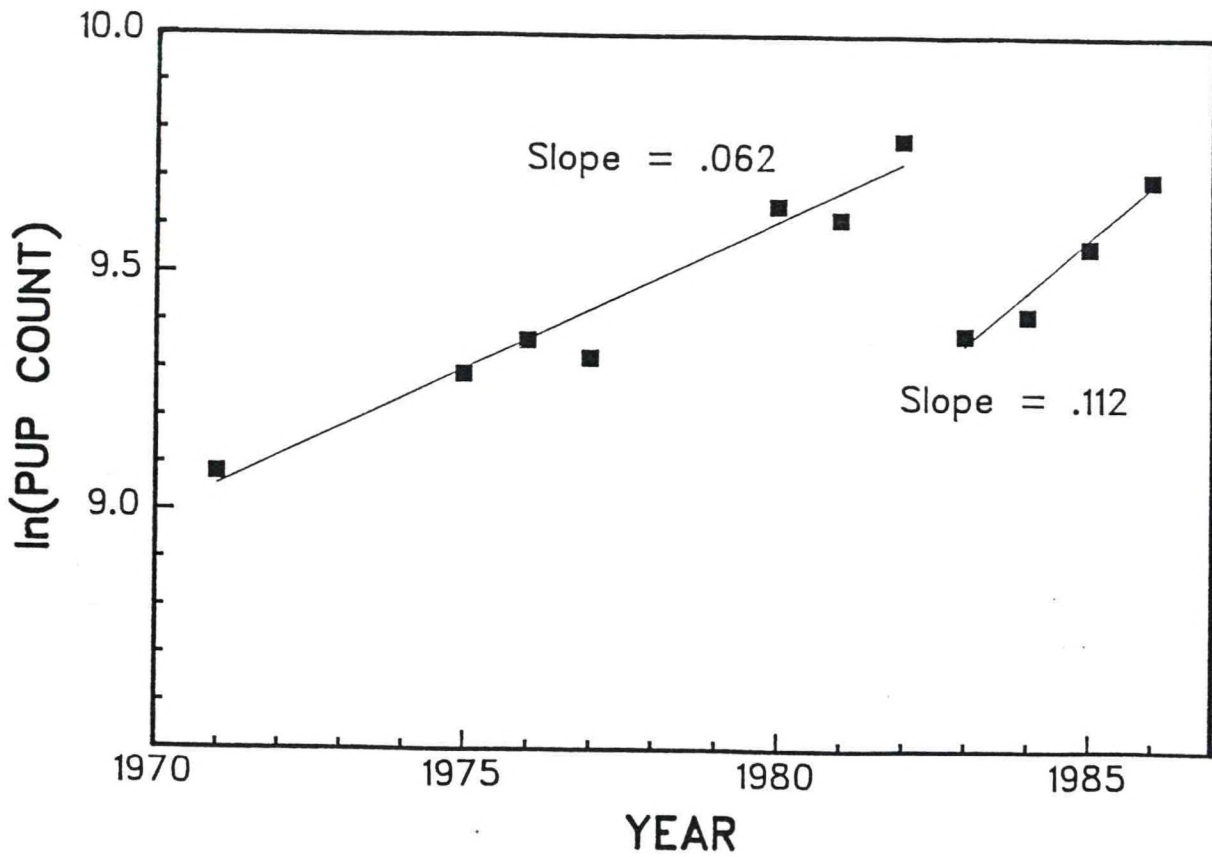
Figure 3. Natural logarithm of total pup counts from San Miguel and San Nicolas Islands plotted against year, showing regression lines and slopes.
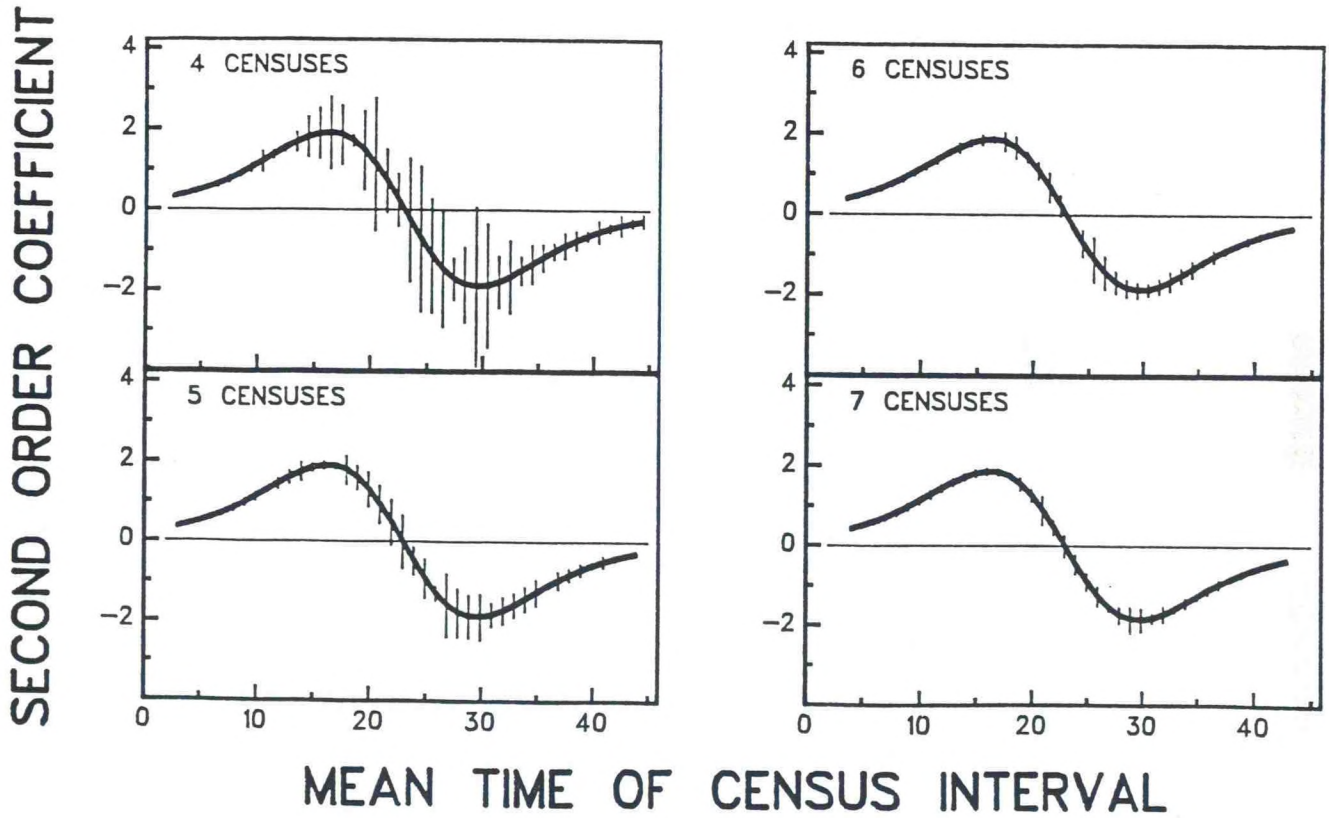
Figure 4.   Second order regression coefficients computed from a hypothetical logistic population trajectory using intervals of from 4 to 7 censuses.   The coefficients were computed by moving the intervals along the trajectory in steps, computing a regression at each step.   Error bars are ± 2 standard errors of the coefficients.
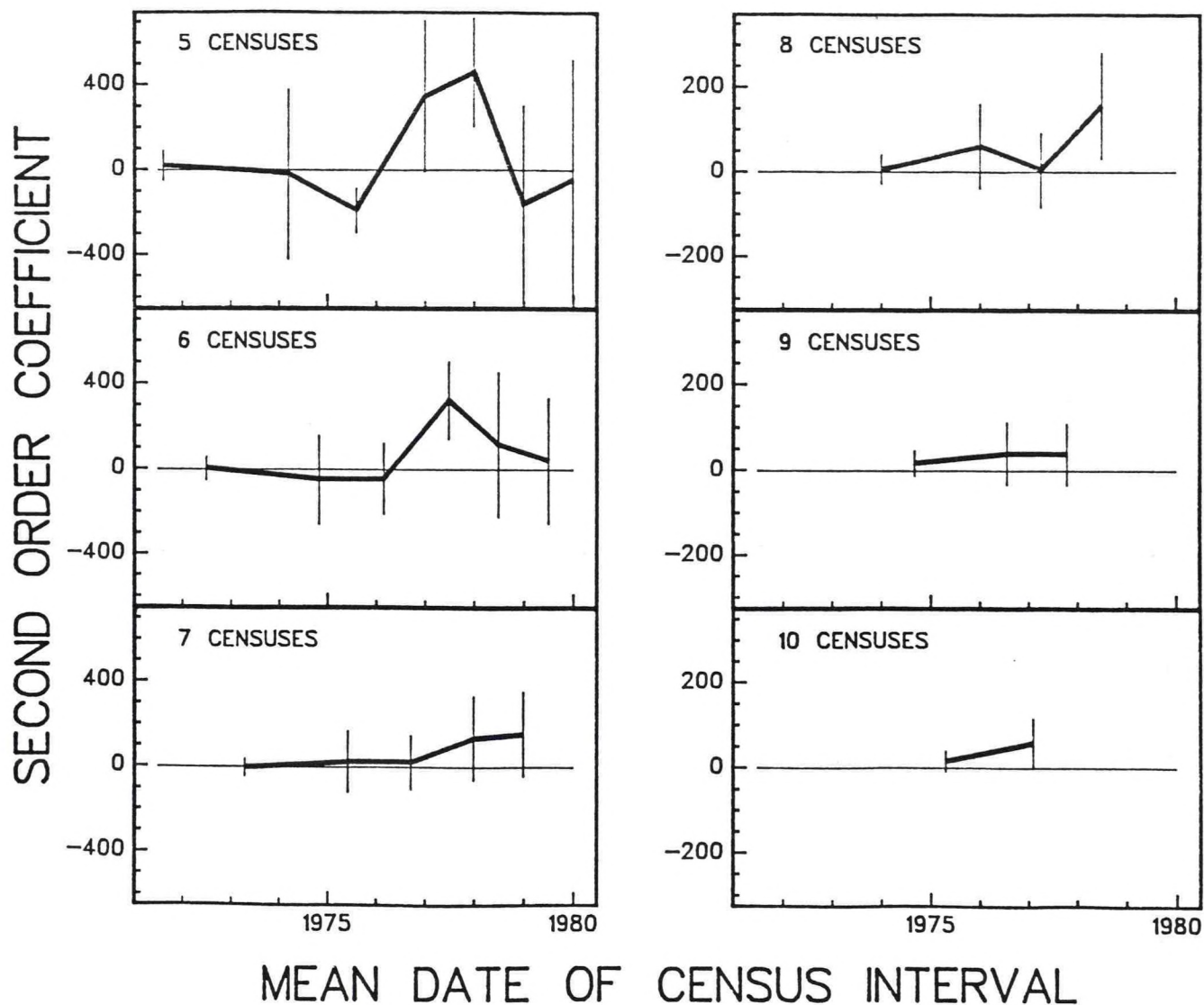
Figure 5. Second order regression coefficients computed from the trajectory of total pup counts in Figure 1. Regressions were computed for intervals ranging in size from 5 to 10 censuses. The intervals were moved along the trajectory in steps, computing a new regression at each step. Error bars are ± 2 standard errors of the coefficients.