

Review of Selected Marine Mammals in the North Pacific 1978-1997

Sarah Courbis May, 1998

Hatfield Marine Science Center, 2030 S. Marine Science Dr., Newport, OR 97365

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PINNIPEDIA (SEALS AND SEA LIONS)

Review of Northern Elephant Seals (Mirounga angustirostris) in the North Pacific 1978-1997

Population, Abundance, and Distribution

Range:

Males:

Gulf of CA to Baja CA north to the eastern Aleutian Islands and Gulf of AK and have been found as far west as Midway Atoll and Niihima Island, 5,000 km west of Midway; haul out irregularly from north of rookeries

Females:

Gulf of CA to Baja CA north to south of 56°N; have been recorded in the Gulf of AK; generally below 45°N-50°N along the coasts of CA, OR, WA, and British Columbia and west to about 150°W; haul out irregularly north of rookeries

Breeding Range:

Historically bred from Point Reyes peninsula, CA to Cabo San Lazaro at the tip of Baja CA; breed from Isla Cedros and Isla Natividad, Baja CA north to the Farallon Islands, CA, including the Channel Islands San Miguel, San Nicolas, Año Nuevo, Santa Barbara, and San Clemente Islands and on mainland rookeries at Point Reyes, Año Nuevo Point, Point Saint George, Cape San Martin, and marginal rookeries at Crescent City, CA and Cape Arago, OR; at least 13 island and 4 mainland sites

Preferred Habitat:

Sandy beaches

Weight

Males: 1,000-2,700 kg

Females: 250-900 kg

Length

Males: 5.0-6 m

Females: 3-3.4 m

For management purposes, elephants are separated into CA breeding and Baja CA breeding stocks (Barlow et al, 1997). However, there is genetic exchange between the stocks, and in fact, all elephants are practically genetically identical due to a genetic bottleneck (Barlow et al, 1997; Le Boeuf, 1977; Stewart et al, 1994). Antonelis et al (1981) state that breeding populations can be divided into three subpopulations by centers of distribution: Baja CA; CA Channel Islands, including Islas Los Coronados, Baja CA; and central CA.

Elephants were hunted to near extinction in the late 1800's (Antonelis et al, 1981; Allen et al, 1989; Barlow et al, 1997; Campbell, 1987; Kajimura and Loughlin, 1988). They were thought to be extinct in 1880 but were rediscovered on Isla de Guadalupe in 1892 (Cooper and Stewart, 1983; Stewart et al, 1994). In the early 1900's, there were less than 100 remaining in

the remnant population occupying Isla de Guadalupe, Mexico (Allen et al, 1989; Barlow et al, 1997; Brueggeman et al, 1990; Cooper and Stewart, 1983; Kajimura and Loughlin, 1988; Le Boeuf, 1977).

After elephants received protection in Mexico in 1922 and shortly after in the U.S., population began to increase (Allen et al, 1989; Campbell, 1987; King et al, 1989; Le Boeuf, 1977; Stewart et al, 1994). As their population increased, beginning in the 1930's, elephants began to re-establish themselves on progressively more northern islands (Brueggeman et al, 1990; Antonelis et al, 1981; Mesnick et al, 1998; Stewart and Yochem, 1986; Bodkin et al, 1985). Most elephants return to their natal rookery (Barlow et al, 1997) and display site fidelity (Campbell, 1987; Stewart and DeLong, 1994), but dispersal of juveniles from crowded rookeries into new areas does occur and initiates colony formation (Brueggeman et al, 1990; Le Boeuf and Panken, 1977; Mesnick, 1998; Reiter et al, 1981; Stewart et al, 1994; Clinton and Le Boeuf, 1993). All rookeries established since 1975 were first utilized by juveniles, and juveniles appeared at these sites 6-16 years before breeding began (Mesnick, 1998).

Elephants bred only at Isla de Guadalupe from the late 1890's to the 1920's (Stewart et al, 1994). They were first seen on Islas San Benito in 1918, on San Miguel Island in 1925, on Los Coronados and Santa Barbara Islands in 1948, on San Nicolas Island in 1949, and on Año Nuevo Island in 1955 (Stewart et al, 1994; Antonelis et al, 1981). Breeding began on Islas San Benito in the 1930's, on San Miguel Island in 1958, on San Nicolas Island in 1959, on Santa Barbara Island in the late 1950's, and on Año Nuevo Island in 1961 (Antonelis et al, 1981; Stewart et al, 1994; Stewart and Yochem, 1986). Pups were first noted on the Farallon Islands in 1972 (Stewart and Yochem, 1986). Since 1960, colonies have been founded on the mainland on Cape San Martin/Gorda, Point Reyes Headlands, and on Point St. George (Stewart et al, 1994; Stewart and Yochem, 1986). The first pup born on the CA mainland in the 1900's was born at Año Nuevo Point, CA in 1975 (Allen et al, 1989; Cooper and Stewart, 1983; Le Boeuf and Panken, 1977; Stewart and Yochem, 1986). It is unknown if elephants historically bred on the mainland (Allen et al, 1989). The first pup born on Point Reyes peninsula in the 1900's was born in 1976 (Allen et al, 1989). A female with a pup was seen on San Clemente Island in 1977 (Antonelis et al, 1981). Isla Natividad and Isla San Martin were recognized as breeding areas in 1977 (Le Boeuf and Mate, 1978; Stewart et al, 1994; Stewart and Yochem, 1986). Elephants also have a clearly established breeding colony on Isla Cedros (Stewart et al, 1994). Two newborns were observed with an adult female on Santa Rosa Island in 1985 (Stewart and Yochem, 1986). The Gulf of CA may soon also become a breeding site for elephants, since they began to appear there in 1979 (Mesnick, 1998). Mesnick et al (1998) reports that elephants breed on 13 island and 4 mainland sites from Isla Cedros, Baja CA to Cape Arago, OR.

In 1960, 91% of the population was on Isla de Guadalupe, 8% was on Islas San Benito, and 1% was on the Channel Islands (Stewart et al, 1994). Population centers have shifted to the Southern CA Bight, where over half of all pups are born (Brueggeman et al, 1990). Of all the pups born in CA in 1982, 60% were on San Miguel Island, 20% were on San Nicolas Island, and 13% were on Año Nuevo Island (Cooper and Stewart, 1983). In 1991, Mexican rookeries contributed 25.5% and CA rookeries contributed 74.8% of all births (Stewart et al, 1994). Nearly half (49.3%) were born on San Miguel Island alone (Stewart et al, 1994). The largest colony in the species range is on San Miguel Island (Stewart et al, 1994).

Range-wide population has increased from the 1920's, when it was 20-100, to 13,000-15,000 in 1960, to 60,000 in 1978, to 114,000 in 1988, to 127,000 in 1991 (Allen et al, 1989;

Antonelis et al, 1981; Antonelis and Fiscus, 1980; Antonelis et al, 1994; Barlow et al, 1997; Brueggeman et al, 1990; Campbell, 1987; Consiglieri et al, 1982; Stewart et al, 1994; Le Boeuf, 1977; Antonelis and Fiscus, 1980). Population doubled every five years for two decades up to 1982 (Cooper and Stewart, 1983). Highest growth rate measured for the range-wide population was 8.3%/yr. from 1965 to 1977 (Barlow et al, 1997).

The CA breeding stock population was 84,000 in 1996 and was increasing through 1994 (Barlow et al, 1997). Population of this stock appears to have leveled off from 1994 to 1996 (Barlow et al, 1997). Growth rates as high as 16%/yr. were documented on U.S. rookeries from 1959 to 1981; however, much growth was created by immigration from Mexican rookeries (Barlow et al, 1997). Minimum population of this stock is 51,625 (Barlow et al, 1997). Population of the Baja CA breeding stock was stable or slowly decreasing through 1994 (Barlow et al, 1997).

Elephants make two migrations from their rookeries each year (Barlow et al, 1997; Braham, 1992; Condit and Le Boeuf, 1984; Le boeuf, 1994; Stewart and DeLong, 1993). They travel to pelagic feeding areas between the spring/summer molting and winter breeding seasons (Barlow et al, 1997). Females molt in May, and males molt in August (Stewart and DeLong, 1993). Adult females and males spend 84% and 67% of the year at sea respectively (Le Boeuf et al, 1986). Females spend about 10 months of every year at sea (Le Boeuf et al, 1988; 1989).

Tagging and attaching time-depth recorders to elephants has revealed much about their extensive pelagic migrations (Condit and Le Boeuf, 1984; Le Boeuf, 1994; Le Boeuf et al, 1986; 1988; 1993 Stewart, 1992; Stewart and DeLong, 1990). Most elephants migrate north, but some migrate south (Condit and Le Boeuf, 1984; Mesnick et al, 1998). Condit and Le Boeuf (1984) found that elephants born in central CA were commonly seen as far north as British Columbia and concentrated in two areas, northern CA and around the southern end of Vancouver Island. They found that elephants born in southern CA were commonly seen in central CA and scattered much farther north. They also found that elephant born in Mexico congregated in southern CA. Overall, juveniles born on northern rookeries were seen farther north than those born on southern rookeries (Condit and Le Boeuf, 1984). Condit and Le Boeuf (1984) also noted that juveniles segregate by birth site on feeding grounds. However, Le Boeuf (1994) and Stewart and DeLong (1994) state that adults from different rookeries migrate to the same feeding areas. Feeding areas are generally far offshore, primarily off the continental shelf (Le Boeuf et al, 1988).

Adult males migrate north in the spring and fall, hauling out in July and August to molt and from December to March to breed (Condit and Le Boeuf, 1984). Adult females are at sea ten weeks during the spring and again for about seven months in the summer and fall, hauling out in April and May to molt and January and February to breed (Condit and Le Boeuf, 1984). Pregnant females migrate farther from the rookery than non-pregnant females (Le Boeuf, 1994), and males range farther north and west than females (Brueggeman et al, 1990; Stewart and DeLong, 1990; Stewart and DeLong, 1993; 1995).

Le Boeuf (1994) found that the migratory paths of males and females overlapped in the eastern Pacific, especially along the coast up to about 58°N. He found that the most northerly and westerly migrations were made by males. One male traveled as far as the eastern Aleutian Islands (7,500 km round-trip) and some traveled no farther north than about 49°N near WA (Le Boeuf, 1994). Post-breeding females moved from near the coastline to as far west as 150°W (Le Boeuf, 1994). The longest round-trip migrations of post-breeding females were 6,800 km (Le Boeuf, 1994). A juvenile female traveled to the Gulf of AK, exceeding the distances traveled by

most post-breeding adult females (Le Boeuf, 1994). Le Boeuf et al (1994) found that the average distance traveled by juveniles was 2,239 km, by adult females, was 1,910 km, and by adult males, was 2,353 km. Females on San Miguel Island outfitted with dive and location recorders in 1989 to 1992 were at sea an average of 66-73 days after breeding and 120-234 days after molting (Stewart and DeLong, 1993; 1994; 1995). Males were at sea an average of 119-124 days after breeding and 122-126 days after molting (Stewart and DeLong, 1993; 1995). Females covered at least 5,500-6,300 km during the post-breeding migration, and males covered at least 11,100-12,000 km (Stewart and DeLong, 1994; 1995). Over an entire year, females were at sea for about 307 days and traveled minimum distances of 18,553 km, and males were at sea about 250 days and traveled minimum distances of 21,584 km (Stewart and De Long, 1995).

Elephants' annual movements of 18,000-21,000 km/yr. are the greatest reported for a mammal (Stewart and DeLong, 1995).

Unusually long travels have been reported for several elephants; a nine month old born at Año Nuevo Island was found dead 4,000 km north on Amaknak Island, Aleutian Islands, AK, a yearling born on San Miguel Island was seen on Midway Island, HI, 4,700 km west of its birthplace, a yearling was observed midway up in the Gulf of CA (southernmost sighting), and a young male was observed on a sandy beach on Niihima Island, 40 km off the Pacific coast of Japan, 5,000 km west of Midway (Condit and Le Boeuf, 1984; Kiyota et al, 1992).

Elephants are not considered an endangered or threatened species under the ESA, nor are they considered depleted or strategic under the MMPA (Barlow et al, 1997). There are no international agreements between the U.S. and Mexico for the management of elephants (Barlow et al, 1997).

Food and Feeding

Elephants have a diverse diet and are capable of changing their diet to suit the abundance of various prey species (Antonelis et al, 1994). They prey primarily on epi- and mesopelagic cephalopods, particularly vertically migrating squid (Antonelis et al, 1994; Stewart and DeLong, 1990; 1993; 1994; Antonelis et al, 1987; 1994; Condit and Le Boeuf, 1984), feeding in water as deep as 200-1,000 m and as far offshore as 16-27 km (Kajimura and Loughlin, 1988; Le Boeuf et al, 1988). They also prey on fish to a lesser extent (Stewart and DeLong, 1990; Antonelis et al, 1987; Condit and Le Boeuf, 1984). Pacific whiting (Merluccius productus) is often the predominant fish prey species (Antonelis et al, 1987; 1994; Condit and Le Boeuf, 1984). Crustaceans, particularly the pelagic red crab (Pleuroncodes planipes), were found to be frequent prey during 1984, probably due to the effects of El Niño (Antonelis et al, 1987; 1994). Antonelis et al (1994) found that 70% of 53 prey taxa identified in stomach lavages on San Miguel Island from 1984 to 1990 were pelagic and 30% were neritic or benthic. Elephants have been observed feeding during the day and are assumed to feed at night as well (Kajimura and Loughlin, 1988).

Elephants leave haul-outs twice each year to feed for long periods (Condit and Le Boeuf, 1984). They fast while on land during the breeding and molting seasons (DeLong and Stewart, 1991). Two dive types are thought to be associated with foraging: Type D and Type E (Asaga, 1994). Type D dives are characterized by a direct descent to a depth, followed by 2-12 vertical excursions, followed by direct ascent to the surface (Crocker et al, 1994). Type E dives are characterized by direct descent to a flat bottom, followed by direct ascent to the surface (Crocker et al, 1994). The most common dive for females is the Type D dive, which they use for pelagic

foraging at an average depth of about 545 m for an average duration of 26.48 min. (Le Boeuf, 1994; Le Boeuf et al, 1993). Males generally forage using the Type E dive at an average depth of about 331 m for an average duration of 24.20 minutes (Le Boeuf, 1994; Le Boeuf et al, 1993). Females typically forage continually en route to their migration destination, which is in the general range of 44°N-52°N in the open ocean (Le Boeuf, 1994; Le Boeuf et al, 1993). They tend to feed deep in the water column (Le Boeuf, 1994). Males generally move directly to a foraging area along the northern edge of the north Pacific from WA west to the eastern Aleutian Islands and then begin feeding (Le Boeuf, 1994; Le Boeuf et al, 1993). They tend to dive in areas where they can reach the bottom to feed (Le Boeuf, 1994; Le Boeuf et al, 1993). Dive patterns of females tend to correspond to diel vertical migrations of prey, but dive patterns of males tend to follow a crepuscular pattern (Le Boeuf et al, 1988; 1993). Females gain about 1 kg/day while at sea and consume 6.2% of their mass daily (Le Boeuf et al, 1988). Males consume about three times more prey/day than females (Le Boeuf et al, 1993).

The differences in feeding techniques suggests that males and females may feed on different prey (Le Boeuf, 1994; Le Boeuf et al, 1993). Males may feed on more benthic prey, such as cyclostomes and elasmobranchs, and females may feed on more pelagic prey (Le Boeuf, 1994; Le Boeuf et al, 1993). Some analyses of stomach contents has indicated that this is the case (Le Boeuf et al, 1993). However, Stewart and DeLong (1994) state that males and females in the CA current eat similar prey but may forage on different size classes of that prey. Antonelis et al (1987) found no differences between prey of males and females whose stomachs were lavaged on San Miguel Island in 1984. The following prey species have been determined for elephants:

<u>Category</u>	<u>Scientific Name</u>	<u>Common Name</u>
Molluska	<u>Abraliopsis felis</u>	Enope squid
	<u>Architeuthis japonica</u>	Giant squid
	<u>Berryteuthis magister</u>	Magistrate armhook squid
	Chroteuthidae	Whiplash squid
	<u>Chroteuthis calyx</u>	Whiplash squid
	<u>Chroteuthis</u> sp.	Whiplash squid
	<u>Cranchia scabra</u>	Cranch squid
	Cranchiidae	Cranch squid
	<u>Dosidicus gigas</u>	Arrow squid
	<u>Galiteuthis pacifica</u>	Cranch squid
	<u>Galiteuthis phyllura</u>	Cranch squid
	<u>Galiteuthis</u> sp.	Cranch squid
	Gastropoda	Gastropod shells
	Gonatidae	Gonata squid
	<u>Gonatopsis borealis</u>	Gonata squid
	<u>Gonatopsis</u> sp.	Gonata squid
	<u>Gonatus berri</u>	Gonata squid
	<u>Gonatus onyx</u>	Gonata squid
	<u>Gonatus pyros</u>	Gonata squid
	<u>Gonatus</u> sp.	Gonata squid

	<u>Histioteuthis dofleini</u>	Umbrella or Jewel squid
	<u>Histioteuthis heteropsis</u>	Umbrella or Jewel squid
	<u>Histioteuthis sp.</u>	Umbrella or Jewel squid
	<u>Japetella heathi</u>	
	<u>Loligo opalescens</u>	Market squid
	<u>Megalocranchia sp.</u>	Cranch squid
	<u>Moroteuthis robusta</u>	Giant squid
	<u>Octopoda</u>	Octopus
	<u>Octopoteuthis deletron</u>	Octopus squid
	<u>Octopoteuthis sp.</u>	Octopus squid
	<u>Octopus bimaculatus</u>	Two-spotted octopus
	<u>Octopus dofleini</u>	Giant octopus
	<u>Octopus rubescens</u>	Red octopus
	<u>Octopus sp.</u>	Octopodid octopus
	<u>Ocythoe tuberculata</u>	Ocythoid octopus
	<u>Ommastrephes bartrami</u>	Arrow squid
	<u>Ommastrephidae</u>	Arrow squid
	<u>Onychoteuthis borealjaponicus</u>	Hooked squid
	<u>Pelecypoda</u>	Bivalve shells
	<u>Rossia pacifica</u>	Cuttlefish/stubby squid
	<u>Taningia danae</u>	Giant squid
	<u>Taonius pavo</u>	Cranch squid
	<u>Vampyroteuthis infernalis</u>	Vampire squid
Crustacea		
	<u>Euphausia sp.</u>	Euphausiids, krill
	<u>Hemisquilla ensigera californiensis</u>	Hemisquillid crustacean
	<u>Pasiphaea pacifica</u>	Ammonotheid isopod
	<u>Pleuroncodes planipes</u>	Pelagic red crab
Urochordata		
	<u>Pyrosoma atlanticum</u>	Tunicate
	<u>Thetys vagina</u>	Tunicate
Agnatha		
	<u>Eptatretus sp.</u>	Hagfish
	<u>Eptatretus stouti</u>	Pacific hagfish
	<u>(Polistotrema stouti)</u>	
	<u>Lampetra tridentata</u>	Pacific lamprey
	<u>(Entosphenus tridentatus)</u>	
Chondrichthys		
	<u>Apristurus brunneus</u>	Brown cat shark
	<u>Cephaloscyllium ventriosum</u>	Pacific rattail/swell shark
	<u>(Catulus ater)</u>	
	<u>Coryphaenoides acrolepis</u>	Rattail
	<u>Heterodontus francisci</u>	Horn shark
	<u>Hydrolagus collieri</u>	Ratfish
	<u>Platyrrhinoides triseriata</u>	Thornback

Osteichthys	<u>Prionace glauca</u>	Blue shark
	<u>Raja binoculata</u>	Big skate
	<u>Raja sp.</u>	Skate
	<u>Squalus acanthias</u>	Spiny dogfish
	<u>Squatina californica</u>	Angel shark
	<u>Urolophus halleri</u>	Stingray
	<u>Anoplopoma fimbria</u>	Sablefish
	<u>Chilara taylori</u>	Spotted cusk eel
	<u>(Otophidium taylori)</u>	
	<u>Citharichthys sordidus</u>	Pacific sanddab
Other	<u>Citarichthys sp.</u>	Sanddab
	<u>Clupea pallasii</u>	Pacific herring
	<u>Damalichthys vacca</u>	Pileperch
	<u>Engraulis mordax</u>	Northern anchovy
	<u>Icichthys lockingtoni</u>	Medusafish
	<u>Lycodes diapterus</u>	Black eelpout
	<u>Merluccius productus</u>	Pacific whiting/hake
	Pleuronectidae	Flounders and soles
	<u>Porichthys notatus</u>	Plainfin midshipman
	<u>Sardinops sagax</u>	Pacific sardine
	<u>Sebastes eos</u>	Pink rockfish
	<u>Sebastes sp.</u>	Rockfish
	<u>Sebastolobus alascanus</u>	Thornyhead
		Broken shells
		Sand
		Rocks

Habits

When at sea, elephants dive continually with only rare interruptions (DeLong and Stewart, 1991; Le Boeuf et al, 1988; 1989; Stewart, 1992; Stewart and DeLong, 1990; 1993; 1994; 1995). Young elephants are capable of continuous diving to depths over several hundred meters within days of leaving the rookery (Sinclair, 1994). Elephants display five different types of dives (Crocker et al, 1994; Asaga et al, 1994). Type A dives are characterized by direct descent to a sharp inflection point, followed by direct ascent to the surface; Type B dives are characterized by direct descent ending in a rounded bottom, followed by direct ascent to the surface; Type C dives are characterized by direct descent to depth, at which point, descent rate decreases noticeably but proceeds directly to the bottom of the dive, followed by direct ascent to the surface; Type D dives are characterized by direct descent to a depth, followed by 2-12 vertical excursions, followed by direct ascent to the surface; Type E dives are characterized by direct descent to a flat bottom followed by direct ascent to the surface (Crocker et al, 1994). Type A and B dives may serve as transit dives as elephants travel; Type C dives may be processing dives, in which behavior and energy utilization are focused on internal processes such

as digestion and clearing of metabolites; Type D dives may be for pelagic foraging; Type E dives may be for benthic foraging (Asaga et al, 1994; Crocker et al, 1994). Females tend to use Type D dives, and males tend to use Type E dives while foraging (Asaga et al, 1994; Le Boeuf, 1993). Le Boeuf et al, (1993) found that Type E dives are shorter (average 24.20 minutes) and follow a crepuscular pattern, and Type D dives are longer (average 26.48 minutes) and follow a diel pattern. Crocker et al (1994) found that, on average, angles of dive descent were shallower (30°-56°) than angles of ascent (52°-82°). They also found that average descent speeds were 27% faster than average ascent speeds.

The fastest long distance movement recorded for male elephants by Stewart (1992) was 102 km/day. Le Boeuf et al (1993) report that average speed of travel is not significantly different between adult males and females, averaging 77.2-89.4 km/day and average speed of travel for juveniles is lower, averaging 58.3 km/day. However, according to Stewart and DeLong (1994), elephants generally travel at speeds of about 90-100 km/day. Crocker et al (1994) recorded average swim speeds of 0.91-1.66 m/sec. Elephants are submerged for about 83-90% of the time they are at sea (DeLong and Stewart, 1991; Le Boeuf, 1994; Le Boeuf et al, 1988; 1989) and spend about 25-35% of their time at the bottom of dives (DeLong and Stewart, 1991; Stewart and DeLong, 1993). Stewart and DeLong (1993) found that females were submerged an average of 91% of the time and males were submerged an average of 88% of the time. Average vertical travel distances have been recorded as 6,800 km/yr. for males and 9,000 km/yr. for females (Stewart and DeLong, 1995).

Dive depth for males averages 331-393 m (DeLong and Stewart, 1991; Le Boeuf, 1994; Stewart and DeLong, 1990; 1993; 1994). Dive depth for females averages 333-520 m (Le Boeuf et al, 1986; 1988; Stewart and DeLong, 1993; 1994). Maximum depths of 1,581 m for a male and 1,567 m for a female have been recorded (Stewart and DeLong, 1990; 1994). In general, males dive shallower than females (Le Boeuf, 1994; Le Boeuf et al, 1993; Stewart and DeLong, 1994).

Dive duration for males averages 21-23.1 (DeLong and Stewart 1991; Stewart, 1992; Stewart and DeLong, 1993; 1994), with the longest dive recorded lasting 77 minutes (DeLong and Stewart, 1991; Stewart and DeLong, 1990). Surface interval for males averages 3-3.6 minutes and also varies seasonally, as do rates of ascent and descent (DeLong and Stewart, 1991; Stewart and DeLong, 1993; 1994). Dive duration for females averages 17-23.6 minutes (Le Boeuf et al, 1986; 1988; Stewart, 1992; Stewart and DeLong, 1993; 1994), with the longest dive recorded lasting 62 minutes (Le Boeuf et al, 1989). Surface interval for females averages 2-4.2 minutes (Le Boeuf et al, 1986; 1988; 1989; Stewart and DeLong, 1993; 1994). Le Boeuf (1994) found that pregnant elephants in the third trimester had 36% longer dives than non-pregnant females and had maximum dive durations over an hour long. Unpredictable surface intervals more than 10 minutes long occur occasionally while elephants are at sea (Le Boeuf, 1994; Le Boeuf et al, 1988; 1989).

Dive depths and durations vary seasonally and daily (DeLong and Stewart, 1991; Stewart and DeLong, 1990) but do not vary with mass or age (Le Boeuf, 1994; Le Boeuf et al, 1988). Dives are more numerous, shallower, and of shorter duration at night than during the day (Le Boeuf et al, 1988; 1989; Stewart and DeLong, 1993).

Overall, elephants generally descend rapidly to depths greater than 150 m but less than 800 m, spend about 5-34 minutes at maximum depth with little vertical movement, then ascend quickly for a brief surface interval of 2-4 minutes before diving again (DeLong and Stewart, 1991; Stewart and DeLong, 1990; 1993; 1994; 1995). DeLong and Stewart (1991) found no

evidence as to whether or not elephants sleep at sea. Sleep may be unihemispheric or may occur during descent or on the bottom (DeLong and Stewart, 1991; Le Boeuf et al, 1986; 1988).

Elephants have been observed interacting with California sea lions (Webster and Baird, 1998). An elephant was seen chasing California sea lions both on land and in the water on Isla Los Islotes in January, 1996 (Webster and Baird, 1998).

Reproduction

Breeding season: Dec. to mid-Mar.

Pupping season: Dec. to Feb.

Sexual maturity:

Females: 2-6 (ave. 3-4) yrs. old

Males: 5-10 (ave. 8) yrs. old, most do not mate until 8-9 yrs. old

Lactation: 25-34 days

Neonate length: 1.0 m

Neonate weight: 35 kg

Elephants are extremely polygynous (Clinton and Le Boeuf, 1993; Campbell, 1987; Cox, 1981; Deutsch, 1991; Deutsch et al, 1994; Le Boeuf, 1977; Le Boeuf and Mesnick, 1991; Reidman, 1983). Males and females return to rookeries in the southern part of their range to breed (Allen et al, 1989; Barlow et al, 1997; U.S. Air Force, 1996). 95% of copulations occur on land (Le Boeuf and Mesnick, 1991). Females arrive on the rookery pregnant, join a harem, and give birth to a single pup after 6 days (Le Boeuf and Mesnick, 1991). The pup is nursed daily for 4 weeks (Le Boeuf and Mesnick, 1991). Each female remains on land for about 35 days, and during the last 3-5 days on land, she mates with one or more males (Cox, 1981). Multiple copulations and copulations with as many as three different males are common (Le Boeuf and Mesnick, 1991). Dominant males control harems of females for mating (Cox, 1981; Le Boeuf and Mesnick, 1991). Harem size varies from a few females up to a thousand (Le Boeuf and Mesnick, 1991). One male alone may keep all competitors out of a harem of 50 or less, and if entrance is restricted by terrain, a single male may control a harem of 100 females (Le Boeuf and Mesnick, 1991). As harem size increases, number of males within the harem increases (Le Boeuf and Mesnick, 1991). In a harem of 1,000 females, 20-30 males may reside within the harem (Le Boeuf and Mesnick, 1991).

Movement and genetic exchange occurs between rookeries, but most elephants return to natal rookeries when they start breeding (Barlow et al, 1997). It should be noted though that elephants are genetically identical because of a genetic bottleneck created by near extinction in the 1800's (Le Boeuf, 1977; Stewart et al, 1994). Some females emigrate to other rookeries when natal rookeries become crowded (Huber et al, 1991; Reiter et al, 1981). All rookeries established since 1975 were first utilized by juveniles, and juveniles appeared at these sites 6-16 years before breeding began (Mesnick et al, 1998). Females display fidelity to a rookery once they begin breeding (Campbell, 1987; Stewart and DeLong, 1994; Sydeman and Nur, 1994).

During the breeding season, males take part in only three activities: threatening or avoiding the threats of other males, mounting and mating with females, and resting on the beach (Cox, 1981). A variety of movements, postures, and vocalizations are used by males to signal aggressive intent and establish dominance (Cox, 1981). Usually, the recipient of a threat moves away before physical contact becomes immanent, or occasionally the challenged male responds with a threat and the challenger moves off; if neither male retreats, direct combat occurs (Cox, 1981). A linear social dominance hierarchy of males is established with the threats and fights (Cox, 1981). Each male tries to mate with as many females as possible and to prevent others from doing the same (Cox, 1981). There is a strong positive correlation between aggressiveness, social rank, and the number of copulations obtained (Cox, 1981). Cox (1981) found that males on Año Nuevo Island from 1973 to 1975 made an average of 2.7 threats/hr. with a mean duration of 11.9 seconds for interactions. This suggests that males spend about 0.9% of their time threatening or fighting with other males (Cox, 1981). Cox (1981) also found that the alpha male was issued more threats than any other male. On a typical day, the alpha male spent 4.2% of his time threatening other males, 8.3% attempting to mate, and 87.5% resting (Cox, 1981). Less than 5% of the threats issued by subadults were directed toward adults (Cox, 1981). In more than 98% of the interactions observed, the challenger threatened a male who was about the same age or younger than himself (Cox, 1981). 93.9% of challenges were initiated with postural or vocal threats rather than by direct attack; 2.3% of challenges initiated by threat rather than attack subsequently escalated to blows (Cox, 1981). Males within the harem were threatened considerably more frequently than those at a greater distance, even though males outside the harem were spatially closer to each other (Cox, 1981). Males in the harem spent about 5% of their time engaged in sexual activity (Cox, 1981).

Males on the periphery of a harem wait for an opportunity to mate by sneaking into the harem or attempting to mate with females as they exit the harem (Le Boeuf and Mesnick, 1991). The number of peripheral males around a harem varies from 0 to 60 or more (Le Boeuf and Mesnick, 1991). When a female leaves the harem, she is chased and intercepted by several peripheral males and perhaps one or more high ranking harem males, who compete aggressively to copulate with her (Le Boeuf and Mesnick, 1991; Mesnick and Le Boeuf, 1991). Several males may bite her on the neck and copulate with her on the beach or in the water before she gets away (Le Boeuf and Mesnick, 1991). It is easier for the female to elude her pursuers once she reaches the water, but sometimes, males will chase a female for 500 m or more (Le Boeuf and Mesnick, 1991; Mesnick and Le Boeuf, 1991).

Male courtship is usually direct, aggressive, and persistent (Le Boeuf and Mesnick, 1991). A male moves directly to the side of a female, puts a foreflipper over her back, bites her on the neck, pulls her to him with his foreflippers, and attempts intromission (Le Boeuf and Mesnick, 1991). If the female resists or moves away, the male pins her down by slamming the full weight of his head and forequarters on her back one or more times and bites her on the neck more vigorously (Le Boeuf and Mesnick, 1991). Males often try to physically overpower resistant females (Le Boeuf and Mesnick, 1991). This high level of aggression can be dangerous to females (Le Boeuf and Mesnick, 1991). It can be particularly dangerous during the keen competition for a female departing from a rookery (Le Boeuf and Mesnick, 1991). Le Boeuf and Mesnick (1991) found that females received over 20 times more blows, mounts, and copulations while departing from the Año Nuevo Island and adjacent mainland rookeries than they did within the harems. Even after death, female carcasses are objects of avid competition among males (Le

Boeuf and Mesnick, 1991). Minimum male induced female mortality was 0.008, and possibly 0.001, on Año Nuevo Island and adjacent mainland rookeries from 1976 to 1986 (Le Boeuf and Mesnick, 1991). This may be a significant selection pressure (Le Boeuf and Mesnick, 1991). Females appear to submit to copulation more readily when leaving a rookery (Le Boeuf and Mesnick, 1991). This behavior may allow them safer passage to the water (Le Boeuf and Mesnick, 1991).

Males attempt to mate with a variety of inappropriate partners, including pups, yearlings, 2 yr. olds, pregnant or otherwise non-receptive females, and dead females of all ages (Mortenson and Follis, 1997; Rose et al, 1991). Rose et al (1991) found that up to 50% of weaned pups on Año Nuevo Island at the end of breeding season show evidence of having been mounted by males. They also found that 91% of these deviant mounters were subadults. A subadult male elephant was observed mounting harbor seal pups, both dead and alive, in Russian River, central CA (Mortenson and Follis, 1997). It is possible a juvenile male observed interacting with human swimmers was attempting to mate, as his penis was extruded and erect during the interaction (Webster and Baird, 1998). Possible reasons for inappropriate mounting are male sexual inexperience, high libido, limited access to females, and stimulus generalization (Rose et al, 1991).

Females employ a different mating strategy than males. Non-estrous females always respond with total resistance to males attempting to mate with them (Cox, 1981). Females in estrous generally also resist males, and female protests alerts other males in the area of the attempted copulation, giving more dominant males the chance to displace the male attempting to mate (Cox, 1981; Le Boeuf and Mesnick, 1991). A protested mount is far more likely to be detected and interrupted than a mount in which the female responds passively (Cox, 1981; Le Boeuf and Mesnick, 1991). Females are more likely to respond receptively to older, larger males than to younger, smaller males regardless of their behavior (Cox., 1981). Females are also more likely to respond receptively to an adult male who has just dominated another male than to one who has been dominated (Cox, 1981). Males usually attempt to mount a female more than once when the female resists (Cox, 1981). Cox (1981) found that, on Año Nuevo Island from 1973 to 1975, in 44.4% of the interludes in which males threatened and dominated another before attempting another mount, the female showed increased receptivity when remounted. Females became more receptive in only 6.6% of remounts when the male rested or was dominated in the interlude (Cox, 1981).

Cox (1981) found that, on Año Nuevo Island from 1973 to 1975, 78.1% of unprotested mounts, 65.5% of partially protested mounts, and 19.3% of totally protested mounts led to intromission. Average time from initiation of mount to intromission was 93.6 seconds for unprotested mounts, 125.4 seconds for partially protested mounts, and 165.6 seconds for totally protested mounts (Cox, 1981).

Elephants fast during the breeding season (Deutsch, 1991; Le Boeuf et al, 1988; Stewart and DeLong, 1994). The length of the fast increases sharply for males at 6 yrs. old (Clinton, 1994). Males fast three times longer than females (Deutsch et al, 1994). Males lose 20-50% of their body mass (Deutsch, 1991; Deutsch et al, 1994; Stewart and DeLong, 1994). Females lose 36-42% of their body mass during lactation (Le Boeuf et al, 1989; Le Boeuf and Mesnick, 1991; Stewart and DeLong, 1994). Deutsch et al (1994) estimates that adults of both sexes lose 36-37% of their mass during the breeding season.

Fecundity of males increases up to 12 yrs. old (Clinton, 1994). Mating success of males also increases with age (Clinton and Le Boeuf, 1993) and with increased dominance rank (Haley et al, 1994). Pups receive no paternal care (Reidman, 1983).

Females generally produce one pup per year with occasional skipping (Deutsch et al, 1994; Reidman, 1983). Sydemann and Nur (1994) report that skipping is fairly regular among multiparous females, although it is apparently most common for young 4-5 yr. olds. Females that begin breeding at an early age are more likely to skip breeding in subsequent years (Sydemann and Nur, 1994). Female effort during lactation is not correlated with age or mass (Deutsch et al, 1994). Absolute measures of investment in offspring are directly proportional to maternal mass, which increases from 3-6 yrs. old before reaching an asymptote (Deutsch et al, 1994). Maternal investment in male offspring is similar to that in female offspring (Deutsch et al, 1994). Average lactation duration increases with age (Deutsch et al, 1994).

Females primiparous at 4 or 5 yrs. old have higher reproductive success and subsequent natality and weaning success than those primiparous at 3 yrs. old (Huber, 1987; Sydemann and Nur, 1994). Also, females primiparous at 4 yrs. old have higher survivorship up to 8 yrs. old than females primiparous at 3 yrs. old (Reiter and Le Boeuf, 1991). Probability of weaning a healthy pup increases with age for females (Reiter et al, 1981; Sydemann and Nur, 1994). Age-specific weaning success is greatest in low-density rookeries, and density effects are most influential on young elephants (Sydemann and Nur, 1994). Females that fail to wean a pup in one year are less likely to rear a pup in the next year (Sydemann and Nur, 1994).

Older females are better mothers because they are larger, have higher dominance, have more experience, and are more likely to mate with high ranking males and give birth in optimal time and place (Reiter et al, 1981). However, Sydemann et al, (1994) found that females on the Farallon Islands from 1974 to 1989 increased reproductive success from 3 to 7 yrs. old and then success leveled off. Controlling for previous experience showed reproductive success increased linearly through the lifetime, but controlling for age showed reproductive success was related to experience in a parabolic fashion (Sydemann et al, 1994). Experience is beneficial for young females but is deleterious among old females (Sydemann et al, 1994; Sydemann and Nur, 1994). Arrival and parturition dates are later for inexperienced and very experienced females and are earlier for moderately experienced females (Sydemann et al, 1994). Young females that give birth later in the season have lower reproductive success, but there is no such decline for older females (Sydemann et al, 1994).

Average natality by parous females in the Farallon and several Channel Islands from 1975 to 1983 was 86% (Huber, 1987). Average natality of females 5-10 yrs. old on southeast Farallon Island is 80%, indicating 20% do not give birth in any given year (Sydemann and Nur, 1994). Age-specific natality was 13% for 3 yr. olds, 75% for 4 yr. olds, and 80% for 5-10 yr. olds in the Farallon and Channel Islands from 1970 to 1986 (Huber et al, 1991). Huber et al (1991) found that the average age of females giving birth rose from 4 yrs. old in 1975 to 7 yrs. old in 1986. Pregnancy rates declined temporarily at some rookeries in 1984 and 1985, following the 1982-83 El Niño (Stewart et al, 1994). Age of primipary was also delayed for cohorts affected by the 1982-83 El Niño (Sydemann and Nur, 1994).

Mortality and Economics

Natural Mortality

White sharks (Carcharodon carcharias) and orcas prey on elephants (Ainley et al, 1985; Antonelis and Fiscus, 1980; Barrett-Lennard et al, 1995; Le Boeuf et al, 1982; Le Boeuf et al, 1986; Le Boeuf et al, 1988; Le Boeuf and Mesnick, 1990; Stewart and Yochem, 1985; Heimlich-Boran, 1988; Baird and Dill, 1996; Felleman et al, 1991; Hoyt, 1984; Sobolevsky and Mathisen, 1996). Most shark attacks on elephants occur near the surface, so elephants' deep diving and little time spent at the surface may reduce predation (Le Boeuf et al, 1986; 1988).

Caliciviruses isolated from elephants may be associated with pup orphaning (Smith et al, 1986). At least 14 parasites, 17 bacteria, and 6 viruses have been found in or on elephants:

Bacteria

<u>Aeromonas hydrophilia</u>	Gulland et al, 1997
<u>Bacillus</u> sp.	Gulland et al, 1997
<u>Citrobacter</u> sp.	Gulland et al, 1997
<u>Corynebacterium</u> sp.	Gulland et al, 1997
<u>Enterobacter cloacae</u>	Gulland et al, 1997
<u>Enterococcus</u> sp.	Gulland et al, 1997
<u>Escherichia coli</u>	Gulland et al, 1997
<u>Klebsilla</u> sp.	Gulland et al, 1997
<u>Moraxella</u> sp.	Gulland et al, 1997
<u>Morganella morganii</u>	Gulland et al, 1997
NESSD	Mesnick et al, 1998
<u>Proteus vulgaris</u>	Gulland et al, 1997
<u>Pseudomonas fluorescens</u>	Gulland et al, 1997
<u>Psudomonas putida</u>	Gulland et al, 1997
<u>Salmonella newport</u>	Gulland et al, 1997
<u>Staphylococcus</u> sp.	Gulland et al, 1997
B-hemolytic <u>Streptococcus</u> sp.	Gulland et al, 1997
<u>Streptococcus viridens</u>	Gulland et al, 1997

Viruses

Caliciviruses	Smith et al, 1978
SMSV-2	Berry et al, 1990
SMSV-4	Smith et al, 1978
SMSV-5	Smith et al, 1978
SMSV-7	Smith et al, 1986

Parasites

Trematoda	
<u>Cryptocotyle lingua</u>	Creplin, 1825; Delyamure, 1955
<u>Zalphotrema hepaticum</u>	Stunkard and Alvey, 1929; King, 1964
Nematoda	
<u>Anisakis similis</u>	Baird, 1853; Caballero and Peregrina, 1938
<u>Contracaecum osculatum</u>	Rudolphi, 1802; Caballero and Peregrina, 1938

<u>Otostrongylus circumlitus</u>	Gulland et al, 1997
<u>Porrocaecum decipiens</u>	Krabbe, 1878; Baylis, 1916
Acanthocephala	
<u>Corynosoma</u> sp.	King, 1964
Acarina	
<u>Halarachne miroungae</u>	Ferris, 1925

Also, ectoparasites have been found on elephants, including a colonial diatom, Navicula grevilli, and goose barnacles, Lepas pacifica (Baldrige, 1977). Stalked barnacles, Conchoderma auritum and C. virgatum and goose barnacles, Lepas hilli have also been recorded on dead elephants (Joseph et al, 1986). Also, cookiecutter (or cigar) shark (Isistius brasiliensis) wounds have been found on elephants (Deutsch, 1991; Gallo-Reynoso and Figueroa-Carranza, 1992; Le Boeuf et al, 1987).

Bodkin et al (1985) reports that pup mortality was 1.6-3.7%/yr. on San Nicolas Island 1969 to 1971, ranged from 13.0-40.0%/yr. on Año Nuevo Island from 1968 to 1980, and ranged from 7-71%/yr. on the Farallon Islands 1974 to 1976. Pup mortality was 3-8% on San Miguel Island from 1979 to 1982 and 4% on San Nicolas Island from 1971 to 1982 (Cooper and Stewart, 1983). Le Boeuf et al (1994) found that average percentage survival to 1 yr. old was 36.8%, to 2 yrs. old was 26.3%, to 3 yrs. old was 19.4%, and to 4 yrs. old was 16.3% on Año Nuevo, CA from 1971 to 1988. As colony size increased, survivorship to 4 yrs. old decreased significantly on Año Nuevo, CA (Le Boeuf et al, 1994). Le Boeuf et al (1994) report that most of the first-year mortality on Año Nuevo, CA from 1971 to 1988 occurred at sea, with an average of 31.5% of first-year mortality due to neonate death on the rookery.

Early reproduction increases mortality for males 6-8 yrs. old (Clinton, 1994). Age-specific mortality rates for males decrease from weaning to 4 yrs. old, then rise steadily until 10 yrs. old (Clinton and Le Boeuf, 1993). Mortality rates for males drop at 11-12 yrs. old (Clinton, 1994; Clinton and Le Boeuf, 1993). About 10% of males live to 8 yrs. old and about 1% live to 12-13 yrs. old (Clinton and Le Boeuf, 1993). Mortality of males 5-10 yrs. old on Año Nuevo Island increased from the 1970's to the 1980's (Clinton and Le Boeuf, 1993). Females primiparous at 3 yrs. old have lower survivorship up to 8 yrs. old than females primiparous at 4 yrs. old (Reiter and Le Boeuf, 1991; Sydeman and Nur, 1994). Maternal survival is inversely related to age of primipary (Sydeman and Nur, 1994).

Mortality is also caused by males killing females and pups (Le Boeuf and Mesnick, 1991; Mesnick and Le Boeuf, 1991). At least 0.2% of elephant pups on Año Nuevo Island are killed each year by males attempting to mount them (Mortenson and Follis, 1997). Rose et al (1991) state that 0.5% of weaned pups died on Año Nuevo Island from 1969 to 1990. About 35% of those were obviously killed by males, and males were suspected to have killed the rest (Rose et al, 1991).

Storms and heavy swells can also cause pup mortality (Bodkin et al, 1985; Lowry et al, 1996). Le Boeuf et al (1994) report that juvenile survivorship rates were lowest in El Niño years (1978, 1983, and 1986). Males usually live a maximum of 14 years (Campbell, 1987; Clinton and Le Boeuf, 1993; Deutsch et al, 1994). Females can live up to 19 years (Deutsch et al, 1994).

Anthropogenic Interactions and Mortality

Pollution may detrimentally affect elephants. PCBs and DDT and its metabolites have been isolated from elephant tissues (Bacon et al, 1992; Hong et al, 1993; Newman et al, 1994).

Human disturbance may keep elephants from colonizing some islands (Antonelis et al, 1981). U.S. Air Force (1996) states that human disturbance could lead to disruption of social structure and mother/pup separations. However, Le Boeuf and Panken (1977) report that elephants on Año Nuevo Point, CA from 1975 to 1977 seemed undisturbed by people in plain view at distances of 10 m or more.

Unusual interactions have occurred at Isla Los Islotes (Mesnick et al, 1998; Webster and Baird, 1998). A young male elephant has been observed trying to mate with humans and to establish dominance over sea lions and small boats (Mesnick et al, 1998; Webster and Baird, 1998).

Elephants were commercially harvested for oil in the 1800's (Allen et al, 1989; Bodkin et al, 1985; Campbell, 1987; King et al, 1989; LeBoeuf, 1977). This harvest led to the near extinction of the species (Allen et al, 1989; Bodkin et al, 1985; Le Boeuf, 1977).

Fisheries are an anthropogenic source of mortality for elephants. Elephants have been found tangled in fishing gear (Campbell, 1987; Condit and Le Boeuf, 1984; Kajimura and Loughlin, 1988; Loughlin et al, 1983) and specifically in longline gear set for sablefish (*Anoplopoma fimbria*) (Antonelis and Fiscus, 1980). Foreign and joint venture trawl vessels have taken elephants incidentally (Perez and Loughlin, 1991). Elephants were observed incidentally taken in the Japanese high seas squid fishery (Hobbs and Jones, 1993). Most elephants caught in fishing gear are caught about 200 m below the surface (Condit and Le Boeuf, 1984; Le Boeuf et al, 1988). Elephants were one of the most common pinnipeds observed taken in the CA drift gillnet fishery for swordfish and sharks from 1990 to 1993 (Barlow et al, 1994). Fisheries that take elephants and annual mean mortality are as follows (Barlow et al, 1994; 1997):

CA breeding stock:

CA/OR thresher shark/swordfish drift gillnet	106 (CV=0.11)
CA angel shark/halibut and other species large mesh (>3.5) set gillnet	1.5 (CV=0.26)
WA, OR, CA groundfish trawl	0
WA Willapa Bay drit gillnet (salmon)	0.4
Chehalis River salmon setnet	1
Yakutat and southeastern AK gillnet for salmon	
AK trawl for pollock and other groundfish	
Longline for sablefish	

Minimum annual incidental fisheries mortality of the CA breeding stock of elephants is 145/yr. (Barlow et al, 1997). This does not exceed 10% of PBR (214) and is considered insignificant (Barlow et al, 1997). Drift gillnet fisheries for swordfish and sharks similar to U.S. fisheries occur along the entire Pacific coast of Baja CA and probably take elephants in the Baja CA breeding stock (Barlow et al, 1997). One elephant was found dead in a gillnet in Bahía de los Angeles in June, 1986 (Mesnick et al, 1998). There is no subsistence harvest of elephants (Barlow et al, 1997).

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Review of Pacific Harbor Seals (Phoca vitulina) in the North Pacific 1978-1997

Population, Abundance, and Distribution

Range:

Subspecies: P. v. stejnergeri

Eastern coast of Hokkaido, Japan, eastern limit of Point Erimo; Nemuro Peninsula on Hokkaido north along the Kuril Islands and eastern Kamchatka coast to the Commander Islands

P. v. richardsi

Pribilof, Kuril, Commander, and Aleutian Islands, northern limit of Herschel Island and Point Barrow, through AK to CA, the Channel Islands, and Mexico; southern limit is Cedros Island (28°12' N) or San Roque Island, Guadalupe

Breeding Range:

Subspecies: P.v. stejnergeri

Nine rocky haul-outs on the eastern coast of Hokkaido, Japan; Nemuro Peninsula on Hokkaido northward along the Kuril Islands and eastern Kamchatka coast to the Commander Islands

P.v. richardsi

Bahia San Quintin, Mexico (30 °N) to Prince William Sound, Alaska (61 °N), including all the Channel Islands except maybe Santa Barbara Island

Preferred Habitat:

Offshore rocks and sandy beaches and mudflats or sandbars at river mouths; haul-out on rocks, reefs, beaches, and drifting glacial ice

Harbor seal populations show varying trends depending upon location. Generally, Pacific harbor seals are divided into two subspecies; P. v. stejnergeri inhabits Japan, and P. v. richardsi ranges from Baja CA to northern AK (Barlow et al, 1997; Temte, 1985; Wada et al, 1991). The latter subspecies is separated into six overall stocks. Three stocks are along the west coast of the continental U.S.: CA, OR/WA coastal, and WA inland (Barlow et al, 1997; U.S. Dept. of Comm., 1996). There is a small number of harbor seals in Baja CA, but they are not considered part of the CA stock because no international agreement for harbor seal management exists between the U.S. and Mexico (Barlow et al, 1997). Phylogenetic and other evidence indicates that harbor seals in AK make up three additional stocks: southeastern AK, which ranges from the AK/British Columbia border to Cape Suckling, AK (144°W); Gulf of AK, which ranges from Cape Suckling to Unimak Pass, including the Aleutian Islands; and Bering Sea, which ranges in all waters north of Unimak Pass (Hill et al, 1997; Barlow et al, 1997; MMC, 1997; U.S. Dept. of Comm., 1996).

Population in southeastern AK appears relatively stable or increasing (Regelin et al, 1996; Hill et al, 1997; Pitcher, 1989). Counts decreased at Sitka and Ketchikan trend sites from 1983-1993, but counts were higher in 1995 than they were in the early 1980's (Hill et al, 1997; MMC, 1997). Population of the southeastern AK stock was 37,450 in 1993 (Hill et al, 1997). N_{\min} for the stock is 35,226, and PBR is 2,114.

Population of the Gulf of AK stock has been declining (Balsiger, 1995; Hobson et al, 1997; MMC, 1997; Perez, 1990). NMFS (1993) assert that population decreased significantly from the 1970's to 1992. Hill et al (1997) state that trends in the Aleutian Islands are unclear. Population declined 57% in Prince William Sound from 1984 to 1992 (Hill et al, 1997; Loughlin et al, 1996; MMC, 1997) and 46% from 1988 to 1994 (Regelin et al, 1996). This decline began before the 1989 Exxon Valdez oil spill (Hill et al, 1997; Loughlin et al, 1996; MMC, 1997). Also, population has declined steadily throughout the Kodiak Archipelago since 1976 (Hill et al, 1997; Miles et al, 1992; MMC, 1997; Pitcher, 1989). However, Regelin et al (1996) indicate that population in the Kodiak region may be increasing based upon 1992-1994 surveys. Counts from 1976 to 1988 declined 85% and counts from 1988 to 1994 declined 33% (MMC, 1997) on southwestern Tugidak Island, formally the site of one of the largest concentrations of harbor seals in the world (Hill et al, 1997; MMC, 1997; Pitcher, 1989). However, counts increased from 1992 to 1994 (Hill et al, 1997). Despite population growth in some areas, overall stock size remains low compared to its size in the 1970's and 1980's (Hill et al, 1997; MMC, 1997). Population of the Gulf of AK stock was 23,504 in 1994 (Hill et al, 1997). N_{\min} of the stock is 22,427, and PBR is 673 (Hill et al, 1997).

Population of the Bering Sea stock is thought to be declining (Hill et al, 1997; Hobson et al, 1997; MMC, 1997; Pitcher, 1989; Swartzman and Hogman, 1991). MMC (1997) asserts that population declined substantially in the southeastern Bering Sea in the 1980's. However, Regelin et al (1996) state population is simply fluctuating in the Bering Sea and there is no strong evidence of decline. Counts declined 83% on Otter Island from 1974 to 1995, declined 42% on the northern side of the AK peninsula from 1975 to 1995, and declined in northern Bristol Bay from the 1970's to the 1990's; however, Bristol Bay counts remained stable from 1990 to 1995 (Hill et al, 1997; MMC, 1997). Population of the Bering Sea stock was 13,323 in 1995 (Hill et al, 1997). N_{\min} of the stock is 12,648 and PBR is 379 (Hill et al, 1997).

Population of the CA stock seems to be increasing (with the exception of El Niño years) (Barlow et al, 1997; NMFS, 1993; Regelin et al, 1996). Annual growth rate was 3.5%/yr. from 1982 to 1995 (Barlow et al, 1997). Population increased rapidly on the Channel Islands in the 1960's and 1970's, possibly as a result of immigration from the north during recovery from bounty hunting (Stewart et al, 1988). Population increased 14.7%/yr. on mainland CA and 9.1%/yr. on the Channel Islands from 1965 to 1986 (Olesiuk et al, 1990a). Population of the CA stock was 30,293 in 1995 (Barlow et al, 1997). N_{\min} of the stock is 27,962 and PBR is 1,678 (Barlow et al, 1997).

Population levels of the OR/WA coastal stock were depressed in the 1940's and 1950's due to bounty hunting (Barlow et al, 1997). Currently, population of this stock appears to be increasing, although the rate of increase may be slowing (Barlow et al, 1997; NMFS, 1991; 1992; 1993; Regelin et al, 1996). Counts of this stock increased 11%/yr. from 1977 to 1982 and 5.5%/yr. from 1983 to 1992 (Barlow et al, 1997). Population increased 6-9%/yr. in OR from 1975 to 1983, increased 19.1%/yr. in waters adjacent to the Columbia River from 1976 to 1983, and increased 10.7%/yr. in the Columbia River from 1976 to 1982 (Olesiuk et al, 1990a).

Growth rates of 6%/yr. and 8%/yr. were observed in OR and WA respectively from 1991 to 1992 (Hill et al, 1997). Population of the OR/WA coastal stock was 27,131 in 1993 (Barlow et al, 1997). N_{min} of this stock is 25,665, and PBR is 1,540 (Barlow et al, 1997).

Population of the WA inland stock decreased in the 1940's and 1950's due to bounty hunting and remained relatively low in the 1970's (Barlow et al, 1997). Currently, population is thought to be increasing (Barlow et al, 1997). Population of the WA inland stock was 16,253 in 1993. N_{min} of this stock is 15,349, and PBR is 921 (Barlow et al, 1997). Tagging studies have shown no movement of harbor seals between WA coastal and inland waters (Barlow et al, 1997; Temte, 1985).

Populations in Baja CA and British Columbia are generally not considered in U.S. management stocks. Count was 600 in Baja CA in the mid-1970's (Brueggeman et al, 1990). Population increased 12.5%/yr. in British Columbia from 1973 to 1988 (Olesiuk, 1993; Hill et al, 1997; Olesiuk et al, 1990a) and is still thought to be increasing (Regelin et al, 1996). Population in British Columbia was 9,000-10,500 when protection began in 1970; population was 95,500 in 1988 (Olesiuk et al, 1990a).

According to Wada et al (1991), population along the eastern coast of Hokkaido, Japan was steady at about 350 from 1974 to the early 1990's. Population has decreased since the 1960's because of strong hunting pressure and incidental mortality (Wada et al, 1991). Population was 10,000-15,000 in the western north Pacific in the early 1980's (Brueggeman et al, 1990).

Range-wide population was 312,000-317,000 (302,000 in the eastern north Pacific and 10,000-15,000 in the western north Pacific) in the early 1980's (Jeffries, 1984; Brueggeman et al, 1990). Fourteen percent of the population occurs south of AK (Brueggeman et al, 1990). Fluctuations in population levels may be caused by changes in prey availability, quality of pupping habitat, weather, tides, or season (Barlow et al, 1997; Bayer, 1985; Bigg et al, 1990; Brueggeman et al, 1990; Jeffries, 1984). Population increases might reflect recovery from previous heavy hunting (Olesiuk et al, 1990a), and southern population increases might reflect immigration from the north as northern populations recover (Allen et al, 1989; Stewart et al, 1988). Population decreases may be a result of natural population cycles, disease, predation, past commercial harvesting, subsistence take by natives, pollution, changes in important habitat components, or increased commercial fishing (MMC, 1997).

Harbor seals are generally not considered migratory and are thought to be somewhat site specific; however, they can display seasonal abundances and local movements of considerable distances (Barlow et al, 1997; Brueggeman et al, 1990; Hill et al, 1997; Naito, 1982; U.S. Air Force, 1996a; 1996c), occasionally traveling as far as 300-500 km (Barlow et al, 1997). Most long distance movements are interseasonal or inter-annual; intraseasonal movements are usually shorter (Yochem et al, 1987). Harbor seals have displayed strong fidelity to haul-out sites (Barlow et al, 1997; Hill et al, 1997; Temte, 1985) and are usually found within 10-20 km of shore (Temte, 1985; MMC, 1997). Harbor seals are somewhat polygynous, and females and males aggregate at different sites (Slater and Markowitz, 1983). Males may actively exclude other males from female areas, and females may form nursery herds (Slater and Markowitz, 1983).

Harbor seals are not considered endangered or threatened under the MMPA (Barlow et al, 1997; Hill et al, 1997). However, the MMPA was amended in 1994 to include provisions for establishing agreements between the NMFS and the AK Native organizations for the cooperative

management of marine mammals, including harbor seals, in AK (MMC, 1997). As a result, the AK Native Harbor Seal Commission was formed (MMC, 1997).

Food and Feeding

Harbor seals are opportunistic predators that consume a wide variety of fish, crustaceans, mollusks, birds, and other organisms (Antonelis and Fiscus, 1980; Everitt et al, 1981; Haaker et al, 1984; Harvey et al, 1995; Temte, 1985; Olesiuk, 1993; Jones, 1981). They have different prey preferences depending upon location and season (Antonelis and Fiscus, 1980; Harvey, 1987; Johnson and Jeffries, 1983; Pitcher, 1980a; Sease, 1992; Jones, 1981). Harbor seals take advantage of seasonal and geographical abundances of prey species, and some populations may follow certain anadromous prey species, including lamprey as well as salmonids (Bigg et al, 1990; Harvey et al, 1995; Hobson et al, 1997; Jeffries, 1984). According to Brown (1980) and Brown and Mate (1983), harbor seals prefer benthic and epibenthic fish.

Harbor seals are of concern to fisherpeople (Springer, 1992; Lowry, 1982; Hobson et al, 1997; Harvey, 1987). However, harbor seals do not actively seek out commercially valuable species but take advantage of seasonal abundances (Bigg et al, 1990; Harvey et al, 1995; Hobson et al, 1997; Jeffries, 1984; Roffe, 1980). Harbor seals also prey on competitors and other predators of commercially valuable species (Jeffries, 1984; Johnson and Jeffries, 1983; Roffe, 1980). For example, harbor seals prey on large quantities of lamprey, which parasitize salmonids and other valuable species (Jeffries, 1984; Johnson and Jeffries, 1983; Roffe, 1980; Roffe and Mate, 1984).

Bigg et al (1990) observed harbor seals in British Columbia and found that they forage cooperatively in small groups in some areas and tend to forage in shallow waters, close to land. Generally, harbor seals are described as solitary hunters (Bigg et al, 1990). They feed in marine, estuarine, and occasionally fresh water (Barlow et al, 1997; Hill et al, 1997). Much foraging is done at night (Haaker et al, 1984; Johnson and Jeffries, 1983). Kajimura and Loughlin (1988) state that feeding occurs in late afternoon and twilight but tidal changes and diel photoperiods influence feeding times. Small prey is usually consumed beneath the surface, and larger prey is often brought to the surface (Baldrige and Rogers, 1991). The following prey species have been determined for harbor seals:

<u>Category</u>	<u>Scientific Name</u>	<u>Common Name</u>
Molluska	Decapoda	Squid
	Gastropoda	Gastropods
	Gastropoda	Whelks
	Gonatidae	Gonata squid
	<u>Gonatus magister</u>	Gonata squid
	<u>Gonatus</u> sp.	Gonata squid
	<u>Loligo opalescens</u>	Market squid
	<u>Octopus</u> sp.	Octopodid octopus
	Octopoda	Octopi
	Pelecypoda	Clams
	Pelecypoda	Mussels

Crustacea	Pelecypoda	Bivalves/shellfish
	Cirripedia	Barnacles
	Amphipoda	Amphipods
	<u>Callianassa californianus</u>	Ghost shrimp
	<u>Callianassa</u> sp.	Ghost shrimp
	<u>Cancer magister</u>	Dungeness crab
	<u>Cancer</u> sp.	Crab
	<u>Crago</u> sp.	Sand shrimp
	<u>Crangon</u> sp.	Crangon shrimp
	Decapoda	Crabs
	Decapoda	Crayfish
	Decapoda	Shrimp
	<u>Emerita analoga</u>	Brachyuran crustacean
Echinodermata		
Agnatha	Echinoidea	Sea urchin
	<u>Eptatretus</u> sp.	Hagfish
	<u>Polistotrema stouti</u>	Hagfish
	<u>Lampetra ayresi</u>	River lamprey
	<u>Lampetra</u> sp.	Lamprey
	<u>Lampetra tridentata</u>	Pacific lamprey
	(<u>Entosphenus tridentatus</u>)	
	Petromyzonidae	Lampreys
	<u>Polistotrema stouti</u>	Hagfish
Chondrichthys		
	<u>Hydrolagus collieri</u>	Ratfish
	<u>Raja</u> sp.	Skate
	Rajidae	Skates
	<u>Squalus acanthias</u>	Spring dogfish
Osteichthys		
	<u>Acanthogobius flavimanus</u>	Yellowfin goby
	<u>Acipenser transmontanus</u>	White sturgeon
	<u>Allosmerus elongatus</u>	White smelt
	<u>Alosa sapidissima</u>	American shad
	<u>Ammodytes hexapterus</u>	Pacific sand lance
	Ammodyidae	Sand lances
	<u>Amphistichus rhodopterus</u>	Redtail surfperch
	Anarchichadidae	Wolffishes
	<u>Anarrhichthys ocellatus</u>	Wolf-eel
	<u>Anoplopoma fimbria</u>	Sablefish
	Anoplopomatidae	Sablefishes
	<u>Apodichthys flavidus</u>	Penpoint gunnel
	<u>Atheresthes stomias</u>	Arrowtooth flounder
	<u>Atherinops affinis</u>	Topsmelt

<u>Atherinops californiensis</u>	Jacksmelt
<u>B. marginata</u>	
Batrachoididae	Midshipmen
<u>Bathymaster signatus</u>	Searcher
Bathymasteridae	Ronquills
Blenniidae	Blennies
<u>Boreogadus saida</u>	Arctic cod
Bothidae	Left-eye flounders
<u>Brachyistius frenatus</u>	Kelp perch
<u>C. ios</u>	
<u>C. nicholsi</u>	
<u>C. pugetensis</u>	
<u>Chilara taylori</u>	Spotted cusk-eel
<u>Chromis punctipinnis</u>	Blacksmith fish
<u>Citharichthys sordidus</u>	Pacific sanddab
<u>Citharichthys stigmaeus</u>	Speckled sanddab
Clupeidae	Herrings
<u>Clupea pallasii</u>	Pacific herring
Cottidae	Sculpins
<u>Cottus asper</u>	Prickly sculpin
<u>Cymatogaster aggregata</u>	Shiner perch
<u>Cyprinus carpio</u>	Common carp
<u>Damalichthys vacca</u>	Pile perch
<u>Dasycottus setiger</u>	Spinyhead sculpin
<u>Eleginus gracilis</u>	Saffron cod
<u>Embiotica jacksoni</u>	Black perch
<u>Embiotoca lateralis</u>	Striped seaperch
Embiotocidae	Surf perches
Engraulidae	Anchovies
<u>Engraulis mordax</u>	Northern anchovy
<u>Enophris bison</u>	Buffalo sculpin
<u>Enophris</u> sp.	Sculpin
<u>Eopsetta jordani</u>	Petrale sole
Gadidae	Cods
<u>Gadus macrocephalus</u>	Pacific cod
Gasterosteidae	Sticklebacks
<u>Gasterosteus aculeatus</u>	Three-spined stickleback
<u>Genyonemus lineatus</u>	White croaker
<u>Glyptocephalus zachirus</u>	Rex sole
Gobiesocidae	Clingfishes
<u>Hemilepidotus</u> sp.	Irish lord
<u>Hemilepidotus spinosus</u>	Brown Irish lord
Hexagrammidae	Greenlings
<u>Hexagrammos decagrammus</u>	Kelp greenling
<u>Hexagrammos</u> sp.	Greenling

<u>Hexagrammus stelleri</u>	Greenling
<u>Hippoglossoides classodon</u>	Flathead sole
<u>Hyperprosopon argenteum</u>	Walleye surfperch
<u>Hyperprosopon ellipticum</u>	Silver surfperch
<u>Hypomesus pretiosus</u>	Surf smelt
<u>Icelus</u> sp.	Sculpin
<u>Imiastes hexapterus</u>	Sandlance
<u>Isopsetta isolepsis</u>	Butter sole
<u>Lepidogobius lepidus</u>	Bay goby
<u>Lepidopsetta bilineata</u>	Rock sole
<u>Leptocottus armatus</u>	Pacific staghorn sculpin
<u>Limanda aspera</u>	Yellowfin sole
<u>Lumpenus sagitta</u>	Snake prickleback
<u>Lycodes diapterus</u>	Black eelpout
<u>Lycodes</u> sp.	Eelpouts
<u>Lycodopsis pacifica</u>	Blackbelly eelpout
<u>Lyopsetta exilis</u>	Slender sole
<u>Malacocottus kincaidi</u>	Blackfin sculpin
<u>Mallotus villosus</u>	Capelin
<u>Merluccius productus</u>	Pacific hake
<u>Microgadus proximus</u>	Pacific tomcod
<u>Microstomus pacificus</u>	Dover sole
<u>Myoxocephalus polyacanthocephalus</u>	Great sculpin
<u>Myoxocephalus</u> sp.	Sculpin
<u>Nantichthys oculo fasciatus</u>	Sailfin sculpin
<u>Oligocottus rimensis</u>	Prickly sculpin
<u>Oligocottus</u> sp.	Tidepool/Fluffy/ Saddleback
<u>Oncorhynchus gorbuscha</u>	Pink salmon
<u>Oncorhynchus keta</u>	Chum salmon
<u>Oncorhynchus kisutch</u>	Coho salmon
<u>Oncorhynchus nerka</u>	Sockeye salmon/Red salmon
<u>Oncorhynchus</u> sp.	Salmon
<u>Oncorhynchus tshawytscha</u>	Chinook salmon
<u>Ophiodon elongatus</u>	Lingcod
Osmeridae	Smelts
<u>Osmerus mordax dentex</u>	Rainbow smelt
<u>Oxylebius</u> sp.	Greenling
<u>Parophrys vetulus</u>	English sole
<u>Peprilus simmillimus</u>	Pacific pompano
<u>Phanerodon furcatus</u>	White surfperch
Pholidae	Gunnels
<u>Pholis ornata</u>	Saddleback gunnel
<u>Pholis</u> sp.	Gunnel
<u>Platichthys stellatus</u>	Starry flounder
<u>Pleurogrammus monopterygius</u>	Atka mackerel

	<u>Pleuronectidae</u>	Right-eye flounders
	<u>Pleuronichthys decurrens</u>	Curlfin sole
	<u>Pleuronichthys</u> sp.	Sole
	<u>Porichthys notatus</u>	Plainfin midshipman
	<u>Poroclinus rothrocki</u>	Whitebarred prickleback
	<u>Psettichthys melanostictus</u>	Sand sole
	<u>Ptereleotris evides</u>	Arrow goby
	<u>Radulinus asprellus</u>	Slim sculpin
	<u>Rhacochilus</u> sp.	Surfperch
	<u>Rhacochilus toxotes</u>	Rubberlip surfperch
	<u>Rhacochilus vacca</u>	Pile perch
	<u>Salmo gairdneri</u>	Steelhead
	<u>Salmo/Salvelinus</u> sp.	Trout
	<u>Salmonidae</u>	Salmon
	<u>Scorpaenichthys marmoratus</u>	Cabezon
	<u>Scorpaenidae</u>	Rockfishes
	<u>Sebastes melanops</u>	Black rockfish
	<u>Sebastes paucispinus</u>	Bocaccio
	<u>Sebastes</u> sp.	Rockfish
	<u>Sebastolobus alascanus</u>	Short-spine thornyhead
	<u>Spirinchus starksi</u>	Night smelt
	<u>Spirinchus thaleichthys</u>	Longfin smelt
	<u>Steichaeidae</u>	Pricklebacks
	<u>Thaleichthys pacificus</u>	Eulachon
	<u>Theragra chalcogramma</u>	Walleye pollock/Whiting
	<u>Trachurus symmetricus</u>	Jack mackerel
	<u>Trichodon trichodon</u>	Pacific sandfish
	<u>Zalemibus rosaceus</u>	Pink seaperch
	<u>Zoracidae</u>	Eelpouts
Aves		
	<u>Brachyramphus marmoratus</u>	Marbled murrelet
	<u>Ptychoramphus aleuticus</u>	Cassin's auklet
	<u>Synthliboramphus antiquus</u>	Ancient murrelet
Mammalia		
Other		
		Algae
		Milk

Habits

Harbor seals are generally found in small, loosely structured aggregations occupying bays, estuaries, and near-shore rocks (Temte, 1985). They haul out along shorelines and on ice (MMC, 1997). Haul-out behavior of harbor seals may be influenced by time of day, tide height, sea state, weather, daily activity cycles, lunar cycles, and disturbance (Brueggeman et al, 1990; Olesiuk et al, 1990a; Payne et al, 1996; Stewart, 1984; Stewart et al, 1988; Watts, 1992; 1993).

Most haul out during mid-day and under low-tide conditions (Brueggeman et al, 1990; Stewart, 1984; Stewart et al, 1988; Steart and Yochem, 1984; Watts, 1992). Harbor seals spend most of the evening and night in the ocean (NMFS, 1996; Torok, 1994). Individuals may not haul-out each day (Brueggeman et al, 1990; Payne et al, 1996). Harbor seals haul out less frequently and for shorter durations after molting (Mathews and Kelly, 1996).

Harbor seals are usually found in water less than 55 m deep (Perez, 1990) and are usually within 10-20 km of shore (Temte, 1985; MMC, 1997). Mean dive times of radio-tagged harbor seals in south San Francisco Bay, CA were 0.5-3.3 min., and the maximum dive duration was 12.7 min. (Torok, 1994).

Harbor seals have been observed interacting with other species of marine mammals. A subadult male elephant was observed mounting harbor seal pups, both dead and alive, in central CA in 1994 and 1995 (Mortenson and Follis, 1997). Harbor seal pups have also been observed being mounted by a male sea otter (Mortenson and Follis, 1997). Harbor seals have been observed interacting with orcas with no apparent predator response from the orcas (Jacobsen, 1986; Felleman et al, 1991).

Reproduction

(P. v. richardsi):

Breeding season: Mar. to July

Implantation: 2 ½ months after estrous; 3 ½ months after parturition in multiparous females

Pupping season: Pupping clines: start in Mid-Feb. in the south and finish in Sept. in the north; 3 clusters: Mexico to Washington: 4.14 days/degree; Puget Sound, Washington and Vancouver Island, British Columbia; northern British Columbia to Alaska (latitudinal dependence below 50 °N); two linear models for mean pupping dates show 2.78 days/degree and 4.06 days/degree respectively; despite clines, parturition is synchronized within populations; harbor seals in Puget Sound and the Strait of Georgia pup 3 months later than coastal seals at the same latitudes; pupping dates of P. v. stejnegeri (mid-May to late June) fall along the cline for P. v. richardsi

Sexual maturity:

Females: 5 yrs. old

Males: 5-6 yrs. old

Gestation: 9 months

Lactation: 2-6 wks.

Neonate length: 80-89 cm (ave. 84 cm)
Neonate weight: 10.7-21.3 kg (ave. 14.2 kg)

The mating system is somewhat polygynous (Slater and Markowitz, 1983), and females that have copulated with one male are no longer receptive to other males that season (Allen, 1985). Males and females aggregate at different sites, and males may actively exclude other males from female areas (Slater and Markowitz, 1983). Females form nursery herds (Slater and Markowitz, 1983). Site fidelity of females over multiple breeding seasons has not been reported (Allen et al, 1989). Harbor seals copulate almost exclusively in the water (Allen, 1985). Copulations that have been observed on land have shown that receptivity of the female seems to dictate whether copulation occurs (Allen, 1985).

Females give birth on isolated haul-outs or in the water (Temte, 1985). Air and water temperature in late pregnancy may affect time of birth (Temte, 1985). Harbor seals display latitudinal pupping clines, with pupping occurring earlier in the southern part of the range and later in the northern part (Brueggeman et al, 1990; Temte, 1985; Temte et al, 1991). Temte (1985) suggests that specific photoperiod, 13.80 hrs./day, occurring at 68 days post-partum, defines and is causative to the cline in the pupping season. He also speculates that photoperiod may act to trigger implantation.

Mortality and Economics

Natural Mortality

Harbor seals are preyed upon by orcas (Baird and Dill, 1995; 1996; Frost et al, 1992; Heimlich-Boran, 1988; Jacobsen, 1986; Felleman et al, 1991; Hoyt, 1984; Perez, 1990; Matkin and Dahlheim, 1995). Calculations suggest harbor seals in British Columbia have a 50-80% chance of being eaten by an orca by age five (Watts, 1992). Barrett-Lennard et al (1995) found that all of 8 orca stomachs found to contain marine mammal remains contained harbor seal remains. They also found remains of 20 harbor seals in one stomach and remains of 18 harbor seals in another stomach. Matkin and Dahlheim (1995) report that 35% of identified transient predation in their study in northern southeastern AK was on harbor seals.

Another predator of harbor seals is the white shark (Carcharodon carcharias) (Ainley et al, 1980; LeBoeuf et al, 1982; Stewart and Yochem, 1985). Also, coyotes (Canis latrans) prey on harbor seal pups (Calambokidis et al, 1985).

Estimated annual mortality rate in British Columbia was 10.0% from 1966 to 1988 (Olesiuk et al, 1990a). Pup mortality on San Nicholas Island was 7% in 1980, 12% in 1981, and 7% in 1982 (Stewart and Yochem, 1984). In 1977, 50% of pups born at two sites in south Puget Sound, WA were reported dead within a few months of the pupping season (Calambokidis et al, 1984). Calambokidis et al (1984) also reported that unusual incidences of premature births of harbor seals were seen in the early 1970's in San Francisco Bay. Birth defects may affect pup survival. For example, a dead male P. v. stejnegeri pup was observed with a unilateral and total cleft of the primary palate on Kuril Island in 1992 (Suzuki et al, 1992). Also, some pups may have a persistent open ductus arteriosus, which causes adverse effects (Bruce Mate, pers. comm.).

Disease may be linked to neonatal mortality (Calambokidis et al, 1985). At least 32 parasites, 12 bacteria, and 7 viruses have been found in or on harbor seals:

Bacteria

<u>Acinetobacter</u> sp.	Calambokidis et al, 1985
<u>Corynebacterium</u> sp.	Calambokidis et al, 1985
<u>Enterobacter</u> sp.	Calambokidis et al, 1985
<u>Escherichia coli</u>	Calambokidis et al, 1985
<u>Leptospira</u>	Calambokidis et al, 1985
<u>Pasteurella</u> sp.	Calambokidis et al, 1985
Pneumonia	Calambokidis et al, 1985; Suzuki et al, 1992
<u>Proteus</u> sp.	Calambokidis et al, 1985
<u>Pseudomonas</u> sp.	Calambokidis et al, 1985
Septicemia	Calambokidis et al, 1985
a- <u>streptococcus</u> sp.	Calambokidis et al, 1985
b- <u>streptococcus</u> sp.	Calambokidis et al, 1985

Viruses

Canine Distemper Encephalitis	Hofmeister et al, 1988
Influenza-like virus	Calambokidis et al, 1985
Lymphosarcoma (cancer)	Stroud and Sevens, 1980
Phocid herpesvirus 1	Pitcher, 1990
Reo-like virus	Calambokidis et al, 1985
SMSV	Calambokidis et al, 1985

Parasites

Trematoda	
<u>Cryptocotyle lingua</u>	Creplin, 1825; Ransom, 1920
<u>Echinostoma acanthoides</u>	Rudolphi, 1819
<u>Phocitrema fusiforme</u>	Goto and Ozaki, 1930
<u>Pseudamphistomum truncatus</u>	Rudolphi, 1819; Delyamure, 1955
<u>Rossicotrema venustum</u>	Ransom, 1920
<u>Zalophotrema hepaticum</u>	Stunkard and Alvey, 1929; King, 1964
Cestoda	
<u>Diphyllbothrium alascense</u>	Jeffries and Johnson, 1990
<u>Diphyllbothrium cordatum</u>	Leuckart, 1863; Delyamure, 1955
<u>Diphyllbothrium hians</u>	Diesing, 1850; Delyamure, 1955
<u>Diphyllbothrium latum</u>	Linnaeus, 1758; Delyamure, 1955
<u>Diphyllbothrium osmeri</u>	Neiland, 1962
<u>Diphyllbothrium schistochilos</u>	Germanos, 1895; King, 1964
<u>Diplogonoporus tetrapterus</u>	Siebold, 1848; Delyamure, 1955
<u>Schistocephalus solidus</u>	Müller, 1776; Delyamure, 1955
Nematoda	

<u>Anisakis similis</u>	Baird, 1853; King, 1964
<u>Anisakis simplex</u>	Jeffries and Johnson, 1990
<u>Contracaecum osculatum</u>	Rudolphi, 1802; Baylis, 1920; Jeffries and Johnson, 1990
<u>Dipetalonema spirocauda</u>	Conlogue et al, 1980; Eley, 1981; Jeffries and Johnson, 1990
<u>Dirofilaria spirocauda</u>	Leidy, 1858; Taylor et al, 1961
<u>Otostrongylus circumlitus</u>	Railliet, 1899; Bruyn, 1933; Jeffries and Johnson, 1990; Gulland et al, 1998
<u>Parafilaroides gymnurus</u>	Railliet, 1899; Bougherty, 1946
<u>Phocanema decipiens</u>	Jeffries and Johnson, 1990
<u>Phocascaris netsiki</u>	Lyster, 1940
<u>Porrocaecum decipiens</u>	Krabbe, 1878; Stiles and Hassall, 1899
<u>Skjabinaria spirocauda</u>	Leidy, 1858; Lubimov, 1927
Acanthocephala	
<u>Corynosoma falcatum</u>	van Cleave, 1953
<u>Corynosoma magdaleni</u>	Montreuil, 1958
<u>Corynosoma semerme</u>	Forssell, 1904; Neiland, 1962
<u>Corynosoma strumosum</u>	Rudolphi, 1802; Ball, 1930; Jeffries and Johnson, 1990
Acarina	
<u>Halarachne halichoeri</u>	Fay and Furman, 1982
<u>Halarachne miroungae</u>	Ferris, 1925; Conlogue et al, 1980
Anoplura	
<u>Echinophthirius horridus</u>	Olfers, 1816; Ferris, 1934; Conlogue et al, 1980; Jeffries and Johnson, 1990

Anthropogenic Interactions and Mortality

Pollution is a serious threat to pinniped populations (Mate, 1982). PCB's, DDT, DDE, and heavy metals have been detected in harbor seal tissues (Addison et al, 1996; Calambokidis et al, 1984; Miles et al, 1992). Dioxins, furans, non-ortho and mono ortho-substituted PCB's were detected in the blubber of harbor seals shot in the Strait of Georgia and Quatsino Sound, British Columbia in 1991 and 1992 (Addison et al, 1996). All samples of blubber from 73 harbor seals in WA in the early 1980's contained detectible PCB and DDE (Calambokidis et al, 1984). In Grays Harbor, WA, PCB levels were found to be highest in adults, second highest in subadults, and lowest in pups (Calambokidis et al, 1984). In Puget Sound, WA, PCB and DDE levels were found to be higher in adult males than in females but higher in subadult females than in males (Calambokidis et al, 1984). PCB and DDE levels were also found to be lower in pregnant and lactating females than in non-pregnant and non-lactating females (Calambokidis et al, 1984). Harbor seals in WA have also been found to carry a variety of heavy metals (Calambokidis et al, 1984). Tissues of harbor seals shot in the Gulf of AK from 1976 to 1978 were found to contain

PCB's, DDT, DDE, oxychlordane, and heavy metals (Miles et al, 1992). Cd levels were significantly higher in males than in females (Miles et al, 1992). Hg, Pb, Se, and As levels were similar in males and females (Miles et al, 1992). PCB levels were similar between males and females; DDT levels were significantly higher in males than in females; and DDE and oxychlordane levels tended to be higher in males than in females (Miles et al, 1992). Hg, Pb, and Cd levels in the liver correlated with age in male harbor seals (Miles et al, 1992). Cd levels in the liver and Cd, Hg, and Zn levels in the kidneys have also been shown to increase with age in harbor seals from Hokkaido, Japan (Tohyama et al, 1986).

Polution may contribute to premature birth, birth defects, pelage anomalies, umbilical lesions, and high pup mortality in harbor seals (Calambokidis et al, 1984; 1985). It may be responsible for the high incidence of premature births observed in San Francisco Bay in the early 1970's and for the unusually high level of pup mortality in south Puget Sound in 1977 (Calambokidis et al, 1984).

The Exxon Valdez oil spill in Prince William Sound in March, 1989 may have affected harbor seal populations (Loughlin et al, 1996; Regelin et al, 1996). A minimum of 302 harbor seals were missing from Prince William Sound after the spill (Loughlin et al, 1996). The oil from the spill caused skin irritation, conjunctivitis, and liver lesions (Loughlin et al, 1996). Schneider (1993) states that 345 harbor seals died immediately after the spill and 80% of adult harbor seals in the path of the oil slick became oiled, along with many pups.

Humans also cause mortality by incidental and direct kills. Prior to protection, harbor seal populations along the west coast of North America were greatly reduced by commercial hunting (Barlow et al, 1997). 17,133 were killed in WA by bounty hunters from 1943 to 1960 (Barlow et al, 1997), and 146,000 were killed in OR by a state-hired seal hunter and bounty hunters from 1914 to 1972 (Harvey, 1987). 18,000 harbor seals were harvested on Tugidak Island from the 1960's to 1972 (Pitcher, 1990).

Subsistence harvest is allowed under the MMPA (Barlow et al, 1997; Hill et al, 1997; Regelin et al, 1996). Several northwestern Indian tribes are developing or have developed subsistence harvests of harbor seals in the OR/WA coastal and WA inland stocks (Barlow et al, 1997). Mean annual subsistence harvest was 1,688/yr., 850/yr., and 178/yr. from 1993 to 1995 from the southeastern AK, Gulf of AK, and Bering Sea stocks respectively (Hill et al, 1997). Other non-fishery sources of anthropogenic mortality include boat collisions, entrainment in power plants, and shooting (Barlow et al, 1997). Entanglement may affect harbor seals (Pitcher, 1990). Also, harassment and disturbance may lower reproductive rate of females (Harvey, 1987; Slater and Markowitz, 1983).

Fisheries cause mortality directly through incidental and illegal kills and indirectly through competition with harbor seals for their prey species. Decreased prey availability due to fisheries may have contributed to population declines in the Bering Sea and Gulf of AK (Hobson et al, 1997; Swartzman and Hogman, 1991). Harbor seal population declines in AK began at the same time commercial walleye pollock (*Theragra chalcogramma*) catch began to decline (Springer, 1992). Fisheries that take harbor seals incidentally and annual mean mortality are as follows (Hill et al, 1997; Barlow et al, 1997; U.S. Dept. of Comm., 1996; Wada et al, 1991):

CA stock

CA/OR thresher shark/swordfish drift gillnet	0
CA angel shark/halibut and other species large mesh (>3.5") set gillnet	228

CA, OR, and WA salmon troll	7.33
CA herring purse seine	0
CA anchovy, mackerel, and tuna purse seine	0.67
WA, OR, CA groundfish trawl	0
CA squid purse seine	0
Unknown net and hook	6
OR/WA coastal stock	
Northern WA marine set gillnet	4.3 (CV=0.68)
WA/OR lower Columbia River drift gillnet	N/A
WA Grays Harbor salmon drift gillnet	6.7 (CV=0.50)
WA Willapa Bay drift gillnet	0
WA Willapa Bay drift gillnet (logbooks)	>3.5
WA/OR/CA groundfish trawl (Pacific whiting)	0
WA/OR salmon net pens	>0.5
WA inland stock	
Northern WA marine set gillnet	11 (CV=0.53)
WA Puget Sound Region salmon set/drift gillnet	unknown
WA Puget Sound Region salmon set/drift gillnet (logbooks)	N/A
Puget Sound non-treaty salmon gillnet	N/A
Puget Sound non-treaty chum salmon gillnet	10
Puget Sound treaty chum salmon gillnet	0
Puget Sound treaty chum and sockeye salmon gillnet	0
Southeastern AK stock	
Gulf of AK groundfish longline	4 (CV=1.0)
Southeastern AK salmon drift gillnet	3.75
Yakutat salmon set gillnet	27.5
Gulf of AK stock	
Gulf of AK groundfish trawl	1 (CV=0.63)
Gulf of AK finfish pot	0.2 (CV=1.0)
Prince William Sound salmon drift gillnet	24 (CV=0.50)
Cook Inlet salmon set gillnet	1.75
Cook Inlet drift gillnet	
Prince William Sound set gillnet	0.25
Kodiak salmon set gillnet	0.75
AK salmon purse seine	0.5
AK peninsula/Aleutian Island salmon drift gillnet	7.0
Bering Sea stock	
AK Peninsula/Aleutians salmon drift gillnet	
Bering Sea/Aleutian Islands groundfish trawl	1.8 (CV=0.54)
Bering Sea/Aleutian Islands longline	0.6 (CV=1.0)
Bering Sea/Aleutian Islands finfish pot	1.2 (CV=0.81)
Bristol Bay salmon drift gillnet	26.25
Bristol Bay salmon set gillnet	0.5
Western Pacific	
Nemuro peninsula salmon trap net	163

Minimum annual incidental fisheries mortality of harbor seals in each stock is as follows (Hill et al, 1997; Barlow et al, 1997):

CA	234/yr. (ave. 1994-1995 = 228/yr.)
OR/WA coastal	15/yr.
WA inland	36/yr.
Southeastern AK	36/yr.
Gulf of AK	36/yr.
Bering Sea	31/yr.

Annual total human caused mortality of harbor seals in each stock is as follows (Hill et al, 1997; Barlow et al, 1997):

CA	234/yr.
OR/WA coastal	15/yr.
WA inland	36/yr.
Southeastern AK	1,704/yr.
Gulf of AK	886/yr.
Bering Sea	209/yr.

Annual total human caused mortality exceeds 10% of PBR for the CA stock (168), southeastern AK stock (211), Gulf of AK stock (673), and Bering Sea stock (38) and is considered significant for these stocks (Barlow et al, 1997; Hill et al, 1997). Annual total human caused mortality does not exceed 10% of PBR for the OR/WA coastal stock (154) and WA inland stock (92) and is considered insignificant for these stocks (Barlow et al, 1997). No PBR estimates have been determined for western Pacific harbor seals, but Wada et al (1991) state that the population has decreased since the 1960's due to strong hunting pressure and incidental catch mortality. The amount of illegal kill of harbor seals is unknown (Hill et al, 1997).

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Review of Steller (Northern) Sea Lions (Eumetopias jubatus) 1978-1997

Population, Abundance, and Distribution

Range:

Hokkaido, Japan to Kuril Islands and Sea of Okhotsk through the Aleutian Islands and south Bering Sea, along Alaska's southern coast, and south to California

Breeding Range:

Chernyye Brat'ya ostrova in the Kuril Islands to Año Nuevo Island, central California (37°06' N) (two Stellers were born on San Miguel Island in 1982, but none have been seen there since 1984)

Preferred Habitat:

Rocky islands with isolated sloping areas, flat rocky areas with boulders, cobbles, and coarse, sandy beaches with free access to the sea

Stellers in the U.S. are separated into two stocks: west of Cape Suckling (144 °W) (U.S. Western stock) and east of Cape Suckling (U.S. Eastern stock). Phylogenetic studies support this separation (Small and DeMaster, 1995; Bickham et al, 1996). The population of the western stock is decreasing at a rapid rate and the population of the eastern stock is increasing slightly or remaining stable (Small and DeMaster, 1995; Hill et al, 1997). Boltnev and Mathisen (1996) suggested that increases in the population of Stellers in the Commander Islands up to the mid-1970's may have been due to immigration of Stellers in search of fat fish, such as herring, which were decreasing in the east. Immigration may be playing a part in the current decline of Stellers in the west, but Merrick et al (1987) state that no increases in the east are offsetting the rapid decline in the west. Stellers were listed as a threatened species under the U.S. ESA effective December 4, 1990 (Payne et al, 1996a; Fritz et al, 1995). The western stock was reclassified as endangered effective May 5, 1997 (Rogers, 1997). The N_{min} and PBR of the western stock are 42,536 and 383 respectively; for the eastern stock, N_{min} is 37,166 and PBR is 1,672 (Hill et al, 1997). The range-wide population was estimated to be 52,200 in 1994 (Regelin and Calkins, 1995). Hill et al (1997) estimated the western stock population to be 43,200 and the eastern stock population to be 23,900 (37,746 if British Columbia is included) in 1994. MMC's (1996) estimates were 33,600 and 18,600 respectively. NOAA's (1995) estimate for the western stock was 24,104 in 1994.

The range-wide population of Stellers has decreased considerably, from hundreds of thousands to tens of thousands from the 1950's and 1960's to the 1990's. At one time, Stellers were the most abundant sea lion species in North America (Braham et al, 1980). Regelin and Calkins' (1995) estimated range-wide population of 52,200 for 1994 is about 1/9 the estimated population of the late 1950's (240,000-300,00 (Mate, 1982)). Steller populations declined in Russia, the Aleutian Islands, the Bering Sea, Alaska, the Gulf of Alaska, and California from the 1950's and 1960's to the 1990's (Loughlin et al, 1993a; Lowry et al, 1989; Mate, 1982, NMFS, 1991). Severe declines occurred in the eastern Aleutian Islands, resulting in about 1/10 the population of the 1960's by 1994, but small increases were noted in the population in the mid-1990's (6.6% 1994-96) (MMC, 1997). MMC (1997) stated that populations at some major

rookeries and haul-outs in the western Gulf of Alaska and the eastern Aleutian Islands and Russia declined more than 90% from the 1960's to 1994. Springer (1992) states that Steller populations declined by 50-80% at rookeries throughout the Bering Sea and Gulf of Alaska from the early 1970's to the early 1990's. Counts in the region from Kenai peninsula to Kiska Island, AK declined from 140,000 in the late 1950's to 68,000 in 1985 to 25,000 in 1989 and 1990 (Alverson, 1992). Loughlin et al (1990) suggest that the Steller decline began in the eastern Aleutian Islands in the mid-1970's and spread east to Kodiak Island in the late 1970's and early 1980's and west to the central and western Aleutian Islands in the early and mid-1980's. Counts of adults and juveniles decreased 21% and pup counts decreased 28% for the western stock 1990-94; counts of juveniles and adults increased 17% for the eastern stock 1990-94 (MMC, 1997). Population decreased 34.7 % in the western stock from 1989 to 1994 (Hill et al, 1997).

Counts of adults and juveniles in CA decreased 19% 1990-94, and the range of the eastern stock may be receding northward (MMC, 1997). Steller populations have increased or remained stable in Washington and Oregon from the 1950's and 1960's to the 1990's (NMFS, 1991; NOAA, 1995; Hill et al, 1997). Populations in British Columbia decreased from the early 1900's to the 1970's but increased in the 1980's and early 1990's (NOAA, 1995). Bigg (1984) reports the summer population is 6,000 in British Columbia.

There appears to be some debate as to whether or not Stellers migrate. Stellers display seasonal abundances and local movements of considerable distances. If they do not migrate, they do disperse widely (Loughlin et al, 1993a; Loughlin and Nelson, 1986; Kajimura and Loughlin, 1988). The males generally move northward after breeding (Bigg, 1988; Brueggeman et al, 1990; Mate, 1982). Females also seem to move northward slightly later than males; some females in the Pribilof Islands may move southward (Mate, 1982). Hamanaka et al (1982) mention that Stellers may migrate seasonally along the southern coast of Hokkaido from the Kuril Islands and spend several months in southern Hokkaido waters. The bulk of the Steller breeding population is in the Aleutian Islands. Females may copulate at different colonies but deliver calves only at specific colonies (Kastelein and Weltz, 1990). Payne et al (1996a) also noted that females tend to have high fidelity to their native rookeries.

Food and Feeding

Stellers are opportunistic feeders and may have different prey preferences depending upon location, season, and perhaps even time of day (Antonelis and Fiscus, 1980; Fritz et al, 1995; Merrick, 1995; Alverson, 1992; Jones, 1981). Stellers take advantage of seasonal and geographical abundances of prey species, and some populations may follow certain anadromous prey species. This includes lamprey as well as salmonids (Brueggeman et al, 1990; Bigg et al, 1990). Stellers have also been observed eating other pinnipeds (Byrnes and Hood, 1994; Lander and Kajimura, 1982; National Advisory Commission on Oceans and Atmosphere, 1985) and birds (O'Daniel and Schneeweis, 1992). Primarily, Stellers feed on semi-demersal schooling fish (Fritz et al, 1995). They also feed heavily on commercially valuable species such as walleye pollock (Lowry, 1982; Lowry et al, 1989; Merrick, 1995). Springer (1992) estimates that Stellers consume 120,000 tons of walleye pollock/yr. (33% of their diet) in the Bering Sea. However, Boltnev and Mathisen (1996) state that Stellers fed from fish processing boats in the western Bering Sea in the 1960's refused walleye pollock but never refused herring. They suggest that Stellers shifted their diet to walleye pollock as herring populations decreased from

fishing pressure. Merrick (1995) also suggests that Steller prey preferences shifted to increased consumption of walleye pollock (65% increased consumption by juveniles from 1975-78 to 1985-86) due to declines in alternative prey. He also found that diets tend to be dominated by one or two taxa in each foraging area. He indicates that decreases in walleye pollock populations may have caused an increase in the foraging effort of Stellers, thereby decreasing survival of Stellers. Alverson (1992) suggests that walleye pollock may have less nutritional value than previously preferred prey species, perhaps causing health declines that have contributed to population declines.

Stellers forage mostly over the continental shelf, usually at between 4 and 50 m deep (Kajimura and Loughlin, 1988; Loughlin 1993b). They generally do not forage any deeper than 200 m (Loughlin et al, 1993). Adult females generally depart from rookeries to feed in the evening and return early in the morning (Merrick et al, 1988; Merrick, 1995). Kajimura and Loughlin (1988) also state that Stellers feed mostly at night. Merrick (1995) found that overall average foraging effort for adult females in AK was 3.24 hrs/day in summer and 4.30 hrs./day in winter. He found that overall average foraging effort of young-of-the-year in AK was 1.48 hrs./day. He also found that home range for foraging increases in the winter. The following prey species have been determined for Stellers:

<u>Category</u>	<u>Scientific Name</u>	<u>Common Name</u>
Molluska	<u>Chiroteuthis</u> sp.	Whiplash squid
	Decapoda	Crabs
	Decapoda	Squid
	Gastropoda	Snails
	Gonatidae	Gonata squid
	<u>Gonatus magister</u>	Gonata squid
	<u>Loligo opalescens</u>	Market squid
	Octopoda	Octopi
	<u>Octopus</u> sp.	Octopodid octopus
	<u>Onychoteuthis</u> sp.	Hooked squid
	<u>Paroctopus dofleini dofleini</u>	Giant octopus
	Pelecypoda	Clams
	Pelecypoda	Mussels
	Pelecypoda	Shellfishes
Crustacea	Decapoda	Crabs
	Decapoda	Shrimps
	<u>Chionoecetes</u> sp.	Snow crab
	<u>Hyas</u> sp.	Spider crab
Agnatha	<u>Lampetra</u> sp.	Lamprey
	<u>Lampetra tridentata</u>	Pacific lamprey
	<u>(Entosphenus tridentatus)</u>	
Chondrichthys	Petromyzontidae	Lamprey

Osteichthys

Apristurus brunneus
Hydrolagus coliei
Raja sp.
 Rajidae
 Scylinorhinidae
Squalus acanthias
Squalus suckleyi

Agonidae
Agonus acipenserinus
Ammodytes hexapterus
Ammodytes personatus
Anoplopoma fimbria
Aptocyclus ventricosus
Atheresthes sp.
Atheresthes stomias
Careproctus melanurus
Careproctus sp.
Chilara sp.
Chilara taylori
Citharichthys sp.
Clupea pallasii
 Cottidae
Eleginus gracilis
 Embiotocidae
Engraulis mordax
Engraulis sp.
Eopsetta sp.
 Gadidae
Gadus macrocephalus
Glyptocephalus sp.
Gymnocanthus sp.
Hemilepidotus hemilepidotus
Hemilepidotus jordani
Hemilepidotus sp.
 Hexagrammidae
Hexagrammos sp.
Hippoglossus stenolepis
Limanda aspera
Lycodes pacificus
Lycodes sp.
Lyopsetta sp.

Sharks
 Brown cat shark
 Ratfish
 Skate
 Skates
 Cat sharks
 Spiny dogfish
 Dogfish

Graycod
 Lumpfish
 Poachers
 Sturgeon poacher
 Sand lance
 Sand lance
 Sablefish
 Smooth lumpsucker
 Arrowtooth or Kamchatka flounder
 Arrowtooth flounder
 Blacktail snailfish
 Snailfish
 Cusk eel
 Spotted cusk eel
 Flounder
 Pacific herring
 Sculpins
 Saffron cod
 Surfperch
 Northern anchovy
 Anchovy
 Sole
 Cods
 Pacific cod
 Flounder
 Sculpins
 Red Irish lord
 Yellow Irish lord
 Irish lord
 Greenlings
 Greenling
 Pacific halibut
 Yellowfin sole
 Blackbelly eelpout
 Eelpout
 Sole

	<u>Mallotus villosus</u>	Capelin
	<u>Merluccius productus</u>	Pacific hake
	<u>Microgadus proximus</u>	Pacific tomcod
	<u>Microstomus</u> sp.	Sole
	<u>Oncorhynchus keta</u>	Chum salmon
	<u>Oncorhynchus kisutch</u>	Coho salmon
	<u>Oncorhynchus</u> sp.	Salmon
	<u>Ophiodon elongatus</u>	Lingcod
	Osmeridae	Smelts
	<u>Parophrys</u> sp.	Sole
	<u>Platichthys stellatus</u>	Starry flounder
	<u>Pleurogrammus monopterygius</u>	Atka mackerel
	<u>Pleuronectes bilineatus</u>	Rock sole
	Pleuronectidae	Flatfishes
	<u>Pleuronichthys</u> sp.	Turbot
	<u>Porichthys notatus</u>	Plainfin midshipman
	<u>Porichthys</u> sp.	Midshipman
	<u>Sardinops sagax</u>	Sardine
	Salmonidae	Salmon
	Scorpaenidae	Rockfishes
	<u>Sebastes</u> sp.	Rockfish
	<u>Spirinchus</u> sp.	Smelt
	Stichaeidae	Pricklebacks
	<u>Thaleichthys pacificus</u>	Eulachon
	<u>Theragra chalcogramma</u>	Walleye pollock/Whiting
	<u>Trachurus symmetricus</u>	Jack mackerel
	<u>Trichodon trichodon</u>	Pacific sandfish
Aves		
	<u>Larus glaucescens</u>	Glaucous winged gull
Mammalia		
	<u>Callorhinus ursinus</u>	Northern fur seal
	<u>Phoca vitulina richardsi</u>	Harbor seal
	<u>Zalophus californianus</u>	California sea lion
Other		
		Kelp
		Milk
		Stones/gravel
		Sand

Habits

Haul-out behavior of Stellers may be location, age, and sex dependent and may be affected by breeding or foraging patters (Katstelein and Weltz, 1991). Haul-out patterns are also dependent upon tides (Kastelein and Weltz, 1991).

Adult female Stellers in AK and the Aleutian Islands were found to spend about 1/3 of their time at sea (Loughlin et al, 1993). Merrick (1995) found that adult females in AK dove to an average depth of 24.4 m, with a maximum of more than 250 m, in winter and to an average depth of 21.0 m, with a maximum of more than 150 m, in summer. He also found that young-of-the-year in AK dove to an average depth of 8.5 m, with a maximum of 72 m. He determined that average dive depth was 18.0 m, and 59.4% of dives were less than 10 m. Average dive duration was 1.45 min, and young-of-the-year made shorter dives than adult females (Merrick, 1995). Average trip distance was 17.1 km, and average home range for adult females in summer was 639 km², in winter was 6,569 km², and for young-of-the-year was 7,558 km² (Merrick, 1995). Loughlin et al (1993) also found that average dive depth for pups was 7.8 m and for adult females, was 23.4 m; pups did not dive deeper than 50 m, but adults dove close to 200 m. Also, they found that average dive duration for pups was 2.1 min. and for adults, was 2.5 min. Dives usually lasted about 1-6 min, with a maximum of 11 min. recorded (Loughlin et al, 1993). The average amount of time spent at sea per trip was 10.2 hrs. for pups and 39.9 hrs. for adults (Loughlin et al, 1993).

Stellers are not considered to occur in deep pelagic waters, but a few large groups were seen farther than 18-30 km offshore in the 1950's and 1960's (Kajimura and Loughlin, 1988). Large groups are now more frequently seen far out to sea associated with foreign and domestic commercial fishing vessels (Kajimura and Loughlin, 1988).

Stellers have been observed interacting with orcas with no apparent predator response from the orcas (Jacobsen, 1986). Stellers and northern fur seals fight on shared rookeries on Mednyi Island (Chenokov, 1984).

Reproduction

Breeding season: Late May to July (uniform throughout range (Bigg, 1985; 1988; Merrick et al, 1995))

Implantation: Late Sept. to early Oct.

Pupping season: **Late May to mid-July**

Sexual maturity:

Females: 4-5 yrs. old
Males: 5-7 yrs. old (obtain territories at 9-13 yrs. old)

Lactation: 11-12 months

Stellers are polygynous, and males establish territories with harems of females with whom to mate (Brueggeman et al, 1990; Kajimura and Loughlin, 1988). Females have strong site fidelity to their native rookeries (Fritz et al, 1995; Kastelein and Weltz, 1990; Payne et al, 1996a). Reproductive senescence may occur at about 20 yrs. old (Lee et al, 1996). One instance of twinning in Stellers was reported in July, 1968 on Año Nuevo Island, CA (Spotte, 1982).

60-65% of adult females in AK give birth each year, although at least 90% implant the preceding November (Merrick, 1995). The first trip to sea for the female is usually 11-14 days

after giving birth (Merrick, 1995). Pups first enter the water 2-4 weeks after birth (Merrick, 1995).

On one occasion, a male Steller killed a female during a mating attempt on Año Nuevo Island in 1968 (LeBoeuf and Mesnick, 1991). The female resisted being mounted and kept turning to bite the male. Her neck was broken as the male attempted to pin her down. When high surf washed the body to sea, the male retrieved the corpse and copulated with it.

Mortality and Economics

Natural Mortality

Lidicker et al (1981) found extremely low genetic variability and genetic heterozygosity in Stellers in the northeastern Gulf of Alaska. This may be a factor contributing to the Steller population decline. However, Bickham et al (1996) stated that, according to their analyses, Stellers have not undergone a genetic bottleneck. Steller decline, particularly in the Channel Islands, may be partly due to competition from other pinnipeds, such as California sea lions, northern fur seals, and elephant seals (Payne et al, 1996a; NOAA, 1995).

Also, fecundity and/or juvenile survival may be lower than in years past (Strick et al, 1997), as evidenced by a decrease in the mean age of females (York, 1994). Castellini et al (1993) found that Stellers on Marmot Island, Alaska were leaner and that females were smaller and more anemic in 1985-86 than in the 1970's, and Calkins et al (1998) report that female Stellers in the Gulf of AK were shorter, thinner, and had lower mass in 1985-1986 than in 1975-1978. Regelin and Calkins (1995) found that Stellers in the Kodiak area were significantly smaller at age in the 1980's than in the 1970's. Blood analyses of Stellers in AK have revealed effects of stress (Bishop and Morado, 1995). Also, Steller pup survival decreased from the 1970's to the 1980's (MMC, 1997). However, Merrick et al (1988) state that declines in population are probably not due to declining reproductive rates or reduced pup survival. Rather, they suggest the declines are a result of a factor or agent that decreases juvenile survival or has no sex specific effect on adult survival. Merrick (1995) states that pups in decline areas are larger than those in southeastern AK, where population is stable or increasing. Alverson (1992) suggests that walleye pollock, which became a more predominant prey species in Stellers' diets in the 1980's, may have less nutritional value than previously preferred prey species, causing lowered health and fitness levels. Environmental perturbations may also be affecting populations. Counting errors and changes in survey techniques do not explain the declines, although, changes in haul-out behavior may have affected counts. A population viability analysis from 1993 based on data from 1985 to 1992 indicates a high probability of Steller extinction within 60 to 100 years (MMC, 1997).

Sharks and orcas prey on Stellers (NOAA, 1995; Hall, 1986; Barrett-Lennard et al, 1995; Frost et al, 1992; Baird and Dill, 1995; Baird and Stacey, 1989; Jacobsen, 1986; Hoyt, 1984; Sobolevsky and Mathisen, 1996; Kajimura and Loughlin, 1988; Perez, 1990; Matkin and Dahlheim, 1995). One orca's stomach was found to contain 15 Steller flipper tags (Barrett-Lennard et al, 1995). Matkin and Dahlheim (1995) found that 15% of identified transient predation in their study in northern southeastern AK was on Stellers.

Male Stellers may kill females in mating attempts (LeBoeuf and Mesnick, 1991). On one occasion, a male Steller killed a female during a mating attempt on Año Nuevo Island in 1968 (LeBoeuf and Mesnick, 1991).

At least 23 parasites, 2 bacteria, and 6 viruses have been found in or on Stellers:

Bacteria

Chlamydia

Chlamydia psittaci

Leptospira

Merrick et al, 1987

Pitcher, 1990

Merrick et al, 1987; Pitcher, 1990

Viruses

SMSV 2, 5, 6, 7, 13

Smith et al, 1986; Barlough et al, 1988; Berry et al, 1990

Tillamook calicivirus

Smith et al, 1986

Parasites

Cestoda

Delyamure and Krotov, 1955

Diphyllbothrium pacificum

Diplogonopories violettae

Diplogonoporus fasciatus

Adenocephalus pacificus

Pyramicocephalus phocarum

Anophryocephalus ochotensis

Nybelin, 1931; Margolis, 1956

Yurakhno, 1986

Krabbe, 1865; Stunkard, 1948

Nybelin, 1931; King, 1964

Fabricius, 1780; Monticelli, 1890

Nematoda

Anisakis simplex

Anisakis tridentata

Contracaecum ogmorhinic

Contracaecum osculatum

Porrocaecum decipiens

Parafilaroides nanus

Parafilaroides prolificus

Anisakis similis

Uncinaria hamiltoni

Uncinaria lucasi

Rudolphi, 1809; Baylis, 1920

Kreis, 1938

D'Amelio et al, 1994

Rudolphi, 1802; Baylin, 1920

Krabbe, 1878; Baylis, 1916

Dougherty and Herman, 1947

Dougherty and Herman, 1947

Baird, 1853; Baylis, 1920

Baylis, 1933

Loughlin and Merrick, 1989

Acanthocephala

Bolbosoma bobrovi

Corynosoma strumosum

Corynosoma villosum

Krotov and Delyamure, 1952

Rudolphi, 1802; King, 1964

van Cleave, 1953

Acarina

Orthohalarachne attenuata

Orthohalarachne diminuata

Orthohalarachne zalophi (zoo)

Banks, 1910; Newell, 1947

Fay and Furman, 1982

Oudemans, 1916; Newell, 1947

Anoplura

It is hard to determine how significant diseases are as a cause of mortality. Not all diseases are fatal, and some bacteria and parasites may be normally present in Stellers without causing any serious health problems. Diseases may affect fecundity and other aspects of reproductive success. Populations stressed for other reasons may be more susceptible to negative effects from disease.

Anthropogenic Interactions and Mortality

Pollution is a serious threat to pinniped populations (Mate, 1982). Organochlorines, PCB's, DDE, DDT, and heavy metals are commonly found in Steller tissue and blood samples (Loughlin et al, 1996; Kim et al, 1996a; 1996b; Tanabe et al, 1994). OC levels were found to be higher in males than in females sampled in AK from 1976-78 (Loughlin et al, 1996). Loughlin et al (1996) also found that OC levels increased with age in males and increased with age in females to age five, then decreased to age 25. Females may lose 80% of their body burden of PCBs and 79% of their body burden of DDT to their first pup through gestation and lactation (Loughlin et al, 1996). In Loughlin et al's (1996) sample, PCB>DDT>CHL>HCH. Tanabe et al (1994) also found PCB>DDT>HCH. BT has been found in Stellers from Hokkaido, Japan (Kim et al, 1996a; 1996b). Kim et al (1996b) found in a 1994-95 sample that BT is excreted through shedding hair, among other methods. No sex or age differences were found in levels of BT, so transfer through gestation and lactation probably do not occur in Stellers (Kim et al, 1996b). Mate (1982) states, that probably the greatest single threat to Stellers is one shared by all marine mammals: increased concentrations of man-made toxicants.

Humans also cause mortality by incidental and direct kills. A small number of Stellers have been collected annually for research and display (about 12 Stellers/yr. according to Mate (1982)). Before the MMPA of 1972 and the Canadian Federal Fisheries Act of 1970, hunting was a very significant source of mortality for Stellers. At least 45,943 Stellers were killed in British Columbia between 1912 and 1968 (Bigg, 1984).

Stellers were commercially harvested in the Gulf of AK and eastern Aleutian Islands from 1959 to 1972 (Alverson, 1992). 45,788 pups were harvested from 1963 to 1972 (Alverson, 1992).

Subsistence harvest is allowed under the MMPA (Hill et al, 1997). Ninety-nine percent of the 1993-95 subsistence harvest was from the western stock (Hill et al, 1997). The estimated harvest of Stellers from the western stock was 339 in 1995 (MMC, 1997). The mean annual harvest from 1993-95 was 412/yr. Since the PBR of the western stock was lowered from 776 (Small and DeMaster, 1995) to 383 (Hill et al, 1997) in 1996, this level of mortality exceeds the PBR and is considered significant. The mean annual harvest of the eastern stock from 1993-95 was 2/yr., but harvest in British Columbia is unknown (Hill et al, 1997). Other non-fishery, human activities that affect populations are oil and gas industry activities (e.g. exploration, lease sales, contingency plans, spills), hunting, tourism, dredging, logging, research activity, and disturbance (Pascual and Adkison, 1994; MMC, 1997; Castellini et al, 1993).

Fisheries cause mortality directly through incidental and illegal kills and indirectly through competition with Stellers for their prey species. Changes in walleye pollock and other

prey availability due to competition with the fishery may be a significant factor in Steller decline (Shima et al, 1994; Alverson, 1992). Steller consumption of walleye pollock decreased from 1976 to 1986 as the fishery catch increased (Lowry et al, 1989). Also, Steller dependence of walleye pollock may be a result of previous fishing pressure on their preferred prey species, herring (Boltnev and Mathisen, 1996). Ninety percent of marine mammals incidentally taken by foreign vessel fisheries in the north Pacific and Bering Sea from 1978 to 1981 were Stellers (Loughlin et al, 1983). Over 20,000 Stellers were killed in the Gulf of AK, Bering Sea, and Aleutian Islands from 1966 to 1988 in foreign and joint venture trawl fisheries (MMC, 1997). Stellers are also shot by fisherpeople protecting their gear and their catch (Alverson, 1992). Direct fishery related mortality (including commercial and subsistence harvest) ranged from 1.1% to 4.9% of the population from 1960 to 1990 (Alverson, 1992). Steller population declines from 1985 to 1989 far exceed direct losses from fishing related activities (Alverson, 1992). Fisheries that take Stellers incidentally and annual mean mortality are as follows (Hill et al, 1997; MMC, 1997; Alverson, 1992):

Western U.S. stock

Bering Sea/Aleutian Island groundfish trawl	12
Gulf of AK groundfish trawl	1.2
Bering Sea/Aleutian Islands groundfish longline	0.2
Gulf of AK groundfish longline	1.0
Prince William Sound salmon drift gillnet	14.5
AK peninsula/Aleutian Islands salmon set gillnet	0.75
Cook Inlet salmon drift gillnet	0.5
Bristol Bay salmon drift gillnet	3.5
Prince William Sound set gillnet	0.5
AK miscellaneous finfish set gillnet	0.25
AK halibut longline	0.2
Unimak salmon driftnet	
Russian salmon coastal trap-net	
Japanese, U.S. and Soviet herring gillnet and seine	
Foreign and U.S. crab tanglenet, trawl, and pot	
Foreign and U.S. shrimp	

Eastern U.S. stock

CA/OR thresher shark and swordfish drift gillnet	2.8
WA/OR/CA groundfish trawl	0.2
British Columbia aquaculture predator control program	9.8
Southeast AK salmon drift gillnet	1.25
Foreign and U.S. crab tanglenet, trawl, and pot	
Foreign and U.S. shrimp	

Incidental mortality of Stellers in the western stock is about 41/yr. according to MMC (1997) and about 35/yr. according to Hill et al (1997). This is close to or exceeds 10% of the PBR (38) and is likely significant (Hill et al, 1997). Incidental mortality of Stellers in the eastern stock is about 14/yr. (Hill et al, 1997). This is well below 10% of the PBR (167) and is considered insignificant (Hill et al, 1997).

Estimated annual total human-caused mortality and serious injury of Stellers in the western stock is 447/yr. (Hill et al, 1997); this level of mortality exceeds 10% of the PBR (38) (Hill et al, 1997) and is considered significant. For the eastern stock, the estimate was 17/yr.; this level of mortality does not exceed 10% of the PBR (167) (Hill et al, 1997) and is considered insignificant. The amount of illegal kill is unknown (Hill et al, 1997; Alverson, 1992).

Stellers may negatively impact some fisheries. Stellers prey on commercially valuable species such as walleye pollock (Loughlin and Merrick, 1989; Smith, 1995; Springer, 1992; Lowry, 1982) and salmon (Roffe, 1980; Roffe and Mate, 1984; Fiscus, 1980). However, walleye pollock did not become a major prey item of Stellers until the 1980's, possibly due to reduction of preferred prey species by commercial fisheries (Alverson, 1992; Boltnev and Mathisen, 1996). Roffe and Mate (1984) state that consumption of salmon and steelhead by pinnipeds is not a threat to the fish stocks they observed in the Rogue River, OR. Also, Stellers may benefit salmon by preying heavily on lamprey, a parasite of salmon (Roffe, 1980; Roffe and Mate, 1984). In British Columbia, Stellers interfere with local commercial fisheries by preying on salmon and halibut and damaging gear and captured fish (Bigg, 1984). They disperse schools of herring and enter nets in herring gillnet and seine fisheries (Bigg, 1984). Also, they are suspected of feeding on inshore stocks of cod (Bigg, 1984).

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Review of Northern Fur Seals (Callorhinus ursinus) in the North Pacific 1978-1997

Population, Abundance, and Distribution

Range:

Across the subarctic north Pacific, into the Bering and Okhotsk Seas and the Sea of Japan up to about 60°N, south to Honshu, Japan (36°N) in the western Pacific and to the U.S.-Mexico border (30°N-32°N) in the eastern Pacific

Breeding Range:

Breed on the Pribilof and Commander Islands and Bogoslof Island in the Bering Sea, Robben Island in the Okhotsk Sea, the Kuril Islands in the western Pacific, and Adams Cove and Castle Rock on San Miguel Island off CA; there were 32 breeding colonies in 1992

Preferred Habitat:

Offshore rocks, sloping rock outcroppings, and sandy or cobble beaches

Fur seals have high site fidelity to natal rookeries (Vladimirov, 1978; MMC, 1997; Loughlin, 1993) and can be roughly divided into at least five breeding stocks (Bigg, 1990; Loughlin, 1993). However, there is intermixing of fur seals between breeding sites (Lander and Kauimura, 1982; Kajimura, 1984), and Hill et al (1997) and Barlow et al (1997) go so far as to state that there is considerable interchange between rookeries. Sobolevskij (1984) suggests that variations in the shapes of foreflippers in fur seals from Robben Island and the Commander Islands indicate separate stocks. Fur seals breed on the Pribilof Islands, Bogoslof Island, the Commander Islands, Robben Island, the Kuril Islands, and San Miguel Island (Antonelis and Fiscus, 1980; Bigg, 1990; Braham, 1992; Consiglieri et al, 1982; Fowler and Kozloff, 1988; Kajimura, 1984). Most, 74-80%, breed on the Pribilof Islands; 15% breed on the Commander Islands; 9% breed on Robben Island; 2% breed on the Kuril Islands; and less than 1% breed on San Miguel Island and on Bogoslof Island (Consiglieri et al, 1982; Loughlin, 1993; MMC, 1997; U.S. Air Force, 1996; Barlow et al, 1997; NMFS, 1991; 1992; 1993; Kajimura, 1984). For management purposes, fur seals are separated into two stocks in the U.S.: eastern Pacific, which includes breeding colonies on the Pribilof Islands and Bogoslof Island; and San Miguel Island, which includes breeding colonies on San Miguel Island in the CA Channel Islands (Barlow et al, 1997; Hill et al, 1997).

Fur seals were discovered on the Pribilof Islands in 1786 (Loughlin, 1993). Pre-exploitation population on the Pribilof Islands was about 2-2.25 million (Kajimura, 1985; Brueggeman et al, 1990; Lander and Kajimura, 1982; Manning, 1989; MMC, 1997; NACOA, 1985; Young, 1981; Kajimura, 1984). Population was severely reduced by harvesting to a low of 200,000-300,000 in 1911-1912 (Brueggeman et al, 1990; Lander and Kajimura, 1982; Loughlin, 1993; Trites and York, 1993; Young, 1981), but recovered to 1.8-2.1 million in the 1950's and leveled off (Boltnev, 1996; Loughlin, 1993; Merrick, 1995; MMC, 1997). Harvest of females beginning in 1956 decreased the population again in the late 1950's and 1960's (Fowler, 1982; Loughlin, 1993; MMC, 1997; Springer, 1992), and the population did not recover as well as expected with the termination of female harvest in 1968 (Swartzman and Haar, 1983; Fowler,

1982). However, population reached about 1.25 million by 1974 (Hill et al, 1997; Loughlin, 1993). From 1975 into the 1980's, population decreased about 8%/yr. (Brueggeman et al, 1990; Burd et al, 1992; Hill et al, 1997; MMC, 1997; Perez, 1990), with pup production declining 6.5-7.8%/yr. (Hill et al, 1997; Kajimura and Loughlin, 1988; Loughlin, 1993). Number of pups born on the Pribilof Islands declined 70% from 1950 to 1981 (Fowler, 1982; York and Hartley, 1981). Population was 877,000 in 1983 (Hill et al, 1997; Loughlin, 1993; MMC, 1997). Population was relatively stable from 1983 to the early 1990's (Loughlin et al, 1993; MMC, 1994). It increased to 1.01 million in 1990 but decreased to 984,000 in 1992 (Hill et al, 1997; Loughlin, 1993). Population on the Pribilof Islands was 1,014,000 in 1994 (MMC, 1997). Population on St. Paul Island was stable and on St. George Island was decreasing in 1992 (Wickens and York, 1997).

Fur seals were discovered on Bogoslof Island in 1978 (Loughlin, 1993) and were first observed breeding there in 1980 (Lloyd et al, 1981; Fowler and Kozloff, 1988; Loughlin, 1993; Loughlin and Miller, 1989). Population was up to 112 in 1985 (Fowler and Kozloff, 1988; Loughlin et al, 1988). From 1987 to 1988, population grew 57% to over 400 (Loughlin and Miller, 1989). NMFS (1991; 1992; 1993) reports that the population on Bogoslof Island was 1,500 in the early 1990's. Population on Bogoslof Island was still increasing in 1992 (Wickens and York, 1997). It was 5,173 in 1994 (Payne et al, 1996a), and pup counts increased 67% increase from 1993 to 1994 (Piatt and Goley, 1996). Pup production on Bogoslof Island increased 57%/yr. up to 1988 and 130%/yr. from 1990 to 1993 (Piatt and Goley, 1996). Non-pup population may have peaked in 1991 and then began declining or stabilizing (Piatt and Goley, 1996). Population of the eastern Pacific stock, which includes the Pribilof Islands and Bogoslof Island, was 1,019,192 in 1994 (MMC, 1997; Hill et al, 1997). N_{min} of this stock is 969,595, and PBR is 20,846 (Hill et al, 1997).

A small breeding colony of less than 100 fur seals was discovered on San Miguel Island in 1968 (Antonelis et al, 1990b; Braham, 1996; Brueggeman et al, 1990; Lander and Kajimura, 1982; Melin et al, 1996; Kajimura, 1984). Population on the island has steadily increased since the 1970's, except during the 1982-1983 El Niño, during which pup production declined 63% (Barlow et al, 1997) and population of adult females declined 48% in Adams Cove (Antonelis et al, 1988). Population decreased from over 3,600 in 1982 to 1,500 in 1983 (Brueggeman et al, 1990). Live pup counts increased 24%/yr. from 1972 to 1982 and have steadily increased since 1983 (Barlow et al, 1997). Counts were below the 1982 level until 1990 (Barlow et al, 1997). Counts of territorial males grew 28.6% and of pups grew 16% from 1993 to 1994 (Melin et al, 1996). NMFS (1993) reports that the population on San Miguel Island was 6,000 in the early 1990's. In addition to births in CA on San Miguel Island, one pup was born in College Grove, Trinidad, CA in July, 1983 (Stein et al, 1986). This was the first known birth on the CA mainland (Stein et al, 1986). Stein et al (1986) also report that a pup was born on the WA mainland in 1959. U.S. Air Force (1996) estimates that the San Miguel Island stock has been growing at about 10%/yr. since it was discovered. Population of the stock was 10,036 in 1995 (Barlow et al, 1997). N_{min} is 5,018, and PBR is 216 (Barlow et al, 1997).

Population on the Commander Islands was at its lowest level at 11,000 in 1911 (Boltnev, 1996). Population grew to 15,000-17,000 in 1917 and to 30,000 in 1934 (Boltnev, 1996). Population growth slowed in the mid-1960's, and trends differed among rookeries (Boltnev, 1996). In the late 1970's, population on the Commander Islands was 265,000 (Antonelis and Fiscus, 1980). In the late 1980's, population was 200,000-220,000 (Bigg, 1990; Perez, 1990). Overall, there was a period of rapid population growth from 1958 to 1966, a decline from 1967

to 1975, a period of slow growth from 1976 to 1986, and another decline from 1987 to 1992 (Boltnev, 1996). Population on the Commander Islands was stable or increasing in 1992 (Wickens and York, 1997).

In the late 1970's, population on Robben Island was 165,000 (Antonelis and Fiscus, 1980). In the late 1980's, population was 70,000-80,000 (Bigg, 1990; Perez, 1990). Population decreased at a rate of about 10%/yr. from 1967 to 1992 (Wickens and York, 1997).

In the late 1970's, population on the Kuril Islands was 33,000 (Antonelis and Fiscus, 1980). In the late 1980's, population was 45,000-50,000 (Bigg, 1990; Perez, 1990). Population was increasing or stable in 1992 (Wickens and York, 1997). Mednyi Island in the Kuril Islands is inhabited by both fur seals and Steller sea lions during the breeding season (Pitcher and Calkins, 1981; Chelnokov, 1984).

Population growth on the Commander Islands in the late 1950's and early 1960's may have been due to immigration from other populations, especially the Pribilof Islands (Boltnev, 1996; Boltnev and Mathisen, 1996). Pribilof Island fur seals comprised up to 20% of the population on the Commander Islands in the late 1950's and early 1960's (Boltnev, 1996). Other Islands also experience immigration and emigration; for example, on Robben Island, 98% of fur seals were of local origin, and emigration rate was 0-0.89% in the mid-1970's (Vladimirov, 1978). Also, most growth in pup production on Bogoslof Island is due to immigration of breeding adults from the Commander and Pribilof Islands (Piatt and Goley, 1996).

Range-wide population was 1,765,000 in the early 1970's (Lander and Kajimura, 1982). Population was 1.7 million in 1982, and fur seals were described as the most abundant pinniped in the north Pacific (Manning, 1989). Population was more or less growing exponentially in 1982 (Manning, 1989). However, in 1983, population was 1,190,000-1,225,000 (NMFS, 1991; 1992; Perez, 1990) and was 40% below the level in the mid-1950's (NMFS, 1991; 1992; 1993). Population was about 1,427,000 in 1992 (Wickens and York, 1997) and 1,267,500 in 1994 (Hill et al, 1997; MMC, 1997).

Fur seals migrate south as far as the waters offshore of CA beginning in late October (Antonelis and Fiscus; Loughlin et al, 1987). Abundance from 34°N-42°N is maximum from January to March (Antonelis and Fiscus, 1980; Brueggeman et al, 1990; U.S. Air Force, 1996; Kajimura, 1984). Both sexes spend 7-8 months at sea (Barlow et al, 1997; Loughlin, 1993), rarely approach land during the winter months (Antonelis and Fiscus, 1980), and are typically seen over the continental shelf and slope or over underwater ridges and sea mounts (Brueggeman et al, 1990; Loughlin et al, 1987; 1993; U.S. Air Force, 1996; Kajimura, 1984), generally 20-200 km from shore (Gentry and Holt, 1986). However, fur seals may temporarily haul out onto land in AK, British Columbia, OR, and on islets along the coast of the U.S (Hill et al, 1997; Barlow et al, 1997; Loughlin, 1993; Courbis, pers. obs.). Adult females and pups from the Pribilof Islands migrate through the Aleutian Islands into the north Pacific to offshore OR and CA (Barlow et al, 1997; Loughlin, 1993). Pups may stay at sea for over 2 years before returning to their natal rookery (Barlow et al, 1997; Hill et al, 1997; Loughlin, 1993; Kajimura, 1984). Most of the fur seals that winter in CA are females and males 2-4 years old (Antonelis and Fiscus, 1980; Loughlin et al, 1987). Older females (5 years old and older) are generally more abundant off CA than off WA (Kajimura, 1984). Adult males generally migrate as far south as the Gulf of AK (Barlow et al, 1997; Bigg, 1990; Hill et al, 1997; Loughlin, 1993; U.S. Air Force, 1996; Kajimura, 1984; Fiscus and Kajimura, 1981), with a peak in abundance from February to March (Brueggeman et al, 1990). In the west, adult males migrate south to northern Japan (Loughlin,

1993). Harem bulls winter closest to breeding islands (Lander and Kajimura, 1982). Only younger, immature males (1-5 years old) have been found south of AK with a few exceptions (Kajimura, 1984).

Northward migration from CA begins in March, and most have left the CA area by early June, except for the fur seals in the San Miguel Island stock (Antonelis and Fiscus, 1980). Males and females begin arriving on San Miguel Island in May (Melin et al, 1996). U.S. Air Force (1996) states that fur seals that breed on San Miguel Island may not migrate out of CA at all. Bigg (1988) states that after breeding off CA, females stay in the south and males migrate north. Mature females from northern rookeries generally depart CA in March or April, while younger, non-breeding females depart in May or June (Brueggeman, 1990). Adult males and females usually occur ashore at different, though overlapping, times (Barlow et al, 1997; Hill et al, 1997). Adult males occur on shore during the four month period from May to August, though some may be present until November; adult females occur on shore as long as six months, from June to November (Barlow et al, 1997; Hill et al, 1997; Brueggeman et al, 1990). The first to arrive on rookeries are older pregnant females, followed by intermediate age pregnant females, then barren old females, then young and first time breeders and immature females at the end of the season (Kuzin and Panina, 1977).

There is one record of a fur seal in inland waters of the west coast (Dierauf, 1984). A small female about six months old was sighted 144 km upstream on the Sacramento River across from the Port of Stockton turning basin in 1981 (Dierauf, 1984).

International fur seal management by the U.S., Japan, Russia, and Canada began in 1911 (Kajimura, 1985; Manning, 1989) with the Convention for the Preservation and Protection of Fur Seals, which prohibited pelagic sealing (Kajimura, 1984). The Interim Convention for the Conservation of North Pacific Fur Seals was signed in 1957 by the U.S., Japan, Canada, and Russia (Boltnev, 1996; MMC, 1997; Kajimura, 1984). A Protocol to the Convention was signed to extend the Convention four years to 1984 (NACOA, 1985; Young, 1981). The Fur Seal Act was passed in 1966 (MMC, 1997; NACOA, 1985). Fur seals in the eastern Pacific stock were listed as depleted and strategic under the MMPA in 1988 (Hill et al, 1997; Balsiger, 1995; Loughlin et al, 1993; NMFS, 1991; 1992; 1993; Pitcher, 1990; Sinclair, 1996). Fur seals in the San Miguel Island stock are not considered depleted or strategic under the MMPA (U.S. Dept. of Comm., 1996). Fur seals are not considered endangered or threatened under the ESA (Hill et al, 1997; Barlow et al, 1997).

Food and Feeding

Fur seals are opportunistic feeders that change their diets as they follow their migratory routes (Fowler, 1982; Kajimura, 1984; 1985; Lander and Kajimura, 1982; NACOA, 1985; Sinclair et al, 1994; Perez and Bigg, 1986; Fiscus and Kajimura, 1981). For example, northern anchovies are about 50% of the fur seal diet in CA in January and February but are down to 25% of the diet by March and 10% by April (Bigg, 1990). Small schooling fish are the principal prey over the continental shelf, and oceanic squid are the principal prey seaward of the continental slope (Kajimura, 1984; Fiscus and Kajimura, 1981). Primary prey in CA are northern anchovy, Pacific saury, Pacific hake, jacksmelt, rockfish, and squid, particularly market squid; primary prey in OR are northern anchovy, Pacific hake, and squid; primary prey in WA are northern anchovy, Pacific hake, sablefish, and squid, Onychoteuthis sp.; primary prey in WA are northern

anchovy, Pacific herring, eulachon, salmon, capelin, Pacific hake, sablefish, rockfish, and squid, Onychoteuthis sp.; primary prey in British Columbia are Pacific herring, eulachon, salmon, sablefish, rockfish, walleye pollock, Pacific cod, Pacific hake, Pacific saury, and squid, particularly market squid and nail squid; primary prey in AK are Pacific herring, capelin, sablefish, salmon, walleye pollock, Atka mackerel, Pacific sand lance, deep sea smelt, rockfish, and squid, particularly Gonatus spp. and Berryteuthis magister; primary prey in the Bering Sea are Pacific herring, capelin, walleye pollock, Atka mackerel, Greenland turbot, salmon, deepsea smelts, and squid, including Gonatus spp., Berryteuthis magister, and Gonatopsis borealis; primary prey in the western Pacific are cod-like fish, lanternfish, chub mackerel, walleye pollock, and squid; primary prey in the Sea of Japan are walleye pollock and squid; and primary prey in the Sea of Okhotsk are walleye pollock (Lander and Kajimura, 1982; Perez and Bigg, 1986; Kajimura, 1984). Walleye pollock became a larger part of the fur seal diet in the 1970's after the establishment of the fishery (Swartzman and Haar, 1983). Time of day and season are important factors in fur seal diets (Fowler, 1982).

Fur seals prey on over 100 species of fish and cephalopods (Fowler, 1982). Birds and isopods have also been found among stomach contents of fur seals (Jones et al, 1981). Hobson et al (1997) report that fur seals do not simply feed on the most abundant prey, rather they are size-selective feeders. Sinclair et al, (1994) state that fur seals generally concentrate on three primary prey species in each oceanographic subregion. Fur seals feed mainly in the oceanic zone (Antonelis et al, 1990b) and mostly at night (Antonelis and Fiscus, 1980; Kajimura and Loughlin, 1988; NACOA, 1985; Kajimura, 1984; Fiscus and Kajimura, 1981). They often feed over or beyond the continental slope (Brueggeman et al, 1990; Kajimura and Loughlin, 1988; Lander and Kajimura, 1982; Gentry et al, 1986; Fiscus and Kajimura, 1981). They probably feed at least twice per day (Lander and Kajimura, 1982). They usually swallow whole fish or squid up to 25 cm long but will often surface and break up larger prey before eating it (Antonelis and Fiscus, 1980; Kajimura, 1984).

Antonelis et al (1990b) found that postpartum females on San Miguel Island forage at an average distance of 72.3 km at depths ranging from 37-2,562 m, with an average of 933 m. Average duration of postpartum feeding trips has been reported as 3.5-8 days on San Miguel Island and 4.5-9.7 days on the Pribilof Islands (Antonelis et al, 1990b; Brueggeman et al, 1990). Females on the Pribilof Islands make 8-12 postpartum feeding trips (Brueggeman et al, 1990). Duration of postpartum trips increases with each successive trip for the first 3-4 trips (Antonelis et al, 1990b). Duration of the first three feeding trips increased by about two days on San Miguel island during the 1982-1983 El Niño (Loughlin et al, 1987). Loughlin et al (1987) found that lactating females consume as much as 1.5-1.8 times more food than non-lactating females. They also report that females feeding over the shallow continental shelf dive over 100 m and feed throughout the day and females that feed over deep water in the Aleutian Basin feed primarily at night with average dive depths of less than 100 m. Loughlin et al (1993) found that there is little overlap of foraging activities of males and females in the Bering Sea. They suggest that adult males and pups probably feed on different size prey and dive to different depths to feed.

Fur seals eat less than 1% of the standing fish stock in the eastern Bering Sea and the Aleutian islands (NACOA, 1985). Food shortage does not appear to be a problem for fur seals (NACOA, 1985). The following prey species have been determined for fur seals:

<u>Category</u>	<u>Scientific Name</u>	<u>Common Name</u>
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Molluska

Abraliopsis felis
Abraliopsis sp.
Berryteuthis magister
 Chroteuthidae
Chroteuthis sp.
 Decapoda
Dosidicus gigas
Galiteuthis phyllura
 Gonatidae
Gonatopsis borealis
Gonatus berryi
Gonatus madokai-middendorffi
Gonatus pyros
Gonatus sp.
Gonatus tinro
Loligo opalescens
Moroteuthis robusta
 Octopoda
Octopoteuthis sp.
Ocythoe tuberculata
Ommastrephes bartrami
Ommastrephes sloani pacificus
 Onychoteuthidae
Onychoteuthis borealijaponicas
Onychoteuthis sp.
Watasenia scintillans

Enope squid
 Enope squid
 Magistrate armhook squid
 Whiplash squid
 Whiplash squid
 Squid
 Arrow squid
 Cranch squid
 Gonata squid
 Gonata squid
 Gonata squid
 Gonata squid
 Gonata squid
 Gonata squid
 Market squid
 Giant squid
 Octopus
 Octopus squid
 Ocythoid Octopus
 Arrow squid
 Arrow squid
 Hooked squid
 Nail squid
 Hooked squid
 Firefly squid

Crustacea

Riggia sp.

Isopod

Agnatha

Lampetra tridentata
(Entosphenus tridentatus)

Pacific lamprey

Chondrichthys

Hydrolagus colliei
Squalis acanthias

Spotted ratfish
 Spiny dogfish

Osteichthys

Alosa sapidissima
Ammodytes hexapterus
 Anarhichadidae
Anarhichas orientalis
Anoplopoma fimbria
Anopterus pharao
Aptocyclus ventricosus
Atheresthes stomias
Atherinopsis californiensis

American shad
 Pacific sand lance
 Wolffish
 Bering wolffish
 Sablefish
 Daggertooth
 Smooth lumpsucker
 Arrowtooth flounder
 Jacksmelt

Bathylagidae	Deep sea smelts
<u>Bathymaster signatus</u>	Searcher
Bathymasteridae	Ronquils
<u>Brama japonica</u>	Pacific pomfret
<u>Ceratoscopelas</u> sp.	Lampfish
<u>Citharichthys</u> sp.	Flounder
<u>Clupea pallasi</u>	Pacific herring
Clupeidae	Herring
<u>Cololabis saira</u>	Pacific saury
Cottidae	Sculpins
Cyclopteridae	Snailfish
<u>Diaphus gigas</u>	Headlightfish
<u>Diaphus theta</u>	California headlightfish
<u>Engraulis japonicus</u>	Japanese anchovy
<u>Engraulis mordax</u>	Northern anchovy
Gadidae	Cods
<u>Gadus macrocephalus</u>	Pacific cod
<u>Gasterosteus aculeatus</u>	Threespine stickleback
<u>Genyonemus lineatus</u>	White croaker
<u>Hemilepidotus hemilepidotus</u>	Red Irish lord
Hexagrammidae	Greenlings and lingcod
<u>Hippoglossus stenolepis</u>	Pacific halibut
<u>Hypomesus pretiosus</u>	Surf smelt
<u>Imiastes hexapterus</u>	Sandlance
<u>Laemonema longipes</u>	Cod-like fish
<u>Lampanyctus jordani</u>	Brokenline lampfish
<u>Lampanyctus</u> sp.	Lampfish
<u>Leuroglossus schmidtii</u>	Northern smoothtongue
<u>Lycodes</u> sp.	Eelpouts
<u>Lyopsetta exilis</u>	Slender sole
<u>Mallotus villosus</u>	Capelin
<u>Medialuna californiensis</u>	Halfmoon
<u>Merluccius productus</u>	Pacific whiting/hake
<u>Microgradus proximus</u>	Pacific tomcod
Myctophidae	Lanternfish
<u>Notoscopelas japonicus</u>	Lampfish
<u>Oncorhynchus gorbuscha</u>	Pink salmon
<u>Oncorhynchus keta</u>	Chum salmon
<u>Oncorhynchus kisutch</u>	Coho salmon
<u>Oncorhynchus mykiss</u>	Steelhead
(<u>Salmo gairdneri</u>)	
<u>Oncorhynchus nerka</u>	Sockeye salmon/red salmon
<u>Oncorhynchus</u> sp.	Salmon
<u>Oncorhynchus tshawytscha</u>	Chinook salmon
Osmeridae	Smelts

<u>Paralepis atlantica</u>	Barracudinas
<u>Pentaceros richardsoni</u>	Pelagic armorhead
Percoid	
Pleuronectidae	Flatfish
<u>Pleurogrammus monopterygius</u>	Atka mackerel
<u>Porichthys notatus</u>	Plainfin midshipman
<u>Reinhardtius hippoglossoides</u>	Greenland turbot
Salmonidae	Salmon
<u>Sardinops melanosticta</u>	Japanese sardine
Sciaenidae	Croakers
<u>Scomber japonicus</u>	Chub mackerel
<u>Scopelosaurus</u> sp.	Paperbone
Scorpaenidae	Rockfish
<u>Sebastes alutus</u>	Pacific ocean perch
<u>Sebastes entomelas</u>	Widow rockfish
<u>Sebastes jordani</u>	Shortbelly rockfish
<u>Sebastes</u> sp.	Rockfish
<u>Seriplus politus</u>	Queenfish
<u>Sygnathus californiensis</u>	Kelp pipefish
<u>Symbolophorus californiensis</u>	California lanternfish
<u>Symbolophorus</u> sp.	Lanternfish
<u>Tactostoma macropus</u>	Longfin dragonfish
<u>Tarletonbeania crenularis</u>	Blue lanternfish
<u>Tarletonbeania</u> sp.	Lanternfish
<u>Tetragonurus cuvieri</u>	Smalleye squaretail
<u>Thaleichthys pacificus</u>	Eulachon
<u>Theragra chalcogramma</u>	Walleye pollock
Trachipteridae	Dealfish
<u>Trachipterus altivelis</u>	King-of-the-salmon
<u>Trachurus symmetricus</u>	Jack mackerel
<u>Trichodon</u> sp.	Sandfish
<u>Trichodon trichodon</u>	Pacific sandfish
Trichodontidae	Sandfish
Zoarcidae	Eelpouts
	Unidentified bird

Aves

Habits

Fur seals are usually alone at sea, but pairs and groups of three are fairly common (Antonelis and Fiscus, 1980; Brueggeman et al, 1990; Kajimura and Loughlin, 1988; Kajimura, 1984; Fiscus and Kajimura, 1981). The largest group seen included about 100 fur seals (Antonelis and Fiscus, 1990b; Kajimura and Loughlin, 1988). Fur seals segregate on land and at sea by age, sex, and maturity (Lander and Kajimura, 1982).

Average duration of time females spend ashore on San Miguel Island between feeding trips is 1.7 days (Antonelis et al, 1990b). Average duration of time ashore on St. Paul Island is 1.9 days (Loughlin et al, 1987) and on the Pribilof Islands together is 2.2 days (Gentry and Holt, 1986). On the Pribilof Islands, the last visit to shore before weaning is significantly longer than previous visits (Gentry and Holt, 1986). Duration of time spent ashore is determined by contact with the pup rather than the female's inability to remain ashore longer (Gentry and Holt, 1986). Duration of time ashore on the Pribilof Islands is shorter for suckling females than for non-suckling females (Gentry and Holt, 1986). However, pregnancy is not an important determinant in shore visit duration or time spent at sea (Gentry and Holt, 1986). Gentry and Holt (1986) found that non-suckling females on the Pribilof Islands make 5-6 visits to shore and 5-6 trips to sea and suckling females make 12-14 visits to shore and 11-12 trips to sea. Peak departure time on San Miguel Island is in late morning and early afternoon, and peak arrival time is in early morning (Antonelis et al, 1990b). Departures on St. Paul Island are fewest in the early morning, and arrivals show no particular timing pattern (Loughlin et al, 1987). Abundance on the Pribilof Islands peaks in afternoon and declines at night (Gentry and Holt, 1986). Average duration of trips to sea is 7.5 days, with a range of 5.5 to almost 10 days on St. George Island (Gentry et al, 1986). Duration of females' trips to sea increases an average of 1.2 days for every 30 days postpartum on the Pribilof Islands (Gentry and Holt, 1986).

Female fur seals exhibit two types of diving patterns: shallow and deep (Goebel et al, 1991; Gentry et al, 1986). Females may be exclusively deep or shallow divers or may use both patterns (Gentry et al, 1986). The deep diving pattern is typically to depths greater than 75-100 m without changing depth during a diving bout and diving at all hours of the day (Loughlin et al, 1987; Goebel et al, 1991; Gentry et al, 1986). The shallow diving pattern is typically to depths less than 100 m and primarily diving at night (Loughlin et al, 1987; Goebel et al, 1991; Gentry et al, 1986). Dives tend to be simple spikes with no time spent at maximum depth (Gentry et al, 1986). Number of dives per hour is greater for shallow divers than deep divers (Goebel et al, 1991). Both types of divers dive more at night than during the day (Goebel et al, 1991). Females diving in deep water beyond the continental shelf primarily exhibit the shallow diving pattern, and females diving at or near the shelf break commonly exhibit the deep diving pattern (Goebel et al, 1991). Fur seals rarely dive below 200 m (Loughlin et al, 1987). Near the Pribilof Islands, average dive depth is 68 m, with an average dive duration of 2.6 minutes (Kajimura and Loughlin, 1988). Diving bouts average 18.2 dives each, and bout durations average 2.2 hours (Gentry et al, 1986). Gentry et al (1986) found that, while at sea, females spend about 57% of the time active at the surface without diving, 26% diving, and 17% resting. They also found that 80% of resting occurred during the day and 72% of periods of uninterrupted resting were four hours or less. Additionally, 77% of periods of uninterrupted diving were eight hours or less, and 76% of active periods without diving were 10 hours or less (Gentry et al, 1986). Fastest estimated swimming speed of female fur seals on San Miguel Island, CA is 13.7 km/hr. (Antonelis et al, 1990b).

Male fur seals near St. Paul Island generally dive 4-100 m (Loughlin et al, 1993). No dives recorded near St. Paul Island by Loughlin et al (1993) were deeper than 350 m. Of the dives recorded by Loughlin et al (1993), 90% were less than 6 minutes long and 43% were one minute or less. Dive depth and duration varies among individuals (Loughlin et al, 1993). Time of day males dive also varies among individuals (Loughlin et al, 1993).

Fur seals interact with other marine mammals. Fur seals fight with Steller sea lions on overlapping rookeries on Mednyi Island (Chelnokov, 1984).

Reproduction

Breeding season:	May-November
Implantation:	November
Pupping season:	June to August
Sexual maturity:	
Females:	3-6 yrs. old
Males:	4 yrs. old; attain territories 7 yrs. old and over
Gestation:	1 year
Lactation:	4-5 months

Fur seals are polygynous, colonial breeders (Goebel et al, 1991). Males arrive on rookeries before females and establish territories (Brueggeman et al, 1990). When females arrive, they pup within a few days and breed within the following week (Brueggeman et al, 1990; Goebel et al, 1991; Temte, 1985; Gentry and Holt, 1986). Implantation occurs about 120 days after copulation (Temte, 1985). Nulliparous females may ovulate two months after parous females (Temte, 1985). Pup rearing lasts 111-125 days, with females on shore suckling 31-35 days (Gentry and Holt, 1986). Males compete for females on the border between territories (Le Boeuf and Mesnick, 1991; Wickens and York, 1997). Occasionally, two males will seize a female with their teeth and injure or kill her by pulling in opposite directions (Le Boeuf and Mesnick, 1991). Territorial males are not the only males to mate (Wickens and York, 1997). It is not uncommon for relatively younger males without territories to mate with females on the periphery (Wickens and York, 1997).

Generally, females give birth to one pup each year, but twinning has been recorded in the eastern north Pacific at least 11 times (Spotte, 1982). Females have high site fidelity to rookeries (Goebel et al, 1991; NOCOA, 1985).

Loughlin (1993) states that timing of reproduction is essentially the same on the Pribilof Islands, Bogoslof Island, and San Miguel Island. However, Wickens and York (1997) found the median pupping date was July 9 on the Pribilof Islands in 1951, 1962, 1963, and 1983 and was June 24 on San Miguel Island from 1969 to 1978. Temte (1985) suggests that there is a relationship between photoperiod and reproductive timing. Photoperiod may act to trigger implantation (Temte, 1985; Melin et al, 1996). Temte (1985) found that the average date of parturition in 1979 was June 26 on San Miguel Island and July 10 on St. George Island. He suggests that a specific photoperiod occurring about 62 days after parturition (65 days before implantation) acts to synchronize the timing of parturition in fur seals at a given latitude. Onset of pupping on San Miguel Island occurred after the first week of June, one week later than normal, in 1983 and 1993 (El Niño years) (Melin et al, 1996). In El Niño years, warmer sea

surface temperature may shift distribution of fur seals farther north, changing the photoperiod necessary to initiate implantation to a week later (Melin et al, 1996).

Males fast during the breeding season (Baker et al, 1994). Immature males observed on St. Paul Island lost 20-30% of their body mass (Baker et al, 1994). Females begin making feeding trips 1-2 days after giving birth but return to nurse their pups (Goebel et al, 1991; Loughlin et al, 1987).

Fur seals are primiparous as young as 4 yrs. old (Trites and York, 1993). Rate of primiparity rises up to 8 yrs. old, levels to 14 yrs. old, and declines to none at 25 yrs. old (Trites and York, 1993; Wickens and York, 1997). Pregnancy rate peaks at about 89% at 11 yrs. old (York and Hartley, 1981). Age classes with increased juvenile survival start reproducing younger (York, 1983). Age of primiparity was 5.5-6.5 yrs. old from 1958 to 1974 (York, 1983). Pregnancy rates of females 8-13 yrs. old declined in the Pribilof Islands and from CA to British Columbia from 1958 to 1972 (Trites and York, 1993). The proportion of 6 and 7 yr. old multiparous females declined from 1958 to 1974, and age of primiparity increased from 1959 to 1974 (Trites and York, 1993). Natality rate of head branded females on St. George Island decreased from 93% in 1986 to 74% in 1989 (Gentry and Goebel-Diaz, 1990). Efficiency of reproduction decreased on Robben Island from 1957 to the mid-1970's (Kolesnik et al, 1977). Pup production decreased on San Miguel Island in 1992 due to El Niño but increased again in 1993 (Melin et al, 1996). Pregnancy rate was 72% in the western Pacific from 1958 to 1974 (Wickens and York, 1997).

Hybrids have occurred between fur seals and California sea lions on San Miguel Island (DeLong and Melin, 1992).

Mortality and Economics

Natural Mortality

Fur seals are preyed upon by orcas (Hoyt, 1984; Kajimura and Loughlin, 1988; Lander and Kajimura, 1982; Perez, 1990). They are also preyed upon by blue foxes and sharks and occasionally by sea lions (Lander and Kajimura, 1982; NACOA, 1985).

Fifty percent of pup mortality occurs within the first week of life (Melin et al, 1996). Pup mortality on the Pribilof Islands increased in the 1950's and gradually decreased after the mid-1960's (Boltnev, 1996). It declined in parallel with the decline of pup numbers on the Pribilof Islands (Fowler, 1982). Pup mortality was 5.3% on St. Paul Island in 1989 (Antonelis et al, 1990a) and 4.5% on the Pribilof Islands in 1990 (Melin et al, 1996). On the Pribilof Islands, mortality is 50% in the first year of life, 20% from 1-3 years old, and 20% for males 3-7 years old (Lander and Kajimura, 1982). Male mortality may exceed female mortality up to 3 years old (Lander and Kajimura, 1982). Pup mortality decreased from 9.9% in 1984 to 3.7% in 1985 in Adams Cove on San Miguel Island (Antonelis et al, 1988). Minimum pup mortality was 7.1% in Adams Cove and 6.6% on Castle Rock on San Miguel Island in 1994 (Melin et al, 1996). Pup mortality was up to 24% in Adams Cove during the 1992 El Niño (Melin et al, 1996). On the Commander Islands, pup mortality increased in the mid-1960's, then fluctuated at 10-16% until it decreased to 4-8% from about 1988 to 1994 (Boltnev, 1996). As density on rookeries increases, pup mortality increases (Lander and Kajimura, 1982).

Baker and Fowler (1992) found that male fur seals that survived at least two years after weaning had been significantly heavier than the average for their cohort as pups. They found no significant relationship between pup weight and survival of females. Weight of newborn pups on the Commander Islands was greatest in 1983, decreased through the late 1980's, reached its lowest in 1989, and increased from 1989 to 1994 (Boltnev, 1996). Declines were due to inadequate nutrition of females during lactation (Boltnev, 1996). Pups born during the first major reduction in population on the Pribilof Islands were born at greater weights than pups born during the period when population was at its peak (Fowler, 1982). Also, growth rate of females was more rapid during the years following the decline created by female harvest on the Pribilof Islands (Fowler, 1982). Pup growth rates on St. George Island exceed those of same-sex pups on St. Paul Island (Gentry and Holt, 1986). Growth rate of males on the Pribilof Islands increased from 1962 to 1971 (Fowler, 1982). Pup mortality increased and weight decreased on San Miguel Island during the 1992 El Niño (Melin et al, 1996). However, average pup weights of both sexes on San Miguel Island increased from 1993 to 1994 (Melin et al, 1996). Boltnev (1990) found that pup mortality is higher among pups born at the beginning of the reproductive period and among smaller pups. Boltnev (1990) also found that pups of younger females and immigrant females have lower survival. Pup survival is lower in peripheral than in central areas of the rookery (Boltnev, 1990).

As mentioned, El Niño can decrease pup survival and weight (Melin et al, 1996). El Niño may be responsible for the unusually high number of stranded fur seals in CA in 1997 (Anonymous, 1997a; 1997b). It may also have increased pup mortality to up to 66% in 1997, three times the normal rate, in some areas of San Miguel Island (Anonymous, 1997b). Reduced food supply brought on by El Niño causes females to spend more time foraging and less time feeding pups (Anonymous, 1997a; 1997b). As a result, pups are undernourished and mortality increases (Anonymous, 1997a; 1997b). Antonelis et al (1988) state that El Niño may decrease fur seal populations.

Steller sea lions breeding on Mednyi Island may inhibit fur seal reproduction and affect population on overlapping rookeries (Chelnokov, 1984). However, fur seals appear to outcompete Steller sea lions (Chelnokov, 1984). Mortality on rookeries can also occur when male fur seals accidentally injure and kill a female by seizing her with their teeth and pulling her in opposite directions (Le Boeuf and Mesnick, 1991).

Diseases and parasites affect fur seals. An epizootic of hookworm, Uncinaria lucasi, occurred in the early 1970's on Northwestern rookery on the Commander Islands, causing pup mortality of up to 40% (Boltnev, 1996). Lander and Kajimura (1982) state that hookworm, emaciation syndrome, and infection by a spirochaete bacterium account for about 2/3 of all pup mortality on the Pribilof Islands. 371 pups were found dead of a previously unrecorded disease, white muscle disease, which causes severe muscle lesions, on St. Paul Island in July and August, 1990 (Braham, 1992). Leptospira pomona has been isolated in fur seals and has been associated with reproductive failure (Braham et al, 1980). Caliciviruses are also suspected of being associated with reproductive failure (Smith et al, 1986). Average weight of pups infected with Salmonella or with Acinetobacter lwoffii is less than that of uninfected pups (Burd et al, 1992). Burd et al (1992) state that mortality from Salmonella is not a problem on San Miguel Island but may be a problem on the Pribilof Islands. Salmonella may cause mild gastroenteritis and may become a disease problem in pups or others that become debilitated for other reasons (Gilmartin et al, 1979). Salmonella may also cause abortions (Gilmartin et al, 1979). Burd et al (1992)

report that Pseudomonas sp. causes bacteremia and pneumonia but does not seem to be a serious threat. However, opportunistic pathogens may be an important source of mortality (Burd et al, 1992). Pox was first described in a fur seal in 1951 (Hadlow et al 1980). Pox is characterized by multiple 3-8 mm nodules in the skin of the nose and flippers (Hadlow et al, 1980). Ke Chung Kim and Haas (1980) found that nasal mites, Orthohalarachne attenuata, can impair respiration and cause lesions in the lungs and secondary alveolar emphysemas. At least 26 parasites, 20 bacteria, and 9 viruses have been found in or on fur seals:

Bacteria

<u>Acinetobacter lwoffii</u>	Burd et al, 1992
<u>Brucella</u> sp.	Spraker, 1993
<u>Citrobacter</u> sp.	Burd et al, 1992
<u>Clostridia chauvoei</u>	Smith et al, 1978
<u>Enterobacter</u> sp.	Burd et al, 1992
<u>Erysipelothrix</u> sp.	Spraker, 1993
<u>Escherichia coli</u>	Burd et al, 1992
beta-hemolytic <u>Escherichia coli</u>	Spraker, 1993
<u>Hafnia alvei</u>	Burd et al, 1992
<u>Klebsiella</u> sp.	Burd et al, 1992
<u>Leptospira interrogans pomona</u>	Braham et al, 1980; Smith et al, 1978; Spraker, 1993
<u>Morganella morganii</u>	Burd et al, 1992
<u>Proteus</u> sp.	Burd et al, 1992
<u>Pseudomonas aeruginosa</u>	Burd et al, 1992
<u>Salmonella enteritidis</u>	Gilmartin et al, 1979; Stroud and Roelke, 1980
<u>Salmonella heidelberg</u>	Gilmartin et al, 1979
<u>Salmonella newport</u>	Gilmartin et al, 1979
<u>Salmonella oranienburg</u>	Gilmartin et al, 1979
<u>Salmonella</u> sp.	Burd et al, 1992; Gilmartin et al, 1979; Smith et al, 1978
<u>Serratia</u> sp.	Burd et al, 1992
<u>Staphylococcus aureus</u>	Burd et al, 1992
<u>Streptococcus</u> sp.	Burd et al, 1992

Viruses

Caliciviruses	Smith et al, 1978; 1986
SMSV-1	Smith et al, 1986
SMSV-2	Smith et al, 1986
SMSV-4	Smith et al, 1978
SMSV-5	Smith et al, 1978; 1986
SMSV-8	Smith et al, 1981; 1986
SMSV-10	Smith et al, 1981; 1986
SMSV-11	Smith et al, 1981; 1986
SMSV-12	Smith et al, 1981

Parasites

Trematoda

<u>Cryptocotyle jejuna</u>	Nicholl, 1907; Neiland, 1961
<u>Phocitrema fusiforme</u>	Goto and Ozaki, 1930
<u>Pricitrema callorhini</u>	Yurakhno, 1986
<u>Pricitrema zalophi</u>	Price, 1932; Neiland, 1961

Cestoda

<u>Adenocephalus septentrionacis</u>	Nybelin, 1931; Delyamure, 1955
<u>Diphyllbothrium glaciale</u>	Cholodkovski, 1914; Stunkard, 1948
<u>Diphyllbothrium krooyi</u>	Delyamure, 1955
<u>Diphyllbothrium macrocephalos</u>	Linstow, 1905; Stiles and Hassall, 1912
<u>Diphyllbothrium pacificum</u>	Keyes, 1965
<u>Diplogonoporus tetrapterus</u>	Seibold, 1948; Stunkard, 1948

Nematoda

<u>Contracaecum callotariae</u>	King, 1964
<u>Contracaecum ogmorhini</u>	D'Amelio et al, 1994
<u>Contracaecum osculatum</u>	Rudolphi, 1802; Keyes, 1965
<u>Dipetalonema spirocauda</u>	Leidy, 1858; Anderson, 1959
<u>Porrocaecum callotariae</u>	King, 1964
<u>Porrocaecum decipiens</u>	Krabbe, 1878; Stiles and Hassall, 1899
<u>Uncinaria lucasi</u>	Stiles, 1901; Boltnev, 1996; Lyons et al, 1978; Lyons and Keyes, 1978

Acanthocephala

<u>Bolbosoma bobrovi</u>	Krotov and Delyamure, 1952
<u>Bolbosoma nipponicum</u>	Yamaguti, 1939
<u>Corynosoma semerme</u>	Forssell, 1904; van Cleave, 1953
<u>Corynosoma strumosum</u>	Rudolphi, 1802; van Cleave, 1953
<u>Corynosoma villosum</u>	van Cleave, 1953

Acarina

<u>Orthohalarachne attenuata</u>	Newell, 1947; Ke Chung Kim and Haas, 1980; Kikuchi and Okuyama, 1987
<u>Orthohalarachne diminuata</u>	Doetschman, 1944; Newell, 1947; Ke Chung Kim and Haas, 1980

Ectoparasitic suckling lice, Proechinophthirus fluctus and Anarctophthirus callorhini have also been found on fur seals (Lyons et al, 1978; Lyons and Kim, 1980).

Fur seals can live a maximum of 26 years (Kajimura and Loughlin, 1988; NACOA, 1985). Average lifespan is 2 years for males and 4.6 years for females (NACOA, 1985). There is a record of a fur seal possibly living to 35 years old (Wickens, 1993).

Anthropogenic Interactions and Mortality

Pollution may affect fur seals. PCBs, DDT, DDE, HCH, and other OCs have been detected in fur seal tissues (Bacon et al, 1992; Norstrom and Muir, 1994; Tanabe et al, 1994; Wagemann and Muir, 1984). Schantz et al (1993) found that OC levels are higher in fur seal blubber than in liver, kidney, and muscle. Tanabe et al (1994) reports that PCBs and DDTs

increase in females until six years old and then drastically decrease. Residue levels in females increase slightly after 20 years old (Tanabe et al, 1994). Decreases are due to lactational transfer (Tanabe et al, 1994). Schantz et al (1993) found that DDT>PCB>HCH on St. Paul Island in 1987, but Tanabe et al (1994) found that PCB>DDT>HCH off Sanriku, Japan from 1971 to 1988. Hg, Fe, Mn, Cu, Cd, Pb, and Zn have also been found in tissues (Noda et al, 1995; Wagemann and Muir, 1984). Noda et al (1995) found that Cd levels were higher in fur seals than other marine mammals studied. Cd can cause renal dysfunction (Noda et al, 1995). Heavy metal levels in fur seals vary geographically (Noda et al, 1995). Noda et al (1995) found that Hg in muscle, liver, and kidney, Cd in muscle, and Fe in muscle and liver increase with age and Mn in muscle and kidney and Cu in kidney decrease with age. Fowler (1982) states that increasing levels of toxins in the environment may be causing fur seal declines on the Pribilof Islands, and Young (1981) suggests that increases in toxic substances in the Bering Sea ecosystem may make it unlikely that herds on the Pribilof Island can be rebuilt to their pre-1956 levels.

More than 20,000 fur seals were killed for research on western and eastern North Robben Island from 1958 to 1985 (Chugunkov, 1990). 16,000 females were taken for research on the Pribilof Islands from 1958 to 1974 (York and Hartley, 1981). Air traffic may disturb fur seals if it is too close to haul-outs (Insley, 1992). Ship traffic and construction may also disturb fur seals (Insley, 1992). A fur seal was hit by a car in Berkley, CA in 1982 (Dierauf, 1984).

Fur seals have been harvested since the discovery of their breeding grounds in 1786 (Manning, 1989; Wickens and York, 1997). Products from fur seals include fur, mixed feed for fish, poultry, and fur-bearing animals, fertilizers, glycerin for munitions, oil for tanning, and oriental pharmaceuticals (Lander and Kajimura, 1982). Almost 1 million fur seals, mostly pregnant females, were harvested at sea from 1879 to 1909 (Kajimura, 1984). Commercial harvest of fur seals took place on St. Paul Island until 1984 and on St. George Island until 1972 (Antonelis et al, 1990a; Fowler and Sinclair, 1993; Payne et al, 1996a; Sinclair, 1996). Over 100,000 fur seals, including 30,000-40,000 females, were harvested each year on the Pribilof Islands from the late 1950's to the early 1960's (Boltnev, 1996). Females were targeted for harvest on the Pribilof Islands from 1956 to 1968 (Burd et al, 1992; Fowler, 1982; MMC, 1997; Swartzman and Haar, 1983). About 23,000 females/yr. were killed in the harvest during those years (Loughlin, 1993; Merrick, 1995), for a total of over 300,000 females (NACOA, 1985; Trites and York, 1993; York, 1983; York and Hartley, 1981). Otherwise, harvest was limited to males 2-6 years old (NACOA, 1985). At least 269,685 (at least 24,615/yr.) males 2-5 years old were harvested on St. Paul Island from 1971 to 1983 (Kozloff and York, 1988). Harvest on the Commander Islands exceeded 11,000/yr. through 1973, when lack of males forced hunters to stop harvesting on Bering Island and limit the hunt to Mednyi Island (Boltnev, 1996).

Fur seals are harvested for subsistence (Hill et al, 1997). They are not harvested on Bogoslof Island or San Miguel Island (Fowler and Kozloff, 1988). Subsistence harvest averaged 1,394/yr. on St. Paul Island and 189/yr. on St. George Island from 1986 to 1995 (Hill et al, 1997). Average annual subsistence harvest of the eastern Pacific stock was 1,713 from 1993 to 1995, and only juvenile males were harvested (Hill et al, 1997). NMFS estimates the annual subsistence needs on each of the Pribilof Islands and when the lower limit of harvest is met, harvest is suspended until it can be determined if additional fur seals are needed (MMC, 1997). Stunning followed by exsanguination was determined to be the most humane method of killing fur seals (Lander and Kajimura, 1982; Young, 1981).

Fisheries also cause fur seal mortality (Barlow et al, 1994; 1997; Hill et al, 1997). Commercial fisheries may disturb fur seals by forcing them to move from traditional areas of occupancy (Boltnev, 1996). Fisheries may deplete fur seal prey abundance (Fowler, 1982; Hobson et al, 1997; Loughlin et al, 1987). However, increased pup weights and growth rates of males and females, as well as no increase in foraging time of females, suggests that food shortages from the walleye pollock fishery are not a factor in population declines on the Pribilof Islands (Fowler, 1982; Pitcher, 1990). Fur seals in the CA current area annually consume amounts of northern anchovies, Pacific whiting, Pacific herring, and market squid that are equal to about 13%, 10%, 15%, and 50% of the commercial catch respectively.

Entanglement in discarded trawl net fragments may be an important factor in the decline fur seals on the Pribilof Islands and may account for an extra 15-20% of juvenile mortality (Barlow et al, 1994). Rates of entanglement for males on the Pribilof Islands increased from the mid-1960's to the mid-1970's, reaching a peak of 0.76% among subadult males in 1976 (Robson et al, 1996). Rates of entanglement seem to generally fluctuate between 0.3% and 0.4% on the Pribilof Islands (Fowler and Baba, 1992; Fowler et al, 1993; Fowler and Ragen, 1990; NACOA, 1985; Braham, 1992; Scordino et al, 1988). Fowler (1982) states that about 80% of entangled fur seals may die and about 5.5% of the population on the Pribilof Islands die each year due to net fragments alone. Entanglement survival rate is probably about 50% (Fowler and Baba, 1992; Fowler et al, 1993; Fowler and Ragen, 1990). Male fur seals on the Pribilof Islands are most frequently observed entangled in trawl netting, followed by plastic packing bands; string, small line, and cords; and miscellaneous debris (Fowler and Baba, 1992; Fowler et al, 1993; Fowler and Ragen, 1990; Robson et al, 1996).

Fur seals were the second most commonly incidentally caught pinniped by foreign vessels in the north Pacific and Bering Sea from 1978 to 1981 (Loughlin et al, 1983). 3,150-3,750 females were killed each year in the Japanese mothership fishery for salmon from 1956 to 1963 (Lander and Kajimura, 1982). 100-1,000 fur seals were incidentally taken each year in the Japanese high seas salmon gillnet fishery in the early 1980's (Fowler, 1982). Over 50,000 fur seals may die each year on the Pribilof Islands due to direct fisheries effects (Fowler, 1982). About 22/yr. were killed incidental to both the foreign and joint U.S.-foreign commercial groundfish trawl fisheries in the north Pacific from 1978 to 1988 (Hill et al, 1997). An estimated 5,313 were taken in the Japanese, Taiwanese, and Korean high seas squid fisheries and the Japanese and Taiwanese large-mesh fisheries in 1990 (Hobbs and Jones, 1993). 5,200 were killed incidentally by the foreign high seas driftnet fisheries in 1991 (Hill et al, 1997). However, the foreign and joint U.S.-foreign groundfish trawl fisheries and the foreign high seas driftnet fisheries are no longer in operation (Hill et al, 1997). Fisheries that may take fur seals incidentally are as follows (Barlow et al, 1994; 1997; Baba et al, 1993; Wada et al, 1991; Hill et al, 1997):

Eastern Pacific stock:

- Kodiak, AK peninsula, and south Unimak salmon driftnet and set net
- AK pollock and other groundfish trawl
- Japanese high seas salmon gillnet
- Japanese salmon trap net
- Bering Sea/Aleutian Island groundfish trawl
- Prince William Sound salmon drift gillnet
- AK peninsula/Aleutian Island salmon drift gillnet

Bristol Bay salmon drift gillnet
San Miguel Island stock:
CA/OR thresher shark and swordfish drift gillnet
CA angel shark/halibut set gillnet
WA/OR/CA groundfish trawl (Pacific whiting component)
CA/OR thresher shark and swordfish drift gillnet
CA angel shark/halibut set gillnet

No mortalities of the San Miguel Island stock were reported in any observed fishery from 1990 to 1995 (Barlow et al, 1997). Logbooks indicate 3 mortalities from 1990 to 1993 (Barlow et al, 1997). Minimum annual incidental fisheries mortality of fur seals in the San Miguel Island stock is 0/yr. (Barlow et al, 1997). This is less than 10% of PBR (21.6) and is considered insignificant (Barlow et al, 1997). Total minimum annual human-caused mortality for this stock is 0/yr. (Barlow et al, 1997). This is less than PBR (216) and is considered insignificant (Barlow et al, 1997). Minimum annual incidental fisheries mortality of fur seals in the eastern Pacific stock is 18/yr. (Hill et al, 1997). This is less than 10% of PBR (2,085) and is considered insignificant (Hill et al, 1997). Total minimum annual human-caused mortality for this stock is 1,731/yr. (Hill et al, 1997). This is less than PBR (20,846) and is considered insignificant (Hill et al, 1997). Intentional killing of fur seals may occur in fisheries, but the magnitude of this mortality is unknown (Hill et al, 1997; NACOA, 1985).

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Review of California sea lions (Zalophus californianus) in the North Pacific 1978-1997

Population, Abundance, and Distribution

Range:

Subspecies: Z. c. wolfebaeki

Galapagos Islands (1 °S); one sighted at Isla del Coco 500 km southwest of Costa Rica

Z. c. japonicus

Sea of Japan (34-37 °N) (may be extinct)

Z. c. californianus

Males: From about 19 °N up to Bull Harbor, Vancouver Island, British Columbia (51 °N); one was seen in Acapulco, Mexico; one was seen on Erlington Island, Prince William Sound, Gulf of AK

Females: Baja CA, Mexico up to Año Nuevo Island, CA (37 °N)

Breeding Range (Z. c. californianus):

Isla San Margarita, Baja California, Mexico to Año Nuevo Island, central California; mostly in Channel Islands; northern most rookery is on San Miguel Island

Preferred Habitat (Z. c. californianus):

Rocky beaches on offshore islands

California sea lions are generally divided into three subspecies: Z. c. wolfebaeki inhabits the Galapagos Islands, Z. c. japonicus inhabits Japan (but may be extinct), and Z. c. californianus ranges from Baja CA to British Columbia (Barlow et al, 1997; Mate, 1982). Phylogenetic evidence suggests that the third subspecies may be broken down into at least two stocks: Gulf of CA and Pacific coast (Maldonado et al, 1995). Barlow et al (1997) separate Z. c. californianus into three stocks: U.S. stock, which ranges from the U.S./Mexico border to northern Canada; western Baja CA stock, which ranges from the U.S./Mexico border to the southern tip of Baja CA; and Gulf of CA stock, which ranges from the Gulf of CA to the southern tip of Baja CA across to mainland southern Mexico. Some movement has been documented between stocks, but rookeries in the U.S. are widely separated from major rookeries of western Baja CA, Mexico (Barlow et al, 1997).

The California sea lion is considered the most abundant pinniped in coastal CA waters (Antonelis et al, 1984). Population was less than 800 in CA by 1927 due to extensive hunting (Brueggeman et al, 1990). Population also decreased in the late 1960's and early 1970's due to Leptospira pomona infection (Braham et al, 1980). However, pup production of the U.S. stock increased 5.4%/yr. from 1975-95, experiencing significant declines only during severe El Niño events, such as those in 1976, 1983, and 1992 (Barlow et al, 1997). Major rookeries are located on the Channel Islands, and population on these islands increased over 133% from 1975 to 1993 (Lieberg-Clark et al, 1995). San Miguel and San Nicholas Islands have the main Channel Island rookeries (Brueggeman et al, 1990; NMFS, 1996). Population of the U.S. stock was 167,000-

188,000 in 1995 (Barlow et al, 1997). N_{min} of this stock is 111,339 and PBR is 6,680 (Barlow et al, 1997).

Some of the population growth in CA may be due to immigration from Mexico. Counts declined 40% at the two largest Mexican rookeries (Islas San Benito and Cedros) from 1976 to 1981 (Bonnell and Ford, 1987). Pup production was 45% lower on Santa Margarita Island, Baja CA in 1983 (a severe El Niño year) than in the years preceding and succeeding; abundance of females decreased 59% on the island from 1982 to 1983 (Aurioles and LeBoeuf, 1991). Zavala-González and Melink (1997) report that studies have determined the population along the Pacific coast of Mexico to be 74,467, and NMFS (1993) reports that population was 74,500 in western Baja CA in 1992.

Zavala-González and Melink (1997) state that population in the Gulf of CA increased 30% at some breeding colonies from the 1960's to the 1980's, and annual increase of those populations was between 2% and 4.7% in the 1980's and early 1990's. They mention that some populations experienced a slight decline and later, a partial recovery in 1991. However, Harcourt et al (1994) report that the population in the Gulf of CA is neither increasing nor decreasing. Population was 6,000 in the Gulf of CA in 1969 and 14,000 in 1979 (Aurioles et al, 1979c). According to Aurioles-Gamboa and Zavala-Gonzalez (1994), population in the Gulf of CA was 31,393 in the 1980's. Zavala-González and Melink (1997) report that studies have determined the population in the Gulf of CA to be 28,220.

Population of the Galapagos subspecies was 20,000 in the early 1960's (Mate, 1982). Little information is available on this subspecies. The Japanese subspecies is now thought to be extinct (Mate, 1982; Barlow et al, 1997).

Typically, California sea lions are found in coastal waters over the continental shelf (Brueggeman et al, 1990). Some local populations fluctuate significantly (Mate, 1982), but the range-wide population seems to be increasing (Barlow et al, 1997). Populations display seasonal abundances (Jeffries, 1984a; 1984b; Huber, 1991; Everitt et al, 1981), and distribution may be affected by concentrations of prey (Aurioles-Gamboa and Zavala-Gonzales, 1994; Antonelis and Fiscus, 1980; Roffe, 1980) or by competition with Steller sea lions (Mate, 1982). As populations increased in the 1970's, immature males began dispersing further north during the non-breeding season (Huber, 1991), and increases in northern populations may be causing range widening dispersals to the south (Gallo and Ortega, 1986). There is no proof, except one skull found in the late 1800's, of California sea lions inhabiting British Columbia in the early 1900's, but population began increasing there in the 1960's (Bigg, 1988). Population increased ten fold on Vancouver Island from 1972 to 1984 (Bigg, 1985; Bigg, 1988). Population in North America was more than 175,000 in 1991 (NMFS, 1992).

California sea lions are considered migratory (Aurioles et al, 1981a; Barlow et al, 1997; Everitt et al, 1981). Most males migrate north after the summer breeding season (Aurioles et al, 1983; Barlow et al, 1997; Brueggeman et al, 1990; Kajimura and Loughlin, 1988), but some Gulf of CA males may migrate south (Mate, 1982). Adult males that breed in Baja CA migrate north to the Channel Islands in the winter (NMFS, 1996), and adult males that breed in the Channel Islands migrate as far north as British Columbia in the winter (NMFS, 1996; Bigg, 1985). Migration only shifted northward to commonly include British Columbia beginning in the middle of this century (Bigg, 1988; Huber, 1991). Females generally do not migrate far outside the breeding range (Mate, 1982; NMFS, 1996; U.S. Air Force, 1996a; 1996b). Males

may migrate more freely between populations than females, and females may display regional philopatry (Maldonado et al, 1995).

California sea lions are not considered endangered or threatened under the MMPA (Barlow et al, 1997).

Food and Feeding

California sea lions are opportunistic predators that consume a wide variety of fish, crustaceans, mollusks, birds, and other organisms (Antonelis and Fiscus, 1980; Antonelis et al, 1984; Long and Gilbert, 1997). They have different prey preferences depending upon location and season (Antonelis et al, 1984). Fluctuations in prey preferences may reflect prey availability (Antonelis et al, 1984). California sea lions take advantage of seasonal and geographical abundances of prey species, and some populations may follow certain anadromous prey species, including lamprey as well as salmonids (Lowry et al, 1991; Bigg et al, 1990; Roffe, 1980). Antonelis et al (1990) state that California sea lions feed principally in the neritic zone. Kajimura and Loughlin (1988) report that California sea lions in the Gulf of AK feed mostly from coastal areas to the continental slope.

California sea lions are of great concern to fisherpeople. However, these pinnipeds do not actively seek out commercially valuable species but take advantage of seasonal abundances (Bigg et al, 1990; Roffe, 1980). In most locations, California sea lions do not seem to be a threat to fish stocks, but there are isolated cases, such as steelhead (Oncorhynchus mykiss, formally Salmo gairdneri) predation at Ballard Locks, WA, where they may be a threat (General Accounting Office, 1993; Hollingshead et al, 1996; MMC, 1997). However, California sea lions also prey on competitors and other predators of commercially valuable species (Roffe and Mate, 1984; Roffe, 1980). For example, California sea lions prey on large quantities of lamprey, which parasitize salmonids and other valuable species (Roffe and mate, 1984; Roffe, 1980; Jeffries, 1984b, Jameson and Kenyon, 1977). California sea lions have also been observed depredating sport fish caught by anglers aboard commercial passenger fishing vessels (Hanan et al, 1989).

Bigg et al, (1990) observed California sea lions in British Columbia and found that they forage individually and tend to forage in deep waters, far from land. Kajimura and Loughlin (1988) state that California sea lions feed throughout the day and night and night day dives are deeper than night dives. Antonelis et al (1990) found that tagged females from San Miguel Island foraged an average of 54.2 km offshore and 323 m deep. They also found that average trip duration was 73.4 hrs. Females were found to start postpartum foraging trips earlier and forage longer during the El Niño of 1983 (Ono et al, 1987). Galapagos sea lions forage during the day, mainly between 20 and 60 m deep for durations of about one day (Trillmich and Limberger, 1985). The following prey species have been determined for California sea lions:

<u>Category</u>	<u>Scientific Name</u>	<u>Common Name</u>
Molluska	<u>Abraliopsis</u> sp.	Enope squid
	<u>Chroteuthis calyx</u>	Whiplash squid
	Gonatidae	Gonata squid
	<u>Gonatopsis borealis</u>	Gonata squid
	<u>Gonatus</u> sp.	Gonata squid

	<u>Histioteuthis</u> sp.	Umbrella or jewel squid
	<u>Loligo opalescens</u>	Market squid
	<u>Moroteuthis robusta</u>	North Pacific giant squid
	Octopoda	Octopi
	<u>Octopoteuthis deletron</u>	Octopus squid
	<u>Octopus bimaculatus</u>	Two-spotted octopus
	<u>Octopus rubescens</u>	Red octopus
	<u>Octopus</u> sp.	Octopodid octopus
	<u>Ocythoe tuberculata</u>	Ocythoid octopus
	<u>Onychoteuthis borealijaponicus</u>	Nail squid
Crustacea		
	<u>Crangon</u> sp.	Crangon shrimp
	<u>Pleuroncodes planipes</u>	Pelagic red crab
Agnatha		
	<u>Lampetra tridentata</u>	Pacific lamprey
	(<u>Entosphenus tridentatus</u>)	
Chondrichthys		
	<u>Galeorhinus zyopterus</u>	Soupfin shark
	<u>Hydrolagus coliei</u>	Ratfish
	<u>Prionace glauca</u>	Blue shark
Osteichthys		
	<u>Allosmerus elongatus</u>	White smelt
	<u>Anoplopoma fimbria</u>	Sablefish
	<u>Antherinopsis californiensis</u>	Jacksnelt
	<u>Anthias</u> sp.	Threadfin bass
	<u>Argentina sialis</u>	Pacific argentine
	Atherinidae	Silversides
	<u>Atherinops affinis</u>	Topsmelt
	<u>Aulopus</u> sp.	Cusk eel
	<u>Bathylagus stilbius</u>	California smoothtongue
	<u>Calamus brachysomus</u>	Pacific porgy
	<u>Careproctus melanurus</u>	Blacktail snailfish
	<u>Ceratoscopelus townsendi</u>	Dogtooth lampfish
	<u>Chilara taylori</u>	Spotted cusk-eel
	<u>Chromis punctipinnis</u>	Blacksmith
	<u>Citharichthys sordidus</u>	Pacific sanddab
	<u>Citharichthys</u> sp.	Sanddab
	<u>Citharichthys xanthostigma</u>	Longfin sanddab
	<u>Clupea pallasii</u>	Pacific herring
	<u>Cololabis saira</u>	Pacific saury
	Cottidae	Sculpins
	<u>Cymatogaster aggregata</u>	Shiner surfperch
	<u>Cypselurus californicus</u>	California flying fish
	<u>Diplectrum</u> sp.	Sea bass
	Embiotocidae	Surfperches

<u>Engraulis mordax</u>	Northern anchovy
<u>Euthynnus lineatus</u>	Benito
Exocoetidae	Flying fish
<u>Genyonemus lineatus</u>	White croaker
<u>Girella nigricans</u>	Opaleye
<u>Glyptocephalus zachirus</u>	Rex sole
<u>Gradus macrocephalus</u>	Pacific cod
<u>Haemulopsis</u> sp.	Sargo
<u>Hemanthias</u> sp.	Splitail bass
<u>Hippoglossina stomata</u>	Bigmouth sole
<u>Icelinus tenuis</u>	Spotfin sculpin
<u>Icichthys lockingtoni</u>	Medusafish
Labridae	Wrasses
<u>Lepophidium</u> sp.	Cusk eel
<u>Lepophidium prorates</u>	Pink cusk eel
<u>Leptocottus armatus</u>	Pacific staghorn sculpin
<u>Leuroglossus stilbius</u>	California smoothtongue
<u>Lycodes cortezianus</u>	Bigfin eelpout
<u>Lyopsetta exilis</u>	Slender sole
<u>Medialuna californiensis</u>	Halfmoon
<u>Merluccius angustimanus</u>	Panama hake
<u>Merluccius productus</u>	Pacific hake/Pacific whiting
<u>Microgadus proximus</u>	Pacific tomcod
<u>Microstomus pacificus</u>	Dover sole
Myctophidae	Lanternfishes
<u>Neobythites</u> sp.	Cusk eel
<u>Oncorhynchus</u> sp.	Salmon
<u>Oncorhynchus keta</u>	Chum salmon
<u>Oncorhynchus kisutch</u>	Coho salmon
<u>Oncorhynchus mykiss</u>	Steelhead salmon/trout
<u>Oncorhynchus tshawytscha</u>	Chinook salmon
Ophidiidae	Cusk eels
<u>Ophiodon elongatus</u>	Lingcod
<u>Ophidion scrippsae</u>	Basketweave cusk eel
Osmeridae	Smelts
<u>Oxyjulis californica</u>	Señorita
<u>Paralabrax clathratus</u>	Kelp bass
<u>Paralabrax</u> sp.	Seabass
<u>Paralichthys californicus</u>	California halibut
<u>Parophrys ventulus</u>	English sole
<u>Peprilus simillimus</u>	Pacific pompano
<u>Phanerodon furcatus</u>	White seaperch
Pleuronecidae	Flatfish
<u>Pontinus</u> sp.	Scorpion fish
<u>Porichthys myriaster</u>	Spekledfin midshipman

	<u>Porichthys notatus</u>	Plainfin midshipman
	<u>Porichthys sp.</u>	Midshipman
	<u>Prionotus stephanophrys</u>	Lumptail searobin
	<u>Pronotogrammus sp.</u>	Sea bass
	<u>Psettichthys melanostictus</u>	Sand sole
	<u>Salmo gairdneri</u>	Steelhead
	<u>Sardinops sagax</u>	Pacific sardine
	<u>Scomber japonicus</u>	Pacific (chub) mackerel
	Scombridae	Mackerels or Tunas
	Scorpaenidae	Rockfish
	<u>Sebastes sp.</u>	Rockfish
	<u>Sebastolobus alascanus</u>	Shortspine thornyhead
	<u>Semicossyphus pulcher</u>	Sheephead
	<u>Seriphus politus</u>	Queenfish
	<u>Serranus sp.</u>	Sea bass
	<u>Serranus aequidens</u>	Serrano
	<u>Stenobranchius leucopsarus</u>	Northern lampfish
	<u>Symbolophorus californiensis</u>	California lanternfish
	<u>Synodus jenkinsi</u>	Yellowbelly lizardfish
	<u>Synodus lucioceps</u>	California lizardfish
	<u>Thaleichthys pacificus</u>	Eulachon
	<u>Theragra chalcogramma</u>	Walleye pollock
	<u>Trachurus symmetricus</u>	Jack mackerel
	<u>Xeneretmus ritteri</u>	Stripefin poacher
	<u>Xeneretmus sp.</u>	Poacher
	<u>Xeneretmus triacanthus</u>	Bluespotted poacher
	<u>Zalembeus rosaceus</u>	Pink sea perch
	<u>Zaniolepis sp.</u>	Combfish
	Zoarcidae	Eelpouts
Aves		
	<u>Uria aalga</u>	Common murre
Mammalia		
	Pinnipedia	Seals and sea lions
Other		
		Algae
		Milk

Habits

Female California sea lions tagged on San Miguel Island were found to dive between 18 and 1,556 m deep (Antonelis et al, 1990). Average depth was 323 m. Average distance from shore was 54.2 m. Average time spent ashore between trips was 50.1 hrs., and average time spent at sea during a trip was 73.4 hrs. 42% of all departures and 34% of all arrivals occurred between 24:00 and 07:00 hrs. The fastest swimming speed recorded by Antonelis et al, (1990) was 11.5 km/hr.

Reproduction

(Z. c. californianus)

Breeding season: May to July; latitudinal pupping clines observed in captives

Pupping season: Mid-May to mid-July

Sexual maturity:

Males: 7-9 yrs. old

Lactation: 9-12 months or longer (rarely exceeds 2 yrs.) (Galapagos sea lion lactates about 1 yr.)

California sea lions gather on rookeries from the Channel Islands to Baja CA to mate and pup (DeLong et al, 1991; Aurioles and LeBoeuf, 1991). There are large rookeries on San Miguel and San Nicholas Islands and smaller rookeries on Santa Barbara and San Clemente Islands (Brueggeman et al, 1990). San Miguel Island has the northern most rookery, although a few pups are occasionally born farther north (Brueggeman et al, 1990). A relationship between photoperiod and reproductive timing has been suggested (Temte, 1985).

Pup production decreased drastically during the 1983 El Niño (Aurioles and LeBoeuf, 1991; DeLong et al, 1991), but pup counts increased an average of 8.3%/yr. 1983-95 (Barlow et al, 1997). In the Gulf of CA, the 1983 El Niño had little effect on pup production (Aurioles and LeBoeuf, 1991). Mate (1982) points out that there is little known about the reproductive biology of California sea lions, and pregnancy rates, ages of sexual maturity, and longevity are poorly known.

California sea lion and northern fur seal hybrids occur on San Miguel Island (DeLong and Melin, 1992).

Mortality and Economics

Natural Mortality

California sea lions are preyed upon by sharks, including white sharks (Carcharodon carcharias) and orcas (Ainley et al, 1980; Baird and Dill, 1995; Baird and Stacey, 1989; Bodkin and Jameson, 1991; Mate, 1982; Scholl, 1983; Hoyt, 1984; Sobolevsky and Mathisen, 1996; Kajimura and Loughlin, 1988). Western gulls (Larus occidentalis) have also been observed preying on pups on San Margarita Island (Aurioles and Llinas, 1987). Also, a Steller sea lion was observed eating a small California sea lion on Año Nuevo Island (Byrnes and Hood, 1994).

Weights of male and female pups declined 25-35% on San Miguel and San Clemente Islands from 1982 to 1986 (DeLong et al, 1991). Neonatal and juvenile mortality increased in 1983 on most of the breeding islands, and adult female mortality probably also increased (DeLong et al, 1991). Pup mortality in the first six months of life was 20% in the Gulf of CA in the mid-1980's, and annual pup mortality was 50-60% (Aurioles and LeBoeuf, 1991). The 1983 El Niño event appeared to decrease health and increase mortality (Francis and Heath, 1991; DeLong et al, 1991). Thirty immature California sea lions washed up dead or died on the South Farallon Islands from March, 1983 to March, 1984; previously, only 1-2 washed ashore dead each year (Huber, 1991). Of 23,000 California sea lions born on San Miguel Island in 1997, 1,200 died by September because of the severe 1997 El Niño (Anonymous, 1997).

Pup mortality of Galapagos sea lions within the first 7 months was found to be 5% on Santiago in 1977 (Trillmich and Limberger, 1985). Pup mortality within the first 2 months was found to be 5% on Fernandina in 1977 (Trillmich and Limberger, 1985). The 1983 El Niño event increased pup mortality of the Galapagos sea lion, causing pup production to be less than 30% of normal on most islands and reducing the yearling age class to 5-20% of its normal size (Trillmich and Limberger, 1985).

Diseases, such as leptospirosis and SMSV, have been linked to such problems as premature births and reproductive failure (Addison, 1989; Calambokidis et al, 1984). California sea lions experienced outbreaks of leptospirosis that affected population numbers in the 1970's (Harvey and Hurley, 1996; Mate, 1982), and Berry et al (1990) state that high antibody presence in San Miguel Island pups indicates a recent epizootic. Gulland et al (1996a) state that leptospirosis epizootics occurred in CA in 1984, 1988, 1991, and 1994. Additionally, salmonella has been found to cause mild gastroenteritis and may be a problem for pups or others that become debilitated for other reasons (Gilmartin et al, 1979). Salmonella may also cause abortion (Gilmartin et al, 1979). At least 24 parasites, 5 bacteria, and 26 viruses have been found in or on California sea lions:

Bacteria

Coccidioidomycosis

Escherichia coli

Leptospira interrogans serovar pomona

Leptospira pomona

Leptospira pomona kenniwicki

Pneumonia

Salmonella

Serotypes:

Oranienburg

Newport

Fauquier et al, 1996

Diamond et al, 1980

Smith et al, 1978

Braham et al, 1980; Calambokidis et al, 1984

Gulland et al, 1996a

Anderson et al, 1990

Gilmartin et al, 1979

Viruses

Adenovirus-like viron

Cancer

Squamous cell carcinoma

Britt et al, 1979

Anderson et al, 1990; Joseph et al, 1986

Metastatic tumors

Cetacean calicivirus

SMSV

SMSV-1

SMSV-2

SMSV-3

SMSV-4

SMSV-5

SMSV-6

SMSV-7

SMSV-8

SMSV-9

SMSV-10

SMSV-11

SMSV-12

SMSV-13

Tillamook calicivirus

VESV-A

VESV-C

VESV-D

VESV-E

VESV-F

VESV-G

VESV-I

VESV-J

VESV-K

Brown et al, 1980; Gulland et al, 1996a

Smith et al, 1986

Barlough et al, 1988

Smith et al, 1978; 1986

Smith et al, 1978; 1986

Smith et al, 1986

Smith et al, 1978; 1986

Smith et al, 1978; 1986

Barlough et al, 1988; Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1981; Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1981; Smith et al, 1986

Berry et al, 1990; Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Smith et al, 1986

Parasites

Trematoda

Heterophyes heterophyes

Pricetrema zalophi

Schistosoma haematobium

Schistosoma mansoni

Stephanoprora denticulata

Stictodera ubelakeri

Zalophotrema hepaticum

Zalophotrema sp.

King, 1964

Price, 1932; Ciurea, 1933

King, 1964

King, 1964

Rudolphi, 1802; Price, 1932

Dailey, 1969

Stunkard and Alvey, 1929; De Leon, 1991; Joseph et al, 1986

Barlough et al, 1988

Cestoda

Diphyllbothrium pacificum

Dailey and Brownell, 1972

Nematoda

Anasakis similis

Contracaecum ogmorhini

Baird, 1853; Herman, 1942

D'Amelio et al, 1994

<u>Contracaecum osculatum</u>	Rudolphi, 1802; Herman, 1942; Alarcon-Gonzalez et al, 1988; Joseph et al, 1986
<u>Dipetalonema odendhali</u>	Perry, 1967
<u>Dirofilaria immitis</u>	Leidy, 1856; Faust, 1937
<u>Dujardinia</u> sp.	Herman, 1942
<u>Parafilaroides decorus</u>	Dougherty and Herman, 1947; Joseph et al, 1986
<u>Porrocaecum</u> sp.	Herman, 1942
<u>Uncinaria</u> sp.	Dailey and Brownell, 1972
Acanthocephala	
<u>Corynosoma obtusens</u>	Lincicome, 1943
<u>Corynosoma osmeri</u>	Fujita, 1921
Acarina	
<u>Orthohalarachne diminuta</u>	Doetschman, 1941
<u>Demodex</u> sp.	Kenney, 1968
<u>Demodex zalophi</u>	Dailey and Nutting, 1980; Nutting and Dailey, 1980
Anoplura	
<u>Antarctophthirius microchir</u>	Enderlein, 1906; Ferris, 1916

Anthropogenic Interactions and Mortality

Pollution is a serious threat to pinniped populations (Mate, 1982). PCBs, DDT, DDE, and heavy metals have been detected in California sea lion tissues (Calambokidis et al, 1985; Lieberg-Clark et al, 1995; Wagemann and Muir, 1984; O'Shea and Brownell, 1994). California sea lions between San Francisco and Los Angeles were found to contain an "extraordinary" amount of DDT in the fall of 1970 (Addison, 1989). Bacon et al (1992) found that DDE levels in California sea lions increased to the south. They also found that California sea lions had the highest DDE levels of several pinniped species tested. High OC levels may be linked to premature birth (Addison, 1989; Corsolini et al, 1995; De Guise et al, 1995), abortion (De Swart et al, 1994), or other reproductive problems in California sea lions (Lieberg-Clark et al, 1995; Reijnders, 1994). OCs may also be related to leptospirosis and SMSV and other calicivirus infections (Addison, 1989; De Swart et al, 1994). Oil spills can also detrimentally affect California sea lions (Bodkin and Jameson, 1991).

Humans also cause mortality by incidental and direct kills. Population was less than 800 in CA by 1927 due to extensive hunting (Brueggeman et al, 1990). Subsistence harvest is allowed under the MMPA (Barlow et al, 1997). Several northwestern Indian tribes are developing or have developed subsistence harvests of California sea lions in the U.S. stock (Barlow et al, 1997).

Other non-fishery sources of anthropogenic mortality include boat collisions, entrainment in power plants, and shooting (Barlow et al, 1997). Of 24 California sea lions found dead and necropsied on Santa Catalina Island from 1983 to 1991, 63% had evidence of gunshot wounds (Shane, 1994). The proportion of U.S. stock California sea lions observed on land as entangled in debris ranges from 0.08-0.35% (Barlow et al, 1997; Harcourt et al, 1994). Harcourt et al

(1994) found that entanglement rate (excluding pups) was 3.9-7.9% on Los Islotes, Baja CA in the summer of 1992. Minimum mortality by guns, boat collisions, entrainment in power plants, marine debris, and gaffs was 52 for the U.S. stock in 1995 (Barlow et al, 1997). El Niño may also have a detrimental effect on populations (Harvey and Hurley, 1996; Ono et al, 1987).

Fisheries cause mortality directly through incidental and illegal kills. Many California sea lions are shot in conjunction with fisheries (Barlow et al, 1997; Jeffries, 1984b). They are deliberately killed illegally in the southern Gulf of CA to use for baiting shark longlines (Zavala-González and Melink, 1997). Fisheries that take California sea lions incidentally and annual mean mortality are as follows (Barlow et al, 1997):

U.S. stock	
CA driftnet for sharks and swordfish	49 (CV=0.21)
CA set gillnet for halibut and angel shark	815 (CV=0.09)
OR Columbia River gillnet	19 (CV=0.54)
CA, OR, and WA salmon troll	
WA Puget Sound salmon drift gillnet	12
CA herring purse seine	
CA anchovy, mackerel, and tuna purse seine	
CA squid purse seine	
WA, OR, CA ground fish trawl	0.4
WA, OR, CA commercial passenger fishing vessel	2
CA salmon net pen	0.2
WA, OR salmon net pen	
Canada BC salmon pen	17
British Columbia herring gillnet and seine	

Minimum annual incidental fisheries mortality of the U.S. stock of California sea lions is 915/yr. (Barlow et al, 1997). Annual human caused mortality is 974/yr. This exceeds 10% of PBR (668) and is considered significant (Barlow et al, 1997).

California sea lions negatively impact some fisheries. In British Columbia, they prey on salmon and halibut and damage gear and captured fish (Bigg, 1984). They also disperse schools of herring and enter nets in herring gillnet and seine fisheries (Bigg, 1984). Additionally, they are suspected of feeding on inshore stocks of cod (Bigg, 1984). In the U.S., California sea lions prey on north Pacific salmon that are caught in fishing gear, causing catch loss and gear damage (Fiscus, 1980). A serious problem has occurred at Ballard Locks, Seattle, WA, where some California sea lions are preying so heavily on steelhead (Oncorhynchus mykiss formerly Salmo gairdneri) as to threaten the existence of steelhead in the area (General Accounting Office, 1993). Relocation and driving off California sea lions do not seem to be successful at thwarting the predation (General Accounting Office, 1993; Braham, 1992). However, the rate of predation at Ballard Locks dropped to 8% of the total wild steelhead run from the 50-65% that were taken from 1986 to 1991 (Hollingshead et al, 1996). In 1994, amendments added to the MMPA set forth a process for authorization of intentional lethal taking of individually identifiable pinnipeds that are having a significant negative impact on salmonids that are either listed or approaching listing under the ESA or migrating through Ballard Locks (Hollingshead et al, 1996; MMC, 1997). On January 4, 1995, NMFS issued a Letter of Authorization to WA state for the lethal removal of individually identified California sea lions from Ballard Locks (Hollingshead et al,

1996; MMC, 1997). However, none were lethally removed in 1995 or 1996 (MMC, 1997). In 1995, one was captured, held, and released and two were captured, transported, and released, and in 1996, California sea lions thought to be the primary cause of predation were captured and removed to permanent captivity at Sea World, Orlando, FL (MMC, 1997).

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