the criteria contained within the CTR may not be reached for up to 10 years. All site-specific criteria must be approved by the EPA and are therefore subject to consultation pursuant to section 7 of the Act.

DESCRIPTION OF THE ACTION AREA

The CTR covers surface waters in California, which are waters of the United States, and which have been designated as inland surface waters or enclosed bays and estuaries. These include all watersheds with their rivers, streams, channels, lakes, ponds, enclosed bays and estuaries in California. Ocean water is not covered by the CTR, because the State of California already has a valid statewide plan to control ocean water quality. This proposed rule does not change or supersede any criteria previously promulgated for the State of Califomia in the NTR, as amended. This proposed rule is not intended to apply to waters within Indian Country (*sic*).

The CTR is a statewide rulemaking process promulgating water quality criteria for all parts of California, with limited exceptions, where water quality criteria have been adopted for specific water bodies. For instance, the selenium criteria for the San Francisco Bay have already been promulgated under the NTR. For a complete list of such exceptions see footnotes “o” through “t” to the table listing all priority toxic pollutants in the CTR itself.

Water quality criteria previously promulgated within the NTR (but not previously consulted on) are considered in this opinion for adequacy of protection of listed species. EPA has not provided the Services with a list of waters for which the CTR does not apply and therefore, the Services have considered all waters within the State equally.

SPECIES DESCRIPTIONS

Aleutian Canada Goose (*Branta canadensis leucoparia*)

Species Description and Life History: The Aleutian Canada goose was listed as threatened on December 12, 1990 (55 **FR** 51112). This subspecies was originally classified as endangered on March 11, 1967.

The Aleutian Canada goose can be distinguished from most other subspecies of Canada geese by their small size (only cackling Canada geese are smaller) and a ring of white feathers at the base of the black neck in birds older than 8 months. Lakes, reservoirs, ponds, large marshes, and flooded fields are used for roosting and loafing (Grinnell and Miller 1944, USDI-FWS 1982a).

Foraging Ecology: Aleutian Canada geese forage in harvested corn fields, newly planted or grazed pastures, or other agricultural fields (e.g., rice stubble and green barley).

Historic and Current Distribution: Historically, Aleutian Canada geese wintered from British

Columbia to California and northwestern Mexico (Delacour 1954). Although they occurred throughout California, the greatest concentrations were found in the Sacramento and San Joaquin Valleys (Grinnell and Miller 1944).

The subspecies nested throughout the Aleutian Islands and into Russia (Springer 1977). Predation by introduced arctic foxes eliminated most breeding colonies of the Aleutian Canada goose, and by 1962 the subspecies was nearly extinct, with only one breeding colony remaining on the tiny island of Buldir. This island was one of the few to escape the introduction of arctic foxes (USDI-FWS 1982a). In 1982, a new or remnant breeding population of Aleutian Canada geese of unknown size was discovered on Chagulak Island in the Islands of the Four Mountains (USDI-FWS 1982a).

The present population of Aleutian Canada geese migrates along the northern California coast and winters in the Central Valley near Colusa, and on scattered feeding and roosting sites along the San Joaquin River from Modesto to Los Banos (Nelson *et al.* 1984). Fall migration usually begins in late August or early September, with birds arriving in the Central Valley between October and early November. Spring migration usually occurs from mid-February to early March.

In California, the Aleutian goose occurs on agricultural lands along the north coast, and throughout the Sacramento and San Joaquin Valleys. Major migration and wintering areas include agricultural lands north of Crescent City in Del Norte County, around the Sutter Buttes in the Sacramento Valley, near El Sobrante in Contra Costa County, and along the San Joaquin River between Modesto and Los Banos.

<u>Reasons for Decline and Threats to Survival</u>: Predation by introduced arctic foxes on the breeding islands is the primary reason for the population decline. Avian cholera is currently a major threat to the concentrations of Aleutian Canada geese in the Central Valley. In 1991, 58 geese died during an outbreak of avian cholera in the San Joaquin Valley (USDI-FWS 1991). This subspecies is particularly vulnerable to cholera outbreaks because most of the population overwinters in a small geographical area. Sport hunting on its wintering grounds in California and by natives on the nesting grounds also contributed to the species' decline (USDI-FWS 1982a). At one time, recreational and subsistence take of this subspecies in the Pacific Flyway may have been a significant factor preventing the remnant breeding segments from recovering.

Changing land use practices in the wintering range, including the conversion of cropland and pastures to housing and other urban development, adversely affect Aleutian geese (USDI-FWS 1991). The lack of adequately protected migration and winter habitat for Aleutian geese is the greatest obstacle to full recovery of this species (USDI-FWS 1991). Habitat quality has also likely declined due to the concentrated effects of pollution, human disturbance, and disease (USDI-FWS 1991).

Bald Eagle (*Haliaeetus leucocephalus*)

Species Description and Life History: The bald eagle was federally listed as endangered on February 14, 1978 (43 **FR** 6233) in all of the coterminous United States except Minnesota, Wisconsin, Michigan, Oregon, and Washington, where it was classified as threatened. On August 15, 1995 (60 **FR** 36010), the bald eagle was down-listed to threatened throughout its range. Critical habitat has not been designated for the bald eagle. On July 6, 1999, the Service published a proposed rule to remove the bald eagle from the federal list of threatened and endangered species (64 **FR** 36454). The recovery plan for the Pacific population of the bald eagle describes the species biology, reasons for decline, and the actions needed for recovery (USDI-FWS 1986b).

The Pacific Recovery Region for the bald eagle includes the States of California, Oregon, Washington, Idaho, Montana, Wyoming, and Nevada. Other recovery plans exist for bald eagle populations in the Southeast, Southwest, Northern States, and Chesapeake Bay. Delisting/reclassification of the bald eagle in the Pacific Recovery Region is not dependent on the status of bald eagle populations covered by these other plans (USDI-FWS 1986b). For this reason, the Pacific Recovery Region for the bald eagle will be viewed as a recovery unit for purposes of this consultation.

Foraging Ecology: The bald eagle is a generalized predator/scavenger primarily adapted to edges of aquatic habitats. Typically fish comprise up to 70% of the nesting eagle diet with mammals, birds, and some amphibians and reptiles providing the balance of the diet. Wintering eagles forage fish, waterfowl, mammals, and a variety of carrion. Bald eagles can maneuver skillfully and frequently hunt from perches. They are also known to hunt by coursing low over the ground or water.

Historic and Current Distribution: The bald eagle is the only North American representative of the fish or sea eagles, and is endemic to North America. The breeding range of the bald eagle includes most of the continent, but they now nest mainly in Alaska, Canada, the Pacific Northwest states, the Great Lake states, Florida, and Chesapeake Bay. The winter range includes most of the breeding range, but extends primarily from southern Alaska and southern Canada, southward.

As of 1996, about 5,068 occupied bald eagle territories were estimated within its range. Of these, 1,274 (25 %) were estimated to occur within the Pacific Recovery Region. Within the 7-State Pacific Recovery Region, 105 occupied territories occurred in California, 90 in Idaho, 165 in Montana, 0 in Nevada, 266 in Oregon, 582 in Washington, and 66 in Wyoming (Jody Millar, Bald Eagle Recovery Coordinator, FWS, pers. comm.). The most recent estimates for Washington are 589 occupied territories (Jim Michaels, FWS, pers. comm.), 308 in Oregon (Diana Wang, FWS, pers. comm.), and 117 occupied territories in California (Maria Boroja, FWS, pers. comm.).

The California bald eagle nesting population has increased in recent years from 40 occupied territories in 1977 to 116 occupied territories in 1995 (Jurek 1995, CDFG data), approximately

800 individuals are known to winter in California in a given year. The majority of nesting eagles occur in the northern one-third of the state, primarily on public lands. Seventy percent of nests surveyed in 1979 were located near reservoirs (Lehman 1979), and this trend has continued, with population increases occurring at several reservoirs since the time of that study. In southern California, nesting eagles occur at Big Bear Lake, Cachuma Lake, Lake Mathews, Nacimiento Reservoir, and San Antonio Reservoir (Zeiner et. al., 1990). The Klamath Basin in northern California and southern Oregon supports the largest wintering population of eagles in the lower 48 states, where up to 400 birds may congregate at one time. Scattered smaller groups of wintering eagles occur throughout the State near reservoirs, and typically in close proximity to large concentrations of overwintering migratory waterfowl. Clear Lake, Lake County, may support up to 60 wintering eagles and is a mercury-impaired water body. San Antonio Reservoir has become an important wintering area for bald eagles. An estimate of 50+ eagles regularly winter there. Lake Nacimiento also supports as many as 14 wintering eagles, and is an identified mercury-impaired water of the State. Women are precautioned against consuming any large mouth bass and no one should eat more than 24 ounces of large mouth bass per month from this lake (Cal EPA public health warnings). The observed increase in populations is believed to be the result of a number of protective measures enacted throughout the range of the species since the early 1970s. These measures included the banning of the pesticide DDT, stringent protection of nest sites, and protection from shooting.

<u>Reasons for Decline and Threats to Survival</u>: The species has suffered population declines throughout most of its range, including California, due primarily to habitat loss, shooting, and environmental pollution (Snow 1973, Detrich 1986, Stalmaster 1987). The use of DDT and its accumulation caused thin shelled eggs in many predatory birds. After the ban of DDT and other organochlorine compounds, the bald eagle populations started to rebound (USDI-FWS 1986a).

Other environmental contaminants represent potentially significant threats to bald eagles. Dioxin, endrin, heptachlor epoxide, mercury, and polychlorinated biphenyls (PCB's) still occur in eagle food supplies; however, their overall effects on eagle populations are poorly understood (USDI-FWS, 1986a).

Bald eagles are sensitive to human disturbances such as recreational activities, home sites, campgrounds, mines, and timber harvest (Thelander 1973, Stalmaster 1976) when roosting, foraging, and nesting areas are located near these sites. The bald eagle is protected under the Migratory Bird Treaty Act of 1918, as amended (16 U.S.C. §§ 703-712) and the Bald Eagle Protection Act of 1940, as amended (16 USC §§ 668-668d).

Olendorff and Lehman (1986) collected reports of bald eagles colliding with transmission lines from around the world and covering the period from 1965-1985. The reported mortality rate for bald eagles was 87%. Olendorff and Lehman (1986) suggest that the heavy weight of eagles could be a factor in the higher mortalities for eagles than for other smaller buteos. Olendorff *et al*. (1986) observed eagle flight patterns in wintering areas in the vicinity of proposed transmission line routes in California. Eagles were observed flying through drainages, canyons and saddles, across low ridges, over valleys, and were concentrated above high ridges. Eagles usually flew above 100 feet from the ground (Olendorff *et al*. 1986).

California Brown Pelican (*Pelecanus occidentalis californicus*)

<u>Species Description and Life History</u>: The brown pelican was federally listed as endangered in 1970 (35 **FR** 16047). The recovery plan describes the biology, reasons for decline, and the actions needed for recovery of the California brown pelican (USDI-FWS 1983).

The brown pelican is a large bird recognized by the long, pouched bill. Brown pelicans nest in colonies on small coastal islands that are free of mammalian predators and human disturbance, and are associated with an adequate and consistent food supply. During the non-breeding season brown pelicans roost communally, generally in areas that are near adequate food supplies, have some type of physical barrier to predation and disturbance, and provide some protection from environmental stresses such as wind and high surf.

Foraging Ecology: The brown pelican uses its pouched bill to catch surface schooling fishes by plunge-diving into the water. The brown pelican feeds exclusively on small schooling animals found in the marine environment. Species that occur in Salton Sea that may serve as pelican prey are *Tilapia* sp., juvenile orange mouth corvina (*Cynoscionxanthalus sp*), sailfin mollies (*Poecilia latipinna*), red shiner (*Notropis umbratilis*), and mosquito fish (*Gambusia sp.*).

<u>Historic and Current Distribution</u>: Nesting colonies range from the Channel Islands in the Southern California Bight to the islands off Nayarit, Mexico. Prior to 1959, intermittent nesting was observed as far north as Point Lobos in Monterey County, California. Dispersal between breeding seasons ranges from British Columbia, Canada, to southern Mexico and possibly to Central America. Variable numbers of brown pelicans also occur at the Salton Sea, Imperial County, California, with maximum numbers present in late July and August (Small 1994). Limited numbers of brown pelicans are known to occasionally winter there (Small 1994). Breeding at the Salton Sea has been recorded only once (16 nests in 1996) at this inland location (Gress, pers. comm. 1996). During the non-breeding season California brown pelicans roost communally, generally near areas with adequate food supplies, physical barriers that offer protection from predation, human disturbance, and environmental stressors such as high surf, and high winds.

<u>Reasons for Decline and Threats to Survival</u>: Brown pelicans experienced widespread reproductive failures in the 1960s and early 1970s. Much of the failure was attributed to eggshell thinning caused by high concentrations of DDE, a metabolite of DDT. Since the listing of the species the EPA has banned the use of DDT in the United States (37 **FR** 13369). Restrictions that banned use of aldrin and dieldrin were imposed in the United States (39 **FR** 37246). Following this ban, the production of California brown pelicans increased and was correlated with an increase in eggshell thickness (Anderson *et al*., 1975). Decline of DDE residues in California brown pelicans began leveling off in 1972, and the improvement

reproductive success began stabilizing in 1974 (Anderson *et al.*, 1977). Other factors implicated in the decline of this subspecies include human disturbance at nesting colonies and food shortages. Brown pelicans have nested sporadically on Bird Island, north of the Channel Islands, since the subspecies' decline in the late 1950s and early 1960s. Oil spills pose a threat to both breeding and wintering birds.

Large die offs, such as those that have occurred at the Salton Sea may have a direct impact on populations of pelicans that nest in the Gulf of California. Long term effects of large die-offs have the potential to effect numbers of pelicans available for dispersal and ultimate recruitment to the Southern California Bight breeding populations.

California Clapper Rail (*Rallus longirostris obsoletus*)

Species Description and Life History: The California clapper rail was federally listed as endangered in 1970 (35 **FR** 1604). A detailed account of the taxonomy, ecology, and biology of the California clapper rail is presented in the approved Recovery Plan for this species (USDI-FWS 1984b). Supplemental information is provided below. Clapper rails are non-migratory and are year-round residents of San Francisco Bay tidal marshes. Evans and Page (1983) concluded from research in a north San Francisco Bay marsh that the clapper rail breeding season, including pair bonding and nest construction, may begin as early as February. Field observations in south San Francisco Bay marshes suggest that pair formation also occurs in February in some areas (J. Takekawa, pers. comm.). The clapper rail breeding season has two nesting peaks, one between mid-April and early-May and another between late-June and early-July. Harvey (1988) and Foerster *et al.* (1990) reported mean clutch sizes of 7.27 and 7.47 for clapper rails, respectively. The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during renesting attempts have hatched and young are mobile.

Foraging Ecology: California clapper rails forage primarily on benthic invertebrates (J. Albertson, pers. comm.; Eddleman and Conway 1994; Varoujean 1972; Test and Test 1942; Moffitt 1941; Applegarth 1938; Williams 1929). The non-migratory nature of the California clapper rail makes them extremely vulnerable to local contamination. A significant portion of the reported prey include algal and detrital foragers, and filter feeders, including bivalves (i.e. *Macoma balthica*, *Ischadium demissum*), crabs (i.e. *Pachygrapsus crassipes*), amphipods, and polychaetes (i.e. *Nereis vexillosa*).

Historic and Current Distribution: Of the 193,800 acres of tidal marsh that bordered San Francisco Bay in 1850, about 30,100 acres currently remain (Dedrick 1993). This represents an 84 percent reduction from historical conditions. Furthermore, a number of factors influencing remaining tidal marshes limit their habitat values for clapper rails. Much of the east San Francisco Bay shoreline from San Leandro to Calaveras Point is rapidly eroding, and many marshes along this shoreline could lose their clapper rail populations in the future, if they have not already. In addition, an estimated 600 acres of former salt marsh along Coyote Creek, Alviso Slough, and Guadalupe Slough, has been converted to fresh- and brackish-water vegetation due to freshwater discharge from south San Francisco Bay wastewater facilities and is of lower quality for clapper rails. This conversion has at least temporarily stabilized as a result of the drought since the early 1990s.

The suitability of many marshes for clapper rails is further limited, and in some cases precluded, by their small size, fragmentation, and lack of tidal channel systems and other micro-habitat features. These limitations render much of the remaining tidal marsh acreage unsuitable or of low value for the species. In addition, tidal amplitudes are much greater in the south Bay than in San Pablo or Suisun bays (Atwater *et al.* 1979). Consequently, many tidal marshes are completely submerged during high tides and lack sufficient escape habitat, likely resulting in nesting failures and high rates of predation. The reductions in carrying capacity in existing marshes necessitate the restoration of larger tracts of habitat to maintain stable populations.

The clapper rail population is estimated to be approximately 500 to 600 individuals in the southern portion of San Francisco Bay, while a conservative estimate of the north San Francisco Bay population, including Suisun Bay, is 195 to 282 pairs. Historic populations at Humboldt Bay, Elkhorn Slough, and Morro Bay are now extinct; therefore, 30,100 acres of tidal marsh remaining in San Francisco Bay represent the current distribution of this subspecies.

<u>Reasons for Decline and Threats to Survival</u>: As described above, the clapper rail's initial decline resulted from habitat loss and degradation, and reduction in range. Throughout San Francisco Bay, the remaining clapper rail population is besieged by a suite of mammalian and avian predators. At least 12 native and 3 non-native predator species are known to prey on various life stages of the clapper rail (Albertson 1995). Artificially high local populations of native predators, especially raccoons, result as development occurs in the habitat of these predators around the Bay margins (J. Takekawa, pers. comm.). Encroaching development not only displaces lower order predators from their natural habitat, but also adversely affects higher order predators, such as coyotes, which would normally limit population levels of lower order native and non-native predators, especially red foxes (Albertson 1995).

Hunting intensity and efficiency by raptors on clapper rails also is increased by electric power transmission lines, which criss-cross tidal marshes and provide otherwise-limited hunting perches (J. Takekawa, pers. comm.). Non-native Norway rats (*Rattus norvegicus*) long have been known to be effective predators of clapper rail nests (DeGroot 1927, Harvey 1988, Foerster *et al.* 1990). Placement of shoreline riprap favors rat populations, which results in greater predation pressure on clapper rails in certain marshes. These predation impacts are exacerbated by a reduction in high marsh and natural high tide cover in marshes.

The proliferation of non-native red foxes into tidal marshes of the south San Francisco Bay since 1986 has had a profound effect on clapper rail populations. As a result of the rapid decline and almost complete elimination of rail populations in certain marshes, the San Francisco Bay National Wildlife Refuge implemented a predator management plan in 1991 (Foerster and Takekawa 1991) with an ultimate goal of increasing rail population levels and nesting success through management of red fox predation. This program has proven successful in increasing the overall south San Francisco Bay populations from an all-time low (see below); however, it has been difficult to effectively conduct predator management over such a large area as the south San Francisco Bay, especially with the many constraints associated with conducting the work in urban environments (J. Takekawa, pers. comm.).

Predator management for clapper rails is not being regularly practiced in the north San Francisco Bay, and rail populations in this area remain susceptible to red fox predation. Red fox activity has been documented west of the Petaluma River and along Dutchman Slough at Cullinan Ranch (J. Collins, pers. comm.). Along Wildcat Creek near Richmond, where recent red fox activity has been observed, the rail population level in one tidal marsh area has declined considerably since 1987 (J. Evens, pers. comm.), even though limited red fox management was performed in 1992 and 1993 (J. Takekawa, pers. comm.).

California Least Tern (*Sterna antillarum browni*)

Species Description and Life History: The California least tern (least tern) was listed as endangered on October 13, 1970 (35 **FR** 16047). A detailed account of the taxonomy, ecology, and biology of the least tern is presented in the approved Recovery Plan for this species (USDI-FWS 1980). The Service is currently developing an updated recovery plan, which incorporates information gathered since the publication of the first Recovery Plan (USDI-FWS 1980). Supplemental or updated information is provided below.

California least terns are migratory. They arrive in California in April to breed and depart to wintering areas in Central and South America by the end of September. Little is known about least tern wintering areas. While in California, least tern adults court, mate, and select nest sites; lay, incubate, and hatch eggs; and raise young to fledging prior to departing from the breeding site.

After their eggs hatch, breeding adults catch and deliver small fish to the flightless young. The adults shift their foraging strategy when chicks hatch in order to obtain the very small sized fish for nestlings (Collins *et al.* 1979, Massey 1988). The young begin to fly at about 20 days of age, but continue to be fed and are taught how to feed by their parents for some time after fledging. Reproductive success is, therefore, closely related to the availability of undisturbed nest sites and nearby waters with adequate supplies of appropriately sized fishes
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Terns typically employ a shallow plunge dive technique to capture fish immediately below the water's surface. Adults usually dive from a hover but occasionally dive directly from flight. Most foraging activity is conducted within a couple miles of the colony (Atwood and Minsky 1983).

California Least Terns are opportunistic in their foraging strategy and are known to take many different species of fish. However, they seem to select fish based on certain morphological characteristics. Massey and Atwood (1981) conclude that prey items are generally less than 9 cm in length and have a body depth of less than 1.5 cm.

Once their eggs hatch, the adult terns must feed their young as well as themselves. The adults shift their foraging strategy when chicks hatch in order to obtain the small fish for nestlings (Collins *et al.* 1979, Massey 1986). The adult terns begin foraging nearer the colony and in water with an abundance of small prey fish.

The adult tern does not dismember larger fish in order to feed its small chick. The adult captures a fish and disables it by shaking, and delivers it whole to the chick. A small, newly hatched least tern chick cannot swallow a fish that is too large or relatively deepbodied. The chicks can only eat small, elongated fish. Despite an abundance of larger fish that may be preferred food for an adult Least Tern, an inadequate supply of smaller fish will reduce chick survival.

After fledging, the young terns do not become fully proficient at capturing fish until after they migrate from the breeding grounds. Consequently, parents continue to feed their young even after they are strong fliers.

Foraging Ecology: Least terns feed exclusively on small fishes captured in shallow, nearshore waters, particularly at or near estuaries and river mouths (Massey 1974, Collins *et al.* 1979, Massey and Atwood 1981a, 1984, Atwood and Minsky 1983, Atwood and Kelly 1984, Minsky 1984, Bailey 1984). While in California during the breeding season, least terns forage for fish in nearshore waters which are generally productive foraging habitat areas. Collins (1995) summarized least tern prey selection studies conducted at Naval Air Station (NAS) Alameda from 1981 through 1995. Researchers counted fish, by species, dropped by least terns flying between foraging and nesting areas. Although studies of dropped fish do not provide direct evidence of prey consumed, they do provide a good indication of least tern diets. Least terns dropped larvae and juveniles of nearly 30 species; however, northern anchovy (*Engraulis mordax*) and silversides (*Atherinidae spp.*) comprised 25% and 60% of all dropped fish, respectively. Silversides included topsmelt (*Atherinops affinis*) and jacksmelt (*Atherinopsis californiensis*). Shiner surf-perch (*Cymatogaster agregata*) comprised approximately 5% of the tern's diet.

Thirty-seven different species of fish dropped by the least tern while breeding at the Venice Beach nesting site, next to the Ballona Creek Channel, Marina del Rey marina in Santa Monica Bay, were recorded by Massey and Atwood (1981). At Venice Beach and Huntington Beach in Orange County next to the Santa Ana River mouth, in 1978-81, northern anchovy (*Engraulis mordax*) and silversides including topsmelt (*Antherinops affinis*), jacksmelt (*Atherinopsis californiensis*), and California grunion (*Leuresthes tenuis*) composed most of the samples of fish found dropped in the nesting areas as well as most of the actually documented food items (Atwood and Kelly 1984). Very small or soft scaled species such as gobies (especially *Clevelandia ios, Quietula y-cauda*, and *Ilypnus gilberti*) are under represented in dropped fish surveys.

The larval and yearling sizes of anchovies and silversides fall well within the size range of fish taken by least terns. Northern anchovy are a planktivorous, schooling fish that broadcast-spawn in the Bay. Larvae begin schooling at 1.1-1.2 cm in length, and larvae and juveniles form tightly packed schools in nearshore areas. Topsmelt are a schooling fish that have a prolonged spawning period from April through October, with a peak in May and June. Moyle (1976a) described topsmelt as bottom feeding omnivores, based upon the organisms, detritus, and sand grains found in their stomachs. Stomach content analyses describe topsmelt diets as consisting of diatoms and filamentous algae (50% by volume), detritus (29%), chironomid midge larvae (10%), and amphipods (10%). Jacksmelt are omnivorous, schooling fish that spawn in late winter and early spring. Large schools of juveniles remain in the Bay through the summer, emigrating to coastal waters in the fall. Juvenile jacksmelt foraging behavior, described by Bane and Bane (1971), is similar to that of topsmelt. Jacksmelt juveniles are bottom feeding omnivores, primarily feeding on algae, detritus, small crustaceans, and amphipods. California least terns can therefore be considered exclusive consumers of trophic level 3 fish.

Historic and Current Distribution: The California least tern continues to occupy nesting sites distributed throughout its historic range. The historic breeding range extended along the Pacific Coast from Moss Landing, Monterey County, California, to San Jose del Cabo, southern Baja California, Mexico (A.O.U 1957, Dawson 1924, Grinnell 1928, Grinnell and Miller 1944). However, least terns were nesting several miles north of Moss Landing at the mouth of the Pajaro River, Santa Cruz County, California, at least from 1939 (W.E. Unglish, Western Foundation of Vertebrate Zoology egg collection) to 1954 (Pray 1954); and although nesting at San Francisco Bay was not confirmed until 1967 (Chandik and Baldridge 1967), numerous spring and summer records for the area suggest nesting may have occurred previously (Allen 1934, Chase and Paxton 1965, Grinnel and Wythe 1927, Sibley 1952). Since 1970, nesting sites have been documented in California from San Francisco Bay to the Tijuana River at the Mexican Border; and in Baja California from Ensenada to San Jose del Cabo at the tip of the peninsula.

There are no reliable estimates describing the historic numbers of California least terns along the Pacific Coast (USDI-FWS 1980). Early accounts describe the existence of substantial colonies along the southern and central California coast (Grinnell 1898; McCormick 1899, as cited in Bent 1921), including a colony of about 600 breeding pairs along a 3-mile stretch of beach in San Diego County (Shepardson 1909). At the time of its Federal listing as endangered in 1970, the U.S. population of the California least tern was estimated to be 600 breeding pairs (Fancher 1992). The dramatic decline in breeding least terns has been attributed to the degradation and loss of breeding sites, colonies, and foraging areas, which resulted from human development and disturbance, and pollution (USDI-FWS 1980).

Since its listing, the statewide population of the least tern has recovered to an estimated 4,009 breeding pairs in 1997 (Ron Jurek, pers. comm). Despite this dramatic increase in breeding pairs, statewide monitoring has revealed threats to the least tern which emphasizes the importance of demography to the least tern's survival and recovery. In 1983, for example, the presence of predators caused most of the NAS Alameda colony to attempt to breed at the Oakland Airport site, where 61 nesting pairs produced only 8 fledglings. This event and other stuff at other colony/nest sites has highlighted the importance of multiple nesting sites available to a colony. The effects of El Nino years on southern CA colonies has highlighted the significance of multiple clusters, distributed along the coast.

The current U.S. population of the California least tern is grouped into 5 geographically discrete clusters, which support multiple active and historic breeding sites. These clusters include: (1) San Diego County, (2) Los Angeles/Orange Counties, (3) Ventura County, (4) San Luis Obispo/Santa Barbara Counties, and (5) San Francisco Bay area. The maintenance of multiple viable clusters and multiple breeding sites within them is important to the least tern's survival and recovery.

San Diego County The San Diego County cluster includes 24 active nest sites and supports the majority of the U.S. population of the California least tern. The active nest sites and number of pairs recorded in 1997 (in parentheses) include White Beach (17), three sites at the Santa Margarita River mouth (728, 41, and 39), five sites in Batiquitos Lagoon (83, 59, 25, 0, and 104), San Elijo Lagoon (9), three sites in Mission Bay (20, 268, and 76), nine sites in San Diego Bay (0, 102, 22, 310, 15, 85, 0, 38, and 36), and the Tijuana River Estuary (211). Least tern foraging has been studied at Mission Bay (ERC 1989, SWRI 1994). Least tern foraging studies or observations in San Diego Bay indicate a very significant reliance upon the Bay's tidal waters (Baird 1993, 1995, Manning 1995). While virtually every coastal area of southern California is vulnerable to exposure to toxic or environmentally contaminating discharges from the intense industrializing/urbanizing influences, San Diego Bay has been particularly developed as a commercial port, major U.S. Navy homeport, and industrial area.

Los Angeles/Orange Counties The Los Angeles County/Orange County cluster includes active nest sites at Venice Beach, Pier 300 (Terminal Island), Pier 400 and TC2 (new harbor sites), Seal Beach National Wildlife Refuge, Bolsa Chica, Huntington Beach, and Upper Newport Bay. In 1997, these sites supported 375, 4, 76, 178, 141, 373, and 82 nests, respectively. Atwood and Minsky (1983) studied the foraging patterns of breeding least terns at Huntington Beach and Venice Beach nesting colonies. Drainage channels from highly urbanized areas discharge near or directly into the least tern foraging areas. San Pedro Bay has been the focus of foraging studies of least terns nesting at the Terminal Island colony (MEC 1988, Keane 1997). The least tern relies upon fish captured in the nearshore zone, and in tidal sloughs and relatively shallow bodies of water that support large numbers of small fish. In highly urban LA and Orange Counties, these are water bodies under the influence of a very wide variety of industrial discharges, particularly San Pedro Bay which is also a commercial port and highly industrialized area.

San Luis Obispo/Santa Barbara Counties The San Luis Obispo County/Santa Barbara County cluster includes active least tern nest sites at Oceano (Pismo) Dunes State Vehicular Recreational Area, Mussel Rock (Guadalupe) Dunes, and Beach 2 and Purisima Point at Vandenberg Air Force Base. In 1997, these sites supported 6, 30, 3, and 25 nesting pairs, respectively. In this portion of their range California least terns are known to forage in the Santa Ynez and Santa

Maria River lagoons in the Pacific Ocean. Least terns also stage at area lagoons prior to post-breeding dispersal.

Ventura County The Ventura County cluster includes seven nest sites at three locations: Point Magu Naval Air Station, Ormond Beach, and McGrath State Beach at the Santa Clara River mouth. In 1997, these three locations supported approximately 74, 63, and 43 nesting pairs, respectively. In this portion of their range California least terns are known to forage in the Ormond, Ventura, and Santa Clara River Lagoons, Mugu Lagoon, Revolon Slough, and in the slough near the Mandalay Generating Station. Least terns also stage at area lagoons prior to post-breeding dispersal.

San Francisco Bay In the San Francisco Bay, least terns have nested at 6 sites in Contra Costa, Alameda, and San Mateo Counties. Most sites in the San Francisco Bay have not been used by breeding least terns in recent years. Presently, only NAS Alameda supports significant numbers of nesting pairs. There are two other minor least tern breeding sites that remain in the San Francisco Bay area, but the Oakland Airport site has not been used in years and the PG&E Pittsburg site supports only 1 to 4 pairs each year, including 4 pairs in 1997. Therefore, the NAS Alameda site currently represents the entire San Francisco Bay area population, and is the most northern of least tern breeding colonies by about 178 miles. Because of its northern location, the NAS Alameda site is relatively unaffected during El Nino years when many southern California sites experience pronounced breeding failure resulting from limited food availability. In the most recent El Nino year, 1992, the NAS Alameda site supported 6 percent of the statewide number of breeding pairs, but produced 16 percent of the total statewide number of fledglings.

According to Caffrey (1995), the least tern breeding site at NAS Alameda has played a significant role in recent increases in the number of least terns throughout California. The NAS Alameda site is consistently one of the most successful sites in California. Between 1987 and 1994, the NAS Alameda site supported 5 to 6 percent of the statewide breeding population out of 35 to 40 sites each year, but produced an average of 10.6 percent of the total number of fledglings produced statewide in each of those years. In 1997, an estimated 244 pairs of least terns nested at the colony out of a total population of over 4,000 nesting pairs at 37 breeding sites along the California and Baja California coasts. In 1997, an estimated 316 young fledged successfully at NAS Alameda; this represented 10.1 percent of the total number of fledglings produced throughout California that year. By consistently producing large numbers of fledglings each year, the colony has added large numbers of potential new breeding birds to the statewide population. Therefore, this site is considered to be one of the most important "source" populations in California serving to balance out losses at many "sink" locations throughout the state.

In San Francisco Bay, post-breeding adults and fledglings move to South San Francisco Bay salt ponds where they may remain for several weeks prior to migrating south (Feeney and Collins 1988, Collins 1989).

Reasons for Decline and Threats to Survival: California least terns were once common along the central and southern California coast. The decline of the California least tern is attributed to prolonged and widespread destruction and degradation of nesting and foraging habitats, and increasing human disturbance to breeding colonies. Conflicting uses of southern and central California beaches during the California least tern nesting season have led to isolated colony sites that are extremely vulnerable to predation from native, feral and exotic species, overwash by high tides, and vandalism and harassment by beach users. Since its classification as a Federal and State endangered species, considerable effort has been expended on annual population surveys, protection and enhancement of existing nesting colonies, and the establishment of new nesting locations. Control of predators constitutes one of the most crucial management responsibilities at California least tern nesting sites.

An important aspect of recovery is the protection of coastal feeding grounds of colonies by maintaining high water quality and preventing tideland fill and drainage projects. Protection of non-nesting, feeding, and roosting habitats from detrimental land or water use changes in San Diego and Los Angeles County is also important for recovery (USDI-FWS 1980).

Light-footed Clapper Rail (*Rallus longirostris levipe*)

Species Description and Life History: The light-footed clapper rail was listed as an endangered species on October 13, 1970 (35 **FR** 16047). A recovery plan for the species was issued in 1979 and revised in 1985 (USDI-FWS 1985a). This recovery plan describes the biology, reasons for decline, and the actions needed for recovery of light-footed clapper rails populations in California (USDI-FWS 1985a). The light-footed clapper rail’s coloration blends with the dense stands of cordgrass (*Spartina foliosa)* dominating its preferred habitat in coastal salt or brackish water marshes. Male rails are approximately 12 inches in length and are slightly larger and more colorful than females. The birds are tawny-breasted with gray-brown backs, vertical white bars on the flanks and show whitish coloration under the short tail, on the chin, and over the eye. The rails' bills are mostly orange and the birds' legs and feet are largely brownish.

Rails breed from mid-March to mid-August, usually selecting dense stands of cordgrass (*Spartina foliosa*) as a nest site, although nest are occasionally observed in pickleweed (*Salicornia virginica*) or other marsh type vegetation. In addition to a brood nest, pairs usually build a number of nests, secured in to surrounding vegetation, to serve as refuges from high tides. Males and females usually share the responsibility for incubation of 4-10 eggs, which hatch in 18-27 days. Hatchling rails are covered in black down and are able to follow along after the adults in the marsh within a few hours of hatching. The young rails are dependent upon the adults for several weeks and are still being fed occasionally up to at least 6 weeks of age (Zembal 1989). Light-footed rails spend much of their time in lower salt marsh habitat, particularly in cordgrass. Although this plant species provides preferred nesting substrates, nest are also built in common pickleweed and other upper marsh plants on hummocks of high ground surrounded by low marsh (Massey *et al.* 1984).

Limited evidence exists for intermarsh movements by rails; this bird is resident in its home marsh except under unusual circumstances. Within-marsh movements are also confined and generally of no greater spread than 400 meters. Minimum home range sizes for 9 rails that were radio-harnessed for telemetry at Upper Newport Bay varied from approximately 0.8 to 4.1 acres. The larger areas and daily movements were by first-year birds attempting to claim their first breeding territories (Zembal 1989).

Foraging Ecology: The rail is an opportunistic omnivore. A wide variety of mostly animal foods is consumed using many different foraging strategies including gleaning, probing, crab hunting, fishing, and scavenging. Over 90% of the observed foraging has been of rails executing hundreds of gleans and usually shallow probes over the marsh substrate per hour and consuming hundreds of prey items. However, crabs are important in the diet, too, along with snails, insects, and invertebrates. Plant foods are uncommon (Zembal 1989).

<u>Historic and Current Distribution</u>: The light-footed clapper rail is a resident of coastal marshes, ranging historically from Carpinteria Marsh in Santa Barbara County, California south to San Quintin, Baja California, Mexico. The current distribution of the light-footed clapper rail is limited to Upper Newport Bay, Anaheim Bay, Tijuana Slough National Wildlife Refuge, and Mugu lagoon. The spring counts in 1997 revealed 307 pairs of rails in 16 marshes in California. Of this total, 48.5 percent of the rails were in Upper Newport Bay, Orange County, California (Zembal unpublished data, 1997).

<u>Reasons for Decline and Threats to Survival</u>: The destruction and degradation of habitat led to small, isolated subpopulations and prompted the listing of this species. The United States population has been censused annually over the past decade and the downward trend has continued. The spring counts in 1989 revealed only 163 pairs of rails in 8 marshes in California. Of this total, 116 pairs or 71.2 percent of were in Upper Newport Bay, Orange County, California (Zembal 1990). The one hundred thirty-six pairs detected in Upper Newport Bay in 1992 (Zembal 1993) may closely approach the maximum number of pairs that can be accommodated at this locale (Richard Zembal, personal communication, 1993).

Marbled Murrelet (*Brachyramphus marmoratus*)

<u>Species Description and Life History:</u> The marbled murrelet was federally listed as a threatened species in Washington, Oregon and California on September 28, 1992 (57 **FR** 45328), primarily due to loss of nesting habitat. The final recovery plan was released in 1997 (USDI-FWS 1997b). Critical habitat was designated in 1996 to include 32 critical habitat units (CHU's) in Washington, Oregon, and California, primarily on Federal lands. Primary constituent elements of the CHU's include 1) individual trees with potential nesting platforms, and 2) forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms and a canopy height of at least one-half the site-potential tree height.

The Recovery Plan for the Marbled Murrelet (USDI-FWS, 1997) establishes six conservation zones for the species throughout its range in Washington, Oregon, and California. Conservation zones 4-6 are located in California. Narratives for each of these zones are included in the recovery plan. Conservation zone four, the Siskiyou Coast Range Zone, extends from North Bend, Oregon to the southern end of Humboldt Bay, California. Conservation zone five, Mendocino Zone, extends from the southern end of Humboldt Bay to the mouth of San Francisco Bay. Zone six, the Santa Cruz Mountains Zone, extends from the mouth of San Francisco Bay to Point Sur, Monterey County. Each of these zones include all nearshore waters, as previously defined, within 1.2 miles of the Pacific shoreline. Waters impacted by the CTR include all freshwater, and estuarine ecosystems coincidental with these conservation zones, including Humboldt, San Francisco, Tomales, Bodega, Half Moon, and Monterey Bays.

The marbled murrelet is a small diving seabird that breeds along the Pacific coast of North America from the Aleutian Archipelago and southern Alaska south to central California (USDI-FWS 1997b). The marbled murrelet is the only member of the Alcidae family known to nest in trees. Preferred nesting habitat for the species is characteristically old-growth, coniferous forests within 50 miles of the coast. Nesting stand characteristics include large, old trees, generally greater than 32 inches diameter at breast height (dbh), with large limbs which provide nest platforms. Nest are typically located near the bole of the tree and are simple depressions sometimes located in clumps of moss and lichens.

Marbled murrelets nest in old-growth forests, generally characterized by large trees ($\geq$ 32 inches dbh), multiple canopy layers, and moderate to high canopy closure. As of April 2, 1996, at least 95 active or previously used tree nests were located in North America: 9 in Washington, 41 in Oregon, and 12 in California (K. Nelson, pers. Comm. 1996; Binford *et al.* 1975; Varoujean *et al.* 1989; Quinlan and Hughes 1990; Hamer and Cummins 1990, 1991; Kuletz 1991; Singer *et al.* 1991, 1992; Hamer and Nelson 1995). All nests in Washington, Oregon, and California were located in old-growth trees that were greater than 32 inches dbh. Most nests were located on large or deformed, moss covered branches; however, a few nests were located on smaller branches, and some nests were situated on duff platforms composed of conifer needles or sticks rather than moss. Such locations allow easy access to the exterior of the forest and provide shelter from potential predators. Nest sites in California were located in stands containing old-growth redwood (*Sequoia sempervirens*) and Douglas-fir. Nest sites in Oregon and Washington were located in stands dominated by Douglas-fir, western hemlock (*Tsuga heterophylla*), and Sitka spruce (*Picea sitchensis*). Suitable marbled murrelet habitat is defined as forest stands with conditions that will support nesting marbled murrelets.

Marbled murrelets appear to be solitary in their nesting and feeding habits, but interact in groups over the forest and at sea (Sealy and Carter 1984, Carter and Sealy 1990, Nelson and Hamer 1995a). They lay on one egg on the limb of a large coniferous tree. Incubation lasts 30 days and fledging takes 28 days. Both sexes incubate the egg (Nelson and Hammer 1995a, Nelson and Peck 1995, Simons 1980, Singer *et al.* 1991, 1992).

Foraging Ecology: The marbled murrelet forages almost exclusively in the nearshore environment, including bays, estuaries, and island groups. Adult marbled murrelets forage on a variety of aquatic organisms including: Pacific sand lance (*Ammodytes hexapterus*), Pacific herring (*Clupea harengus*), northern anchovy (*Engraulis mordax*), capelin (*Clupea* spp.), and smelts (family *Osmeridae*), as well as invertebrates such as *Euphausia pacifica* and *Thysanoessa spinfera*. In the early 1900's, Pacific sardines (*Sardinops sagax*) were also documents as prey in California. Adults, subadults, and hatching year birds feed primarily on larval and juvenile fish, whereas nestlings are most commonly fed larger second year fish. The sand lance is the most common food of the marbled murrelet across its range, comprising up to 52% of the observed prey items, anchovy and herring comprised roughly 29% of observed prey items, and Osmerids comprised the remaining 24% of prey item observations (Burkett 1995). The species is an opportunistic forager, relying on numerous species of fish taken in the nearshore environment. This strategy is believed to have sustained the species after declines in historic prey species (Ralph et al 1995, USDI-FWS 1997b). Marbled murrelets will also forage in fresh water lakes on salmonid fry, fingerlings, and yearlings (Carter and Sealy 1986).

During the breeding season, the marbled murrelet tends to forage in well-defined areas along the coast in relatively shallow marine waters, including enclosed bays and estuaries.

<u>Historic and Current Distribution</u>: The historic distribution of the marbled murrelet within the listed range was continuous in nearshore waters and in coniferous forests near the coast from the Canadian border south to Point Sur, Monterey County, California. Current breeding populations are discontinuous and concentrated at sea in areas adjacent to remaining late-successional, coastal, coniferous forests. Off the California coast, marbled murrelets are concentrated in two areas at sea, corresponding to the three largest remaining blocks of older, coastal forest. These blocks of older forest are separated by areas of little or no habitat, which correspond to locations at sea where few marbled murrelets are found. A large gap (about 300 miles) occurs in the southern portion of the marbled murrelet's breeding range, from San Mateo and Santa Cruz counties north to Humboldt and Del Norte counties, California. Marbled murrelets likely occurred in the gap prior to extensive logging of redwood forests (Paton and Ralph 1988).

Estimates of the marbled murrelet population size in California are based on research over the past 15 years. In 1979-1980, the breeding population was estimated to be about 2,000 birds, based on data collected while conducting surveys of other seabird colonies (Sowls *et al.* 1980). Utilizing Sowls' data and similar information collected in 1989, Carter and Erickson (1992) and Carter *et al.* (1990) estimated the breeding population at 1,650 to 1,821 birds. Ralph and Miller (1995) conducted more intensive at-sea surveys in small portions of the murrelet's range in northern California from 1989 to 1993. These multi-year surveys, specifically designed to estimate population size in California, used different methods and assumptions and estimate a total State population size of approximately 6,000 breeding and non-breeding birds. Ralph and Miller, however, extrapolated results from small areas to estimate numbers of murrelets over much larger areas; the result may be an overestimation of murrelet population size, given the non-uniform distribution of murrelets at sea.

Marbled murrelet populations in California, Oregon, and Washington apparently are declining rapidly. Current estimates of nesting success and recruitment are well below levels required to sustain populations in the Pacific Northwest (USDI-FWS 1997b). A population model which analyzed likely ranges of fecundity and survivorship estimated that murrelets population sizes in Washington, Oregon, and California are most likely declining at a rate between 4 and 6 percent per year (Beissinger 1995).

The distribution of the marbled murrelet in California is limited to three separate areas, primarily associated with remaining contiguous old growth forest habitat (Carter and Erickson 1992), Historically the species was plentiful during the winter months from Monterey county north to the Oregon border. Today the remaining populations of murrelets are disjunct and separated by great distances, largely the result of a lack of suitable breeding habitat. For further information on the status, distribution, and biology of the marbled murrelet refer to the *Ecology and Conservation of the Marbled Murrelet* (Ralph *et al.* 1995), Marshall 1988, and Carter and Morrison 1992.

Reasons for Decline and Threats to Survival: Suitable habitat has declined throughout the range of the marbled murrelet as a result of commercial timber harvest, with some loss attributable to natural disturbance such as fire and windthrow. Timber harvest has eliminated most suitable habitat on private lands within the three state area (Norse 1988, Thomas *et al.* 1990). A total of approximately 2,552,200 acres of suitable marbled murrelet habitat occur on Federal lands in California, Oregon, and Washington.

Marbled murrelet reproductive success may be adversely affected by forest fragmentation and associated effects from excessive amounts of edge. Fragmented forests can have higher numbers of predators that can adapt to the changing environment, leading to increased predation on murrelet nests that may be easier for a predator to locate in a fragmented forest. Relatively high observed predation rates are of great concern and have led the Service to conclude that maintenance and development of suitable habitat in relatively large contiguous blocks will contribute to the recovery of the murrelet (USDI-FWS 1997b).

Spills of oil and other pollutants along the coast of California, Oregon, and Washington can also do local harm to populations. The central California population of marbled murrelets is especially vulnerable to oil spill events. Changes in prey abundance from over-harvest, El Nino events, or pollution related deaths can also cause reproductive failure (USDI-FWS 1997b).

Industrial discharges from the population centers of San Francisco Bay, California, Puget Sound, Washington, and Vancouver, British Columbia, have contaminated estuarine sediments with heavy metals, petroleum hydrocarbons, and PCB. The major rivers with historic pollutant discharges in the murrelet range include the Sacramento-San Joaquin River System (Fry 1995).

Protection of the foraging areas is a critical component to a successful recovery strategy. The main threats to marbled murrelets identified in their marine habitat result in the loss of individuals through death or injury. Marbled murrelets are adversely affected by spills of oil and other pollutants. Given the essential role of the marine environment, protecting the quality of the marine environment and reducing adult and juvenile mortality in the marine environment are integral parts of the recovery effort. Important near-shore environments in California include Cape Mendocino to the Oregon border (including Humboldt and Arcata Bays, and river mouths of Smith, Eel, and Klamath Rivers and Redwood Creek), and central California from San Pedro Point south to the mouth of the Pajaro River, including the mouths of Pescadero and Waddell Creeks, as well as other creeks. Protection of areas where prey may concentrate should extend 2 km offshore and include estuaries, the mouths of bays, and eddies in the vicinity of headlands. Additionally prey breeding areas such as near-shore kelp beds, sand or gravel beaches, and sand banks should be protected (USDI-FWS 1997b).

Pacific Coast Population of the Western Snowy Plover (*Charadrius alexandrinus nivosus*)

Species Distribution and Life History: The Pacific coast population of the western snowy plover (plover) was federally listed as threatened on March 5, 1993 (50 **FR** 12864). A designation of critical habitat for the plover was federally proposed on March 2, 1995 (60 **FR** 11763), final critical habitat for the species was designated on January 6, 2000 (64 **FR** 68508).

The western snowy plover is a small shorebird that forages on invertebrates in areas such as intertidal zones, the wrack line, dry sandy areas above high tide line, salt pans, and the edge of salt marshes. The plover breeds primarily on coastal beaches from southern Washington to southern Baja California, Mexico. Other less common nesting habitat includes salt pans, coastal dredged spoil disposal sites, dry salt ponds, salt pond levees (Widrig 1980, Wilson 1980, Page and Stenzel 1981), and riverine gravel bars (Gary Lester, pers. comm.). Sand spits, dune-backed beaches, unvegetated beach strands, open areas around estuaries, and beaches at river mouths are the preferred coastal habitats for nesting (Stenzel *et al.* 1981, Wilson 1980).

Snowy plovers breed in colonies with the number of adults at coastal breeding sites ranging from 2 to 318 (Page and Stenzel 1981; Oregon Department of Fish and Wildlife 1994; Eric Cummins, pers. comm.). The breeding distribution is skewed towards the southern portion of the western snowy plover's range with the majority of breeding activity occurring in Ventura, Santa Barbara, San Luis Obispo, and Monterey counties (Ray Bransfield pers. comm. 1998). Nest sites typically occur in flat, open areas with sandy or saline substrates; vegetation and driftwood are usually sparse or absent (Widrig 1980, Wilson 1980, Stenzel *et al.* 1981). The majority of snowy plovers are site-faithful, returning to the same breeding site in subsequent breeding seasons (Warriner *et al.* 1986).

The breeding season of the coastal population of the western snowy plover extends from early March through late September. Nest initiation and egg laying occurs from mid March through mid July (Wilson 1980, Warriner *et al.* 1986). The usual clutch size is three eggs. Both sexes participate in incubation, which averages 27 days (Warriner *et al.* 1986). Plover chicks are precocious, leaving the nest within hours after hatching to search for food. Fledging (reaching flying age) requires an average of 31 days (Warriner *et al.* 1986). Broods rarely remain in the nesting territory until fledging (Warriner *et al.* 1986, Stern *et al.* 1990).

Snowy plovers will renest after loss of clutch or brood (Wilson 1980, Warriner *et al.* 1986). Double brooding and polygamy (i.e., the female successfully hatches more than one brood in a nesting season with different mates) have been observed in coastal California (Warriner *et al.* 1986) and also may occur in Oregon (Jacobs 1986). After loss of a clutch or brood or successful hatching of a nest, plovers may renest in the same site or move, sometimes up to several hundred miles, to other colony sites to nest (Gary Page, pers. comm.; Warriner *et al.* 1986).

Foraging Ecology: Snowy plovers forage on invertebrates in the wet sand and amongst surf cast kelp within the intertidal zone; in dry, sandy areas above the high tide; on salt pans; spoil sites; on mudflats; and along the edges of salt marshes and salt ponds. In San Francisco Bay, breeding plovers forage on invertebrates around salt ponds, and on nearby mudflats of tidal creeks and the Bay. Only anecdotal information exists on plover food habits. Page, *et al.* (1995) and Reeder (1951) listed known prey items of plovers on Pacific coast beaches and tidal flats: mole crabs (Emerita analoga), crabs (*Pachygrapsus crassipes*), polychaetes (Neridae, *Lumbrineris zonata*, *Polydora socialis*, *Scoloplos acmaceps*), amphipods (*Corophium* spp., *Ampithoe* spp., *Allorchestes angustus*, and sand hoppers [Orchestoidea]), tanadacians (*leptochelia dubia*, flies (Ephydridae, Dolichopodidae), beetles (Carabidae, Buprestidae, Tenebrionidae), clams (*Transenella* sp.), and ostracods. Feeney (1991) described plover prey items in salt evaporation ponds in South San Francisco Bay: flies (*Ephydra cinerea*), beetles (*Tanarthrus occidentalis*, *Bembidion* sp.), moths (*Perizoma custodiata*) and lepedopteran caterpillars.

Historic and Current Distribution: Snowy plovers occur along coastal beaches and estuaries from Washington to Baja California, Mexico. Based on the most recent surveys, a total of 28 snowy plover breeding sites or areas currently occur on the Pacific Coast of the United States. Two sites occur in southern Washington--one at Leadbetter Point, in Willapa Bay (Widrig 1980), and the other at Damon Point, in Grays Harbor (Anthony 1985). In Oregon, nesting birds were recorded in 6 locations in 1990 with 3 sites (Bayocean Spit, North Spit Coos Bay and spoils, and Bandon State Park-Floras Lake) supporting 81 percent of the total coastal nesting population (Oregon Department of Fish and Wildlife, unpubl. data, 1991). A total of 20 plover breeding areas currently occur in coastal California (Page *et al.* 1991). Eight areas support 78 percent of the California coastal breeding population: San Francisco Bay, Monterey Bay, Morro Bay, the Callendar-Mussel Rock Dunes area, the Point Sal to Point Conception area, the Oxnard lowland, Santa Rosa Island, and San Nicolas Island (Page *et al.* 1991).

The coastal population of the western snowy plover consists of both resident and migratory birds. Some birds winter in the same areas used for breeding (Warriner *et al.* 1986, Wilson-Jacobs, pers. comm. in Page *et al.* 1986). Other birds migrate either north or south to wintering areas (Warriner *et al.* 1986). Plovers occasionally winter in southern coastal Washington (Brittell *et al.* 1976), and about 70 plovers may winter in Oregon (Oregon Department of Fish and Wildlife 1994). The majority of birds, however, winter south of Bodega Bay, California (Page *et al.* 1986), and substantial numbers occur in the San Francisco Bay (Bay). Wintering coastal populations are augmented by individuals of the interior population that breed west of the Rocky Mountains (Page *et al.* 1986, Stern *et al.* 1988).

Reasons for Decline and Threats to Survival: Poor reproductive success, resulting from human disturbance, predation, and inclement weather, combined with permanent or long-term loss of nesting habitat to encroachment of introduced European beachgrass (*Ammophila arenaria*) and urban development has led to a decline in active nesting colonies, as well as an overall decline in the breeding and wintering population of the western snowy plover along the Pacific coast of the United States. Of the 87 historic breeding areas, only 28 remain (Page and Stenzel 1981; Charles Bruce, pers. comm.; E. Cummins, pers. comm.). The nesting population in the three states is estimated to be around 1,500 adults (Page *et al.*, 1991). Page and Stenzel (1981) estimated that the South Bay supports 10% of California's breeding snowy plovers, of which 90% can be found nesting in Alameda County salt pond systems.

Yuma Clapper Rail (*Rallus longirostris yumaensis*)

Species Description and Life History: The Yuma clapper rail was listed as endangered on March 11, 1967 (32 **FR** 4001). The Yuma clapper rail is a chicken-sized bird that is grayish-brown with a tawny breast and barred flanks. They prefer habitat that is densely vegetated with either cattails (*Typha* sp.) or giant bulrush (*Scirpus californicus*). Territories are generally in areas with a transition from standing water to saturated soils, but the presence of pond openings and flowing water are also important for foraging. Yuma clapper rails occur in fresh water marshes (e.g. cattail, alkali bulrush, and reed), within the vicinity of the Salton Sea and the Colorado River. This species is known to occur within agricultural drains which contain suitable habitat. Moreover, this species has been found to use extremely small patches of habitat within agricultural drains, patches which barely provide enough cover for concealment. Further information is found in Bennett and Ohmart 1978, Todd 1986, and Conway *et al.* 1993.

Foraging Ecology: The Yuma clapper rail has been documented to feed on a wide variety of invertebrates and some vegetation. Included in its diet are crayfish, fresh water prawns, weevils, isopods, clams, water beetles, leeches, damselfly nymphs, small fish, tadpoles, seeds and twigs. Based on the available information, crayfish appear to make up the majority of its food intake.

Historic and Current Distribution: The largest single breeding population of Yuma clapper rails in the United States is located in the Wister Unit of the California Department of Fish and Game's Imperial Wildlife Area. In the 1994 census, 309 individuals were located in the ponds of the Wister Unit (Steve Montgomery, SJM Biological Consultants, pers. comm.). In that same year, surveys of the Salton Sea National Wildlife Refuge and adjacent drainages located 95 individuals, most of which were breeding pairs (Ken Sturm, Salton Sea National Wildlife Refuge, pers. comm.). Large populations of this species occur in the Imperial and Palo Verde Valleys.

Additional Yuma clapper rails can be found along the Colorado River during the breeding season. Rails use the Lower Colorado River from the US border north to Topock Marsh. In the last complete census of the Lower Colorado River in 1994, the estimated total population was 1,145. Based on census data from 1990 to 1995, the Yuma clapper rail population along the Colorado River appears to be stable at this time.

Reasons for Decline and Threats to Survival: Significant habitat losses are believed to have occurred in the lower Colorado River and the delta with the construction of large water reclamation projects along the Colorado River. Recent studies of the Yuma clapper rail indicate that this species may be at risk of selenium-induced reproductive impacts (Rusk 1991, Roberts 1996). While census information has not indicated a decline, selenium concentrations in the rail eggs and tissues analyzed are at levels that could result in slight reductions in reproductive success.

Bonytail Chub (*Gila elegans*)

Species Description and Life History: The bonytail chub was first proposed for listing under the ESA on April 24, 1978, as an endangered species. The bonytail chub was listed as an endangered species on April 23, 1980 (45 **FR** 27713), with an effective date of the rule of May 23, 1980. In the final rule, the Service determined that at that time there were no known areas with the necessary requirements to be determined critical habitat. Critical habitat was designated in 1994. Critical habitat for the bonytail chub includes portions of the Colorado, Green, and Yampa Rivers. Critical habitat includes the Colorado River at Lake Havasu to its full pool elevation (USDI-FWS 1993a).

The bonytail chub is one of three closely related members of the genus *Gila* found in the Colorado River. Confusion about the proper taxonomy and the degree of hybridization between the bonytail chub, the humpback chub, (*Gila cypha*), and roundtail chub, (*G. robusta*), has complicated examinations of the status of these fish. The bonytail chub is a highly streamlined fish with a very thin, pencil-like, caudal peduncle and large, falcate fins (Allan and Roden 1978). A nuchal hump may be present behind the head. Maximum length is about 600 millimeters (mm), with 300-350 mm more common (USDI-FWS 1990). Weights are generally less than one kilogram (kg) (Vanicek and Kramer 1969). Bonytail chub are long-lived fish; some have reached at least 49 years of age (Minckley 1985).

With their streamlined bodies, bonytail chub appear to be adapted to the Colorado River and large tributary streams. Even with these adaptations, this species does not select areas of high velocity currents and use of pools and eddies by the fish is significant (Vanicek 1967, Vanicek and Kramer 1969).

Spawning takes place in the late spring to early summer (Jonez and Sumner 1954, Wagner 1955) in water temperatures about 18 degrees C (Vanicek and Kramer 1969). Riverine spawning of the bonytail chub has not been documented; however in reservoirs, gravel bars or shelves are used (Jonez and Sumner 1954).

The bonytail chub is adapted to the widely fluctuating physical environment of the historical Colorado River. Adults can live 45-50 years, and apparently produce viable gametes even when quite old. The ability to spawn in a variety of habitats is also a survival adaption. Fecundity measurements taken on adult females in the hatchery ranged from 1,015 to 10,384 eggs per fish with a mean of 4,677 (USDI-FWS 1990). With the fecundity of the species, it would be possible to quickly repopulate after a catastrophic loss of adults.

Foraging Ecology: Bonytail chub feed mostly on insects, algae, and plant debris.

Historic and Current Distribution: Occupied habitat as of 1993 is approximately 344 miles (15% of the historic range). Populations are generally small and composed of aging individuals. Recovery efforts under the Recovery Implementation Program in the Upper Basin have begun, but significant recovery results have not been seen for this species. In the Lower Basin, augmentation efforts along the Lower Colorado River propose to replace the aging populations in Lakes Havasu and Mohave with young fish from protected-rearing site programs. This may prevent the imminent extinction of the species in the wild, but appears less capable of ensuring long term survival or recovery of the bonytail chub. Overall, the status of the bonytail chub in the wild continues to be precarious.

Reasons for Decline and Threats to Survival: Severe reductions in both population numbers and individual bonytail chub numbers can be traced largely to impounding the lower Colorado River and introducing non-native fish into the modified environment. The bonytail chub was listed as an endangered species due to massive declines in or extirpation of all populations throughout the range of the species. The causes of these declines are changes to biological and physical features of the habitat. The effects of these changes have been most noticeable by the almost complete lack of natural recruitment to any population in the historic range of the species.

Chinook Salmon (Including Central Valley Spring-Run, California Coastal and Sacramento Winter-Run ESUs) (*Oncorhynchus tshawytscha*)

Species Description and Life History: Based on the best available scientific and commercial information, NMFS has identified 17 ESUs of chinook salmon from Washington, Oregon, Idaho, and California, including 11 new ESUs, and one re-defined ESU. Further detailed information on these ESUs is available in the NMFS "Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California" (Myers *et al.*, 1998) and the NMFS proposed rule for listing chinook (63 FR 11482). Four of these are within the action area in California. The Sacramento River Winter-Run ESU was listed as endangered on January 4, 1994 (59 FR 440); critical habitat was designated in an earlier listing of the ESU as threatened (June 16, 1993; 58 FR 33212). On September 16, 1999, NMFS listed (64 FR 50394) the Central Valley Spring-Run ESU as threatened; redefined the Southern Oregon and California Coastal ESU, creating a distinct California Coastal ESU extending from the Russian River, Sonoma County, north to Redwood Creek, Humboldt County, and listed this new ESU as threatened. In the same rulemaking, NMFS also determined that the Central Valley Fall/Late Fall ESU and the Southern

Oregon / Northern California Coastal ESU (including those populations now considered separate from the California Coastal ESU) are not warranted for listing at this time.

Critical Habitat: On February 16, 2000, NMFS designated critical habitat for all ESUs of chinook salmon (except Sacramento River Winter-Run)(65 FR 7764). In evaluating the habitat requirements of listed chinook NMFS decided to designate only the current range of the listed ESUs as critical habitat. The current range encompasses a wide range of habitats, including small tributary reaches as well as mainstem, off-channel, and estuarine areas. Areas excluded from this proposed designation include historically occupied areas above impassible dams and headwater areas above impassable natural barriers (e.g., long-standing, natural waterfalls). NMFS has concluded that at the time of this designation, currently inhabited areas within the range of West Coast chinook salmon are the minimum habitat necessary to ensure conservation and recovery of the species. Critical habitat consists of the water, substrate, and adjacent riparian zone of accessible estuarine and riverine reaches for the following areas for chinook salmon located in California:

1) Central Valley Spring-Run chinook salmon geographic boundaries: Critical habitat is designated to include all river reaches accessible to chinook salmon in the Sacramento River and its tributaries in California. Also included are river reaches and estuarine areas of the Sacramento-San Joaquin Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait, all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

2) California Coastal chinook salmon geographic boundaries: Critical habitat is designated to include all river reaches and estuarine areas accessible to chinook salmon along the California coast from the Russian River, in Sonoma County, north to Redwood Creek, Humboldt County. Also excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

3) Sacramento River Winter-Run chinook geographic boundaries: Critical habitat is designated to include the Sacramento River from Keswick Dam (Shasta County) to Chipps Island at the westward margin of the Sacramento-San Joaquin Delta; all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. In addition, the critical habitat designation identifies those physical and biological features of the habitat that are essential to the conservation of the species and that may require special management considerations or protection. These features include (1) access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River, (2) the availability of clean gravel for spawning substrate, (3) adequate river flows for successful spawning,

incubation of eggs, fry development and emergence, and downstream transport of juveniles, (4) water temperatures between 42.5 and 57.5 degrees Fahrenheit for successful spawning, egg incubation and fry development, (5) habitat areas and adequate prey that are not contaminated, (6) riparian habitat that provides for successful juvenile development and survival, and (7) access downstream so that juveniles can migrate from spawning areas to San Francisco Bay and the Pacific Ocean.

Migration and Spawning (Coastal chinook ESUs): Chinook salmon are easily distinguished from other *Oncorhynchus* species by their large size. Adults weighing over 120 pounds have been caught in North American waters. Chinook salmon are very similar to coho salmon (*O. kisutch*) in appearance while at sea (blue-green back with silver flanks), except for their large size, small black spots on both lobes of the tail, and black pigment along the base of the teeth. Chinook salmon are anadromous and semelparous. This means that as adults they migrate from a marine environment into the fresh water streams and rivers of their birth (anadromous) where they spawn and die (semelparous). Adult female chinook will prepare a spawning bed, called a redd, in a stream area with suitable gravel composition, water depth and velocity. Redds will vary widely in size and in location within the stream or river. The adult female chinook may deposit eggs in 4 to 5 nesting pockets within a single redd. After laying eggs in a redd, adult chinook will guard the redd from 4 to 25 days before dying. Chinook salmon eggs will hatch, depending upon water temperatures, between 90 to 150 days after deposition. Stream flow, gravel quality, and silt load all significantly influence the survival of developing chinook salmon eggs. Juvenile chinook may spend from 3 months to 2 years in freshwater after emergence and before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. Historically, chinook salmon ranged as far south as the Ventura River, California, and their northern extent reaches Alaska and the Russian Far East.

Among chinook salmon, two distinct races have evolved. One race, described as a stream-type chinook, is found most commonly in headwater streams. Stream-type chinook salmon have a longer freshwater residency, and perform extensive offshore migrations before returning to their natal streams in the spring or summer months. The second race is called the ocean-type chinook, which are commonly found in coastal streams or the mainstem portions of larger rivers draining inland basins in North America. Ocean-type chinook typically migrate to sea within the first three months of emergence, but they may spend up to a year in freshwater prior to emigration. They also spend their ocean life in coastal waters. Ocean-type chinook salmon return to their natal streams or rivers as spring, winter, fall, summer, and late-fall runs, but summer and fall runs predominate (Healey 1991). The difference between these life history types is also physical, with both genetic and morphological foundations. Juvenile stream- and ocean-type chinook salmon have adapted to different ecological niches. Ocean-type chinook salmon tend to utilize estuaries and coastal areas more extensively for juvenile rearing. The brackish water areas in estuaries also moderate physiological stress during parr-smolt transition. The development of the ocean-type life history strategy may have been a response to the limited carrying capacity of smaller stream systems and glacially scoured, unproductive, watersheds, or a means of avoiding the impact of seasonal floods in the lower portion of many watersheds (Miller and Brannon

1982). Stream-type juveniles are much more dependent on freshwater stream ecosystems because of their extended residence in these areas. A stream-type life history may be adapted to those watersheds, or parts of watersheds, that are more consistently productive and less susceptible to dramatic changes in water flow, or which have environmental conditions that would severely limit the success of subyearling smolts (Miller and Brannon 1982; Healey 1991). At the time of saltwater entry, stream-type (yearling) smolts are much larger, averaging 73-134 mm depending on the river system, than their ocean- type (subyearling) counterparts and are therefore able to move offshore relatively quickly (Healey 1991).

Coast wide, chinook salmon remain at sea for 1 to 6 years (more commonly 2 to 4 years), with the exception of a small proportion of yearling males (called jack salmon) which mature in freshwater or return after 2 or 3 months in salt water (Rutter 1904; Gilbert 1912; Rich 1920; Mullan *et al.* 1992). Ocean- and stream-type chinook salmon are recovered differentially in coastal and mid-ocean fisheries, indicating divergent migratory routes (Healey 1983 and 1991). Ocean- type chinook salmon tend to migrate along the coast, while stream-type chinook salmon are found far from the coast in the central North Pacific (Healey 1983 and 1991; Myers *et al.* 1984). Differences in the ocean distribution of specific stocks may be indicative of resource partitioning and may be important to the success of the species as a whole.

Migration and Spawning (Sacramento River Winter-Run chinook ESU): The first winter-run chinook upstream migrants appear in the Sacramento-San Joaquin Delta during the early winter months (Skinner 1972). On the upper Sacramento River, the first upstream migrants appear during December (Vogel and Marine 1991). The upstream migration of winter-run chinook typically peaks during the month of March, but may vary with river flow, water-year type, and operation of Red Bluff Diversion Dam. Keswick Dam completely blocks any further upstream migration, forcing adults to migrate to and hold in deep pools downstream, before initiating spawning activities.

Since the construction of Shasta and Keswick Dam, winter-run chinook spawning has primarily occurred between Red Bluff Diversion Dam and Keswick Dam. The spawning period of winter-run chinook generally extends from mid-April to mid-August with peak activity occurring in June (Vogel and Marine 1991). Winter-run chinook may also spawn below Red Bluff in some years. In 1988, for example, winter-run chinook redds were observed as far downstream as Woodson Bridge. Winter-run chinook eggs hatch after an incubation period of about 40-60 days depending on ambient water temperatures. Maximum survival of incubating eggs and pre-emergent fry occurs at water temperatures between 42 degrees F and 56 degrees F with a preferred temperature of 52 degrees F. Mortality of eggs and pre-emergent fry commences at 57.5 degrees F and reaches 100 percent at 62 degrees F (Boles 1988).

The pre-emergent fry remain in the redd and absorb the yolk stored in their yolk-sac as they grow into fry. This period of larval incubation lasts approximately 6 to 8 weeks depending on water temperatures. Emergence of the fry from the gravel begins during late June and continues through September. The fry seek out shallow, nearshore areas with slow current and good cover, and begin feeding on small terrestrial and aquatic insects and aquatic crustaceans. As they grow to 50 to 75 mm in length, the juvenile salmon move out into deeper, swifter water, but continue to use available cover to minimize the risk of predation and reduce energy expenditure.

The emigration of juvenile winter-run chinook from the upper Sacramento River is highly dependent on streamflow conditions and water year type. Peak outmigration from the Delta typically occurs from late January through April. Optimal water temperatures for the growth of juvenile chinook salmon in an estuary are 54 to 57 degrees F (Brett 1952). High river flows in the winter and early spring assist juvenile fish migrating downstream to the estuary, while positive outflow from the Delta improves juvenile survival and migration to the ocean.

Available information on winter-run chinook salmon ocean distribution indicates that marked winter-run chinook salmon are caught between Monterey Bay and Fort Bragg, California. However, this data may be biased towards areas where commercial and recreational fisheries occur.

Migration and Spawning (Central Valley Spring-Run chinook ESU): Impassable dams block access to most of the historical headwater spawning and rearing habitat of Central Valley spring-run chinook salmon. In addition, much of the remaining, accessible spawning and rearing habitat is severely degraded by elevated water temperatures, agricultural and municipal water diversions, unscreened and poorly screen water intakes, restricted and regulated streamflows, levee and bank stabilization, and poor quality and quantity of riparian and shaded riverine aquatic (SRA) cover.

Natural spawning populations of Central Valley spring-run chinook salmon are currently restricted to accessible reaches in the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (DFG 1998; FWS, unpublished data). With the exception of Butte Creek and the Feather River, these populations are relatively small ranging from a few fish to several hundred. Butte Creek returns in 1998 and 1999 numbered approximately 20,000 and 3,600, respectively (DFG unpublished data). On the Feather River, significant numbers of spring-run chinook, as identified by run timing, return to the Feather River Hatchery. However, coded-wire-tag information from these hatchery returns indicates substantial introgression has occurred between fall-run and spring-run chinook populations in the Feather River due to hatchery practices. Over time, the spring-run within the Feather River may become homogeneous with Feather River fall-run fish unless current hatchery practices are changed.

Spring-run chinook salmon adults are estimated to leave the ocean and enter the Sacramento River from March to July (Myers et al. 1998). This run timing is well adapted for gaining access to the upper reaches of river systems, 1,500 to 5,200 feet in elevation, prior to the onset of high water temperatures and low flows that would inhibit access to these areas during the fall. Throughout this upstream migration phase, adults require streamflows sufficient to provide olfactory and other orientation cues used to locate their natal streams. Adequate streamflows are also necessary to allow adult passage to upstream holding habitat in natal tributary streams. The preferred temperature range for spring-run chinook salmon completing their upstream migration is 38° F to 56° F (Bell 1991; DFG 1998).

When they enter freshwater, spring-run chinook salmon are immature and they must stage for several months before spawning. Their gonads mature during their summer holding period in freshwater. Over-summering adults require cold-water refuges such as deep pools to conserve energy for gamete production, redd construction, spawning, and redd guarding. The upper limit of the optimal temperature range for adults holding while eggs are maturing is 59° F to 60° F (Hinz 1959). Unusual stream temperatures during spawning migration and adult holding periods can alter or delay migration timing, accelerate or retard mutations, and increase fish susceptibility to diseases. Sustained water temperatures above 80.6° F are lethal to adults (Cramer and Hammack 1952; DFG 1998).

Adults prefer to hold in deep pools with moderate water velocities and bedrock substrate and avoid cobble, gravel, sand, and especially silt substrate in pools (Sato and Moyle 1989). Optimal water velocities for adult chinook salmon holding pools range between 0.5-1.3 feet-per-second and depths are at least three to ten feet (G. Sato unpublished data, Marcotte 1984). The pools typically have a large bubble curtain at the head, underwater rocky ledges, and shade cover throughout the day (Ekman 1987).

Spawning typically occurs between late-August and early October with a peak in September. Once spawning is completed, adult spring-run chinook salmon die. Spawning typically occurs in gravel beds that are located at the tails of holding pools (USFWS 1995a). Spring-run adults have been observed spawning in water depths of 0.8 feet or more, and water velocities from 1.2-3.5 feet-per-second (Puckett and Hinton 1974). Eggs are deposited within the gravel where incubation, hatching, and subsequent emergence takes place. Optimum substrate for embryos is a mixture of gravel and cobble with a mean diameter of one to four inches with less than 5% fines, which are less than or equal to 0.3 inches in diameter (Platts et al. 1979, Reiser and Bjornn 1979). The upper preferred water temperature for spawning adult chinook salmon is 55° F (Chambers 1956) to 57° F (Reiser and Bjornn 1979).

Length of time required for eggs to develop and hatch is dependant on water temperature and is quite variable, however, hatching generally occurs within 40 to 60 days of fertilization (Vogel and Marine 1991). In Deer and Mill creeks, embryos hatch following a 3-5 month incubation period (USFWS 1995). The optimum temperature range for chinook salmon egg incubation is 44° F to 54° F (Rich 1997). Incubating eggs show reduced egg viability and increased mortality at temperatures greater than 58° F and show 100% mortality for temperatures greater than 63° F (Velson 1987). Velson (1987) and Beacham and Murray (1990) found that developing chinook salmon embryos exposed to water temperatures of 35° F or less before the eyed stage experienced 100% mortality (DFG 1998).

After hatching, pre-emergent fry remain in the gravel living on yolk-sac reserves for another two to four weeks until emergence. Timing of emergence within different drainages is strongly influenced by water temperature. Emergence of spring-run chinook typically occurs from November through January in Butte and Big Chico Creeks and from January through March in Mill and Deer Creeks (DFG 1998).

Post-emergent fry seek out shallow, nearshore areas with slow current and good cover, and begin feeding on small terrestrial and aquatic insects and aquatic crustaceans. As they grow to 50 to 75 mm in length, the juvenile salmon move out into deeper, swifter water, but continue to use available cover to minimize the risk of predation and reduce energy expenditure. The optimum temperature range for rearing chinook salmon fry is 50° F to 55° F (Boles et al. 1988, Rich 1997, Seymour 1956) and for fingerlings is 55° F to 60° F (Rich 1997).

In Deer and Mill creeks, juvenile spring-run chinook, during most years, spend 9-10 months in the streams, although some may spend as long as 18 months in freshwater. Most of these "yearling" spring-run chinook move downstream in the first high flows of the winter from November through January (USFWS 1995, DFG 1998). In Butte and Big Chico creeks, spring-run chinook juveniles typically exit their natal tributaries soon after emergence during December and January, while some remain throughout the summer and exit the following fall as yearlings. In the Sacramento River and other tributaries, juveniles may begin migrating downstream almost immediately following emergence from the gravel with emigration occurring from December through March (Moyle, et al. 1989, Vogel and Marine 1991). Fry and parr may spend time rearing within riverine and/or estuarine habitats including natal tributaries, the Sacramento River, non-natal tributaries to the Sacramento River, and the Delta. In general, emigrating juveniles that are younger (smaller) reside longer in estuaries such as the Delta (Kjelson et al. 1982, Levy and Northcote 1982, Healey 1991). The brackish water areas in estuaries moderate the physiological stress that occurs during parr-smolt transitions. Although fry and fingerlings can enter the Delta as early as January and as late as June, their length of residency within the Delta is unknown but probably lessens as the season progresses into the late spring months (DFG 1998).

Foraging Ecology: In an estuarine environment such as the Delta, juvenile chinook salmon forage in intertidal and shallow subtidal areas, such as marshes, mudflats, channels, and sloughs. These habitats provide protective cover and a rich food supply (McDonald 1960; Dunford 1975). The distribution of the juvenile fish appears to change tidally in an estuarine environment. Large fry and smolts tend to congregate in the surface waters of main and subsidiary sloughs and channels, moving into shallow subtidal areas only to feed (Allen and Hassler 1986).

Genetics: There is a significant genetic influence to the freshwater component of the returning adult migratory process. A number of studies show that chinook salmon return to their natal streams with a high degree of fidelity (Rich and Holmes 1928; Quinn and Fresh 1984; McIsaac and Quinn 1988). Salmon may have evolved this trait as a method of ensuring an adequate incubation and rearing habitat. It also provides a mechanism for reproductive isolation and local adaptation. Conversely, returning to a stream other than that of one's origin is important in colonizing new areas and responding to unfavorable or perturbed conditions at the natal stream (Quinn 1993).

Chinook salmon stocks exhibit considerable variability in size and age of maturation, and at least some portion of this variation is genetically determined. The relationship between size and length of migration may also reflect the earlier timing of river entry and the cessation of feeding for chinook salmon stocks that migrate to the upper reaches of river systems. Body size, which is correlated with age, may be an important factor in migration and redd construction success. Roni and Quinn (1995) reported that under high density conditions on the spawning ground, natural selection may produce stocks with exceptionally large-sized returning adults.

Artificial propagation and other human activities such as harvest and habitat modification can genetically change natural populations so much that they no longer represent an evolutionarily significant component of the biological species (Waples 1991). Artificial propagation is a common practice to supplement chinook salmon stocks for commercial and recreational fisheries. However, in many areas, a significant portion of the naturally spawning population consists of hatchery-produced chinook salmon. In several of the chinook salmon ESUs, over 50 percent of the naturally spawning fish are from hatcheries. Many of these hatchery- produced fish are derived from a few stocks which may or may not have originated from the geographic area where they are released. However, in several of the ESUs analyzed, insufficient or uncertain information exists regarding the interactions between hatchery and natural fish, and the relative abundance of hatchery and natural stocks. See the proposed rule for more information on the effects of artificial propagation on chinook salmon.

Among basins supporting only ocean-type chinook salmon, the Sacramento River system is somewhat unusual in that its large size and ecological diversity historically allowed for substantial spatial as well as temporal separation of different runs. Genetic and life history data both suggest that considerable differentiation among the runs has occurred in this basin. The Klamath River Basin, as well as chinook salmon in Puget Sound, share some features of coastal rivers but historically also provided an opportunity for substantial spatial separation of different temporal runs. As discussed below, the diversity in run timing made identifying ESUs difficult in the Klamath and Sacramento River Basins.

No allozyme data are available for naturally spawning Sacramento River spring chinook salmon. A sample from Feather River Hatchery spring-run fish, which may have undergone substantial hybridization with fall chinook salmon, shows modest (but statistically significant) differences from fall-run hatchery populations. DNA data show moderate genetic differences between the spring and fall/late-fall runs in the Sacramento River; however, these data are difficult to interpret because comparable data are not available for other geographic regions.

<u>Historic and Current Distribution</u>: NMFS considers differences in life history traits as a possible indicator of adaptation to different environmental regimes and resource partitioning within those regimes. The relevance of the ecologic and genetic basis for specific chinook salmon life-history traits as they pertain to each ESU is discussed in the brief summary that follows. NMFS calculated trends from the most recent 10 years using data collected after 1984 for series having at least 7 observations since 1984. No attempt was made to account for the influence of

hatchery-produced fish on these estimates, so the estimated trends include the progeny of naturally spawning hatchery fish. After evaluating patterns of abundance drawn on these quantitative and qualitative assessments, and evaluating other risk factors for chinook salmon from these ESUs, NMFS reached the conclusions summarized below.

Central Valley Spring-Run ESU (Threatened): Existing populations in this ESU spawn in the Sacramento River and its tributaries. Historically, spring chinook salmon were the dominant run in the Sacramento and San Joaquin River Basins (Clark 1929), but native populations in the San Joaquin River have apparently all been extirpated (Campbell and Moyle 1990). This ESU includes chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, subyearlings, and yearlings. Recoveries of hatchery chinook salmon implanted with coded-wire-tags (CWT) are primarily from ocean fisheries off the California and Oregon coast. There were minimal differences in the ocean distribution of fall- and spring-run fish from the Feather River Hatchery (as determined by CWT analysis); however, due to hybridization that may have occurred in the hatchery between these two runs, this similarity in ocean migration may not be representative of wild runs. Substantial ecological differences in the historical spawning habitat for spring-run versus fall- and late-fall-run fish have been recognized. Spring chinook salmon run timing was suited to gaining access to the upper reaches of river systems (up to 1,500 m elevation) prior to the onset of prohibitively high water temperatures and low flows that inhibit access to these areas during the fall. Differences in adult size, fecundity, and smolt size also occur between spring- and fall/late fall-run chinook salmon in the Sacramento River.

Native spring chinook salmon have been extirpated from all tributaries in the San Joaquin River Basin, which represents a large portion of the historic range and abundance of the ESU as a whole. The only streams considered to have wild spring-run chinook salmon are Mill and Deer Creeks, and possibly Butte Creek (tributaries to the Sacramento River), and these are relatively small populations with sharply declining trends. Demographic and genetic risks due to small population sizes are thus considered to be high. Current spawning is restricted to the mainstem and a few river tributaries in the Sacramento River. Most of the fish in this ESU are hatchery produced.

California Coastal ESU (Threatened): This ESU includes all naturally spawned coastal spring and fall chinook salmon spawning from the Russian River, in Sonoma County north to Redwood Creek in Humboldt County. Chinook salmon from the Central Valley and Klamath River Basin upstream form the Trinity River confluence are genetically and ecologically distinguishable from those in this ESU. Chinook salmon in this ESU exhibit an ocean-type life-history; ocean distribution (based on marine CWT recoveries) is predominantly off of the California and Oregon coasts. Life-history information on smaller populations, especially in the southern portion of the ESU, is extremely limited. Additionally, only anecdotal or incomplete information exists on abundance of several spring-run populations including, the Chetco, Winchuck, Smith, Mad, and Eel Rivers. Allozyme data indicate that this ESU is genetically distinguishable from the

Oregon Coast, Upper Klamath and Trinity River, and Central Valley ESUs. Life history differences also exist between spring- and fall-run fish in this ESU, but not to the same extent as is observed in larger inland basins. Ecologically, the majority of the river systems in this ESU are relatively small and heavily influenced by a maritime climate. Low summer flows and high temperatures in many rivers result in seasonal physical and thermal barrier bars that block movement by anadromous fish.

This ESU contains chinook salmon from the Russian River in Sonoma County, north to Redwood Creek in Humboldt County. Chinook salmon spawning abundance in this ESU is highly variable among populations. There is a general pattern of downward trends in abundance in most populations for which data are available, with declines being especially pronounced in spring-run populations. The extremely depressed status of almost all coastal populations south of the Klamath River is an important source of risk to the ESU. NMFS has a general concern that no current information is available for many river systems in the southern portion of this ESU, which historically maintained numerous large populations.

Sacramento River Winter-Run ESU (Endangered): The Sacramento River winter-run chinook salmon is a unique population of chinook salmon in the Sacramento River. It is distinguishable from the other three Sacramento River chinook runs by the timing of its upstream migration and spawning season.

Prior to construction of Shasta and Keswick dams in 1945 and 1950, respectively, winter-run chinook were reported to spawn in the upper reaches of the Little Sacramento, McCloud, and lower Pit rivers (Moyle *et al.* 1989). Specific data relative to the historic run sizes of winter-run chinook prior to 1967 are sparse and anecdotal. Numerous fishery researchers have cited Slater (1963) to indicate that the winter-run chinook population may have been fairly small and limited to the spring-fed areas of the McCloud River before the construction of Shasta Dam. However, recent CDFG research in California State Archives has cited several fisheries chronicles that indicate the winter-run chinook population may have been much larger than previously thought. According to these qualitative and anecdotal accounts, winter-run chinook reproduced in the McCloud, Pit and Little Sacramento rivers and may have numbered over 200,000 (Rectenwald 1989).

Completion of the Red Bluff Diversion Dam in 1966 enabled accurate estimates of all salmon runs to the upper Sacramento River based on fish counts at the fish ladders. These annual fish counts document the dramatic decline of the winter-run chinook population. The estimated number of winter-run chinook passing the dam from 1967 to 1969 averaged 86,509. During 1990, 1991, 1992, 1993, 1994, 1995, 1996, and 1997 the spawning escapement of winter-run chinook past the dam was estimated at 441, 191, 1180, 341, 189, 1361, 940, and 841 adults (including jacks), respectively.

<u>Reasons for Decline and Threats to Survival</u>: *Central Valley Spring-Run ESU*: Habitat problems are the most important source of ongoing risk to the Central Valley spring-run ESU. Spring-run

fish cannot access most of their historical spawning and rearing habitat in the Sacramento and San Joaquin River Basins (which is now above impassable dams). The remaining spawning habitat accessible to fish is severely degraded. Collectively, these habitat problems greatly reduce the resiliency of this ESU to respond to additional stresses in the future. The general degradation of conditions in the Sacramento River Basin (including elevated water temperatures, agricultural and municipal diversions and returns, restricted and regulated flows, entrainment of migrating fish into unscreened or poorly screened diversions, and the poor quality and quantity of remaining habitat) has severely impacted important juvenile rearing habitat and migration corridors. There appears to be serious concern for threats to genetic integrity posed by hatchery programs in the Central Valley. Most of the spring-run chinook salmon production in the Central Valley is of hatchery origin, and naturally spawning populations may be interbreeding with both fall/late fall- and spring-run hatchery fish. Related harvest regimes may not be allowing recovery of this at-risk population.

California Coastal ESU: Habitat loss and/or degradation is widespread throughout the range of the California Coastal ESU. The California Advisory Committee on Salmon and Steelhead Trout (CACSST) reported habitat blockages and fragmentation, logging and agricultural activities, urbanization, and water withdrawals as the most predominant problems for anadromous salmonids in California's coastal basins (CACSST 1988). They identified associated habitat problems for each major river system in California. CDFG (1965, Vol. III, Part B) reported that the most vital habitat factor for coastal California streams was “degradation due to improper logging followed by massive siltation, log jams, etc.” They cited road building as another cause of siltation in some areas. They identified a variety of specific critical habitat problems in individual basins, including extremes of natural flows (Redwood Creek and Eel River), logging practices (Mad, Eel, Mattole, Ten Mile, Noyo, Big, Navarro, Garcia, and Gualala Rivers), and dams with no passage facilities (Eel and Russian Rivers), and water diversions (Eel and Russian Rivers). Recent major flood events (February 1996 and January 1997) have probably affected habitat quality and survival of juveniles within this ESU. Artificial propagation programs in the California Coastal ESU are less extensive than those in Klamath/Trinity or Central Valley ESUs. The Rogue, Chetco and Eel River Basins and Redwood Creek have received considerable releases, derived primarily from local sources. Current hatchery contribution to overall abundance is relatively low except for the Rogue River spring run.

Sacramento River Winter-Run ESU: The main cause of decline of the winter-run chinook salmon was the damming of rivers that prevented instream migration. Associated factors contributing to the decline and threat of survival for winter-run chinook salmon include forestry, agriculture, mining, and urbanization that have degraded, simplified, and fragmented habitat significantly throughout the range of the species. Potential sources of mortality during the incubation period include redd dewatering, insufficient oxygenation, physical disturbance, and water-borne contaminants.

Infectious disease is one of the many factors that can influence adult and juvenile survival.

Chinook salmon are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment, poor water quality within these habitats increase steelhead vulnerability to disease and predation.

Overall Threats to Survival for all ESU's: Chinook salmon on the west coast of the United States have experienced declines in abundance in the past several decades as a result of loss, damage or change to their natural environment. Water diversions for agriculture, flood control, domestic, and hydropower purposes (especially in the Columbia River and Sacramento-San Joaquin Basins) have greatly reduced or eliminated historically accessible habitat and degraded remaining habitat. Forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat. Studies indicate that in most western states, about 80 to 90 percent of the historic riparian habitat has been eliminated (Botkin *et al.*, 1995; Norse, 1990; Kellogg, 1992; California State Lands Commission, 1993). Washington and Oregon wetlands are estimated to have diminished by one-third, while California has experienced a 91 percent loss of its wetland habitat. Loss of habitat complexity and habitat fragmentation have also contributed to the decline of chinook salmon. For example, in national forests within the range of the northern spotted owl in western and eastern Washington, there has been a 58 percent reduction in large, deep pools due to sedimentation and loss of pool-forming structures such as boulders and large wood (Forest Ecosystem Management Assessment Team (FEMAT) 1993). Similar or even an elevated level of effects are likely in California.

Introductions of non-native species and habitat modifications have resulted in increased predator populations in numerous rivers. Predation by marine mammals is also of concern in areas experiencing dwindling chinook salmon run sizes. However, salmonids appear to be a minor component of the diet of marine mammals (Scheffer and Sperry 1931; Jameson and Kenyon 1977; Graybill 1981; Brown and Mate 1983; Roffe and Mate 1984; Hanson 1993). Principal food sources are small pelagic schooling fish, juvenile rockfish, lampreys (Jameson and Kenyon 1977; Roffe and Mate 1984), benthic and epibenthic species (Brown and Mate 1983) and flatfish (Scheffer and Sperry 1931; Graybill 1981). Predation may significantly influence salmonid abundance in some local populations when other prey are absent and physical conditions lead to the concentration of adults and juveniles (Cooper and Johnson 1992).

Infectious disease is one of many factors that can influence adult and juvenile chinook salmon survival. Chinook salmon are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment. Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases for chinook salmon. However, studies have shown that naturally spawned fish tend to be less susceptible to pathogens than hatchery-reared fish (Buchanon *et al.* 1983; Sanders *et al.* 1992).

Competition, genetic introgression, and disease transmission resulting from hatchery introductions may significantly reduce the production and survival of native, naturally-reproducing chinook salmon. Collection of native chinook salmon for hatchery brood stock purposes often harms small or dwindling natural populations. Artificial propagation may play an important role in chinook salmon recovery, and some hatchery populations of chinook salmon may be deemed essential for the recovery of threatened or endangered chinook salmon ESUs. While some limits have been placed on hatchery production of anadromous salmonids, more careful management of current programs and scrutiny of proposed programs is necessary in order to minimize impacts on listed species.

The CWA, enforced in part by the EPA, is intended to protect beneficial uses, including fishery resources. To date, implementation has not been effective in adequately protecting fishery resources, particularly with respect to non-point sources of pollution. In addition, section 404 of the CWA does not adequately address the cumulative and additive effects of loss of habitat through continued development of waterfront, riverine, coastal, and wetland properties that also contribute to the degradation and loss of important aquatic ecosystem components necessary to maintain the functional integrity of these habitat features.

Sections 303 (d) (1) (C) and (D) of the CWA require states to prepare Total Maximum Daily Loads (TMDLs) for all water bodies that do not meet State water quality standards. Development of TMDLs is a method for quantitative assessment of environmental problems in a watershed and identification of pollution reductions needed to protect drinking water, aquatic life, recreation, and other uses of rivers, lakes, and streams. Appropriately protective aquatic life criteria are critical to the TMDL process for affecting the recovery of salmon populations, as the criteria exceedance will determine which waterbodies will engage in the TMDL process and criteria compliance goals are the impetus for developing mass loading strategies. The ability of these TMDLs to protect chinook salmon should be significant in the long term; however, it will be difficult to develop them quickly in the short term, and their efficacy in protecting chinook salmon habitat will be unknown for years to come.

Coho Salmon (Including Central California Coast and Southern Oregon/Northern California Coast ESUs) (*Oncorhynchus kisutch*)

Species Description and Life History: General life history information for coho salmon is summarized below, followed by information on population trends for each coho salmon ESU. Further detailed information on these coho salmon ESUs is available in the NMFS Status Review of coho salmon from Washington, Oregon, and California (Weitkamp *et al.* 1995), the NMFS proposed rule for listing coho (60 **FR** 38011), and the NMFS final listings for the Central California Coast coho ESU (61 **FR** 56138) and the Southern Oregon/Northern California Coast coho ESU (62 **FR** 24588). On May 5, 1999, NMFS designated critical habitat for the Central California Coast and the Southern Oregon/Northern California Coast coho salmon ESUs (64 **FR** 24049). The designation includes all accessible reaches of rivers between the Elk River in Oregon and the San Lorenzo River in Santa Cruz County, California. This designation also includes two rivers entering the San Francisco Bay: Mill Valley Creek and Corte Madera Creek. For both ESUs, critical habitat includes the water, substrate, and adjacent riparian zones.

Critical Habitat: Central California Coast ESU coho geographic boundaries encompass accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda (near the Mattole River, Mendocino County) and the San Lorenzo River (Santa Cruz County), inclusive, and including two streams that enter San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creeks.

Southern Oregon/Northern California Coast ESU coho geographic boundaries encompass accessible reaches of all rivers (including estuarine areas and tributaries) between the Mattole River (Mendocino County) and the Elk River in Oregon, inclusive.

Migration and Spawning: Most coho salmon adults are 3-year-olds, having spent approximately 18 months in freshwater and 18 months in salt water (Gilbert 1912; Pritchard 1940; Briggs 1953; Shapovalov and Taft 1954; Loeffel and Wendler 1968). The primary exception to this pattern are 'jacks', which are sexually mature males that return to freshwater to spawn after only 5-7 months in the ocean.

Most west coast coho salmon enter rivers in October and spawn from November to December and occasionally into January. However, both run and spawn-timing of Central California coho salmon are very late (peaking in January) with little time spent in freshwater between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this area. Many small California systems have sandbars which block their mouths for most of the year except during winter. In these systems, coho salmon and other salmon species are unable to enter the rivers until sufficiently strong freshets break the sandbars (Sandercock 1991).

While central California coho spend little time between river entry and spawning, northern stocks may spend 1 or 2 months in fresh water before spawning (Flint and Zillges 1980; Fraser *et al.* 1983). In larger river systems like the Klamath River, coho salmon have a broad period of freshwater entry spanning from August until December (Leidy and Leidy 1984). In general, earlier migrating fish spawn farther upstream within a basin than later migrating fish, which enter rivers in a more advanced state of sexual maturity (Sandercock 1991). Adult coho salmon normally migrate when water temperatures are 44.96 to 60.08 degrees F, minimum water depth is seven inches and streamflow velocity does not exceed 2.44 m/s (Reiser and Bjornn 1979). If the conditions are not right, coho will wait at the mouth of the river or stream for the correct conditions. Most coho stocks migrate upstream during daylight hours. Generally, the coho build their redds at the head of riffles where there is good intra-gravel flow and oxygenation. Gribanov (1948) found that spawning coho appear to favor areas where the stream velocity is 0.30 to 0.55 m/s. Water quality can be clear or heavily silted with varying substrate of fine gravel to coarse rubble. California coho spawn in water temps of 42.08 to 55.94 degrees F (Briggs 1953).

Coho salmon eggs hatch in approximately 38 days at 51.26 degrees F, but, this duration depends on ambient water temperatures (Shopovalov and Taft 1954). Young fry hide in gravel and under large rocks during daylight hours. After several days growth, they move closer to the banks seeking out quiet backwaters, side channels and small creeks, especially those with overhanging riparian vegetation (Gribanov (1948). As they grow, they move into areas with less cover and higher velocity flows (Lester and Genoe 1970). Most fry move out of the system with winter and early spring freshets; however, some level of emigration may occur all year long. Brett (1952) found that coho salmon juveniles had an upper lethal temperature of 77 degrees F with a preferred rearing and emigration range of 53.6 to 57.2 degrees F. Taking advantage of cooler ambient temperatures and the afforded protection from predators, the bulk of seaward migration occurs at night.

Peak outmigration timing generally occurs in May, about a year after they emerge from the gravel. In California, smolts migrate to the ocean somewhat earlier, from mid-April to mid-May. Most smolts measure 90-115 mm, although Klamath River Basin smolts tend to be larger, but this is possibly due to influences of off-station hatchery plants. After entering the ocean, immature coho salmon initially remain in near-shore waters close to the parent stream. In general, coded-wire tag (CWT) recoveries indicate that coho salmon remain closer to their river of origin than do chinook salmon, but coho may nevertheless travel several hundred miles (Hassler 1987).

Foraging Ecology: Coho salmon fry usually emerge from the gravel at night from March to May. Coho salmon fry begin feeding as soon as they emerge from the gravel, and grow rapidly. In California, fry move into deep pools in July and August, where feeding is reduced and growth rate decreased (Shapovalov and Taft 1954). Between December and February winter rains result in increased stream flows and by March, following peak flows, fish feed heavily again on insects and crustaceans and grow rapidly.

<u>Historic and Current Distribution</u>: *Southern Oregon/Northern California Coast ESU (Threatened):* Recently, most coho salmon production in the Oregon portion of this ESU has been in the Rogue River. Recent run-size estimates (1979-1986) have ranged from about 800 to 19,800 naturally-produced adults, and from 500 to 8,300 hatchery-produced adults (Cramer 1994). Average annual run sizes for this period were 4,900 natural and 3,900 hatchery fish, with the total run averaging 45 percent hatchery fish. Adult passage counts at Gold Ray dam provide a long-term view of coho salmon abundance in the upper Rogue River (Cramer *et al.* 1985). In the 1940s, passage counts averaged about 2,000 adults per year. Numbers declined and fluctuated during the 1950s and early 1960s, then stabilized at an average of fewer than 200 adults during the late 1960s and early 1970s. In the late 1970s, the run increased with returning fish produced at Cole Rivers Hatchery. The remaining data is angler catch, which has ranged from less than 50 during the late 1970s to a peak of about 800 in 1991. Average annual catch over the least 10 years has been about 500 fish.

In the northern California region of this ESU, CDFG reported that coho salmon including hatchery stocks could be less than 6 percent of their abundance during the 1940s and have experienced at least a 70 percent decline in numbers since the 1960s (CDFG 1994). The Klamath River Basin (including the Trinity River) historically supported abundant coho salmon runs. In both systems, runs have greatly diminished and are now composed largely of hatchery fish, although small wild runs may remain in some tributaries (CDFG 1994).

Of 396 streams within the range of this ESU identified as once having coho salmon runs, recent survey information is available for 115 streams (30 percent) (Brown *et al.* 1994). Of these 117 streams, 73 (62 percent) still support coho salmon runs while 42 (36 percent) have lost their coho salmon runs. The rivers and tributaries in the California portion of this ESU were estimated to have average recent runs of 7,080 natural spawners and 17,156 hatchery returns, with 4,480 identified as native fish occurring in tributaries having little history of supplementation with non-native fish. Combining recent run-size estimates for the California portion of this ESU with the Rogue River estimates provides a run-size estimate for the entire ESU of about 12,000 natural fish and 21,000 hatchery fish.

Central California Coast ESU (Threatened): Statewide (including areas outside this ESU) coho salmon spawning escapement in California apparently ranged between 200,000 to 500,000 adults per year in the 1940s (Brown *et al.* 1994). By the mid-1960s, statewide spawning escapement was estimated to have fallen to about 100,000 fish per year (CDFG 1965; California Advisory Committee on Salmon and Steelhead Trout 1988), followed by a further decline to about 30,000 fish in the mid-1980s (Wahle and Pearson 1987; Brown *et al.* 1994). From 1987 to 1991, spawning escapement averaged about 31,000 with hatchery populations composing 57% of this total (Brown *et al.* 1994). Brown *et al.* (1994) estimated that there are probably less than 5,000 naturally-spawning coho salmon spawning in California each year, and many of these fish are in populations that contain less than 100 individuals.

Estimated average coho salmon spawning escapement in the Central California ESU for the period from the early 1980s through 1991 was 6,160 naturally spawning coho salmon and 332 hatchery spawned coho salmon (Brown *et al.*1994). Of the naturally-spawning coho salmon, 3,880 were from the tributaries in which supplementation occurs (the Noyo River and coastal streams south of San Francisco). Only 160 fish in the range of this ESU (all in the Ten Mile River) were identified as “native” fish lacking a history of supplementation with the non-native hatchery stocks. Based on redd counts, the estimated run of coho salmon in the Ten Mile River was 14 to 42 fish during the 1991-1992 spawning season (Maahs and Gilleard 1994).

Of 186 streams in the range of the Central California ESU identified as having historic accounts of adult coho salmon, recent data exist for 133 (72 percent). Of these 133 streams, 62 (47 percent) have recent records of occurrence of adult coho salmon and 71 (53 percent) no longer maintain coho salmon spawning runs.

<u>Reasons for Decline and Threats to Survival</u>: The factors threatening naturally reproducing coho salmon throughout its range are varied and numerous. For coho populations in the Central California coast ESU, the present depressed condition is the result of several long-standing, human induced factors (e.g., habitat degradation, timber harvest, water diversions, and artificial propagation).

Among other factors contributing to the decline and threat of survival for west coast coho, forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat significantly throughout the range of the species. Water diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat. Studies estimate that during the last 200 years, the lower 48 states have lost approximately 53% of all wetlands and the majority of the rest are severely degraded (Dahl, 1990; Tiner, 1991). California has experienced a 91 percent loss of its wetland habitat (Dahl, 1990; Jensen *et al.*,1990; Barbour *et al.*, 1991; Reynolds *et al.*, 1993).

Infectious disease is one of the many factors that can influence adult and juvenile survival. Coho are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment, poor water quality within these habitats increase coho vulnerability to disease and predation.

Implementation of existing regulatory mechanisms, specifically sections 303 (d) (1) (C) and (D) of the CWA, designed to protect beneficial resources including fisheries resources have not been effective in protecting fisheries resources or the aquatic ecosystem on which they depend, particularly with respect to non-point sources of pollution.. In addition, section 404 of the CWA does not adequately address the cumulative and additive effects of loss of habitat through continued development of waterfront, riverine, coastal, and wetland properties that also contribute to the degradation and loss of important aquatic ecosystem components necessary to maintain the functional integrity of these habitat features.

Delta Smelt (*Hypomesus transpacificus*)

Species Description and Life History: The delta smelt was federally listed as a threatened species on March 5, 1993 (58 **FR** 12854). On December 19, 1994, a final rule designating critical habitat for the delta smelt was published in the Federal Register (59 **FR** 65256). Critical habitat for delta smelt was originally proposed in the lower Sacramento-San Joaquin Delta and Suisun and Honker bays. However, after considerable debate, critical habitat was reproposed and is now contained within Contra Costa, Sacramento, San Joaquin, Solano, and Yolo counties.

The delta smelt is a slender-bodied fish with a steely blue sheen on the sides, and appears almost translucent (Moyle 1976a). They have an average length of 60 to 70 mm (about two to 3 inches). The delta smelt is a euryhaline species (tolerant of a wide salinity range) that spawns in fresh water and has been collected from estuarine waters up to 14 parts per thousand (ppt) salinity (Moyle *et al.* 1992). For a large part of its annual life span, this species is associated with the freshwater edge of the mixing zone (a saltwater-freshwater interface; also called X2), where the salinity is approximately two ppt (Ganssle 1966; Moyle *et al.* 1992; Sweetnam and Stevens 1993).

The delta smelt is adapted to living in the highly productive San Francisco Bay/Delta Estuary (Estuary) where salinity varies spatially and temporally according to tidal cycles and the amount of freshwater inflow. Despite this tremendously variable environment, the historical Estuary probably offered relatively constant suitable habitat conditions for the delta smelt because it could move upstream or downstream with the mixing zone (Moyle, pers. comm., 1993).

Feeding ecology: Delta smelt feed primarily on planktonic copepods, cladocerans (small crustaceans), amphipods, and to a lesser extent, insect larvae. Larger fish may also feed on the opossum shrimp (*Neomysis mercedis*). The most important food item for all age classes is the euryhaline copepod (*Eurytemora affinis*). Delta smelt are a pelagic fish and their food source is within the water column.

Spawning behavior: Shortly before spawning, adult delta smelt migrate upstream from the brackish-water habitat associated with the mixing zone to disperse widely into river channels and tidally-influenced backwater sloughs (Radtke 1966; Moyle 1976a; Wang 1991). Migrating adults with nearly mature eggs were taken at the Central Valley Project's (CVP) Tracy Pumping Plant from late December 1990 to April 1991 (Wang 1991). Spawning locations appear to vary widely from year to year (DWR and USDI 1993). Sampling of larval delta smelt in the Delta suggests spawning has occurred in the Sacramento River, Barker, Lindsey, Cache, Georgiana, Prospect, Beaver, Hog, and Sycamore sloughs, in the San Joaquin River off Bradford Island including Fisherman's Cut, False River along the shore zone between Frank's and Webb tracts, and possibly other areas (Dale Sweetnam, Calif. Dept. Of Fish and Game, pers. comm.; Wang 1991). Delta smelt also may spawn north of Suisun Bay in Montezuma and Suisun sloughs and their tributaries (Sweetnam, Calif. Dept. Of Fish and Game, pers. comm.).

Delta smelt spawn in shallow, fresh, or slightly brackish water upstream of the mixing zone (Wang 1991). Most spawning occurs in tidally-influenced backwater sloughs and channel edgewaters (Moyle 1976a; Wang 1986, 1991; Moyle *et al.* 1992). Although delta smelt spawning behavior has not been observed in the wild (Moyle *et al.* 1992), the adhesive, demersal eggs are thought to attach to substrates such as cattails, tules, tree roots, and submerged branches (Moyle 1976a; Wang 1991).

The spawning season varies from year to year, and may occur from late winter (December) to early summer (July). Moyle (1976a) collected gravid adults from December to April, although ripe delta smelt were most common in February and March. In 1989 and 1990, Wang (1991) estimated that spawning had taken place from mid-February to late June or early July, with peak spawning occurring in late April and early May. A recent study of delta smelt eggs and larvae (Wang and Brown 1994 as cited in DWR & USDI 1994) confirmed that spawning may occur from February through June, with a peak in April and May. Spawning has been reported to occur at water temperatures of about 7° to 15° C. Results from a University of California at Davis (UCD) study (Swanson and Cech 1995) indicate that although delta smelt tolerate a wide range of temperatures (<8° C to >25° C), warmer water temperatures restrict their distribution more than colder water temperatures.

Laboratory observations indicate that delta smelt are broadcast spawners that spawn in a current, usually at night, distributing their eggs over a local area (Lindberg 1992 and Mager 1993 as cited in DWR & USDI 1994). The eggs form an adhesive foot that appears to stick to most surfaces. Eggs attach singly to the substrate, and few eggs were found on vertical plants or the sides of a culture tank (Lindberg 1993 as cited in DWR & USDI 1994).

Delta smelt eggs hatched in nine to 14 days at water temperatures ranging from 13° to 16° C during laboratory observations in 1992 (Mager 1992 as cited in Sweetnam and Stevens 1993). In this study, larvae began feeding on phytoplankton on day four, rotifers on day six, and *Artemia nauplii* at day 14. In laboratory studies, yolk-sac fry were found to be positively phototaxic, swimming to the lightest corner of the incubator, and negatively buoyant, actively swimming to the surface. The post-yolk-sac fry were more evenly distributed throughout the water column (Lindberg 1992 as cited in DWR & USDI 1994). After hatching, larvae and juveniles move downstream toward the mixing zone where they are retained by the vertical circulation of fresh and salt waters (Stevens *et al.* 1990). The pelagic larvae and juveniles feed on zooplankton. When the mixing zone is located in Suisun Bay where there is extensive shallow water habitat within the euphotic zone (depths less than four meters), high densities of phytoplankton and zooplankton may accumulate (Arthur and Ball 1978, 1979, 1980). In general, estuaries are among the most productive ecosystems in the world (Goldman and Horne 1993). Estuarine environments produce an abundance of fish and zooplankton as a result of plentiful food and shallow, productive habitat.

Swimming behavior. Observations of delta smelt swimming in the swimming flume and in a large tank show that these fish are unsteady, intermittent, slow-speed swimmers (Swanson and Cech 1995). At low velocities in the swimming flume (<three body lengths per second), and during spontaneous, unrestricted swimming in a 1-meter tank, delta smelt consistently swam with a "stroke and glide" behavior. This type of swimming is very efficient; Weihs (1974) predicted energy savings of about 50 percent for "stroke and glide" swimming compared to steady swimming. However, the maximum speed delta smelt are able to achieve using this preferred mode of swimming, or gait, is less than three body lengths per second, and the fish did not readily or spontaneously swim at this or higher speeds (Swanson and Cech 1995). Juvenile delta smelt proved stronger swimmers than adults. Forced swimming at these speeds in a swimming flume was apparently stressful; the fish were prone to swimming failure and extremely vulnerable to impingement. Unlike fish for which these types of measurements have been made in the past, delta smelt swimming performance was limited by behavioral rather than physiological or metabolic constraints *(e.g.,* metabolic scope for activity; Brett 1976). Please refer to the Service (USDI-FWS 1994a, 1996a) and Department of Water Resources and United States Department of Interior - Bureau of Reclamation (DWR & USDI 1994) for additional information on the biology and ecology of this species.

Primary Constituent Elements of Critical Habitat: In designating critical habitat for the delta smelt, the Service identified the following primary constituent elements essential to the conservation of the species: physical habitat, water, river flow, and salinity concentrations required to maintain delta smelt habitat for spawning, larval and juvenile transport, rearing, and adult migration.

Spawning habitat. Specific areas that have been identified as important delta smelt spawning habitat include Barker, Lindsey, Cache, Prospect, Georgiana, Beaver, Hog, and Sycamore sloughs and the Sacramento River in the Delta, and tributaries of northern Suisun Bay.

Larval and juvenile transport. Adequate river flow is necessary to transport larvae from upstream spawning areas to rearing habitat in Suisun Bay and to ensure that rearing habitat is maintained in Suisun Bay. To ensure this, X2 must be located westward of the confluence of the Sacramento-San Joaquin Rivers, located near Collinsville (Confluence), during the period when larvae or juveniles are being transported, according to historical salinity conditions. X2 is important because the "entrapment zone" or zone where particles, nutrients, and plankton are "trapped", leading to an area of high productivity, is associated with its location. Habitat conditions suitable for transport of larvae and juveniles may be needed by the species as early as February 1 and as late as August 31, because the spawning season varies from year to year and may start as early as December and extend until July.

Rearing habitat. An area extending eastward from Carquinez Strait, including Suisun, Grizzly, and Honker bays, Montezuma Slough and its tributary sloughs, up the Sacramento River to its confluence with Three Mile Slough, and south along the San Joaquin River including Big Break, defines the specific geographic area critical to the maintenance of suitable rearing habitat. Three Mile Slough represents the approximate location of the most upstream extent of historical tidal incursion. Rearing habitat is vulnerable to impacts of export pumping and salinity intrusion from the beginning of February to the end of August.

Adult migration. Adequate flow and suitable water quality are needed to attract migrating adults in the Sacramento and San Joaquin river channels and their associated tributaries, including Cache and Montezuma sloughs and their tributaries. These areas are vulnerable to physical disturbance and flow disruption during migratory periods.

<u>Historic and Current Distribution</u>: The delta smelt is endemic to Suisun Bay upstream of San Francisco Bay through the Delta in Contra Costa, Sacramento, San Joaquin, Solano and Yolo counties, California. Historically, the delta smelt is thought to have occurred from Suisun Bay upstream to at least the city of Sacramento on the Sacramento River, and Mossdale on the San Joaquin River (Moyle *et al.* 1992; Sweetnam and Stevens 1993).

<u>Reasons for Decline and Threats to Survival</u>: The delta smelt is adapted to living in the highly productive Estuary where salinity varies spatially and temporally according to tidal cycles and the amount of freshwater inflow. Despite this tremendously variable environment, the historical Estuary probably offered relatively consistent spring transport flows that moved delta smelt juveniles and larvae downstream to the mixing zone (P. Moyle, pers. comm.). Since the 1850's, however, the amount and extent of suitable habitat for the delta smelt has declined dramatically. The advent in 1853 of hydraulic mining in the Sacramento and San Joaquin rivers led to

increased siltation and alteration of the circulation patterns of the Estuary (Nichols *et al.* 1986; Monroe and Kelly 1992). The reclamation of Merritt Island for agricultural purposes, in the same year, marked the beginning of the present-day cumulative loss of 94 percent of the Estuary's tidal marshes (Nichols *et al.* 1986; Monroe and Kelly 1992).

In addition to the degradation and loss of estuarine habitat, the delta smelt has been increasingly subject to entrainment, upstream or reverse flows of waters in the Delta and San Joaquin River, and constriction of low salinity habitat to deep-water river channels of the interior Delta (Moyle *et al.* 1992). These adverse conditions are primarily a result of drought and the steadily increasing proportion of river flow being diverted from the Delta by the CVP and State Water Project (SWP) (Monroe and Kelly 1992). The relationship between the portion of the delta smelt population west of the Delta as sampled in the summer townet survey and the natural logarithm of Delta outflow from 1959 to 1988 (Department and Reclamation 1994) indicates that the summer townet index increased dramatically when outflow was between 34,000 and 48,000 cfs which placed X2 between Chipps and Roe islands. Placement of X2 downstream of the Confluence, Chipps and Roe islands provides delta smelt with low salinity and protection from entrainment, allowing for productive rearing habitat that increases both smelt abundance and distribution.

Delta smelt critical habitat has been affected by activities that destroy spawning and refugial areas and change hydrology patterns in Delta waterways. Critical habitat also has been affected by diversions that have shifted the position of X2 upstream of the confluence of the Sacramento and San Joaquin rivers. This shift has caused a decreased abundance of delta smelt. Existing baseline conditions and implementation of the Service's 1994 and 1995 biological opinions concerning the operation of the CVP and SWP, provide a substantial part of the necessary positive riverine flows and estuarine outflows to transport delta smelt larvae downstream to suitable rearing habitat in Suisun Bay outside the influence of marinas, agricultural diversions, and Federal and State pumping plants.

The Service's 1994 and 1995 biological opinions provided for adequate larval and juvenile transport flows, rearing habitat, and protection from entrainment for upstream migrating adults (USDI-FWS 1994a). Please refer to 59 **FR** 65255 for additional information on delta smelt critical habitat.

Desert Pupfish (*Cyprinodon macularius*)

<u>Species Description and Life History</u>: On March 31, 1986 (51 **FR** 10850), the Service determined the desert pupfish to be an endangered species and critical habitat was designated for this species in Imperial County, California and Pima County, Arizona.

The desert pupfish is a small laterally compressed fish with a smoothly rounded body shape. Adult fish rarely grow larger than 75 millimeters (3 inches) in total length. Males are larger than females and during the reproductive season become brightly colored with blue on the dorsal portion of the head and sides and yellow on the caudal fin and the posterior part of the caudal peduncle. Females and juveniles typically have tan to olive backs and silvery sides. Most adults have narrow, vertical, dark bars on their sides, which are often interrupted to give the impression of a disjunct, lateral band. They are adapted to harsh desert environments and are capable of surviving extreme environmental conditions (Moyle 1976a; and Lowe *et al.* 1967). Although desert pupfish are extremely hardy in many respects, they cannot tolerate competition or predation and are thus readily displaced by exotic fishes.

Desert pupfish mature rapidly and may produce up to three generations per year. Spawning males typically defend a small spawning and feeding territory in shallow water. The eggs are usually laid and fertilized on a flocculent substrate and hatch within a few days. After a few hours, the young begin to feed on small plants and animals. Spawning occurs throughout the spring and summer months. Individuals typically survive for about a year. Desert pupfish forage on a variety of insects, other invertebrates, algae, and detritus.

Foraging Ecology: Desert pupfish typically occur in shallow water and forage on a variety of insects, other invertebrates, algae, and detritus.

Historic and Current Distribution: The desert pupfish was once common in the desert springs, marshes, and tributary streams of the lower Gila and Colorado River drainages in Arizona, California, and Mexico (Minckly 1973 & 1980; Miller and Fuiman 1987; USDI-FWS 1993b). It also formerly occurred in the slow-moving reaches of some large rivers, including the Colorado, Gila, San Pedro, and Santa Cruz. In California, this species is currently known from only a few historic locations. It still exists in two Salton Sea tributaries (San Felipe Creek system and its associated wetland San Sebastian Marsh, Imperial County, and Salt Creek, Riverside County) and a few shoreline pools and irrigation drains along the Salton Sea in Imperial and Riverside Counties (Nichol *et al.* 1991; USDI-FWS 1993b).

Reasons for Decline and Threats to Survival: There are many reasons for declines of desert pupfish populations. They include habitat loss (dewatering of springs, some headwaters, and lower portions of major streams and marshlands), habitat modification (stream impoundment, channelization, diversion, and regulation of discharge, plus domestic livestock grazing and other watershed uses such as mining, and road construction), pollution, and interactions with non-native species (competition for food and space, and predation) (Matsui 1981; Minckley 1985; Miller and Fuiman 1987; USDI-FWS 1993b).

Many historic pupfish localities have been dried by groundwater pumping, channel erosion or arroyo formation, and water impoundment and diversion (Hastings and Turner 1965, Fradkin 1981, Rea 1983, Hendrickson and Minckley 1985). Impoundment also creates upstream habitat unsuitable for pupfish because of increased depth which, because of its lentic character, is more conducive to occupation by non-native fishes. Grazing by domestic livestock may reduce terrestrial vegetative cover, enhance watershed erosion, exacerbate problems of arroyo cutting, and increase sediment loads and turbidity in receiving waters. Habitats may be further impacted by trampling where cattle feed or drink in or adjacent to water. Contamination of the habitat of desert pupfish may have contributed to its decline.

Non-native fishes pose the greatest threat to extant desert pupfish populations (Minckley and Deacon 1968, Deacon and Minckley 1974, Schoenherr 1981 & 1988, Meffe 1985, Miller and Fuiman 1987). Non-native fishes that occupy habitats also used by pupfish include mosquitofish (*Gambusia affinis*), sailfin molly (*Poecilia latipinna*), largemouth bass (*Micropterus salmoides*), and juvenile cichlids (*Oreochromis ssp.* and *Tilapia ssp.*). Primary mechanisms of replacement include predation and aggression (mosquitofish and largemouth bass) and behavioral activities that interfere with reproduction (mollies and cichlids) (Matsui 1981, Schoenherr 1988).

As part of the National Irrigation Water Quality Program, the Service conducted a study to determine body burdens of contaminants in a surrogate species, sailfin mollies (*Poecilia latipinna*) for the endangered desert pupfish. Sailfin mollies were trapped in 13 agricultural drains. At one drain sampling site both mollies and desert pupfish were collected and submitted for analysis; contaminant levels between the two species were generally in agreement, especially for selenium. Mollies collected from 10 of 13 drains and pupfish contained 3 to 6 ppm dry weight selenium, above the levels of concern for warmwater fishes (CAST, 1994; Gober, 1994; Ohlendorf, 1996). Mollies in two other drains contained 6.4 and 10.2 ppm, dry weight selenium, above thresholds for toxicity for warmwater fish reproductive hazards (Lemly 1993a). Lemly (1993a), concluded that 4 ppm dry weight whole body selenium should be considered the toxic effect threshold for the overall health of and reproductive vigor for freshwater fish. These findings indicate that the desert pupfish is likely at risk to reduced reproductive vigor and condition as a result of elevated levels of selenium in its environment.

Lahontan Cutthroat Trout (*Oncorhynchus clarki henshawi*)

<u>Species Description and Life History</u>: The Lahontan cutthroat trout is an inland subspecies of cutthroat trout endemic to the physiographic Lahontan basin of northern Nevada, eastern California, and southern Oregon. It was listed as endangered by the Service in 1970 (35 **FR** 13520) and subsequently reclassified as threatened in 1975 (40 **FR** 229864). No critical habitat has been designated for this species.

The Lahontan cutthroat trout can be distinguished from other subspecies of cutthroat trout by three characteristics identified by Behnke (1979, 1992). These characteristics include: (1) the pattern of medium-large rounded spots, somewhat evenly distributed over the sides of the body, on the head, and often on the abdomen; (2) the highest number of gill rakers found in any trout, 21 to 28, with mean values ranging from 23 to 26; and (3) a high number of pyloric caeca, 40 to 75 or more, with mean values of more than 50.

Lahontan cutthroat trout inhabit both lakes and streams, but are obligatory stream spawners. Intermittent tributary streams are frequently used as spawning sites (Coffin 1981; Trotter 1987). Spawning generally occurs from April through July, depending on stream flow, elevation, and water temperature (Calhoun 1942; La Rivers 1962; McAfee 1966; Lea 1968; Moyle 1976a). Eggs are deposited in 0.25 to 0.5 inch gravels within riffles, pocket water, or pool crests. Spawning beds must be well oxygenated and relatively silt free for good egg survival. Optimum Lahontan cutthroat trout habitat is characterized by 1:1 pool-riffle ratios, well vegetated stable stream banks, over 25 percent cover, and a relatively silt free rocky substrate (Hickman and Raleigh 1982). They can tolerate much higher alkalinities than other trout and seem to survive daily temperature fluctuations of 14-20 degrees C (57-68 degrees F). They do best in waters with average maximum temperatures of 13 degrees C (55 degrees F).

Foraging Ecology: Lahontan cutthroat trout are opportunistic feeders; in streams they feed on the most common terrestrial and aquatic insects which get caught in the drift (Coffin 1983).

Historic and Current Distribution: Lahontan cutthroat trout historically occupied a wide variety of cold water habitats, including large terminal alkaline lakes, oligotrophic alpine lakes, meandering low-gradient rivers, montane rivers, and small headwater tributary streams. Prior to this century, there were 11 lake populations and an estimated 300 to 600 river populations in more than 3,600 miles of streams (USDI-FWS 1995). The western Lahontan Basin population segment includes the Truckee, Carson, and Walker River basins in California.

Lahontan cutthroat trout currently occupy between 155 and 160 streams as well as six lakes and reservoirs in California, Nevada, Oregon, and Utah. Self-sustaining populations occur in 10.7 percent of fluvial and 0.4 percent of lacustrine historical habitat (USDI-FWS 1995). The species has been introduced outside of its native range, primarily for recreational angling purposes. Three distinct vertebrate population segments have been identified by the Service based on geographical, ecological, behavioral, and genetic factors (USDI-FWS 1995).

Lahontan cutthroat trout were introduced into the Upper Truckee River watershed in 1990 and 1991 as part of the species' recovery program. The Upper Truckee River is within a watershed that historically contained Lahontan cutthroat trout. During the summer and fall of 1990, 5,000 fingerlings and 200 adults were planted. In 1991, 2,000 fingerlings and 110 adults were planted into the Upper Truckee River watershed. Before Lahontan cutthroat trout were introduced into these waters, the streams and lakes were treated by CDFG to remove non-native salmonids. The LTBMU has conducted ocular surveys annually since the introduction. In 1995, just under 250 fish were observed, mostly adults. This is down from the 1994 survey of approximately 360 Lahontan cutthroat trout.

Reasons for Decline and Threats to Survival: Major impacts to Lahontan cutthroat trout habitat and abundance include 1) reduction and alteration of stream discharge; 2) alteration of stream channels and morphology; 3) degradation of water quality ; 4) reduction of lake levels and concentrated chemical components in natural lakes; and 5) introduction of non-native fish species. There alterations are usually associated with agricultural use, livestock and feral horse grazing, mining, and urban development. Alteration and degradation of trout habitat have also resulted from logging, highway and road construction, dam building, and the discharge of effluent form wastewater treatment facilities. All these factors reduce the suitability of habitat for the trout (USDI-FWS 1995).

Little Kern Golden Trout (*Oncorhynchus aquabonita whitei*)

Species Description and Life History: The Little Kern golden trout was federally listed as threatened and critical habitat was designated concurrently on April 13, 1978 (43 **FR** 15427). Critical habitat was defined to include all streams and tributaries in the Little Kern River drainage above a barrier falls on the Little Kern River located one mile below the mouth of Trout Meadows Creek. The CDFG has prepared a management plan that has been accepted by the Service as the official recovery plan for Little Kern golden trout. The fishery objectives for conditions within the proposed project boundaries are restoration of pure strain Little Kern golden trout to its critical habitat, protection of critical habitat, and protection and/or restoration of the native Sacramento sucker (*Catostomus occidentalis*).

The Little Kern golden trout requires diverse habitat composed of pools for refugia, instream cover, shade from bankside vegetation to regulate temperature, and gravel substrates for spawning (USDA-FS 1993). Desired habitat includes deep, narrow channels within low gradient meadow environments. Low width to depth ratios and a large percentage of undercut banks are considered indicators of desirable meadow habitat conditions. Desirable habitat outside meadows contains good cover from cobble and boulders (USDA-FS 1993). Little Kern golden trout reach sexual maturity at three years, although some younger fish do exhibit courtship behavior (Smith 1977). Spawning occurs during the spring. Males establish spawning sites on the downstream edge of pools over gravel substrates. Spawning occurs at a water depth of 5 to 15 cm (Smith 1977).

Foraging Ecology: Little Kern golden trout forage on a variety of invertebrates, eating whatever is most abundant in the water column. Diet includes larval and adult insects and planktonic crustaceans (Moyle 1976a).

Historic and Current Distribution: The historical distribution of Little Kern golden trout was restricted to the Little Kern River drainage down to a barrier falls that isolated Little Kern golden trout from Kern River rainbow trout in the Kern River. Approximately 40 of the estimated 100 miles of suitable trout habitat in the Little Kern River drainage are thought to have supported Little Kern golden trout prior to human influence (USDA-FS 1993). Early activities of settlers in the area included transplanting Little Kern golden trout into many nearby waters (Schreck 1969). After human influence, nearly 90 miles of streams and several lakes contained Little Kern golden trout (USDA-FS 1993). Between 1900 and 1950, rainbow trout and brook trout were also transplanted into the Little Kern River watershed. The Little Kern golden trout does not compete well with other species and also hybridizes with rainbow trout. By 1970, only 10.2 miles of streams in the Little Kern River system contained pure Little Kern golden trout (USDA-FS 1993).

The CDFG has been involved in an intensive program to eradicate the non-native fish species within the Little Kern River system. Over the last 20 years, treatment with antimycin or rotenone (fish toxicants) have been used to treat many of the streams, lakes, and a portion of the Little Kern River. Populations of pure strain Little Kern golden trout are now inhabiting many of the treated sections of streams and lakes. Treatments were completed in 1995, with delisting of the species the future goal once studies determine that the fish are pure and at adequate population levels according to the Revised Plan.

Reasons for Decline and Threats to Survival: Little Kern golden trout do not compete well with other species. Hybridization and interspecific competition result in reduced genetic purity and lower population numbers (USDA-FS 1993).

Lost River Sucker (*Deltistes luxatus*)

Species Description and Life History: The Lost River sucker was described by Cope (1879) from specimens he collected from Upper Klamath Lake. A complete discussion of the taxonomy of the species can be found in the Service's Lost River and Shortnose Sucker Recovery Plan (USDI-FWS 1993c). The Lost River sucker was federally listed as endangered species on July 18, 1988 (53 **FR** 27134). The Clear Lake watershed is considered Unit 1 of the proposed designation of six Critical Habitat Units (CHUs) for Lost River and shortnose suckers. Primary constituent elements include water of sufficient quantity and quality to provide conditions required for the particular life stage of the species; physical habitat inhabited or potentially habitable by shortnose suckers for use as refugia, spawning, nursery, feeding, or rearing areas, or as corridors between these areas; and food supply and a natural scheme of predation, parasitism, and competition in the biological environment.

Scoppettone (1988) found shortnose suckers up to 33 years of age in Copco Reservoir and Lost River suckers to 43 years of age in upper Klamath Lake. In the Clear Lake drainage, Scoppettone (1988) found shortnose suckers from one to 23 years of age, and Lost River suckers from one to 27 years old. Lost River suckers can achieve lengths approaching one meter. Sexual maturity is achieved in approximately nine years for Lost River suckers (Scoppettone, pers. comm., cited in USDI-FWS 1994c).

The role upstream populations of Lost River suckers play in the maintenance and viability of downstream populations is poorly understood at this time.

Foraging Ecology: The diet of Lost River suckers includes detritus, zooplankton, algae, and aquatic insects (Buettner and Scoppetone 1990).

Historic and Current Distribution:. The Lost River sucker (along with the shortnose sucker) is endemic to the upper Klamath Basin, Oregon and California, and were once quite abundant. Cope (1884) noted that Upper Klamath Lake sustained "a great population of fishes" and was "more prolific in animal life" than any body of water known to him at that time. Gilbert (1898) noted that the Lost River sucker was "the most important food-fish of the Klamath Lake region." At that time, spring sucker runs "in incredible numbers" (Gilbert 1898) were relied upon as a food source by the Klamath and Modoc Indians and were taken by local settlers for both human consumption and livestock feed (Cope 1879; Coots 1965; Howe 1968). Sucker runs were so numerous, that a cannery was established on the Lost River (Howe 1968) and several other commercial operations processed "enormous amounts" of suckers into oil, dried fish, and other products (Andreasen 1975).

The Lost River sucker was historically found in Upper Klamath Lake and its tributaries, including the Williamson, Sprague, and Wood rivers (Williams *et al.* 1985), Crooked, Seven Mile, Four Mile, Odessa, and Crystal creeks (Stine 1982). It was also found in the Lost River system, Tule Lake, Lower Klamath Lake, and Sheepy Lake (Moyle 1976a).

In a distributional survey of the Clear Lake watershed conducted in the summers of 1989 and 1990, Lost River suckers were collected in lower Willow Creek and Boles Creek upstream to Avanzino Reservoir (Buettner and Scoppettone 1991). Under higher flow conditions, such as the spring of 1993, the range probably extended upstream in all of the creeks in the Clear Lake watershed (M. Buettner, pers. comm., cited in USDI-FWS 1994c). Lost River and shortnose suckers have been captured in the Lost River below Clear Lake and were taken to Malone Reservoir in 1992 during Reclamation's salvage operation at Clear Lake. Buettner (pers. comm. 1995) believes it is unlikely that many suckers remain in Malone Reservoir. The reservoir is drained each fall to a small pool and most of the fish were likely washed down stream into the Lost River.

<u>Reasons for Decline and Threats to Survival</u>: The factors believed to be responsible for the decline of the Lost River suckers include the damming of rivers, dredging and draining of marshes, instream flow diversions, a shift toward hyper eutrophication in Upper Klamath Lake, and other traditional land use practices. A recent analysis of the population genetics of the shortnose and Lost River suckers (Moyle and Berg 1991) suggested that "if populations continue to decline, these species may cross below the minimum viable population threshold and be lost". Entire stocks may have already been lost [e.g., Harriman Springs (Andreasen 1975)].

Suckers appear to be strongly influenced by poor water quality induced by high water temperatures, nutrient enrichment, algal blooms and die-offs, low dissolved oxygen, high pH, and possibly high ammonia (Kann and Smith 1993; Perkins 1997). Higher recruitment success occurs during above-average water quality years; in contrast, large-scale fish kills of adult suckers in the Upper Klamath Lake and Williamson Rivers appear related to poor water quality (Perkins 1997). Although fish kills have occurred sporadically in the 1900s, they appear to have increased in size, duration, and areal extent in recent years and may be adversely affecting current recovery efforts (Perkins 1997). A 1996 August-September fish kill, consisting almost exclusively of the endangered suckers, had the documented deaths of more than 6049 individuals, with many thousands of additional fish estimated to have been killed (Perkins 1997). Another subsequent kill in the Lake in 1997 involved primarily tui chubs, but more than 1400 endangered suckers deaths were also documented (Mark Buettner, Reclamation, pers. comm.). Although the ultimate causes of these fish kills was identified as the bacterial infections of the skin and gills by *Flavobacterium columnare*, degenerative changes in the intestines, livers and kidneys of many of the fish were also observed in the 1996 fish. Lesions of the kidneys were indicative of toxic tubular necrosis, typically caused by heavy metals, pesticides, and other poisons (Foote 1996). Foote suggested that a likely source of toxins in the Upper Klamath Lake system was *Microcystis*, a cyanobacterium producing the toxin microcystin. This bacterium was in bloom during the 1996 fish kill and its toxin was detected in 3 of 9 dead suckers from the 1996 fish kill (Klamath Falls Fish and Wildlife Office, U.S. Fish and Wildlife Service, unpublished data).

In addition, to fish kills, suckers in the Klamath Basin suffer from abnormally high rates of parasitism and physical deformities (Biological Research Division, U.S. Geological Survey, unpublished) that may be related to water quality, nutritional deficiencies, or contaminant exposures. Fish in the Tule Lake area also suffer very high rates of parasitism and deformities (Littleton 1993), although sucker health has not specifically been documented. Overharvest and chemical contamination may have also contributed to the decline. Reduction and degradation of lake and stream habitats in the upper Klamath Basin is considered to be the most important factor in the decline of the endangered suckers (USDI-FWS 1993c). Very low numbers of benthic organisms in many locations and an overall reduction in numbers of aquatic reptiles in the habitat of the sucker may have been caused by pollution of organochlorine pesticides and other pollutants (USDI-FWS 1993c).

Modoc Sucker (*Catostomus microps*)

<u>Species Description and Life History</u>: The Modoc sucker is a dwarf catostomid. The species was federally listed as endangered, with critical habitat designated on June 11, 1985 (50 **FR** 42530). Critical habitat was described to include the following reaches: Johnson Creek from the confluence with Rush Creek upstream approximately four river miles including two tributaries in Higgins Flat and Rice Flat; Rush Creek from the gaging station on highway 299 upstream to the Upper Rush Creek campground; Turner Creek from its confluence with the Pit River upstream about 4.5 river miles; Washington Creek from its confluence with Turner Creek upstream approximately four river miles, including 1.5 miles of Coffee Mill Creek; and approximately 3.5 miles of Hulbert Creek from its confluence with Turner Creek, including 1.5 miles of Cedar Creek. The Modoc sucker also exists in Coffee Mill, Willow, Ash, and Rush creeks (Studinski 1993) for a total of 25 miles (Gina Sato, BLM, pers. comm. 1991). Previously, the California Department of Fish and Game had classified the Modoc sucker as “rare” in 1973 and “endangered” in 1980.

The Modoc sucker was first described in 1908 by C. Rutter from three paratypes collected from Rush Creek in 1898. Unlike many other native fish species, the Modoc sucker’s nomenclature has never been questioned. *Catostomus* refers to the inferior position of the mouth (Moyle 1976a), and *microps* means “small eye” (Mills 1980). The species can be distinguished from other catostomids by the number of dorsal rays (n = 10-12), the number of scales in the lateral line (n = 79-89), and their small body size (<160 mm) (Mills 1980).

Life history studies (Moyle and Marciochi 1975) indicate Modoc suckers are most successful in small, relatively undisturbed, pool-dominated streams where they are isolated from Sacramento sucker (*Catostomus occidentalis*), with which they can hybridize. Modoc sucker habitat is typified by extreme water flows (Studinski 1993). Flows are very high in winter and spring months, but by mid-summer, large reaches of habitat dry up. During these times, fish populations are confined to relatively small, permanent pools. Adults (>70 - 85 mm TL) prefer pools from one foot to over four feet deep during summer. Smaller fish have been observed in riffles and shallow pools in large schools (Studinski 1993). Moyle and Marciochi (1975) found that Modoc suckers were most abundant in areas with low flows, large shallow pools with muddy bottoms or gravel to cobble substrate, partial shade, and moderately clear water. Studinski (1993) found Modoc sucker in pools with maximum water temperature of less than 21°C with a daily temperature variation of less than 2°C. Little is known about Modoc sucker winter habitat requirements.

Moyle and Marciochi (1975) collected ripe males and females from mid-April to late May. They did not observe actual spawning behavior. Modoc suckers were observed spawning during a 1978 study. Boccone and Mills (1979) observed spawning occurring from mid-April through the first week of June. They reported that spawning behavior of Modoc sucker closely resembled that of the Tahoe sucker, a close relative. Spawning took place over coarse to fine gravel in the lower end of pools. Pools were located in meadow areas with abundant cover. Boccone and Mills (1979) also noted spawning coloration and tubercle development on mature male Modoc suckers, but they further noted that ripe females did not express these characteristics. Water temperature and photoperiod were thought to be factors controlling timing of spawning. Spawning was observed from midmorning to late afternoon with water temperature from 13.3°C to 16.1°C (Boccone and Mills 1979).

Foraging Ecology: The diet of the Modoc sucker consists mostly of detritus and algae, with insects and crustaceans making up 25% of the diet.

Historic and Current Distribution: The Modoc sucker is endemic to small streams tributary to the upper Pit River drainage in Modoc and Lassen counties, California. Its current range is restricted to the Turner and Ash Creek subsystems in Modoc County.

Past habitat and populations surveys gave different estimates to Modoc sucker population size. Moyle (1974) estimated the population of Modoc suckers to be less than 5,000 individuals, with an effective population of 200. Ford (1977) found 2,605 suckers, and estimated the effective population to be 104, based on length-frequency analyses. Mills (1980) estimated that only 1,300 genetically pure Modoc sucker remained. During recent habitat and population surveys for six of the nine known Modoc sucker streams, Scoppettone *et al.* (1994) estimated the population to be 3,000 suckers. Biologists on this research project did not differentiate between Modoc sucker and Sacramento sucker during their visual surveys.

Approximately 50 percent of Modoc sucker habitat lies on Modoc National Forest. Modoc sucker populations are generally considered to be stable to improving. Exclosures protect much of the species habitat. Most recovery actions, as outlined in the Modoc sucker recovery action plan (USDA-FS 1989) have been completed. During a recent drought, Modoc suckers were found in deep perennial pools.

Reasons for Decline and Threats to Survival: Main threats are habitat loss from overgrazing, siltation, channelization, and hybridization with a closely related *Catostomid*. Past and present grazing and channelization on both private and public lands have caused severe erosion and siltation, dramatically degrading the species' habitat. In some streams, erosional cutting of stream banks exposed as much as 10 vertical feet of earth. These habitat changes limited the distribution and abundance of the sucker to a point where, at the time the species was listed, only 1,300 genetically pure individuals were thought to remain (Mills 1980). Besides these changes in the habitat, the extreme erosion and channelization also removed natural barriers separating the Modoc sucker from the Sacramento sucker. Hybridization between these two species has occurred.

Mohave Tui Chub (*Gila bicolor mohavensis*)

Species Description and Life History: The Mohave tui chub was listed as endangered on October 13, 1970, without critical habitat (35 **FR** 16047). This account is based on Moyle 1976a and Moyle *et al.* 1989.

The Mohave tui chub, a member of the minnow family, can reach over 10 inches in length. The Mohave tui chub is the only fish native to the Mohave River basin in California. This species was thought to inhabit the deep pools and slough-like areas of the Mohave River. Mohave tui chubs are adapted to the Mohave River's alkaline, hard water. Mohave tui chubs have survived in habitats where dissolved oxygen was less than one microgram per liter; they also have some tolerance for high salinity and high water temperatures. Mohave tui chubs use aquatic vegetation to attach their eggs and for cover and thermal refuges.

Foraging Ecology: Mohave tui chubs are morphologically adapted for feeding on plankton. However, they readily consume food, such as bread and lunch meat, provided by visitors to their refugia.

Historic and Current Distribution: The Mohave tui chub is native to the Mohave River basin. Currently, the only known genetically pure Mohave tui chub populations are found in three artificial ponds, one natural spring, and a series of constructed drainage channels in San Bernardino County. The pond at the Desert Studies Center at Soda Dry Lake is maintained by groundwater pumping; MC Spring is a natural spring also located at the Desert Studies Center. The water supplying both of these habitats is likely from the underflow of the Mohave River.

The two ponds at Camp Cady receive water pumped from the underflow of the Mohave River. The remaining population at the Naval Air Weapons Station, China Lake, California resides in drainage channels which carry percolating water from a system of sewage ponds. The estimated population at China Lake is between 10,000 and 20,000 fish.

Reasons for Decline and Threats to Survival: The primary causes for the decline of the Mohave tui chub were the introduction of arroyo chubs and other exotic species into the Mohave River system and habitat alteration. The construction of headwater reservoirs altered natural flow regimes and provided favorable habitat for exotic species. Water diversions and pollution have decreased habitat suitability in other locations. Increases in permissible levels of environmental contaminants to the species' restricted habitat may have a deleterious effect on the species. The Mohave tui chub is native to the Mohave River basin, which has been identified as an impaired water body.

Owens Pupfish (*Cyprinodon radiosus*)

Species Description and Life History: The Owens pupfish was listed as endangered on March 11, 1967 (32 **FR** 4001). Population declines attributed to competition and predation by non-native species and habitat modification caused by water diversions from the Owens River and its tributaries were identified as the principal causes of the declines. The following information is summarized from the draft recovery plan for the wetland and aquatic species of the Owens Basin (USDI-FWS 1996a).

The Owens pupfish rarely exceeds 2.5 inches in length. Males can easily be distinguished from females by coloration; males are bright blue, particularly during the breeding season, while females are a dusky olive green.

Owens pupfish occupy habitat where water is relatively warm and food is plentiful. Spawning occurs over soft substrates. Eggs are laid singly and hatch in approximately 6 days when temperatures are from 24 to 27 degrees C. They reach maturity in three to four months and rarely live longer than one year.

Foraging Ecology: The Owens pupfish is an opportunistic omnivore. Their diet changes seasonally to include the most abundant organisms in their habitat. They forage in schools, mostly on insects such as chironomid larvae. They were probably the main predator on mosquito larvae when they were abundant (Moyle 1976a).

Historic and Current Distribution: Owens pupfish were reported as common in habitats throughout the Owens Valley in Inyo and Mono counties from Fish Slough, approximately 12 miles north of Bishop, south to Lone Pine. They were most abundant near the margins of marshes, from shallow sloughs bordering the Owens River, and from springs. They are currently known from four sites, all of which are managed to protect Owens pupfish from non-native fish: Warms Springs and the White Mountain Research Station in Inyo County, and BLM Spring and

Owens Valley Native Fish Sanctuary in Mono County. This species was thought to be extinct in 1942; all of the remaining fish have been propagated from a remnant population found in Fish Slough in 1964.

Reasons for Decline and Threats to Survival: The transfer of Owens River water to the Los Angeles Aqueduct and the subsequent loss of habitat almost caused the extinction of the Owens pupfish. Because all of the remaining Owens pupfish are descendants of one population, this species may lack the genetic variability found in other species of pupfish. This factor, along with the relatively brief life span, should be considered in any analysis of the effects of toxic substances on the Owens pupfish. The Owens River, the primary water course through the valley floor where this species occurs, has been declared an impaired water body.

Owens pupfish are extremely limited in distribution. The recovery plan for the Owen's pupfish determined that a population would be determined to be secure when 1) exotic species are controlled or eliminated, 2) emergent vegetation is controlled, and 3) sufficient water quality is guaranteed (USDI-FWS 1984a).

Owens Tui Chub (*Gila bicolor snyderi*)

Species Description and Life History: The Owens tui chub was listed as endangered on August 5, 1985 (50 **FR** 31592). The introduction of non-native fish that affect the Owens tui chub through competition, predation, and hybridization and diversion of water for agricultural and municipal use were the principal reasons for the listing. Critical habitat was designated for this species along eight miles of the Owens River in the Owens Gorge and at two springs at Hot Creek Fish Hatchery. Both of these locations are in Mono County. The following information is summarized from the draft recovery plan for the wetland and aquatic species of the Owens Basin (USDI-FWS 1996a).

The Owens tui chub may reach a length of 12 inches. Its dorsal coloration ranges from bronze to dusky green; its belly is silver or white. Reproductive information is not well-known for the Owens tui chub; however, information derived from other subspecies of tui chub may be applicable. They prefer pool habitats that provide adequate cover and dense aquatic vegetation. Spawning occurs over aquatic vegetation or gravel. Females can produce large numbers of eggs; an eleven-inch long female from Lake Tahoe contained 11,200 eggs. They reach sexual maturity in 2 years and may live more than 30 years.

Foraging Ecology: Owens tui chubs prey primarily on aquatic insects, although they also consume detritus and aquatic vegetation.

Historic and Current Distribution: Owens tui chubs were reported as common from Long Valley in Mono County south to Owens Lake in Inyo County. Although tui chubs remain common in this area, the only non-introgressed populations of the Owens tui chub occur in the headsprings at the Hot Creek Fish Hatchery, the Owens River downstream from Crowley Lake, ponds at Cabin

Bar Ranch in Olancha, and at Mule Spring near Big Pine in Inyo County.

Reasons for Decline and Threats to Survival: The Owens tui chub declined due to Owens River water diversions and introduction of predatory fishes. Hybridization with other tui chub also threatens the genetic purity of the Owens tui chub. The Owens River, the primary water course through the valley floor where this species occurs, has been declared an impaired water body. The Town of Mammoth Lakes deposits sewage effluent in a percolation pond several miles uphill from the headsprings; however, an influence of this water and a hydrologic connection between the pond and the head springs has not been demonstrated.

The draft recovery plan for the Owens tui chub identifies only one specific water quality issue in its discussions of the threats or recovery of this species. Whitmore Hot Springs currently discharges treated swimming pool water into an area identified in the draft recovery plan as a potential conservation area for the Owens tui chub. Chemicals used to treat the swimming pool could be harmful to Owens tui chubs. The draft recovery plan also calls for the maintenance of water quality in the other natural and artificial springs and ponds where the Owens tui chub currently occurs or could be re-introduced.

Paiute Cutthroat Trout (*Oncorhynchus clarki seleniris*)

Species Description and Life History: The Paiute cutthroat trout is an inland subspecies of cutthroat trout endemic to the Lahontan Basin of eastern California. The species was listed as endangered on October 13, 1970 (35 **FR** 16047) and subsequently reclassified as threatened on July 16, 1975 (40 **FR** 29863). The species is believed to have evolved from Lahontan cutthroat trout during the last 5,000 to 8,000 years (Behnke and Zarn 1976).

Paiute cutthroat trout are distinguished from other subspecies of cutthroat by the absence, or near absence, of body spots, the slender body form, relatively small scales, and vivid coloration (USDI-FWS 1985b). Paiute cutthroat trout life history and spawning requirements are similar to other stream-dwelling cutthroat trout. Paiute cutthroat trout reach sexual maturity at age two and peak spawning occurs in June and July (Wong 1975). To spawn successfully, they must have access to flowing waters with clean gravel substrates (USDI-FWS 1985b). Adults and juveniles favor pools, runs, and backwater pools where current velocities are quite low. Fry are most often found in backwaters and pools (USDA-FS 1994). Paiute cutthroat trout commonly select areas of low water velocities during spring, summer and fall. Their use of habitat in the winter is unknown.

Foraging Ecology: Paiute cutthroat trout are opportunistic, foraging on a variety of invertebrates that are abundant in the water column. Insects make up the bulk of their diet (Moyle 1976a).

Historic and Current Distribution: The Paiute cutthroat has a very limited historical range in the eastern Sierra Nevada river drainage of Silver King Creek, a tributary of the East Fork Carson River drainage. Within the Silver King Creek drainage, populations of Paiute cutthroat trout occur in Fly Valley, Fourmile Canyon, Coyote Valley, and Corral Valley Creeks. Transplanted populations occur in the Sierra and Inyo National Forests, in Stairway, Sharktooth, and Cottonwood Creeks. Populations thought to be introgressed occur at a few additional sites. All current populations are in relatively small tributary creeks that do not support large populations. However, these Paiute cutthroat trout populations appear to have normal age/class distributions (Russ Wickwire and Bill Somer pers comm).

Reasons for Decline and Threats to Survival: The principal threats to the species include habitat loss due to livestock grazing and recreational use, hybridization and competition with non-native trout, and over-exploitation by angling. A Recovery Plan for the species was prepared in 1985. Critical habitat has not been designated. Recovery Plan goals include establishing pure populations and secure habitat for Paiute cutthroat trout in Silver King Creek above Llewellyn Falls, in Cottonwood Creek, and in Stairway Creek.

Razorback Sucker (*Xyrauchen texanus*)

Species Description and Life History: The razorback sucker was first proposed for listing under the ESA on April 24, 1978, as a threatened species (56 **FR** 54967). The proposed rule was withdrawn on May 27, 1980, due to changes to the listing process included in the 1978 amendments to the ESA. In March, 1989, the Service was petitioned by a consortium of environmental groups to list the razorback sucker as an endangered species. The Service made a positive finding on the petition in June, 1989, that was published in the Federal Register on August 15, 1989. The proposed rule to list the species as endangered was published on May 22, 1990, and the final rule was published on October 23, 1991. Critical habitat was designated in 1994. Critical habitat for the razorback sucker includes the Colorado, Gila, Salt, and Verde Rivers in the Lower Basin, including the 100-year floodplain of the Colorado River from Parker Dam to Imperial Dam.

The razorback sucker is the only representative of the genus *Xyrauchen*. This native sucker is distinguished from all others by the sharp edged, bony keel that rises abruptly behind the head. The body is robust with a short and deep caudal peduncle (Bestgen 1990). The razorback sucker may reach lengths of one meter and weigh five to six kg (Minckley 1973). Adult fish in Lake Mohave reached about half this maximum size and weight (Minckley 1983). Razorback suckers are long-lived, reaching the age of at least 40 years (McCarthy and Minckley 1987).

Adult razorback suckers utilize most of the available riverine habitats, although there may be an avoidance of whitewater type habitats. Main channel habitats used tend to be low velocity ones such as pools, eddies, nearshore runs, and channels associated with sand or gravel bars (summarized in Bestgen 1990). Backwaters, oxbows, and sloughs adjacent to the main channel are well-used habitat areas ; flooded bottom lands are important in the spring and early summer (summarized in Bestgen 1990). Razorback suckers may be somewhat sedentary, however considerable movement over a year has been noted in several studies (USDI-FWS 1993a). Spawning migrations have been observed or inferred in several locales (Jordan 1891; Minckley

1973; Osmundson and Kaeding 1989; Bestgen 1990; Tyus and Karp 1990).

Spawning takes place in the late winter to early summer depending upon local water temperatures. In general, temperatures between 10° to 20° C are appropriate (summarized in Bestgen 1990). Spawning areas include gravel bars or rocky runs in the main channel (Tyus and Karp 1990), and flooded bottom lands (Osmundson and Kaeding 1989).

Habitat needs of larval razorback suckers are not well known. Warm, shallow water appears to be important. Shallow shorelines, backwaters, inundated bottom lands and similar areas have been identified (Sigler and Miller 1963; Marsh and Minckley 1989; Tyus and Karp 1989, 1990; Minckley *et al.* 1991). For the first period of life, larval razorbacks are nocturnal and hide during the day. Young fish grow fairly quickly with growth slowing once adult size is reached (McCarthy and Minckley 1987). Little is known of juvenile habitat preferences.

The razorback sucker is adapted to the widely fluctuating physical environment of the historical Colorado River. Adults can live 45-50 years and, once reaching maturity between two and seven years of age (Minckley 1983), apparently produce viable gametes even when quite old. The ability of razorback suckers to spawn in a variety of habitats, flows and over a long season are also survival adaptations. Average fecundity recorded in studies ranged from 10,800 to 46,740 eggs per female (Bestgen 1990). With a varying age of maturity and the fecundity of the species, it would be possible to quickly repopulate after a catastrophic loss of adults.

Foraging Ecology: Young fish eat mostly plankton (Marsh and Langhorst 1988, Papoulias 1988). Adults are bottom dwellers, foraging on a variety of algae, detritus, and invertebrates.

Historic and Current Distribution: Occupied habitat as of 1993 is approximately 1,824 river miles, of which 336 miles are reintroduction habitats (52% of historic range). Populations are generally small and composed of aging individuals. Augmentation efforts along the Lower Colorado River propose to replace the aging populations in Lakes Havasu and Mohave and below Parker Dam with young fish from protected-rearing site programs. This may prevent the imminent extinction of the species in the wild, but appears less capable of ensuring long term survival or recovery. Overall, the status of the razorback sucker in the wild continues to decline.

Reasons for Decline and Threats to Survival: The razorback sucker was listed as an endangered species due to declining or extirpated populations throughout the range of the species. The causes of these declines are changes to biological and physical features of the habitat, largely through impounding of the lower Colorado River and introduction of non-native fish species. The effects of these changes have been most clearly noted by the almost complete lack of natural recruitment to any population in the historic range of the species.

Sacramento Splittail (*Pogonichthys macrolepidotus*)

Species Description and Life History: On January 6, 1994, a proposed rule to list the Sacramento splittail (*Pogonichthys macrolepidotus*) as a threatened species was published in 59 **FR** 862. The final rule listing the Sacramento splittail as a threatened species was published on February 8, 1999, and became effective March 10, 1999 (64 **FR** 5963).

The Sacramento splittail is a large cyprinid that can reach greater than 12 inches in length (Moyle 1976a). Adults are characterized by an elongated body, distinct nuchal hump, and a small blunt head with barbels usually present at the corners of the slightly subterminal mouth. This species can be distinguished from other minnows in the Central Valley of California by the enlarged dorsal lobe of the caudal fin. Sacramento splittail are a dull, silvery-gold on the sides and olive-grey dorsally. During the spawning season, the pectoral, pelvic and caudal fins are tinged with an orange-red color. Males develop small white nuptial tubercles on the head.

Feeding Ecology: Sacramento splittail are benthic foragers that feed on opossum shrimp, although detrital material makes up a large percentage of their stomach contents (Daniels and Moyle 1983). Earthworms, clams, insect larvae, and other invertebrates are also found in the diet. Predators include striped bass and other piscivores. Sacramento splittail are sometimes used as bait for striped bass.

Spawning behavior: Sacramento splittail are long-lived, frequently reaching five to seven years of age. Generally, females are highly fecund, producing more than 100,000 eggs each year (Daniels and Moyle 1983). Populations fluctuate annually depending on spawning success. Spawning success is highly correlated with freshwater outflow and the availability of shallow-water habitat with submersed, aquatic vegetation (Daniels and Moyle 1983). Sacramento splittail usually reach sexual maturity by the end of their second year at which time they have attained a body length of 180 to 200 mm. There is some variability in the reproductive period because older fish reproduce before younger individuals (Caywood 1974). The largest recorded individuals of the Sacramento splittail have measured between 380 and 400 mm (Caywood 1974; Daniels and Moyle 1983). Adults migrate into fresh water in late fall and early winter prior to spawning. The onset of spawning is associated with rising water temperature, lengthening photoperiod, seasonal runoff, and possibly endogenous factors from the months of March through May, although there are records of spawning from late January to early July (Wang 1986). Spawning occurs in water temperatures from 9° to 20° C over flooded vegetation in tidal freshwater and euryhaline habitats of estuarine marshes and sloughs, and slow-moving reaches of large rivers. The eggs are adhesive or become adhesive soon after contacting water (Caywood 1974; Bailey, UCD, pers. comm., 1994, as cited in DWR & USDI 1994). Larvae remain in shallow, weedy areas close to spawning sites and move into deeper water as they mature (Wang 1986).

Sacramento splittail can tolerate salinities as high as 10 to 18 ppt (Moyle 1976a; Moyle and Yoshiyama 1992). Sacramento splittail are found throughout the Delta (Turner 1966), Suisun Bay, and the Suisun and Napa marshes. They migrate upstream from brackish areas to spawn in freshwater. Because they require flooded vegetation for spawning and rearing, Sacramento splittail are frequently found in areas subject to flooding. Please refer to the Service (USDI-FWS

1994c, 1996c), and Department of Water Resources and United States Department of Interior - Bureau of Reclamation (DWR & USDI 1994) for additional information on the biology and ecology of the Sacramento splittail.

Historic and Current Distribution: Sacramento splittail are endemic to California's Central Valley where they were once widely distributed in lakes and rivers (Moyle 1976a). Historically, Sacramento splittail were found as far north as Redding on the Sacramento River and as far south as the site of Friant Dam on the San Joaquin River (Rutter 1908). Rutter (1908) also found Sacramento splittail as far upstream as the current Oroville Dam site on the Feather River and Folsom Dam site on the American River. Anglers in Sacramento reported catches of 50 or more Sacramento splittail per day prior to damming of these rivers (Caywood 1974). Sacramento splittail were common in San Pablo Bay and Carquinez Strait following high winter flows up until about 1985 (Messersmith 1966; Moyle 1976a; and Wang 1986 as cited in DWR & USDI 1994).

In recent times, dams and diversions have increasingly prevented upstream access to large rivers and the species is restricted to a small portion of its former range (Moyle and Yoshiyama 1989). Sacramento splittail enter the lower reaches of the Feather (Jones and Stokes 1993) and American rivers on occasion, but the species is now largely confined to the Delta, Suisun Bay, and Suisun Marsh (USDI-FWS 1994c). Stream surveys in the San Joaquin Valley reported observations of Sacramento splittail in the San Joaquin River below the mouth of the Merced River and upstream of the confluence of the Tuolumne River (Saiki 1984 as cited in DWR & USDI 1994).

Reasons for Decline and Threats to Survival: The decline of the Sacramento splittail has been documented over the past 10 years using fall midwater trawl data. This decline is due to hydrologic changes in the Estuary and loss of shallow water habitat due to dredging and filling (Monroe and Kelly, 1992). These changes include increases in water diversions during the spawning period of January through July. Most of the factors that caused delta smelt to decline have also caused the decline of this species. Diversions, dams and reduced outflow, coupled with severe drought years, introduced aquatic species such as the Asiatic clam (Nichols *et al.* 1986), and loss of wetlands and shallow-water habitat apparently have perpetuated the species' decline.

Sources of selenium contamination into the habitat of Sacramento splittail include: subsurface agricultural drainwater from westside San Joaquin Valley agricultural lands, non-point source runoff from Coast Range ephemeral streams flowing into the westside San Joaquin Valley (exacerbated by overgrazing of livestock), oil refinery wastewater disposal in San Francisco Bay and west Delta, and concentrated animal feeding operations (where feedlots supplement animal food with selenium) upstream of the Delta.

Santa Ana Sucker (*Catostomus santaanae*)

Species Description and Life History: The Santa Ana sucker was originally described by Snyder

(1908) from specimens collected in the Santa Ana River, hence its name. The Santa Ana sucker, a small, short-lived sucker, was proposed for threatened status by the Service on January 26, 1999 (64 FR 3915). Moyle (1976) described the Santa Ana sucker as less than 16 centimeters (cm) (6.3 inches (in)) in length. The Santa Ana sucker is silvery below, darker along the back with irregular blotches, and the membranes connecting the rays of the tail are pigmented (Moyle 1976).

The Santa Ana sucker inhabits streams that are generally small and shallow, with currents ranging from swift (in canyons) to sluggish (in the bottomlands). All the streams are subject to periodic severe flooding (Moyle 1976). Santa Ana suckers appear to be most abundant where the water is cool (less than 22° Celsius) (72° Fahrenheit), unpolluted and clear, although they can tolerate and survive in seasonally turbid water. Santa Ana suckers feed mostly on detritus, algae, and diatoms which they scrape off of rocks and other hard substrates, with aquatic insects making up a very small component of their diet. Larger fish generally feed more on insects than do smaller fish (Greenfield et al. 1970).

Santa Ana suckers usually live no more than 3 years (Greenfield et al. 1970). Spawning generally occurs from early April to early July, with a peak in late May and June (Greenfield et al. 1970, Moyle 1976). Spawning period may be variable and protracted, however. Recent field surveys on the East Fork of the San Gabriel River, found evidence of an extended spawning period. These surveys found small juveniles (<30 mm standard length (1.2 in)) in December 1998, and March of 1999 (U. S. Geological Survey (USGS) data in litt. 1999). This data indicates that spawning may be very protracted in this stream, and begin as early as November. Fecundity appears to be exceptionally high for a small sucker species (Moyle 1976). The combination of early sexual maturity, protracted spawning period, and high fecundity should allow the Santa Ana sucker to quickly repopulate streams following periodic flood events that can decimate populations (Moyle 1976).

Historic and Current Distribution: The Santa Ana sucker is one of seven native freshwater fishes that occurred historically in the Los Angeles Basin of California. Of these seven species, the Santa Ana sucker is the most common in the basin today. Four of the native Los Angeles Basin fishes are extinct within the basin, and two are very rare. Historically, the Santa Ana sucker occurred form near the Pacific Ocean to the headwaters of Los Angeles Basin streams. Urbanization and the associated anthropogenic impacts to habitats in the Los Angeles megalopolis have reduced the Santa Ana sucker's range to small reaches of Big Tujunga Creek (a tributary of the Los Angeles River), the headwaters of the San Gabriel River, and a lowland reach of the Santa Ana River, in Los Angeles, San Bernardino, Riverside and Orange counties (Swift et al. 1993).

A population also occurs throughout portions of the Santa Clara River drainage system, in Ventura and Los Angeles counties. The Santa Clara population is presumed to be an introduced population, although this presumption is based entirely on negative data (its absence from early collections), and not on a documented record of introduction (Bell 1978, Hubbs et al. 1943, Miller 1968, Moyle 1976). The Santa Clara River population was not included in the proposal to list the Santa Ana sucker as threatened because of its presumed introduced status (64 FR 3915).

Reasons for Decline and Threats to Survival: Moyle and Yoshiyama (1992) concluded that the native range of the Santa Ana sucker is largely coincident with the Los Angeles metropolitan area. Intensive urban development of the area has resulted in water diversions, extreme alteration of stream channels, changes in the watershed that result in erosion and debris torrents, pollution, and the establishment of introduced non-native fishes. Moyle and Yoshiyama (1992) stated, "[e]ven though Santa Ana suckers seem to be quite generalized in their habitat requirements, they are intolerant of polluted or highly modified streams." The impacts associated with urbanization are likely the primary cause of the extirpation of this species from lowland reaches of the Los Angeles, San Gabriel, and Santa Ana rivers.

As the Los Angeles urban area expanded, the rivers of the Los Angeles Basin, the Los Angeles, Santa Ana, and San Gabriel rivers, were highly modified, channelized, or moved in an effort to either capture water runoff or protect property. As Moyle (1976) stated, "[t]he lower Los Angeles River is now little more than a concrete storm drain." The same is true for the Santa Ana and San Gabriel rivers. These channelized rivers and canals with uniform and altered substrates are not suitable for sustaining Santa Ana sucker populations (Chadwick and Associates 1996). Past and continuing projects have resulted (or will result) in channelization and concrete lining of the Santa Ana River channel throughout most of the range of the Santa Ana sucker in Orange County. Urban development threatens the Santa Ana sucker in the Los Angeles and Santa Ana river basins. This urban development has resulted in changes in water quality and quantity, and the hydrologic regime of these rivers. The Santa Ana sucker is one of seven native freshwater fish species of the Los Angeles Basin. Four of these species, the steelhead (Oncorhynchus mykiss), Pacific lamprey (Lampetra tridentata), Pacific brook lamprey (Lampetra cf. pacifica), and the unarmored threespine stickleback (Gasterosteus aculeatus williamsoni) have been extinct within the Los Angeles Basin since the 1950's, and two others are very rare (Santa Ana speckled dace (Rhinichthys osculus ssp.) and arroyo chub (Gila orcutti)) presumably due to the same factors that have caused the decline of the Santa Ana sucker (Swift et al. 1993).

All three river systems within the historic range of the Santa Ana sucker have dams that isolate and fragment fish populations. Dams likely have resulted in some populations being excluded from suitable spawning and rearing tributaries. Reservoirs also provide areas where introduced predators and competitors can live and reproduce (Moyle and Light 1996). The newly completed Seven Oaks Dam, upstream from the present range of Santa Ana sucker in the Santa Ana River, will prevent future upstream movement of fish and further isolate the Santa Ana sucker populations from their native range in the headwaters of that system.

A recent study of environmental variables affecting Santa Ana sucker abundace found some evidence that deteriorating water quality (electrical conductivity and turbidity) negatively impacts Santa Ana suckers. Results from this study also indicated that the presence of non-native introduced fish species was more strongly correlated with the absence of Santa Ana suckers than any water quality variable. Strongly significant negative associations were found with common carp (Cyprinus carpio), largemouth bass (Micropterus salmoides), bluegill (Lepomis macrochirus), and fathead minnow (Pimephales promelas), indicating nonnative fishes may exclude Santa Ana suckers by competition, or eliminate via predation (Mike Saiki, U.S. Geological Survey, pers. com. 1999). Non-native introduced fishes have long been recognized as having far reaching negative impacts to native fishes in North America (Moyle et al. 1986). Accordingly, introduced predators and competitors likely threaten the continued existence of Santa Ana suckers throughout most of the range of the species.

Shortnose Sucker (*Chasmistes brevirostris*)

Species Description and Life History: The shortnose sucker was described by Cope (1879) from specimens he collected from Upper Klamath Lake. A complete discussion of the taxonomy of the species can be found in the Service's Lost River and Shortnose Sucker Recovery Plan (USDI-FWS 1993c). The shortnose sucker was federally listed as endangered species on July 18, 1988 (53 **FR** 27134). The Clear Lake watershed is considered Unit 1 of the proposed designation of six Critical Habitat Units (CHUs) for Lost River and shortnose suckers. Primary constituent elements include water of sufficient quantity and quality to provide conditions required for the particular life stage of the species; physical habitat inhabited or potentially habitable by shortnose suckers for use as refugia, spawning, nursery, feeding, or rearing areas, or as corridors between these areas; and food supply and a natural scheme of predation, parasitism, and competition in the biological environment.

Scoppettone (1988) found shortnose suckers up to 33 years of age in Copco Reservoir and Lost River suckers to 43 years of age in upper Klamath Lake. In the Clear Lake drainage, Scoppettone (1988) found shortnose suckers from one to 23 years old. Shortnose suckers are generally not larger than 50 centimeters (cm). Sexual maturity for shortnose suckers in Clear Lake appears to be five years (CDFG 1993). Buettner and Scoppettone (1990) found that most growth occurred in the first six to eight years of life for female shortnose suckers sampled from Upper Klamath Lake.

The majority of shortnose suckers spawning in the tributaries of Upper Klamath Lake have been observed in water depths ranging from 21 to 60 cm and in water velocities of 41 to 110 centimeters per second. Fecundity for shortnose suckers is reportedly between 18,000 to 46,000 eggs for suckers measuring about 360 millimeters (mm) to 445 mm in fork length (Buettner and Scoppettone 1990). Shortnose suckers have also been observed spawning in lacustrine habitats at Ouxy Springs and springs adjacent to Sucker Springs (L. Dunsmoor, pers. comm., cited in USDI-FWS 1994b), although little is known about the suitability of this habitat for incubation.

Foraging Ecology: The diet of shortnose suckers includes detritus, zooplankton, algae, and aquatic insects (Buettner and Scoppetone 1990).

Historic and Current Distribution: The shortnose sucker is endemic to the upper Klamath Basin, Oregon and California, and were once quite abundant. Cope (1884) noted that Upper Klamath Lake sustained "a great population of fishes" and was "more prolific in animal life" than any body of water known to him at that time.

The historical distribution of the shortnose sucker was Upper Klamath Lake and its tributaries (Miller and Smith 1981; Williams *et al.* 1985), Lake of the Woods (Moyle 1976a), and possibly the Lost River drainage. This species is now found throughout the Upper Klamath Basin, including the Lost River, Clear Lake Reservoir, Gerber Reservoir, and Tule Lake. Shortnose suckers have also been collected on the Upper Klamath River from Copco Reservoir to the Link River Dam. Those found in Gerber Reservoir and Clear Lake show some morphological differences from those in Upper Klamath Lake (Buettner and Scoppettone 1991). The taxonomic status of various shortnose sucker populations is yet to be resolved. Genetic evaluations are in progress by Dr. Don Buth at the University of California, Los Angeles (UCLA). Andreason (1975) included Clear Lake as the upstream limit of the sucker in the Lost River system.

The largest population of shortnose suckers occurs in Upper Klamath Lake and Clear Lake (Scoppettone, pers. comm., cited in USDI-FWS 1994b). Under higher flow conditions, such as the spring of 1993, the range probably extended upstream in all of the creeks in the Clear Lake watershed (M. Buettner, pers. comm., cited in USDI-FWS 1994b). Shortnose suckers have been captured in the Lost River below Clear Lake and were taken to Malone Reservoir in 1992 during Reclamation's salvage operation at Clear Lake. Buettner (pers. comm. 1995) believes it is unlikely that many suckers remain in Malone Reservoir. The reservoir is drained each fall to a small pool and most of the fish were likely washed down stream into the Lost River.

Reasons for Decline and Threats to Survival: The factors believed to be responsible for the decline of the shortnose sucker include the damming of rivers, dredging and draining of marshes, instream flow diversions, a shift toward hyper eutrophication in Upper Klamath Lake, and other traditional land use practices. A recent analysis of the population genetics of the shortnose and Lost River suckers (Moyle and Berg 1991) suggested that "if populations continue to decline, these species may cross below the minimum viable population threshold and be lost". Entire stocks may have already been lost [e.g., Harriman Springs (Andreasen 1975)].

Suckers appear to be strongly influenced by poor water quality induced by high water temperatures, nutrient enrichment, algal blooms and die-offs, low dissolved oxygen, high pH, and possibly high ammonia (Kann and Smith 1993; Perkins 1997). Higher recruitment success occurs during above-average water quality years; in contrast, large-scale fish kills of adult suckers in the Upper Klamath Lake and Williamson Rivers appear related to poor water quality (Perkins 1997). As indicated above, fish kills appear to have increased in size, duration, and areal extent in recent years and may be adversely affecting current recovery efforts (Perkins 1997).

In addition, to fish kills, suckers in the Klamath Basin suffer from abnormally high rates of parasitism and physical deformities (Biological Research Division, U.S. Geological Survey, unpublished) that may be related to water quality, nutritional deficiencies, or contaminant exposures. Fish in the Tule Lake area also suffer very high rates of parasitism and deformities (Littleton 1993), although sucker health has not specifically been documented. Overharvest and chemical contamination may have also contributed to the decline. Reduction and degradation of lake and stream habitats in the upper Klamath Basin is considered to be the most important factor in the decline of the endangered suckers (USDI-FWS 1993a). Very low numbers of benthic organisms in many locations and an overall reduction in numbers of aquatic reptiles in the habitat of the sucker may have been caused by pollution of organochlorine pesticides and other pollutants (USDI-FWS 1993a).

Steelhead Trout(Including all California ESUs) (*Oncorhynchus mykiss*)

Species Description and Life History: General life history information for steelhead is summarized below, followed by more detailed information on each steelhead ESU, including any unique life history traits as well as their population trends. Further detailed information on these steelhead ESUs is available in the NMFS Status Review of west coast steelhead from Washington, Idaho Oregon, and California (Busby *et al.* 1996); the NMFS proposed rule for listing steelhead (61 FR 41541); the NMFS Status Review for Klamath Mountains Province Steelhead (Busby *et al.* 1994), and the NMFS final rule listing the Southern California steelhead ESU as endangered and the South-Central California Coast and the Central California Coast steelhead ESUs as threatened (62 FR 43937). On March 19, 1998, the Central Valley ESU of steelhead was listed as threatened, and the Klamath Mountains Province and Northern California ESUs were deferred for listing (63 FR 13347). The listing decision for the Northern California steelhead ESU was revisited, and on February 11, 2000, this ESU was proposed for listing as threatened (65 FR 6960).

Critical Habitat: Critical habitat was designated on February 16, 2000 (65 FR 7764) for Central Valley, Central California Coast, South-Central California Coast, and Southern California steelhead ESUs. Critical habitat has not been proposed for the Northern California and Klamath Mountain Province steelhead ESUs. Critical habitat has been designated to include all river reaches accessible to listed steelhead within the range of the ESUs listed, except for reaches on Indian lands within Indian Reservations. Critical habitat consists of the water, substrate, and adjacent riparian zone of estuarine and riverine reaches for all of the steelhead ESUs. Accessible reaches are those within the historical range of the ESUs that can still be occupied by any life stage of steelhead. Inaccessible reaches are those above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years) and specific dams within the historical range of each ESU identified in Tables 16 through 19 of the final critical habitat designation.

1. Central California Coast steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Russian River to Aptos Creek, California (inclusive), and the drainages of San

Francisco and San Pablo Bays. Also included are all waters of San Pablo Bay westward of the Carquinez Bridge and all waters of San Francisco Bay from San Pablo Bay to the Golden Gate Bridge. Excluded is the Sacramento-San Joaquin River Basin of the California Central Valley as well as areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

2. South-Central California Coast steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Pajaro River (inclusive) to, but not including, the Santa Maria River, California. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

3. Southern California steelhead geographic boundaries. Critical habitat is designated to include all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Santa Maria River to Malibu Creek, California (inclusive). Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

4. Central Valley steelhead geographic boundaries. Critical habitat is designated to include all river reaches accessible to listed steelhead in the Sacramento and San Joaquin Rivers and their tributaries in California. Also included are river reaches and estuarine areas of the Sacramento-San Joaquin Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait, all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge. Excluded are areas of the San Joaquin River upstream of the Merced River confluence and areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years).

Proposed ESUs: The geographic boundaries of the Northern California ESU, proposed as threatened, include the coastal river basins from Redwood Creek, Humboldt County, to the Gualala River, in Mendocino County, California, inclusive.

Migration and Spawning: The most widespread run type of steelhead is the winter (ocean-maturing) steelhead, while summer (stream-maturing) steelhead (including spring and fall steelhead in southern Oregon and northern California) are less common. The stream-maturing type enters fresh water in a sexually immature condition and requires several months in freshwater to mature and spawn. The ocean-maturing type enters fresh water with well-developed gonads and spawns shortly thereafter (Barnhart 1986). There is a high degree of overlap in spawn timing between populations, regardless of run-type. California steelhead generally spawn earlier than steelhead in northern areas. Both summer and winter steelhead in California generally begin spawning in December, whereas most populations in Washington begin spawning in February or March. Among inland steelhead populations, Columbia River populations from tributaries upstream of the Yakima River spawn later than most downstream populations.

Steelhead spawn in cool, clear streams featuring suitable gravel size, water depth, and current velocity. The timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches, and associated lower water temperatures. Unusual stream temperatures during spawning migration periods can alter or delay migration timing, accelerate or retard mutations, and increase fish susceptibility to diseases. The minimum stream depth necessary for successful upstream migration is 18 cm (Thompson 1972). Reiser and Bjornn (1979) indicated that steelhead preferred a depth of 24 cm or more. The preferred water velocity for upstream migration is in the range of 40-90 cm/second, with a maximum velocity, beyond which upstream migration is not likely to occur, of 2.4 m/second (Thompson 1972, Smith 1973). Intermittent streams may be used for spawning (Barnhart 1986; Everest 1973). Steelhead may spawn more than once before dying, in contrast to other species of the *Oncorhynchus* genus. It is relatively uncommon for steelhead populations north of Oregon to have repeat spawning, and more than two spawning migrations is rare. In Oregon and California, the frequency of two spawning migrations is higher, but more than two is unusual. The number of days required for steelhead eggs to hatch varies from about 19 days at an average temperature of 60 degrees F to about 80 days at an average of 42 degrees F. Fry typically emerge from the gravel two to three weeks after hatching (Barnhart 1986).

After emergence, steelhead fry usually inhabit shallow water along perennial stream banks. Older fry establish territories which they defend. Stream side vegetation and cover are essential. Steelhead juveniles are usually associated with the bottom of the stream. In winter, they become inactive and hide in any available cover, including gravel or woody debris. Juvenile steelhead live in freshwater between one and four years and then become smolts and migrate to the sea from November through May with peaks in March, April, and May. The smolts can range from 14 to 21 cm in length. Steelhead spend between one and four years in the ocean (usually two years in the Pacific Southwest) (Barnhart 1986). Water temperatures influence the growth rate, population density, swimming ability, ability to capture and metabolize food, and ability to withstand disease of these rearing juveniles.

Reiser and Bjornn (1979) recommended that dissolved oxygen concentrations remain at or near saturation levels with temporary reductions to not less than 5.0 mg/L for successful rearing of juvenile steelhead. Low dissolved oxygen levels decrease the rate of metabolism, swimming speed, growth rate, food consumption rate, efficiency of food utilization, behavior, and ultimately the survival of the juveniles.

North American steelhead typically spend two years in the ocean before entering freshwater to spawn. The distribution of steelhead in the ocean is not well known. Coded wire tag recoveries indicate that most steelhead tend to migrate north and south along the Continental Shelf (Barnhart 1986). Steelhead stocks from the Klamath and Rogue rivers probably mix together in a nearshore ocean staging area along the northern California before they migrate upriver (Everest

1973).

All Central Valley steelhead are currently considered winter steelhead, although three distinct runs, including summer steelhead, may have occurred as recently as 1947 (CDFG 1995; McEwan and Jackson 1996). Steelhead within this ESU have the longest freshwater migration of any population of winter steelhead. There is essentially a single continuous run of steelhead in the upper Sacramento river. River entry ranges from July through May, with peaks in September and February; spawning begins in late December and can extend into April (McEwan and Jackson 1996).

There are two recognized forms of native *O. mykiss* within the Sacramento River Basin: coastal steelhead/rainbow trout (*O. m. irideus*, Behnke 1992) and Sacramento redband trout (*O. m. stonei*, Behnke 1992). It is not clear how the coastal and Sacramento forms of *O. mykiss* interacted in the Sacramento River prior to construction of Shasta Dam in the 1940s which blocked anadromous fish passage. Behnke (1992) reported that coastal and resident redband trout were spawned together at the McCloud River egg-taking station (1879-1888). Therefore, it appears the two forms co-occurred historically at spawning time, but may have maintained reproductive isolation. In addition, the relationship between anadromous and non-anadromous forms of coastal *O. mykiss,* including possible residualized fish upstream from dams, is unclear.

Migration and life history patterns of southern California steelhead depend more strongly on rainfall and streamflow than is the case for steelhead populations farther north (Moore 1980; Titus *et al.* in press). Average rainfall is substantially lower and more variable in southern California than in regions to the north, resulting in increased duration of sand berms across the mouths of streams and rivers and, in some cases, complete dewatering of the lower reaches of these streams from late spring through fall. Environmental conditions in marginal habitats may be extreme (e.g., elevated water temperatures, droughts, floods, and fires) and presumably impose selective pressures on steelhead populations. Their utilization of southern California streams and rivers with elevated temperatures (in some cases much higher than the preferred range for steelhead) suggests that steelhead within this ESU are able to withstand higher temperatures than populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead than occurs in more northerly populations (Moore 1980; Titus *et al.* in press; McEwan and Jackson 1996). However, we have relatively little life history information for steelhead from this ESU.

Large rivers, such as the Klamath and Rogue rivers, may have adult steelhead migrating throughout the year (Shapovalov and Taft 1954; Rivers 1957; Barnhart 1986). For example, summer steelhead in the Rogue River were historically divided into spring and fall steelhead (Rivers 1963). More recently, some researchers contend spring and fall steelhead of the Rogue, Klamath, Mad and Eel rivers are summer steelhead (Everest 1973; Roelofs 1983), while others classify fall steelhead separately (Heubach 1992) or as winter steelhead.

Foraging Ecology: Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects, and emerging fry are sometimes preyed upon by older juveniles.

<u>Historic and Current Distribution</u>: *Central Valley ESU (Threatened)* (63 FR 13347): Historical abundance estimates are available for some stocks within this ESU, but no overall estimates are available prior to 1961. In the Sacramento River including San Francisco Bay, the total run-size of steelhead was estimated at 40,000 in 1961 (Hallock *et al.* 1961). In the mid-1960s, steelhead spawning populations in this ESU were estimated at 27,000 fish (CDFG 1965). The present total run size for this ESU is probably less than 10,000 fish based on dam counts, hatchery returns and past spawning surveys.

At the Red Bluff Diversion Dam, counts have averaged 1,400 fish over the last 5 years, compared with runs in excess of 10,000 in the late 1960s. In the American River, estimates of hatchery produced fish average less than 1,000 fish, compared to 12,000 to 19,000 in the early 1970s (McEwan and Jackson 1996). Data to estimate population trends at the Red Bluff Diversion Dam show a significant decline of 9 percent per year from 1966 to 1992.

The majority of native, natural steelhead production in this ESU occurs in the upper Sacramento tributaries (Antelope, Deer, Mill, and other creeks), but these populations are nearly extirpated. The American, Feather, and Yuba rivers (and possibly the upper Sacramento and Mokelumne rivers) also have naturally-spawning populations (CDFG 1995). However, these rivers have also had substantial hatchery influence, and their ancestry is unknown. In the San Joaquin River Basin, there are reports of: (1) a small remnant steelhead run in the Stanislaus River (McEwan and Jackson 1996); (2) observations of steelhead in the Tuolumne River; and (3) large rainbow trout (possibly steelhead) at the Merced River hatchery.

Southern California ESU (Endangered) (62 FR 43937): The Southern California ESU of steelhead trout occupies rivers from the Santa Maria River to the southern extent of the species range. Historically, *O. mykiss* occurred at least as far south as Rio del Presidio in Mexico (Behnke 1992, Burgner *et al.* 1992). Spawning populations of steelhead did not occur that far south but may have extended to the Santo Domingo River in Mexico (Barnhart 1986); however, some reports state that steelhead may not have existed south of the U.S.-Mexico border (Behnke 1992; Burgner *et al.* 1992). The present southernmost stream used by steelhead for spawning is generally thought to be Malibu Creek, California (Behnke 1992; Burgner *et al.* 1992); however, in years of substantial rainfall, spawning steelhead can be found as far south as the Santa Margarita River, San Diego County (Barnhart 1986; Higgins 1991).

Previous assessments within this ESU have identified several stocks as being at risk or of special concern. Nehlsen *et al.* (1991) identified 11 stocks as extinct and 4 as at high risk. Titus *et al.* (in press) provided a more detailed analysis of these stocks and identified stocks within 14 drainages in this ESU as extinct, at risk, or of concern. They identified only two stocks, those in Arroyo Sequit and Topanga Creek, as showing no significant change in production from historical levels.

Historically, steelhead may have occurred naturally as far south as Baja California. Estimates of historical (pre-1960s) abundance are available for several rivers in this ESU: Santa Ynez River, before 1950, 20,000-30,000; Ventura River, pre-1960, 4,000-6,000; Santa Clara River, pre-1960, 7,000-9,000; Malibu Creek, pre-1960, 1,000. In the mid-1960s, CDFG (1965) estimated steelhead spawning populations for smaller tributaries in San Luis Obispo County as 20,000, but they provided no estimates for streams farther south.

The present total run sizes for 6 streams in this ESU were summarized by Titus *et al.* (in press); all were less than 200 adults. Titus *et al.* (in press) concluded that populations have been extirpated from all streams south of Ventura County, with the exception of Malibu Creek in Los Angeles County. However, steelhead are still occasionally reported in streams where stocks were identified by these authors as extirpated.

Of the populations south of San Francisco Bay (including part of the Central California Coast ESU) for which past and recent information was available, they concluded that 20% had no discernible change, 45% had declined, and 35% were extinct.

Central California Coast ESU (Threatened) (62 FR 43937): Only two estimates of historical (pre-1960s) abundance specific to this ESU are available: an average of about 500 adults in Waddell Creek in the 1930s and early 1940s (Shapovalov and Taft 1954), and 20,000 steelhead in the San Lorenzo River before 1965 (Johnson 1964). In the mid-1960s, 94,000 steelhead adults were estimated to spawn in the rivers of this ESU, including 50,000 and 19,000 fish in the Russian and San Lorenzo rivers, respectively (CDFG 1965). Recent estimates indicate an abundance of about 7,000 fish in the Russian River (including hatchery steelhead) and about 500 fish in the San Lorenzo River. These estimates suggest that recent total abundance of steelhead in these two rivers is less than 15 percent of their abundance 30 years ago. Recent estimates for several other streams (Lagunitas Creek, Waddell Creek, Scott Creek, San Vincente Creek, Soquel Creek, and Aptos Creek) indicate individual run sizes of 500 fish or less. Steelhead in most tributaries to San Francisco and San Pablo bays have been extirpated (McEwan and Jackson 1996). Fair to good runs of steelhead still apparently occur in coastal Marin County tributaries.

Little information is available regarding the contribution of hatchery fish to natural spawning, and little information on present run sizes or trends for this ESU exists. However, given the substantial rates of declines for stocks where data do exist, the majority of natural production in this ESU is likely not self-sustaining.

South-Central California Coast ESU (Threatened) (62 FR 43937): In the mid-1960s, total spawning populations of steelhead in the rivers in this ESU were estimated as 27,750 (CDFG 1965). Recent estimates for those rivers show a substantial decline during the past 30 years. Other estimates of steelhead include 1,000 to 2,000 in the Pajaro River in the early 1960s (McEwan and Jackson 1996), and about 3,200 steelhead for the Carmel River for the 1964-1975 period (Snider 1983). No recent estimates for total run size exist for this ESU. However, recent run-size estimates are available for five streams (Pajaro River, Salinas River, Carmel River, Little Sur River, and Big Sur River). The total of these estimates is less than 500 fish, compared with a total of 4,750 fish for the same streams in 1965.

Adequate adult escapement information was available to compute a trend for only one stock within this ESU (Carmel River above San Clemente Dam). This data series shows a significant decline of 22 percent per year from 1963 to 1993, with a recent 5-year average count of only 16 adult steelhead at the dam. In 1996, however, 700 adults were reported to have passed the ladder at San Clemente Dam.

Little information exists regarding the actual contribution of hatchery fish to natural spawning, and little information on present total run sizes or trends are available for this ESU. However, given the substantial reductions from historical abundance or recent negative trends in the stocks for which data exist, it is likely that the majority of natural production in this ESU is not self-sustaining.

Northern California ESU (Proposed Threatened) (65 FR7764): Population abundance has been determined to be very low relative to historical estimates (1930's dam counts), and recent trends are downward in stocks for which data were available, with the exception of two summer steelhead stocks. Summer steelhead abundance in particular is very low in this ESU. The most complete data set available in this ESU is a time series of winter steelhead counts on the Eel River at Cape Horn Dam. The updated abundance data (through 1997) showed moderately declining long-term and short-term trends in abundance, and the vast majority of these fish were believed to be of hatchery origin. These data show a strong decline in abundance prior to 1970, but no significant trend thereafter. Additional winter steelhead data are available for Sweasy Dam on the Mad River which show a significant decline, but that data set ends in 1963. For the seven populations where recent trend data were available, the only runs showing recent increases in abundance in the ESU were the relatively small populations of summer steelhead in the Mad River, which has had high hatchery production, and winter steelhead in Prairie Creek where the increase may be due to increased monitoring or mitigation efforts.

Reasons for Decline and Threats to Survival: (*All ESUs*) Steelhead on the West Coast have experienced declines in abundance in the past several decades as a result of natural and human factors. Forestry, agriculture, mining, and urbanization have degraded, simplified, and fragmented habitat. Water diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat. Among other factors, NMFS specifically identified timber harvest, agriculture, mining, habitat blockages, and water diversions as important factors for the decline of steelhead.

The status reviews and listing notices have cited extensive loss of steelhead habitat due to water development, including impassable dams and dewatering of portions of rivers, as principal threats to the steelhead. They also reported that of 32 tributaries for the southern California ESU, 21 have blockages due to dams, and 29 have impaired mainstem passage. Habitat problems in these ESUs relate primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions (McEwan and Jackson 1996, Titus *et al.* in press).

Other problems related to land use practices and urbanization also certainly contribute to depressed stock conditions. Habitat fragmentation and population declines have also resulted in small, isolated populations that may face genetic risk from inbreeding, loss of rare alleles, and genetic drift.

During rearing, suspended and deposited fine sediments can directly affect salmonids by abrading and clogging gills, and indirectly cause reduced feeding, avoidance reactions, destruction of food supplies, reduced egg and alevin survival, and changed rearing habitat (Reiser and Bjornn 1979). See also Reasons for Decline and Threats to Survival for chinook and coho salmon sections of this biological opinion for further information on factors affecting steelhead trout.

Tidewater Goby (*Eucyclogobius newberryi*)

Species Description and Life History: The tidewater goby was listed by the Service as endangered on March 7, 1994 (59 **FR** 10584). A recovery plan has not been published, and critical habitat has not been proposed. On June 24, 1999, the Service published a proposed rule to remove northern populations of the tidewater goby from the federal list of threatened and endangered species (64 **FR** 33816). This proposed rule identifies a distinct population segment (DPS) of tidewater goby known from six locations in Orange and San Diego counties, and would remove protection for all populations of tidewater goby north of these locations. On August 3, 1999, the Service published a proposed rule to designate critical habitat for this DPS (64 **FR** 42250). Detailed information regarding the biology of the tidewater goby can be found in Wang (1982), Irwin and Soltz (1984), Swift *et al.* (1989), Worcester (1992), and Swenson (1995).

The tidewater goby rarely exceeds 50 millimeters standard length. The species, which is endemic to California, is found primarily in waters of coastal lagoons, estuaries, and marshes. Its habitat is characterized by brackish shallow lagoons and lower stream reaches where the water is fairly still but not stagnant (Miller and Lea 1972; Moyle 1976a; Swift 1980; Wang 1982; Irwin and Soltz 1984). Tidewater gobies have been documented in waters with salinity levels from 0 to 42 parts per thousand, temperature levels from 8 to 25° Celsius, and water depths from 25 to 200 centimeters (Irwin and Soltz 1984; Swift *et al.* 1989; Worcester 1992; Swenson 1994; Lafferty 1997; Smith 1998). The species can withstand very low dissolved oxygen levels, and is regularly collected in waters with levels below 1 mg/l (Worcester 1992; Swift *et al.* 1997).

The tidewater goby appears to spend all life stages in lagoons. It may enter the marine environment only when flushed out of the lagoon by normal breaching of the sandbars following storm events. These events are important in the normal metapopulation dynamics and distribution of the species (Swift *et al.* 1989; Lafferty *et al.* 1997; Swift *et al.* 1997; Lafferty *et al.* in review). The tidewater goby seems to be an annual species although some variation has been observed (Swift 1980; Wang 1982; Irwin and Soltz 1984). Reproduction can occur year-round although distinct peaks in spawning, often in late spring and late summer or early fall, do occur. Both males and females can breed more than once in a season, with a lifetime reproductive potential of 3 - 12 spawning events. Females deposit an average of 400 eggs (range 100 - 1000) per spawning effort (Swenson 1995, in press). When breeding, males dig vertical burrows for females to deposit eggs. Within nine to ten days larvae emerge and are approximately five to seven mm in length. The larvae live in vegetated areas within the lagoon until they are 15 to 18 mm long (Wang 1982; Swift *et al.* 1989; Swenson 1994).

<u>Historic and Current Distribution</u>: The tidewater goby historically occurred in at least 110 California coastal lagoons (USDI-FWS in prep.) from the Smith River, Del Norte County, to Agua Hedionda Lagoon in San Diego County. The southern extent of its distribution has been reduced by approximately 13 kilometers (8 miles), and the species is currently known to occur in about 85 locations. Exact numbers of sites fluctuate with normal climatic conditions.

<u>Reasons for Decline and Threats to Survival</u>: The decline of the tidewater goby can be attributed primarily to urban, agricultural and industrial development in and surrounding the coastal wetlands and alteration of habitats from seasonally closed lagoons to tidal bays and harbors. The extent and magnitude of these threats has diminished since the promulgation of protective environmental legislation. Some extirpations are believed to be related to pollution, upstream water diversions, and the introduction of exotic fish species. These threats continue to affect remaining populations of tidewater gobies. Tidewater gobies have been extirpated from several impaired water bodies (e.g., Mugu Lagoon, Ventura County), but still occur in others (e.g., Santa Clara River, Ventura County). Lagoons where the goby resides receive municipal and industrial contaminated run-off from coastal streams. The short life-cycle of the species leaves it vulnerable to stochastic events. A single pulse of a contaminant may inhibit growth, survival, and reproduction of an entire cohort.

Unarmored Threespine Stickleback (*Gasterosteus aculeatus williamsoni*)

<u>Species Description and Life History</u>: The unarmored threespine stickleback was listed as endangered in 1970 (35 **FR** 16047). The following information is summarized from the recovery plan for the unarmored threespine stickleback (USDI-FWS 1985d). Two reaches of the Santa Clara River, and a single reach of both San Francisquito Creek and San Antonio Creeks were proposed as critical habitat in 1980 (45 **FR** 76012). However, critical habitat has not been designated.

Unarmored threespine sticklebacks are small fish (up to 6 centimeters) inhabiting slow moving reaches or quiet water microhabitats of streams and rivers. Favorable habitats usually are shaded by dense and abundant vegetation but in more open reaches algal mats or barriers may provide refuge for the species. Unarmored threespine sticklebacks reproduce throughout the year with a minimum of breeding activity occurring from October to January. Unarmored threespine sticklebacks are believed to live for only one year (USDI-FWS 1985d).

Foraging Ecology: Unarmored threespine sticklebacks feed on insects, small crustaceans, and snails, and to a lesser degree, on flat worms and nematodes.

Historic and Current Distribution: Unarmored threespine sticklebacks historically were distributed throughout southern California but are now restricted to the upper Santa Clara River and its tributaries in Los Angeles and Ventura counties, San Antonio and Canada Honda creeks on Vandenberg Air Force Base, Shay Creek in San Bernardino County, and San Felipe Creek in San Diego County. The population in Canada Honda Creek on Vandenberg Air Force Base is a transplanted population, as is the population that may persist in San Felipe Creek.

Reasons for Decline and Threats to Survival: Competition with non-native fish, introgression with other subspecies of sticklebacks, and loss of habitat to urbanization were contributing factors that led to the decline of the unarmored threespine stickleback. The greatest risk of continued urbanization of the Santa Clara River watershed is the degradation of water quality (USDI-FWS 1977). In the Santa Clara River, populations of unarmored threespine sticklebacks are affected by effluent from the Saugus and Valencia water reclamation plants, operated by the County Sanitation Districts of Los Angeles County. Pending modifications to the Valencia Water Reclamation Plant would improve the quality of effluent waters by removing ammonia. Effluent from this plant currently contains concentrations of ammonia that approach the toxic level for some aquatic species. Recovery plan objectives for this species include the regulation, maintenance, and restoration of water quality and quantity to ensure the survival and recovery of the species (USDI-FWS 1977).

Potential for Exposure and Adverse Effects: Contaminants associated with effluent discharges may have contributed to the decline of the unarmored threespine stickleback and may preclude recovery.

Arroyo Toad (*Bufo microscaphus californicus*)

Species Description and Life History: The arroyo toad was listed as endangered on December 16, 1994 (59 **FR** 64589). A draft recovery plan is in preparation, but has not yet been published. Critical habitat has not been proposed. Information regarding the biology of the arroyo toad can be found in Sweet (1992) and Campbell *et al.* (1996). The arroyo toad is a small (adults: snout-urostyle length (SUL) (2.2 to 2.9 inches), light-olive green or gray to tan, dark-spotted toad with a distinctive light-colored, V-shaped stripe across the head and the eyelids.

Arroyo toads are restricted to perennial and intermittent rivers and streams that have shallow, sandy to gravelly pools adjacent to sand or fine gravel terraces. Breeding occurs from March until mid-June (Sweet 1992). Eggs are deposited and larvae develop in shallow pools with minimal current, little or no emergent vegetation, and sand or pea gravel substrate. After metamorphosis from June to August, juveniles remain on the bordering gravel bars until the pool no longer persists (Sweet 1992). Juveniles spend more time exposed on these terraces during the daytime than do adults, and are thus vulnerable to diurnal predators. Adults excavate shallow burrows which are used for shelter during the day when the surface is damp or during longer intervals in the dry season (Sweet 1992). Sexual maturity is reached in one to two years, and toads may live for as few as five years (Sweet 1993). Little is known about movements or other behavior in the non-breeding season.

Foraging Ecology: Juveniles and adults forage for insects, especially ants and small beetles, on sandy stream terraces. Subadults and adults move into surrounding riparian and upland areas to forage.

Historic and Current Distribution: Arroyo toads historically were known to occur in coastal drainages in southern California from San Luis Obispo County to San Diego County and in Baja California, Mexico. In Orange and San Diego Counties, it occurred from the estuaries to the headwaters. The species also was reported from fewer than half a dozen desert slope drainages (USDI in preparation). In 1996, arroyo toads were discovered on Fort Hunter Liggett, Monterey County. This discovery constituted a northern range expansion for the species. Arroyo toads now survive primarily in the headwaters of coastal streams as small isolated populations (Sweet 1992), having been extirpated from much of their historic habitat.

Reasons for Decline and Threats to Survival: Urbanization, agriculture, dam construction, water manipulation, mining, livestock grazing and recreational activities in riparian areas have caused extensive habitat degradation leading to the decline and isolation of the remaining populations of arroyo toads. The introduction of bullfrogs and exotic fish may have severe impacts on toad populations due to predation. Exotic plant species degrade arroyo toad habitat, making it unsuitable, and may cause changes in the invertebrate fauna upon which the toad feeds. Changes in hydrologic regimes and loss of overwintering habitat as streamside areas are developed are probably the most important factors in the decline of arroyo toads.

California Red-Legged Frog (*Rana aurora draytonii*)

Species Description and Life History: The California red-legged frog was federally listed as threatened on May 23, 1996, (61 **FR** 25813). Critical habitat has not been proposed for the species. The Service is currently developing a recovery plan for the species. This species is the largest native frog in the western United States (Wright and Wright 1949), ranging from 4 to 13 centimeters (1.5 to 5.1 inches) in length (Stebbins 1985). The abdomen and hind legs of adults are largely red; the back is characterized by small black flecks and larger irregular dark blotches with indistinct outlines on a brown, gray, olive, or reddish background color. Dorsal spots usually have light centers (Stebbins 1985), and dorsolateral folds are prominent on the back. Larvae (*i.e.,* tadpoles) range from 14 to 80 millimeters (mm) (0.6 to 3.1 inches) in length, and the background color of the body is dark brown and yellow with darker spots (Storer 1925).

California red-legged frogs have paired vocal sacs and vocalize in air (Hayes and Krempels 1986). Female frogs deposit egg masses on emergent vegetation so that the egg mass floats on the surface of the water (Hayes and Miyamoto 1984). California red-legged frogs breed from November through March with earlier breeding records occurring in southern localities (Storer 1925). California red-legged frogs found in coastal drainages are active year-round (Jennings *et al.* 1992), whereas those found in interior sites may be more seasonally inactive.

California red-legged frogs spend most of their lives in and near sheltered backwaters of ponds, marshes, springs, streams, and reservoirs. The largest densities of California red-legged frogs currently are associated with deep pools with dense stands of overhanging willows (*Salix spp.*) and an intermixed fringe of cattails (*Typha latifolia*) (Hayes and Jennings 1988, Jennings 1988). This is considered optimal habitat. California red-legged frog eggs, larvae, transformed juveniles, and adults also have been found in ephemeral creeks and drainages and in ponds that do not have riparian vegetation. Accessability to sheltering habitat is essential for the survival of California red-legged frogs within a watershed, and can be a factor limiting frog population numbers and survival. Sheltering habitat includes mammal burrows, damp leaf litter, downed wood and other cover objects, both natural and manmade, and dense shrubbery up to several hundred meters distant from aquatic sites. California red-legged frogs may shelter in such places for weeks at a time in the wet season. During winter rain events, juvenile and adult California red-legged frogs are known to wander perhaps up to 1-2 km from summer aquatic sites (Rathbun and Holland, unpublished data, cited in Rathbun *et al.* 1991).

Egg masses contain about 2,000 to 5,000 moderate-sized (2.0 to 2.8 mm [0.08 to 0.11 inches] in diameter), dark reddish brown eggs and are typically attached to vertical emergent vegetation, such as bulrushes (*Scirpus* spp.) or cattail (Jennings *et al.* 1992). California red-legged frogs are often prolific breeders, laying their eggs during or shortly after large rainfall events in late winter and early spring (Hayes and Miyamoto 1984). Eggs hatch in 6 to 14 days (Jennings 1988). In coastal lagoons, the most significant mortality factor in the pre-hatching stage is water salinity (Jennings *et al.* 1992). One hundred percent mortality occurs in eggs exposed to salinity levels greater than 4.5 parts per thousand (Jennings and Hayes 1990). Increased siltation that occurs during the breeding season can cause asphyxiation of eggs and small larvae. Larvae undergo metamorphosis 3.5 to 7 months after hatching (Storer 1925, Wright and Wright 1949, Jennings and Hayes 1990). Of the various life stages, larvae probably experience the highest mortality rates, with less than 1 percent of eggs laid reaching metamorphosis (Jennings *et al.* 1992). Sexual maturity normally is reached at 3 to 4 years of age (Storer 1925, Jennings and Hayes 1985). California red-legged frogs may live 8 to 10 years (Jennings *et al.* 1992).

Foraging Ecology: The diet of California red-legged frogs is highly variable. Hayes and Tennant (1985) found invertebrates to be the most common food items. Vertebrates, such as Pacific tree frogs (*Pseudacris* (= *Pseudacris (= Hyla) regilla*) and California mice (*Peromyscus californicus*), represented over half of the prey mass eaten by larger frogs (Hayes and Tennant 1985). Hayes and Tennant (1985) found juvenile frogs to be active diurnally and nocturnally, whereas adult frogs were largely nocturnal. Feeding activity probably occurs along the shoreline and on the surface of the water (Hayes and Tennant 1985). Larvae likely eat algae (Jennings *et al.* 1992).

<u>Historic and Current Distribution</u>: The California red-legged frog has been extirpated or nearly extirpated from 70 percent of its former range. Historically, this species was found throughout the Central Valley and Sierra Nevada foothills. At present, California red-legged frogs are known to occur in 243 streams or drainages from 22 counties, primarily in central coastal

California. The most secure aggregations of California red-legged frogs are found in aquatic sites that support substantial riparian and aquatic vegetation and lack non-native predators [*e.g.*, bullfrogs (*Rana catesbeiana*), bass (*Micropterus spp.*), and sunfish (*Lepomis spp.*)].

Reasons for Decline and Threats to Survival: Over-harvesting, habitat loss, non-native species introduction, and urban encroachment are the primary factors that have negatively affected the California red-legged frog throughout its range (Jennings and Hayes 1985, Hayes and Jennings 1988). Ongoing causes of decline include direct habitat loss due to stream alteration and disturbance to wetland areas, indirect effects of expanding urbanization, and competition or predation from non-native species.

Giant Garter Snake (*Thamnophis gigas*)

Species Description and Life History: The Service published a proposal to list the giant garter snake as an endangered species on December 27, 1991 (56 **FR** 67046). The Service reevaluated the status of the giant garter snake before adopting the final rule. The giant garter snake was listed as a threatened species October 20, 1993 (58 **FR** 54053).

The giant garter snake is one of the largest garter snakes, reaching a total length of at least 64 inches (160 centimeters). Females tend to be slightly longer and proportionately heavier than males. The weight of adult female giant garter snakes is typically 1.1-1.5 pounds (500-700 grams). Dorsal background coloration varies from brownish to olive with a checkered pattern of black spots, separated by a yellow dorsal stripe and two light colored lateral stripes. Background coloration and prominence of the black checkered pattern and the three yellow stripes are geographically and individually variable (Hansen 1980). The ventral surface is cream to olive or brown and sometimes infused with orange, especially in northern populations.

Endemic to wetlands in the Sacramento and San Joaquin valleys, the giant garter snake inhabits marshes, sloughs, ponds, small lakes, low gradient streams, and other waterways and agricultural wetlands, such as irrigation and drainage canals and rice fields, and the adjacent uplands. Giant garter snakes feed on small fishes, tadpoles, and frogs (Fitch 1941, Hansen 1980, Hansen 1988). Essential habitat components consist of: (1) adequate water during the snake's active season (early-spring through mid-fall) to provide food and cover; (2) emergent, herbaceous wetland vegetation, such as cattails and bulrushes, for escape cover and foraging habitat during the active season; (3) upland habitat with grassy banks and openings in waterside vegetation for basking; and (4) higher elevation uplands for cover and refuge from flood waters during the snake's dormant season in the winter (Hansen 1980). Giant garter snakes are typically absent from larger rivers and other water bodies that support introduced populations of large, predatory fish, and from wetlands with sand, gravel, or rock substrates (Hansen 1980, Rossman and Stewart 1987, Brode 1988, Hansen 1988). Riparian woodlands do not typically provide suitable habitat because of excessive shade, lack of basking sites, and absence of prey populations (Hansen 1980).

Foraging ecology - Giant garter snakes are extremely aquatic, are rarely found away from water, forage in the water for food, and will retreat to water to escape predators and disturbance. This species occupies a niche similar to some eastern water snakes (*Nerodia* spp.). Giant garter snakes are active foragers, feeding primarily on aquatic prey such as fish and amphibians. Historically, prey likely consisted of Sacramento blackfish (*Orthodon microlepidotus*), thick-tailed chub (*Gila crassicauda*), and red-legged frog (*Rana aurora*). Because these species are no longer available (the thick-tailed chub is extinct, the red-legged frog is extirpated from the Central Valley, and the blackfish is declining/in low numbers), the predominant food items are now introduced species such as carp (*Cyprinus carpio*), mosquito-fish (*Gambusia affinis*), bullfrogs (*Rana catesbiana*), and Pacific treefrogs (*Pseudacris regilla*) (Fitch 1941, Rossman et al, 1996).

The breeding season extends through March and April, and females give birth to live young from late July through early September (Hansen and Hansen 1990). Brood size is variable, ranging from 10 to 46 young, with a mean of 23 (Hansen and Hansen 1990). At birth young average about 20.6 cm snout-vent length and 3-5 g. Young immediately scatter into dense cover and absorb their yolk sacs, after which they begin feeding on their own. Although growth rates are variable, young typically more than double in size by one year of age (G. Hansen, pers. comm.). Sexual maturity averages three years in males and five years for females (G. Hansen, pers. comm.).

The giant garter snake inhabits small mammal burrows and other soil crevices above prevailing flood elevations throughout its winter dormancy period (i.e., November to mid-March). Giant garter snakes typically select burrows with sunny exposure along south and west facing slopes. Giant garter snakes also use burrows as refuge from extreme heat during their active period. The Biological Resources Division (BRD) of the USGS (Wylie *et al.* 1997) has documented giant garter snakes using burrows in the summer as much as 165 feet (50 meters) away from the marsh edge. Overwintering snakes have been documented using burrows as far as 820 feet (250 meters) from the edge of marsh habitat.

During radio-telemetry studies conducted by the BRD giant garter snakes typically moved little from day to day. However, total activity varied widely between individuals. Snakes have been documented moving up to 5 miles (8 kilometers) over the period of a few days (Wylie *et al.* 1997). In agricultural areas, giant garter snakes were documented using rice fields 19-20% of the observations, marsh habitat 20-23% of observations, and canal and agricultural waterway habitats 50-56% of the observations (Wylie *et al.* 1997). Within canal and agricultural waterway habitats, giant garter snakes are likely to prefer drainage rather than delivery canals, because drainage canals are often less heavily maintained and are allowed to become vegetated.

Historic and Current Distribution: Fitch (1940) described the historical range of the species as extending from the vicinity of Sacramento and Contra Costa Counties southward to Buena Vista Lake, near Bakersfield, in Kern County. Prior to 1970, the giant garter snake was recorded historically from 17 localities (Hansen and Brode 1980). Five of these localities were clustered in and around Los Banos, Merced County, and the paucity of information makes it difficult to determine precisely the species' former range. Nonetheless, these records coincide with the historical distribution of large flood basins, fresh water marshes, and tributary streams. Reclamation of wetlands for agriculture and other purposes apparently extirpated the species from the southern one-third of its range by the 1940's-1950's, including the former Buena Vista Lake and Kern Lake in Kern County, and the historic Tulare Lake and other wetlands in Kings and Tulare Counties (Hansen and Brode 1980, Hansen 1980). Surveys over the last two decades have located the giant garter snake as far north as the Butte Basin in the Sacramento Valley.

As recently as the 1970s, the range of the giant garter snake extended from near Burrel, Fresno County (Hansen and Brode 1980), northward to the vicinity of Chico, Butte County (Rossman and Stewart 1987). California Department of Fish and Game (CDFG) studies (Hansen 1988) indicate that giant garter snake populations currently are distributed in portions of the rice production zones of Sacramento, Sutter, Butte, Colusa, and Glenn Counties; along the western border of the Yolo Bypass in Yolo County; and along the eastern fringes of the Sacramento-San Joaquin River delta from the Laguna Creek-Elk Grove region of central Sacramento County southward to the Stockton area of San Joaquin County. This distribution largely corresponds with agricultural land uses throughout the Central Valley.

Surveys over the last two decades have located the giant garter snake as far north as the Butte Basin in the Sacramento Valley. Currently, the Service recognizes 13 separate populations of giant garter snakes, with each population representing a cluster of discrete locality records (58 **FR** 54053). The 13 extant populational clusters largely coincide with historical riverine flood basins and tributary streams throughout the Central Valley (Hansen 1980, Brode and Hansen 1992): (1) Butte Basin, (2) Colusa Basin, (3) Sutter Basin, (4) American Basin, (5) Yolo Basin--Willow Slough, (6) Yolo Basin--Liberty Farms, (7) Sacramento Basin, (8) Badger Creek--Willow Creek, (9) Caldoni Marsh, (10) East Stockton--Diverting Canal and Duck Creek, (11) North and South Grasslands, (12) Mendota, and (13) Burrel/Lanare. These populations span the Central Valley from just southwest of Fresno (i.e., Burrel-Lanare) north to Chico (i.e., Hamilton Slough). The 11 counties where the giant garter snake is still presumed to occur are: Butte, Colusa, Glenn, Fresno, Merced, Sacramento, San Joaquin, Solano, Stanislaus, Sutter and Yolo.

In 1994, the BRD (formerly the National Biological Survey [NBS]) began a study of the life history and habitat requirements of the giant garter snake in response to an interagency submittal for consideration as an NBS Ecosystem Initiative. Since April of 1995, the BRD has further documented occurrences of giant garter snakes within some of the 13 populations identified in the final rule. The BRD has studied populations of giant garter snakes at the Sacramento and Colusa National Wildlife Refuges within the Colusa Basin, at Gilsizer Slough within the Sutter Basin, and at the Badger Creek area of the Cosumnes River Preserve within the Badger Creek-Willow Creek area (Wylie et al, 1997). These populations, along with the American Basin population of giant garter snakes represent the largest extant populations. With the exception of the American Basin, these populations are largely protected from many of the threats to the species. Outside of these protected areas, giant garter snakes in these population clusters are still subject to all threats identified in the final rule. The remaining nine population clusters identified in the final rule are distributed discontinuously in small isolated patches and are vulnerable to extirpation by stochastic environmental, demographic, and genetic processes. All 13 population clusters are isolated from each other with no protected dispersal corridors. Opportunities for recolonization of small populations which may become extirpated are unlikely given the isolation from larger populations and lack of dispersal corridors between them.

Further descriptions of the status of the thirteen subpopulations are given in Table 4 and in Appendix A.

<u>Reasons for Decline and Threats to Survival</u>: The current distribution and abundance of the giant garter snake is much reduced from former times. Agricultural and flood control activities have extirpated the giant garter snake from the southern one third of its range in former wetlands associated with the historic Buena Vista, Tulare, and Kern lakebeds. These lakebeds once supported vast expanses of ideal giant garter snake habitat, consisting of cattail and bulrush dominated marshes. Vast expanses of bulrush and cattail floodplain habitat also typified much of the Sacramento Valley historically (Hinds 1952). Prior to reclamation activities beginning in the mid to late 1800's, about 60 percent of the Sacramento Valley was subject to seasonal overflow flooding in broad, shallow flood basins that provided expansive areas of giant garter snake habitat (<u>ibid</u>.). All natural habitats have been lost and an unquantifiable small percentage of semi-natural wetlands remain extant. Only a small percentage of extant wetlands currently provide habitat suitable for the giant garter snake. Valley floor wetlands are also subject to the cumulative effects of upstream watershed modifications, water storage and diversion projects, as well as urban and agricultural development. Although some giant garter snake populations have persisted at low levels in artificial wetlands associated with agricultural and flood control activities, many of these altered wetlands are now threatened with urban development. Cities within the current range of the giant garter snake that are rapidly expanding include: (1) Chico, (2) Yuba City, (3) Sacramento, (4) Galt, (5) Stockton, (6) Gustine, and (7) Los Banos.

A number of land use practices and other human activities currently threaten the survival of the giant garter snake throughout the remainder of its range. Ongoing maintenance of aquatic habitats for flood control and agricultural purposes eliminate or prevent the establishment of habitat characteristics required by giant garter snakes and can fragment and isolate available habitat, prevent dispersal of snakes among habitat units, and adversely affect the availability of the garter snake's food items (Hansen 1988, Brode and Hansen 1992). Livestock grazing along the edges of water sources degrades habitat quality in a number of ways: (1) eating and trampling aquatic and riparian vegetation needed for cover from predators, (2) changes in plant species composition, (3) trampling snakes, (4) water pollution, (5) and reducing or eliminating fish and amphibian prey populations. Overall, grazing has contributed to the elimination and reduction of the quality of available habitat at four known locations (Hansen 1982, 1986).

In many areas, the restriction of suitable habitat to water canals bordered by roadways and levee tops renders giant garter snakes vulnerable to vehicular mortality. Fluctuation in rice and agricultural production affects stability and availability of habitat. Recreational activities, such as fishing, may disturb snakes and disrupt basking and foraging activities. Non-native predators, including introduced predatory gamefish, bullfrogs, and domestic cats also threaten giant garter snake populations. While large areas of seemingly suitable giant garter snake habitat exist in the form of duck clubs and waterfowl management areas, water management of these areas typically does not provide summer water needed by giant garter snakes. Although giant garter snakes on NWRs are relatively protected from many of the threats to the species, water quality continues to be a threat to the species both on and off NWRs.

Documented declines due to selenium contamination - San Joaquin Valley subpopulations of giant garter snakes have suffered severe declines and possible extirpations over the last two decades. Prior to 1980, several areas within the San Joaquin Valley supported populations of giant garter snakes. Until recently, there were no post-1980 sightings from Stockton, San Joaquin County, southward, despite several survey efforts (G. Hansen, 1988). Surveys during 1986 of prior localities did not detect any giant garter snakes. During 1995 surveys of prior locality records and adjacent waterways, one road killed giant garter snake was found, and three presumed giant garter snakes were observed but not captured (G. Hansen, 1996). Two sightings occurred at Mendota Wildlife Area, and two occurred several miles south of the town of Los Banos. These data indicate that giant garter snakes are still extant in two localities within the San Joaquin, but in extremely low to undetectable numbers.

Although habitat has been lost or degraded throughout the Central Valley, there have been many recent sightings of giant garter snakes in the Sacramento Valley while there have been very few recent sightings within the San Joaquin Valley. The 1995 report on the status of giant garter snakes in the San Joaquin Valley (G. Hansen, 1996) indicates that Central San Joaquin Valley giant garter snake numbers appear to have declined even more dramatically than has apparently suitable habitat. Factors in addition to habitat loss may be contributing to the decline. These are factors which affect giant garter snakes within suitable habitat and include interrupted water supply, poor water quality, and contaminants (G. Hansen, 1996).

Selenium contamination and impaired water quality have been identified in the final rule listing the giant garter snake as a threat to the species and a contributing factor in the decline of giant garter snake populations, particularly for the North and South Grasslands subpopulation (i.e., Kesterson NWR area). The bioaccumulative food chain threat of selenium contamination on fish, frogs, and fish-eating birds has been well documented. Though there is little data specifically addressing toxicity of selenium, Hg, or metals to reptiles, it is expected that reptiles would have toxicity thresholds similar to those of fish and birds. (58 **FR** 54053 under Factor E - Contaminants)

Threats due to contaminants and impaired water quality - The range of the giant garter snake occurs entirely within the Central Valley of California, putting giant garter snakes at risk of exposure to numerous contaminants from agricultural, urban, and industrial/mining runoff. Current water sources and supplies to areas supporting giant garter snakes indicate that the species is at risk of exposure to both mercury and selenium. Many areas supporting populations of giant garter snake receive water from agricultural drainage, which may contain elevated levels of selenium and other contaminants. Selenium contamination of drainwater has been identified in the San Joaquin Valley giant garter snake subpopulations (58 **FR** 54053 and references therein). However, refuges in the Sacramento Valley which currently support giant garter snakes also receive agricultural return flows as part of their water supplies. These include Gray Lodge Wildlife Area, Sacramento NWR, Delevan NWR, Colusa NWR, and Sutter NWR (USDI 1997). In addition, streams draining the coastal ranges may contribute selenium to aquatic systems within the Central Valley.

Mercury also is present in numerous drainages in the Central Valley due to past mercury and gold mining activity. Sacramento Valley refuges and other areas supporting giant garter snake populations also receive water from drainages which may contribute mercury to the aquatic systems. These drainages include the Sacramento, Feather, American, and Cosumnes Rivers, and Laguna, Morrison, Stony, Auburn Ravine, Putah, and Cache Creeks.

Table 4 describes known giant garter snake locations within the thirteen giant garter snake subpopulations, the status of the subpopulations, the potential for exposure to selenium and mercury, and the potential for synergistic effects of selenium and mercury. Appendix A further describes the status of the thirteen subpopulations, and also describes some water supply sources to refuges and other areas that support giant garter snakes. Although giant garter snake populations on refuges may be protected from many of the threats to the species, they are not protected from exposure to poor water quality and contaminants introduced from water supply sources.

Water quality impairment of aquatic habitat that supports giant garter snakes could reduce the prey base, contribute to bioaccumulation, impair essential behaviors, and reduce reproductive success. Appendix A lists existing impaired water bodies (from California Impaired Waterbodies list) that either currently support giant garter snakes or supply water to areas that support giant garter snakes. Although the level of impairment and specific contaminants were not listed, this information identifies that significant water quality impairment already exists. The list of water bodies that may support or supply giant garter snake populations indicates that the species is currently challenged with poor water quality. Unprotective water quality standards proposed in the CTR could further impair water quality within these giant garter snake subpopulations and represent the potential for cumulative and synergistic effects of contaminants and poor water quality.

Summary of contaminants threats to giant garter snakes - The giant garter snake has a restricted distribution and is entirely dependent on its aquatic ecosystem. The thirteen population clusters identified in the final rule are distributed discontinuously in small isolated patches and are vulnerable to extirpation by stochastic environmental, demographic, and genetic processes. It is probable that elevated selenium levels in the San Joaquin Valley contributed to the severe decline and possible extirpation of the giant garter snake from the majority of this area. The remaining giant garter snake populations are exposed to impaired waterbodies and existing or potential sources of selenium and mercury. As top predators, giant garter snakes are at risk of exposure to elevated levels of contaminants such as mercury and selenium. Over the life of the giant garter snake it is possible to accumulate contaminants that can impact the growth, survival, and reproduction of individuals, leading to declines in distribution.

Mountain Yellow-legged Frog: Southern California Distinct Population Segment (*Rana muscosa*)

Species Description and Life History: The mountain yellow-legged frog is a true frog in the family Ranidae. Mountain yellow-legged frogs were originally described by Camp in 1917 (as cited by Zweifel 1955) as a subspecies of Rana boylii. Zweifel (1955) demonstrated that frogs from the high Sierra and the mountains of southern California were somewhat similar to each other yet were distinct from the rest of the R. boylii (= boylei) group. Since that time, most authors have followed Zweifel, treating the mountain yellow-legged frog as a full species, Rana muscosa.

Mountain yellow-legged frogs are moderately sized, about 40 to 80 millimeters (mm) (1.5 to 3 inches (in)) from snout to urostyle (the pointed bone at the base of the backbone) (Jennings and Hayes 1994; Zweifel 1955). The pattern is variable, ranging from discrete dark spots that can be few and large, to smaller and more numerous spots with a mixture of sizes and shapes, to irregular lichen-like patches or a poorly defined network (Zweifel 1955). The body color is also variable, usually a mix of brown and yellow, but often with gray, red, or green-brown. Some individuals may be dark brown with little pattern (Jennings and Hayes 1994). The back half of the upper lip is pale. Folds are present on each side of the back, but usually they are not prominent (Stebbins 1985). The throat is white or yellow, sometimes with mottling of dark pigment (Zweifel 1955). The belly and undersurface of the high limbs are yellow, which ranges in hue from pale lemon yellow to an intense sun yellow. The iris is gold with a horizontal, black counter shading stripe (Jennings and Hayes 1994).

In the Sierra Nevada Mountains of California, the mountain yellow-legged frog ranges from southern Plumas County to southern Tulare County (Jennings and Hayes 1994), at elevations mostly above 1,820 meters (m) (6,000 feet (ft)). The frogs of the Sierra Nevada are isolated from the frogs of the mountains of southern California by the Tehachapi Mountains and a distance of about 225 kilometers (km) (140 miles (mi)). The southern California frogs now occupy portions of the San Gabriel, San Bernardino, and San Jacinto Mountains. Zweifel (1955) noted the presence of an isolated southern population on Mt. Palomar in northern San Diego County, but this population appears to be extinct (Jennings and Hayes 1994). In southern California, the elevation range reported by Stebbins (1985) is 182 m (600 ft) to 2,273 m (7,500 ft). Representative localities, including some that are no longer occupied, which demonstrate the wide elevation range that mountain yellow-legged frogs inhabited in southern California, include Eaton Canyon, Los Angeles County (370 m (1,220 ft)) and Bluff Lake, San Bernardino County (2,290 m (7,560 ft)). The southern California locations now occupied by mountain yellow-legged frogs range from City Creek, in the San Bernardino Mountains (760 m (2,500 ft)), to Dark

Canyon in the San Jacinto Mountains (1,820 m (6,000 ft)).

Southern California mountain yellow-legged frogs are diurnal, highly aquatic frogs, occupying rocky and shaded streams with cool waters originating from springs and snowmelt. In these areas, juveniles and adults feed on small, streamside arthropods (Jennings and Hayes 1994). They do not occur in the smallest creeks. The coldest winter months are spent in hibernation, probably under water or in crevices in the bank. Mountain yellow-legged frogs emerge from overwintering sites in early spring, and breeding soon follows. Eggs are deposited in shallow water where the egg mass is attached to vegetation or the substrate. In the Sierra Nevada, larvae select warm microhabitats (Bradford 1984 cited in Jennings and Hayes 1994), and the time to develop from fertilization to metamorphosis reportedly varies from 1 to 2.5 years (Jennings and Hayes 1994).

Prior to the late 1960s, mountain yellow-legged frogs were abundant in many southern California streams (G. Stewart, in litt. 1995), but they now appear to be absent from most places in which they previously occurred. Jennings and Hayes (1994) believe that mountain yellow-legged frogs are now absent from more than 99 percent of their previous range in southern California. This decline is part of a well-known larger pattern of declines among native ranid frogs in the western United States (Hayes and Jennings 1986; Drost and Fellers 1996). Some of the western ranid frog species experiencing noticeable declines are the California red-legged frog (Rana aurora draytonii) (61 FR 25813), the spotted frog (R. pretiosa and R. luteventris), the Cascades frog (R. cascadae), and the Chiricahua leopard frog (R. chiricauhensis) (62 FR 49398). Nowhere have the declines been any more pronounced than in southern California, where, besides declines in mountain yellow-legged frogs, the California red-legged frog has been reduced to a few small remnants (61 FR 25813), and the foothill yellow-legged frog (R. boylii) may be extinct (Jennings and Hayes 1994.)

Distinct Vertebrate Population Segment: We analyzed the mountain yellow-legged frog according to the joint Service and National Marine Fisheries Service Policy Regarding the Recognition of Distinct Vertebrate Populations, published in the Federal Register on February 7, 1996 (61 FR 4722). We consider three elements in determining whether a vertebrate population segment could be treated as threatened or endangered under the Act: discreteness, significance, and conservation status in relation to the standards for listing. Discreteness refers to the isolation of a population from other members of the species and is based on two criteria: (1) Marked separation from other populations of the same taxon resulting from physical, physiological, ecological, or behavioral factors, including genetic discontinuity, or (2) populations delimited by international boundaries. We determine significance either by the importance or contribution, or both, of a discrete population to the species throughout its range. Our policy lists four examples of factors that may be used to determine significance: (1) Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon; (2) evidence that loss of the discrete population segment would result in a significant gap in the range of the taxon; (3) evidence that the discrete population segment represents the only surviving natural occurrence of the taxon that may be more abundant elsewhere as an introduced population outside its historic range; and (4) evidence that the discrete population segment differs markedly from other populations of the taxon in its genetic characteristics. If we determine that a population segment is discrete and significant, we evaluate it for endangered or threatened status based on the Act's standards.

Discreteness: The range of the mountain yellow-legged frog is divided by a natural geographic barrier, the Tehachapi Mountains, which isolate Sierran frogs from those in the mountains of southern California. The distance of the separation is about 225 km (140 mi), but the separation may not have been this great in the recent past because a frog collected in 1952 on Breckenridge Mountain in Kern County was identified by Jennings and Hayes (1994) as a mountain yellow-legged frog. The geographic separation of the Sierran and southern California frogs was recognized in the earliest description of the species by Camp (1917, cited in Zweifel 1955), who treated frogs from the two localities as separate subspecies within the R. boylii group. He designated the Sierran frogs R. b. sierrae and the southern California frogs R. b. muscosa, based on geography and subtle morphological differences. Zweifel (1955) reevaluated the morphological evidence and found it insufficient to warrant Camp's recognition of two subspecies, the chief difference between the two being hind-limb length.

More recently, Ziesmer (1997) analyzed the calls of Sierran (Alpine and Mariposa Counties) and southern California (San Jacinto Mountains and Riverside County) mountain yellow-legged frogs. He found that the calls of Sierran frogs differed from southern California frogs in pulse rate, harmonic structure, and dominant frequency. Based on a limited sample, Ziesmer concluded that the results supported the hypothesis that mountain yellow-legged frogs from the Sierra Nevada and southern California are separate species.

Allozyme (a form of an enzyme produced by a gene) variation throughout the range of the mountain yellow-legged frog has been examined, but the results are open to interpretation (Jennings and Hayes 1994 and references therein). In the work most applicable to the question of the distinctiveness of the Sierran and southern California frogs, David Green (pers. comm., 1998) analyzed allozyme variation in central Sierran mountain yellow-legged frogs (four individuals, Tuolumne County) and southern California mountain yellow-legged frogs (two individuals, Riverside County). He found fixed differences at 6 of 28 loci (sites on a chromosome occupied by specific genes). These limited, unpublished data suggest that Sierran and southern California mountain yellow-legged frogs are different at a level that could support the recognition of full species. However, because of the small number of individuals per sample and the limited number of samples, we view these results cautiously. It is possible that existing variation at those six loci may not have been detected with such a small number of individuals sampled. To better understand whether a genetic discontinuity significant enough to warrant full species rank exists between Sierran frogs and those from the mountains of southern California, samples of frogs from the southern Sierra Nevada, especially the Greenhorn Mountains, would be of particular interest.

Although Green's limited allozyme analysis may not be sufficient to support recognizing the Sierran and southern California populations as separate species, it does support the conclusion of significant geographic separation. This conclusion is also supported by earlier observations of morphological differences (Zweifel 1955, and references therein) and differences in vocalizations (Ziesmer 1997). Considered together, the evidence supports an interpretation of isolation between the two populations of frogs over a very long period. We find that the southern California frogs meet the criterion of "marked separation from other populations of the same taxon" and qualify as discrete according to the Policy Regarding the Recognition of Distinct Vertebrate Populations (61 FR 4722).

Significance. One of the most striking differences between Sierran and southern California mountain yellow-legged frogs is the habitats they occupy. Zweifel (1955) observed that the frogs in southern California are typically found in steep gradient streams in the chaparral belt, even though they may range up into small meadow streams at higher elevations. In contrast, Sierran frogs are most abundant in high elevation lakes and slow-moving portions of streams. Bradford's (1989) southern Sierra Nevada study site, for example, was in Sequoia and Kings Canyon National Parks at high elevations (between 2,910-3,430 m (9,600-11,319 ft)). The rugged canyons of the arid mountain ranges of southern California bear little resemblance to the alpine lakes of the Sierra Nevada. On the basis of habitat alone, one might easily conclude that these are two very different frogs.

The mountain yellow-legged frogs of southern California comprise the southern portion of the species' range. The extinction of this southern group would be significant because it would substantially reduce the overall range as it is currently understood, and what is now a gap in the distribution, the Tehachapi Mountains, would become the southern limit of the species' range.

In addition, evidence exists that the mountain yellow-legged frog is not simply a single species with a disjunct distribution (cited in Zweifel 1955; Stebbins 1985). As discussed above, vocal and genetic differences exist between Sierran and southern California mountain yellow-legged frogs. Although the data are limited and some important variation may have been missed, they are consistent with the earlier interpretation by Camp (1917 cited in Zweifel 1955) and numerous other authors prior to Zweifel (e.g., Stebbins 1954) who treated the two forms as taxonomically distinct. If the differences in vocalization described by Ziesmer (1997) and the allozyme variation described by Green (per. comm., 1998) accurately characterize differences between the two forms, then the Sierran and southern California frogs are quite different and have been isolated for a very long time.

Our conclusion that Sierran and southern California frogs are very different from each other, and may even merit recognition as separate subspecies or possibly even species, is based on the cumulative weight of the available evidence. We find that the mountain yellow-legged frogs inhabiting the mountains of southern California meet the significance criteria under our Policy Regarding the Recognition of Distinct Vertebrate Populations (61 FR 4722) on the basis of the geographical, ecological, vocal, and genetic discontinuities described above.

Reasons for Decline and Threats to Survival: The mechanisms causing the declines of western frogs are not well understood and are certain to vary somewhat among species, but the two most common and well-supported hypotheses for widespread declines of western ranid frogs are: (1) Past habitat destruction related to unregulated activities such as logging and mining and more recent habitat conversions for water development, irrigated agriculture, and commercial development (Hayes and Jennings 1986; 61 FR 25813); and (2) alien predators and competitors (Bradford 1989; Knapp 1996; Kupferberg 1997). Natural populations may be killed off directly by these factors operating alone or in combination, or these factors so severely disrupt the normal population dynamics that when local extinctions occur, regardless of the cause, natural recolonization is impossible. Other environmental factors that could have adverse effects over a wide geographic range include pesticides, certain pathogens, and ultraviolet-B (beyond the visible spectrum) radiation, but their role, if any, in amphibian declines is not well understood (Reaser 1996). These factors, acting singly or in combination, may be contributing to widespread, systematic declines of western ranid frogs. Determining their effects, however, is not an easy task (Reaser 1996; Wake 1998), and the Department of the Interior (USDOI) currently supports an initiative to fund research on the causes of amphibian declines (see examples in USDOI 1998).

Some of the same factors that are hypothesized to have caused declines of other western ranid frogs are likely to be responsible for the reduction of the mountain yellow-legged frog in southern California. Because the declines have been so precipitous, and have spared only a small number of frogs in a few localities, the factors, and their interactions, that caused the decline may never be fully understood. We believe that these factors are still operating, and unless reversed, a high probability exists that this frog may be extinct in southern California within a few decades. In the case of the mountain yellow-legged frog, the only factor listed above that we believe can be ruled out as a likely cause of decline is habitat destruction related to activities such as logging, mining, irrigated agriculture, and commercial development. The range of the mountain yellow-legged frog in southern California is mainly on public land administered by the U.S. Forest Service (FS). Most of the rugged canyons and surrounding mountainous terrain have been altered little and look much the same today as they did when earlier naturalists such as Lawrence Klauber collected mountain yellow-legged frogs there in the early decades of the 1900s.

Historic and Current Distribution: In southern California, mountain yellow-legged frogs can still be found in four small streams in the San Gabriel Mountains, the upper reaches of the San Jacinto River system in the San Jacinto Mountains, and at a single locality on City Creek, a tributary of the Santa Ana River, in the San Bernardino Mountains (Jennings and Hayes 1994; M. D. Wilcox in litt., 1998). These areas along with the numbers of frogs most recently observed in each area are described below.

San Gabriel Mountains: Surveys conducted from 1993 to 1997 revealed small isolated populations in the upper reaches of Prairie Creek/Vincent Gulch, Devil's Canyon, and Alder Creek/East Fork, on the East Fork of the San Gabriel River, and Little Rock Creek on the Mojave River (Jennings and Hayes 1994 and references therein; Jennings 1995; Jennings 1998). The surveys involved one to three field biologists and were conducted over 1-5 days per site.

Over the course of these field studies, 15 adults or fewer were observed at any 1 site, and, after the 1995 season, Jennings (1995) concluded that the actual population at each of the sites was only 10-20 adults.

San Jacinto Mountains: Small populations of mountain yellow-legged frogs also occur in four tributaries in the upper reaches of the North Fork, San Jacinto River on Mount San Jacinto: Dark Canyon, Hall Canyon, Fuller Mill Creek, and the main North Fork, San Jacinto River (Jennings and Hayes 1994; Jennings 1995; Jennings 1998). The number of frogs occupying these sites is not known, but fewer than 10 adult frogs per site per year have been observed in surveys from 1995 to the present.

San Bernardino Mountains: A few tadpoles and 26 recently transformed juveniles, but no adults, were rediscovered on a roughly 1-mile reach of the East Fork, City Creek during the summer of 1998 (M. D. Wilcox in litt., 1998). Previous to this finding, mountain yellow-legged frogs had not been observed in the San Bernardino Mountains since the 1970s (Jennings and Hayes 1994), even though surveys were conducted during the summer and fall of 1997 and 1998 (Holland 1997; Tierra Madre 1999).

When frogs were encountered during field surveys accomplished between 1988 and 1995, only a few individuals were observed. Jennings and Hayes (1994) and Jennings (1995) suggested that the entire population of mountain yellow-legged frogs in the San Gabriel and San Jacinto Mountains (8 more or less isolated sites) was probably fewer than 100 adult frogs. Their rough estimate is based on a compilation of the results of visual surveys generally conducted on a single day, not on formal population abundance estimation techniques. While the precise number of adult frogs may be greater than 100, we concur with Jennings and Hayes (1994) that, in the San Gabriel and San Jacinto Mountains, the available data indicate that this once widespread species is now found in only a small number of relatively isolated populations. We do not know the population size of adult frogs at the recently rediscovered site on the east fork of City Creek in the San Bernardino Mountains, but because no adults and only a few juveniles and tadpoles were encountered, the adult population is probably small. Thus, we conclude that each of the three mountain ranges (San Gabriel, San Jacinto, San Bernardino) contains a small number of small, relatively isolated populations.

San Francisco garter snake (*Thamophis sirtalis tetrataenia*)

Species Description and Life History: The San Francisco garter snake was listed as a Federal endangered species in March, 1967 (32 **FR** 4001). The San Francisco garter snake is an extremely colorful snake. It is identified by its burnt orange head, yellow to greenish-yellow dorsal stripe edged in black, and its red lateral stripe which may be continuous or broken with black blotches and edged in black. The belly color varies from greenish-blue to blue. Large adults can reach three feet in length.

The San Francisco garter snakes preferred habitat is a densely vegetated pond near an open hillside where it can sun itself, feed, and find cover in rodent burrows. The snakes are extremely shy, difficult to locate and capture, and quick to flee to water or cover when disturbed (Willy, pers. comm.). Adult snakes may estivate in rodent burrows during summer months when ponds may dry. On the coast snakes hibernate during the winter, but further inland, if the weather is suitable, snakes may be active year round.

San Francisco garter snakes breed in the spring or late fall (Larsen, pers. comm.) and bear live young from May through October (Stebbins 1985). The average litter size is 12-18 (Stebbins 1985). Many species of snakes, including garter snakes, breed adjacent to their hibernacula. Although highly vagile, adults spend considerable time after emergence in their hibernacula.

Foraging Ecology: Although primarily a diurnal species, captive snakes housed in an outside enclosure were observed foraging after dark on warm evenings (Larsen, pers. comm.). Adult snakes feed primarily on California red-legged frogs, and may also feed on juvenile bullfrogs (*Rana catesbeiana*). In laboratory studies, Larsen (1994) fed adult San Francisco garter snakes two year old bullfrog tadpoles and found that only the largest adults could eat and digest the tadpoles; smaller adults regurgitated partially digested tadpoles, apparently unable to fully digest them. Larsen (1994) also found that when these smaller adult snakes were fed bullfrogs and California red-legged frogs of comparable size, they were unable to hold and eat the bullfrogs although they had no trouble with the California red-legged frogs. Newborn and juvenile San Francisco garter snakes depend heavily upon Pacific treefrogs (*Hyla regilla*) as prey (Larsen 1994). If newly metamorphosed Pacific treefrogs are not available, the young snakes may not survive.

<u>Historic</u> <u>and</u> <u>Current</u> <u>Distribution</u>: Historically, San Francisco garter snakes occurred in scattered wetland areas on the San Francisco Peninsula from approximately the San Francisco County line south along the eastern and western bases of the Santa Cruz Mountains, at least to the Upper Crystal Springs Reservoir, and along the coast south to Año Nuevo Point, San Mateo County, and Waddell Creek, Santa Cruz County, California. Currently, the species has been reduced to only six populations in San Mateo County and the extreme northern Santa Cruz County. Sag ponds--small seasonal freshwater ponds formed along the San Andreas fault--historically supported this snake, but most of these former locations have been destroyed by urbanization.

The species has been extirpated from most of its historical distribution in the Skyline Boulevard area of San Mateo County. Fox (1951) reported typical populations of the snake on the coast around Sharp Park (Laguna Salada), and along Skyline Boulevard. Since then, the sag ponds along Skyline Boulevard were drained and filled for urban development and the Sharp Park area has been severely impacted. In 1987, the seawall at Sharp Park failed, allowing the intrusion of salt water into Laguna Salada. In 1989, abandoned quarry ponds adjacent to Calera Creek (over the ridge from Sharp Park) were found to support a small population of snakes. These snakes may have migrated from Laguna Salada after the failure of the sea wall. In August 1989, the quarry ponds were illegally drained and filled. The current population status at the quarry ponds and Sharp Park is unknown. In 1985, the population at Año Nuevo State Reserve was thought to be stable at fewer than 50 snakes, but in 1995 the population appeared to be declining (Paul Keel, pers. comm.). This decline may be caused by inadequate management for the San Francisco garter snake and the recent introduction of bullfrogs.

The Recovery Plan for the San Francisco garter snake (USDI-FWS 1985c) identified six significant populations. These were the Airport (west-of-Bayshore), San Francisco State Fish and Game Refuge (Refuge), Laguna Salada (Pacifica), Pescadero Marsh Natural Preserve (Pescadero) and Año Nuevo State Reserve (Año Nuevo) populations, and an isolated population fragment north of Half Moon Bay. Of the six populations known in 1985, the Pacifica population was heavily impacted in 1989 and is no longer considered significant, four have declined drastically (Airport, Refuge, Pescadero and Año Nuevo). The status of the Half Moon Bay population is unknown.

Reasons for Decline and Threats to Survival: Current threats to the San Francisco garter snakes' existence include reservoir construction and management, agricultural practices, poor management practices on lands where San Francisco garter snakes currently survive, and isolation of populations. Introduced predators such as predatory fish and bullfrogs impact not only the San Francisco garter snake, but also its principal prey species, the Pacific treefrog and the threatened California red-legged frog. Because there are so few remaining populations of the San Francisco garter snake extant populations are extremely vulnerable to local contamination. The San Francisco garter snake has a narrow foraging niche, if contamination of forage species occurs it is likely to significantly impact the species ability to survive. The San Francisco garter snake's beautiful coloration also makes it valuable to both amateur and professional illegal collectors. Extirpation of California red-legged frogs in San Francisco garter snake habitat is likely to cause a local extinction event for the snake.

California Tiger Salamander - Santa Barbara County Distinct Population Segment (*Ambystoma californniense*)

Species Description and Life History: The California tiger salamander is a large, stocky, terrestrial salamander with a broad, rounded snout. This distinct population segment (DPS)of the species was proposed as endangered on January 19, 2000 (65 **FR** 3110). California tiger salamanders are restricted to California, and their range does not overlap with any other species of tiger salamander (Stebbins 1985). Within California, the Santa Barbara County population is separated by the Coast Ranges, particularly the La Panza and Sierra Madre Ranges, and the Carrizo Plain from the closest other population, which extends into the Temblor Range in eastern San Luis Obispo and western Kern Counties (Shaffer, et al. 1993).

Adults may reach a total length of 207 millimeters (mm) (8.2 inches (in)), with males generally averaging about 200 mm (8 in) in total length and females averaging about 170 mm (6.8 in) in total length. For both sexes, the average snout–vent length is approximately 90 mm (3.6 in). The small eyes have black irises and protrude from the head. Coloration consists of white or pale yellow spots or bars on a black background on the back and sides. The belly varies from almost uniform white or pale yellow to a variegated pattern of white or pale yellow and black. Males can be distinguished from females, especially during the breeding season, by their swollen cloacae (a common chamber into which the intestinal, urinary, and reproductive canals discharge), more developed tail fins, and larger overall size (Stebbins 1962; Loredo and Van Vuren 1996).

Subadult and adult California tiger salamanders spend much of their lives in small mammal burrows found in the upland component of their habitat, particularly those of ground squirrels and pocket gophers (Loredo and Van Vuren 1996, Trenham 1998a). During estivation (a state of dormancy or inactivity in response to hot, dry weather), California tiger salamanders eat very little (Shaffer, et al. 1993). Once fall and winter rains begin, they emerge from these retreats on nights of high relative humidity and during rains to feed and to migrate to the breeding ponds (Stebbins 1985, 1989; Shaffer, et al. 1993). The salamanders breeding in and living around a pool or seasonal pond, or a local complex of pools or seasonal ponds, constitute a local subpopulation. The rate of natural movement of salamanders among subpopulations depends on the distance between the ponds or complexes and on the intervening habitat (e.g., salamanders may move more quickly through sparsely covered and more open grassland versus more densely vegetated scrublands).

Adults may migrate up to 2 kilometers (km) (1.2 miles (mi)) from summering to breeding sites. The distance from breeding sites may depend on local topography and vegetation, the distribution of ground squirrel or other rodent burrows, and climatic conditions (Stebbins 1989, Hunt 1998). In Santa Barbara County, juvenile California tiger salamanders have been trapped over 360 m (1,200 ft) while dispersing from their natal (birth) pond (Ted Mullen, Science Applications International Corporation (SAIC), personal communication, 1998), and adults have been found along roads over 2 km (1.2 mi) from breeding ponds (S. Sweet, in litt. 1998a). Migration is concentrated during a few rainy nights early in the winter, with males migrating before females (Twitty 1941; Shaffer, et al. 1993; Loredo and Van Vuren 1996; Trenham 1998b). Males usually remain in the ponds for an average of about 6 to 8 weeks, while females stay for approximately 1 to 2 weeks. In dry years, both sexes may stay for shorter periods (Loredo and Van Vuren 1996, Trenham 1998b). Although most marked salamanders have been recaptured at the pond where they were initially captured, in one study approximately 20 percent were recaptured at different ponds (Trenham 1998b). As with migration distances, the number of ponds used by an individual over its lifetime will be dependent on landscape features.

Female California tiger salamanders mate and lay their eggs singly or in small groups (Twitty 1941; Shaffer, et al. 1993). The number of eggs laid by a single female ranges from approximately 400 to 1,300 per breeding season (Trenham 1998b). The eggs typically are attached to vegetation near the edge of the breeding pond (Storer 1925, Twitty 1941), but in ponds with no or limited vegetation, they may be attached to objects (rocks, boards, etc.) on the bottom (Jennings and Hayes 1994). After breeding, adults leave the pond and typically return to small mammal burrows (Loredo et al. 1996; Trenham 1998a), although they may continue to come out nightly for approximately the next 2 weeks to feed (Shaffer, et al. 1993).

Eggs hatch in 10 to 14 days with newly hatched larvae ranging from 11.5 to 14.2 mm (0.45 to 0.56 in) in total length. Larvae feed on algae, small crustaceans, and mosquito larvae for about 6 weeks after hatching, when they switch to larger prey (P.R. Anderson 1968). Larger larvae have been known to consume smaller tadpoles of Pacific treefrogs (Hyla regilla) and California red-legged frogs (Rana aurora) as well as many aquatic insects and other aquatic invertebrates (J.D. Anderson 1968; P.R. Anderson 1968). Captive salamanders appear to locate food by vision and olfaction (smell) (J.D. Anderson 1968).

Amphibian larvae must grow to a critical minimum body size before they can metamorphose (change into a different physical form) to the terrestrial stage (Wilbur and Collins 1973). Feaver (1971) found that California tiger salamander larvae metamorphosed and left the breeding ponds 60 to 94 days after the eggs had been laid, with larvae developing faster in smaller, more rapidly drying ponds. The longer the ponding duration, the larger the larvae and metamorphosed juveniles are able to grow. The larger juvenile amphibians grow, the more likely they are to survive and reproduce (Semlitsch et al. 1988; Morey 1998).

In the late spring or early summer, before the ponds dry completely, metamorphosed juveniles leave the ponds and enter small mammal burrows after spending up to a few days in mud cracks or tunnels in moist soil near the water (Zeiner et al. 1988; Shaffer, et al. 1993; Loredo et al. 1996). Like the adults, juveniles may emerge from these retreats to feed during nights of high relative humidity (Storer 1925; Shaffer, et al. 1993) before settling in their selected estivation sites for the dry summer months.

Many of the pools California tiger salamanders lay eggs water is not retained water long enough to support successful metamorphosis. Generally, 10 weeks is required to allow sufficient time to metamorphose. The larvae will desiccate (dry out and perish) if a site dries before larvae complete metamorphosis (P.R. Anderson 1968, Feaver 1971). Pechmann et al. (1989) found a strong positive correlation with ponding duration and total number of metamorphosing juveniles in five salamander species. In one study, successful metamorphosis of California tiger salamanders occurred only in larger pools with longer ponding durations (Feaver 1971), which is typical range-wide (Jennings and Hayes 1994). Even though there is little difference in the number of pools used by salamanders between wet and dry years, pool duration is the most important factor to consider in relation to persistence and survival (Feaver 1971; Shaffer, et al. 1993; Seymour and Westphal 1994, 1995).

Lifetime reproductive success for California and other tiger salamanders is typically low, with fewer than 30 metamorphic juveniles per breeding female. While individuals may survive for more than 10 years, many may breed only once, and, in some populations, less than 5 percent of marked juveniles survive to become breeding adults (Trenham 1998b). With such low recruitment, isolated subpopulations can decline greatly from unusual, randomly occurring natural events as well as from human-caused factors that reduce breeding success and individual survival. Factors that repeatedly lower breeding success in isolated ponds that are too far from other ponds for migrating individuals to replenish the population can quickly drive a local population to extinction.

Historic and Current Distribution: The California tiger salamander inhabits low elevation, below 300 meters (m) (1000 feet (ft)), vernal pools and seasonal ponds and the associated coastal scrub, grassland, and oak savannah plant communities of the Santa Maria, Los Alamos, and Santa Rita Valleys in western Santa Barbara County (Shaffer, et al. 1993; Sam Sweet, University of California, Santa Barbara, in litt. 1993, 1998a). Although California tiger salamanders still exist across most of their historic range in Santa Barbara County, the habitat available to them has been reduced greatly. Ponds available to salamanders for breeding have been degraded and reduced in number. In addition, upland habitats inhabited by salamanders for most of their life cycle have been degraded and reduced in area through changes in agriculture practices, urbanization, building of roads and highways, chemical applications, and overgrazing (Gira et al. 1999; S. Sweet, in litt. 1993, 1998a,b).

Currently, California tiger salamanders in Santa Barbara County are found in four discrete regions (S. Sweet, in litt. 1998a). Collectively, salamanders in these regions constitute a single genetic population or DPS, reproductively separate from the rest of the California tiger salamanders (Jones 1993; Shaffer, et al. 1993; Shaffer and McKnight 1996). Ponds and associated uplands in southwestern (West Orcutt) and southeastern (Bradley-Dominion) Santa Maria Valley, Los Alamos Valley, and Santa Rita Valley constitute the four discrete regions or metapopulations where California tiger salamanders now exist in Santa Barbara County (S. Sweet, in litt. 1998a). For the purposes of this account, a metapopulation is defined as a group of subpopulations or "local populations" linked by genetic exchange. Of 14 known breeding sites or subpopulations within this DPS, 1 was destroyed in 1998, the upland habitat around 3 has been converted into more intensive agriculture practices (i.e. vineyards, gladiolus fields, and row crops, which may have eliminated the salamander subpopulations), 1 is surrounded by agriculture and urban development, 2 are affected by overgrazing, 4 are imminently threatened with conversion to vineyards or other intensive agriculture practices, and the remaining 3 are in areas rapidly undergoing conversion to vineyards and row crops (Sweet, et al. 1998; Sweet, in litt. 1998; Santa Barbara County Planning and Development 1998; Grace McLaughlin, Service, personal observations, 1998). Thus, only 6 or 7 of 13 existing ponds potentially provide breeding habitat for viable subpopulations of Santa Barbara County California tiger salamanders. Although other breeding ponds could exist within each of the four metapopulations noted above, searches around extant localities in the county, as well as in other areas with suitable habitat, have not identified additional subpopulations of the species (Paul Collins, Santa Barbara Museum of Natural History, in litt. 1998, pers. comm. 1999; S. Sweet, in litt. 1998a). Four possible breeding ponds or pond complexes (three in the Bradley-Dominion area, one in Santa Rita Valley) have been identified from aerial photography and by finding salamanders on roads in the vicinity (Sweet, et al. 1998) but have not been sampled. Most of the upland habitats around the ponds have been converted to vineyards or row crops within the last 6 years (Santa Barbara County Planning and Development 1998). All of the known and potential localities of the California tiger salamander in Santa Barbara County are on private lands, none are protected by conservation easements or agreements, and access is limited.

Reasons for Decline and Threats to Survival: The factors believed to responsible for the decline of the species are habitat loss due to conversion of natural habitat to intensive agriculture, urban development, habitat fragmentation, and agricultural contaminants.

Santa Cruz Long-Toed Salamander (*Ambystoma macrodactylum croceum*)

Species Description and Life History: The Santa Cruz long-toed salamander was listed on March 11, 1967 (32 **FR** 4001). At that time, only two breeding localities of the Santa Cruz long-toed salamander, Valencia Lagoon and Ellicott Slough, were known. A recovery plan was approved in 1977, and revised in 1985; currently the Service is working on another revision to the existing recovery plan.

The Santa Cruz long-toed salamander spends most of its life underground in small mammal burrows and along the root systems of plants in upland chaparral and woodland areas of coast live oak (*Quercus agrifolia)* or Monterey pine (*Pinus radiata*) as well as riparian strips of arroyo willows (*Salix lasiolepis*). These areas are desirable because they are protected from heat and the drying rays of the sun (Reed 1979, 1981). The breeding ponds are usually shallow, ephemeral, freshwater ponds. The breeding ponds at the Seascape, Larkin Valley, Calabasas, and Buena Vista sites are man-made. The extent of the upland habitat adjacent to the ponds varies from a ring of riparian vegetation on the perimeter of the pond to as far as a mile or more out from the pond (Ruth and Tollestrup 1973). However, examination of all currently available studies on the Santa Cruz long-toed salamander reveals that adult salamanders typically do not move more than 0.6 mile (straight line distance) from a breeding site.

Adult Santa Cruz long-toed salamanders leave their upland chaparral and woodland summer retreats with the onset of the rainy season in mid- to late-November or December and begin their annual nocturnal migration to the breeding pond (Anderson 1960). Adult salamanders migrate primarily on nights of rain, mist, or heavy fog (Anderson 1960, 1967; Ruth and Tollestrup 1973; Reed 1979, 1981). They arrive at the breeding pond from November through March, with most arriving in January and February (Anderson 1967, Reed 1979, Ruth 1988b). Peak breeding occurs during January and February because earlier rains are usually insufficient to fill the breeding ponds (Anderson 1967). Adult salamanders may skip breeding for one or more seasons if no surface water is present during drier years (Russell and Anderson 1956). Female Santa Cruz long-toed salamanders have specialized and selective egg-laying habits. Eggs are laid singly on submerged stalks of spike rush (*Eleocharis* sp.) or other vegetation about one inch apart (Anderson 1960, 1967). Free floating, unattached, and clustered eggs have also been observed (Reed 1981). Each female lays about 300 (range 215 to 411) eggs per year (Anderson 1967). After courtship and egg laying, most adult salamanders leave the pond in March or April and return to the same general areas where they spent the previous summer. Some adults may remain in the vicinity of the breeding site for a year or more before returning to more distant terrestrial retreats (Ruth 1988b). The eggs and the subsequent larvae are left unattended by the adults.

According to Reed (1979, 1981) and Ruth (1988a), eggs usually hatch after 15 to 30 days and enter the aquatic larval stage. The exact amount of time for development depends on water temperature (Anderson 1972). Larvae may metamorphose in a relatively short period of time if the pond environment becomes unsuitable (i.e., dries up, limited food source) for continued larval growth. However, a complex of factors determines the timing of metamorphosis in ambystomatid salamanders (Werner 1986, Wilbur and Collins 1973, Wilbur 1976, Smith-Gill and Berven 1979). Metamorphosis typically occurs from early May to mid-August (Anderson 1967, Reed 1979, 1981; Ruth 1988a). In closely related *A. talpoideum*, metamorphosis can be induced in the laboratory by starvation, pollution of the water, increased water temperatures, or drying of the aquatic habitat (Shoop 1960). If water is available to the larvae for a longer period of time, remaining in the pond may be advantageous for the juveniles. A larger body size at metamorphosis increases resistance to desiccation, makes the individual less vulnerable to predation, and increases the size range of food items that can be eaten (Werner 1986). As the pond begins to dry, the juvenile salamanders move at night and seek underground refuge at or near the pond (Reed 1979, 1981). During the next rainy seasons, these recently metamorphosed juveniles disperse farther away from the pond, not returning until they reach sexual maturity at two to three years (Ruth 1988a).

Adults of closely related *A. m. sigillatum* and *A. m. krausei* are known to have lived over six years in captivity (Snider and Bowler 1992) and ten years in the wild (Russell *et al.* 1995), respectively. An adult *A. m. croceum* confiscated by law enforcement officials was kept in captivity for eight years until its death (Stephen B. Ruth, Science Research and Consulting Services, Marina, California, *in litt.*). Thus, Santa Cruz long-toed salamanders are probably long-lived creatures, possibly living for a decade or more.

Santa Cruz long-toed salamanders are vulnerable to several predators including opossums (*Didelphis virginiana*), striped skunks (*Mephitis mephitis*), and ringneck snakes (*Diadophis punctatus*) (Reed 1979), raccoons (*Procyon lotor*), large California tiger salamanders *(A. californiense*), coast garter snakes (*Thamnophis atratus*), western terrestrial garter snakes (*T. elegans*), and common garter snakes (*T. sirtalis*). Larval *A. m. croceum* are parasitized by a digenetic trematode (Plagiorchiidae) which causes the creation of supernumerary limbs as well as other limb deformities (Sessions and Ruth 1990).

Foraging Ecology: The larvae of Santa Cruz long-toed salamanders subsist largely on aquatic invertebrates, other larval amphibians such as *Hyla regilla*, and conspecifics. Adults often forage for invertebrates, especially isopods (Anderson 1968), on the surface in and around breeding sites during the rainy season.

<u>Historic</u> <u>and</u> <u>Current</u> <u>Distribution</u>: Breeding of Santa Cruz long-toed salamanders have been documented at Valencia Lagoon, Ellicott pond, Seascape pond, Calabasas pond, Buena Vista pond, Green pond, and Rancho Road pond in Santa Cruz County and at McClusky Slough, Moro Cojo Slough, Bennett Slough, and Zmudowski pond in Monterey County. However, many of these sites have not been surveyed recently and may no longer support breeding populations.

Juvenile Santa Cruz long-toed salamanders have also been found at several other sites in Santa Cruz and Monterey counties (California Natural Diversity Data Base, unpubl. data). Whether any of these juveniles represent undiscovered breeding populations or merely wandering individuals from marginal or currently identified breeding habitats is unknown. Further discovery of new breeding sites is likely given the amount of privately owned habitat in the region that has not been surveyed for Santa Cruz long-toed salamanders.

Reasons for Decline and Threats to Survival: The very restricted and disjunct distribution of the Santa Cruz long-toed salamander has made the species particularly susceptible to population declines resulting from both human-associated and natural factors, including habitat loss and degradation, predation by introduced and native organisms, and weather conditions. Highway construction, urban and agricultural development, siltation, vehicles, exotic fish and vegetation, and saltwater intrusion are some of the perturbations affecting Santa Cruz long-toed salamander habitat. Runoff from adjacent agricultural and urban areas into many of the breeding ponds of the Santa Cruz long-toed salamander is a potential threat. Santa Cruz long-toed salamanders occur in several impaired water bodies.

California Freshwater Shrimp (*Syncaris pacifica*)

Species Description and Life History: The California freshwater shrimp was listed as endangered in 1988 (53 **FR** 43889). The California freshwater shrimp is a decapod crustacean of the family Atyidae. Females are generally larger and deeper bodied than males. Shrimp coloration is quite variable. Male shrimp are translucent to nearly transparent, with small surface and internal chromatophores (color-producing cells) clustered in a pattern to help disrupt their body outline and to maximize the illusion that they are submerged, decaying vegetation. Eng (1981) observed that the coloration of female range from a dark brown to a purple color. In some females, a broad tan dorsal band also may be present. Females may change rapidly from this very dark cryptic color to opaque with diffuse chromatophores, a distinctly different coloration. Undisturbed shrimp move slowly and are virtually invisible on submerged leaf and twig substrates, and among the fine, exposed, live roots of trees along undercut stream banks. Atyid shrimps can be separated from others based on the lengths of chelae (pincer-like claws) and presence of terminal setae (bristles) at the tips of the first and second chelae (Eng 1981, Pennak 1989). The presence of a short supraorbital (above the eye) spine on the carapace (body) and the angled articulation of the second chelae with the carpus (wrist) separate the California freshwater shrimp from other shrimp found in California.

Shrimp have been found only in low elevation (less than 16 meters) and low gradient (generally less than 1 percent) streams. With the exception of Yulupa Creek, shrimp have not been found in stream reaches with boulder and bedrock bottoms. In fact, high velocities and turbulent flows in such reaches may hinder upstream movement of shrimp. The California freshwater shrimp has evolved to survive a broad range of stream and water temperature conditions characteristic of small, perennial coastal streams. The shrimp appears to be able to tolerate warm water temperatures (greater than 23 degrees Celsius, 73 degrees Fahrenheit) and low flow conditions that are detrimental or fatal to native salmonids.

The shrimp are generally found in stream reaches where banks are structurally diverse with undercut banks, exposed roots, overhanging woody debris, or overhanging vegetation (Eng 1981, Serpa 1986 and 1991). Excellent habitat conditions for the shrimp involve streams 30 to 90 centimeters (cm) in depth with exposed live roots (e.g., alder and willow trees) along undercut banks (greater than 15 cm) with overhanging stream vegetation and vines (Serpa 1991). During the winter, the shrimp is found in undercut banks with exposed fine root systems or dense, overhanging vegetation. Such microhabitats may provide velocity refugia as well as some protection from high suspended sediment concentrations typically associated with high stream flows.

Habitat preferences apparently change during late-spring and summer months. Eng (1981) rarely found shrimp beneath undercut banks in the summer; submerged leafy branches were the preferred summer habitat. Highest concentrations of shrimp were in reaches with adjacent vegetation comprised of stinging nettles (*Urtica* sp.) grasses, vine maple (Serpa *in litt.* 1994 suspects periwinkle was misidentified as vine maple), and mint (*Mentha* sp.). None were caught from cattails (*Typha* sp.), cottonwood (*Populus fremontii*), or California laurel (*Umbellularia californica*). Serpa also noted that populations of shrimp were proportionately correlated with the quality of summer habitat provided by trailing terrestrial vegetation. However, during summer low flows, shrimp have been found in apparently poor habitat such as isolated pools with minimal cover. In such streams, opaque waters may allow shrimp to escape predation and persist in open pools despite the lack of cover (Serpa 1991).

Although largely absent from existing streams, large, complex organic debris dams may have been prevalent in streams supporting shrimp populations. These structures may have been important feeding and refugial sites for the shrimp. Such structures are known to collect detrital material (shrimp food) as well as leaf litter, which can be later broken down by microbial activity and invertebrates to finer, detrital material (Triska *et al.* 1982). In addition, debris dams may offer refugia during high flow events and reduce displacement of invertebrates (Covich *et al.* 1991).

Adult females produce relatively few eggs, generally, 50 to 120 (Hedgpeth 1968, Eng 1981). The eggs adhere to the pleopods (swimming legs on the abdomen) where they are protected and cared for during the winter incubation. The California freshwater shrimp is one of the few atyid species that breeds during the winter period.

California freshwater shrimp are preyed upon by fish, western pond turtles, salamanders, and newts, which are probably present throughout many of the streams. Invertebrate predators may include water scorpions, predaceous diving beetles, and dragonfly and damselfly nymphs.

Foraging Ecology: Atyid shrimps can be described as collectors feeding upon fine particulate organic matter. The food sources may range from fecal material produced by shredders (a functional group that feeds on coarse particulate organic matter), organic fines produced by physical abrasion and microbial maceration, senescent periphytic algae, planktonic algae, aquatic macrophyte plant fragments, zooplankton, and particles formed by the flocculation of dissolved organic matter. Shrimp observed on pool bottoms, submerged twigs, and vegetation seemed to feed on fine particulate matter (Eng 1981). Atyid shrimp use their claws to scrape and sweep detritus and small organisms from substrates. Much of the material ingested is probably indigestible cellulose. Shrimp may use visual, tactile, or chemical cues in foraging activities (USDI-FWS 1997a).

Historic and Current Distribution: Distribution of the shrimp is assumed, prior to human disturbances, to have been common in low elevation, perennial freshwater streams within Marin, Sonoma, and Napa counties. Today, the shrimp is found in 16 stream segments within these counties. The distribution of the shrimp can be separated into four general geographic regions: 1) tributary streams in the lower Russian River drainage which flows westward into the Pacific Ocean, 2) coastal streams flowing westward directly into the Pacific Ocean, 3) streams draining into a small coastal-embayment (Tomales Bay), and 4) streams flowing southward into northern San Pablo Bay. Many of these streams contain shrimp populations that are now isolated from each other. Distribution of shrimp populations within streams is not expected to be static because of habitat changes by natural or anthropogenic (man made) forces. Distribution within streams may expand and contact depending upon existing conditions. Gradual removal of unnatural barriers to shrimp dispersal and restoration of natural habitat conditions are expected to expand the distribution of shrimp beyond its existing occurrence.

Reasons for Decline and Threats to Survival: Existing populations of the California freshwater shrimp are threatened by introduced fish, deterioration or loss of habitat resulting from water diversion, impoundments, livestock and dairy activities, agricultural activities and developments, flood control activities, gravel mining, timber harvesting, migration barriers, and water pollution.

Fairy Shrimp (Including Conservancy, Longhorn, Riverside, San Diego, and Vernal Pool Fairy Shrimp)

Species Description and Life History: The Riverside fairy shrimp (*Streptocephalus woottoni*) was listed as endangered in 1993 (58 **FR** 41391). The vernal pool fairy shrimp (*Brachinecta lynchi*), conservancy fairy shrimp (*B. conservatio*), longhorn fairy shrimp (*B. longiatenna*), were listed as threatened (vernal pool) or endangered (all others) in 1994 (59 **FR** 48153). The San Diego fairy shrimp (*B. sandiegonensis*) was listed as endangered in 1997 (62 **FR** 4925). Further details on the life history and ecology of the fairy shrimp are provided by Eng *et al.* (1990) and Simovich *et al.* (1992)

Fairy shrimp have a delicate elongate body, large stalked compound eyes, no carapace, and 11 pairs of swimming legs. It swims or glides gracefully upside down by means of complex beating movements of the legs that pass in a wave-like anterior to posterior direction. The females carry the eggs in an oval or elongate ventral brood sac. The eggs are either dropped to the pool bottom or remain in the brood sac until the female dies and sinks. The "resting" or "summer" eggs are capable of withstanding heat, cold, and prolonged desiccation. When the pools fill in the same or subsequent seasons, some, but not all, of the eggs may hatch. The egg bank in the soil may consist of eggs from several years of breeding (Donald 1983). The eggs hatch when the vernal pools fill with rainwater. The early stages of the fairy shrimp develop rapidly into adults. These non-dormant populations often disappear early in the season long before the vernal pools dry up.

The primary historic dispersal method for the fairy shrimp likely was large scale flooding resulting from winter and spring rains which allowed the animals to colonize different individual vernal pools and other vernal pool complexes (J. King, pers. comm., 1995). This dispersal currently is non-functional due to the construction of dams, levees, and other flood control measures, and widespread urbanization within significant portions of the range of this species. Waterfowl and shorebirds likely are now the primary dispersal agents for fairy shrimp (Brusca, in. litt., 1992, King, in. litt., 1992, Simovich, in. litt., 1992). The eggs of these crustaceans are either ingested (Krapu 1974, Swanson *et al.* 1974, Driver 1981, Ahl 1991) and/or adhere to the legs and feathers where they are transported to new habitats.

Fairy shrimp are restricted to vernal pools/swales, an ephemeral freshwater habitat in California that forms in areas with Mediterranean climates where slight depressions become seasonally saturated or inundated following fall and winter rains. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland and Jain 1988). In southern California, these pools/swales typically form on mesa tops or valley floors and are surrounded by very low hills, usually referred to as mima mounds (Zedler 1987). None of these listed branchiopods are known to occur in permanent bodies of water, riverine waters, or marine waters. Water remains in these pools/swales for a few months at a time, due to an impervious layer such as hardpan, claypan, or basalt beneath the soil surface.

The San Diego fairy shrimp is a habitat specialist found in small, shallow vernal pools, which range in depth from 5 to 30 centimeters (cm) (2 to 12 in.) and in water temperature from 10 to 20 degrees Celsius (C)(50 to 68 degrees Fahrenheit (F)) (Simovich and Fugate 1992, Hathaway and Simovich undated). Water chemistry is one of the most important factors in determining the distribution of fairy shrimp (Belk 1977, Branchiopod Research Group 1996). The San Diego fairy shrimp appears to be sensitive to high water temperatures (Branchiopod Research Group 1996). Hathaway and Simovich (undated) presented data indicating that pools located in the inland mountain and desert regions may be too cool (below 5 degrees C (41 degrees F)) or too warm (above 30 degrees C (86 degrees F)) for this species. Adult San Diego fairy shrimp are usually observed from January to March; however, in years with early or late rainfall, the hatching period may be extended.

The vernal pool fairy shrimp inhabits vernal pools with clear to tea-colored water, most commonly in grass or mud-bottomed swales, or basalt flow depression pools in unplowed grasslands, but one population occurs in sandstone rock outcrops and another population in alkaline vernal pools. The vernal pool fairy shrimp has been collected from early December to early May. It can mature quickly, allowing populations to persist in short-lived shallow pools (Simovich *et al.* 1992).

The genetic characteristics of these species, as well as ecological conditions, such as watershed continuity, indicate that populations of these animals are defined by pool complexes rather than by individual vernal pools (Fugate 1992; J. King, pers. comm., 1995). Therefore, the most accurate indication of the distribution and abundance of these species is the number of inhabited vernal pool complexes. Individual vernal pools occupied by these species are most appropriately referred to as subpopulations. The pools and, in some cases, pool complexes supporting these species are usually small.

Foraging Ecology: Fairy shrimp feed on algae, bacteria, protozoa, rotifers, and bits of detritus.

<u>Historic</u> <u>and</u> <u>Current</u> <u>Distribution</u>: These crustaceans are restricted to vernal pools and swales in California. Holland (1978) estimated that between 67 and 88 percent of the area within the Central Valley of California which once supported vernal pools had been destroyed by 1973. However, an analysis of this report by the Service revealed apparent arithmetic errors which resulted in a determination that a historic loss between 60 and 85 percent may be more accurate. Regardless, in the ensuing 23 years, threats to this habitat type have continued and resulted in a substantial amount of vernal pool habitat being converted for human uses in spite of Federal regulations implemented to protect wetlands. For example, the Corps' Sacramento District has authorized the filling of 189 hectares (467 acres) of wetlands between 1987 and 1992 pursuant to Nationwide Permit 26 (USDI-FWS 1992). The Service estimates that a majority of these wetland losses within the Central Valley involved vernal pools. Current rapid urbanization and agricultural conversion throughout the ranges of the species continue to pose the most severe threats to the continued existence of the fairy shrimp. The Corps' Sacramento District has several thousand vernal pools under its jurisdiction (Coe 1988), which includes most of the known populations of the vernal pool fairy shrimp. It is estimated that within 20 years 60 to 70 percent of these pools will be destroyed by human activities (Coe 1988).

Conservancy Fairy Shrimp (Endangered): The Conservancy fairy shrimp inhabits vernal pools with highly turbid water. The species is known from six disjunct populations: Vina Plains, north of Chico, Tehama County; south of Chico, Butte County; Jepson Prairie, Solano County; Sacramento National Wildlife Refuge, Glenn County; near Haystack Mountain northeast of Merced in Merced County; and the Lockwood Valley of northern Ventura County.

Longhorn Fairy Shrimp (Endangered): The longhorn fairy shrimp inhabits clear to turbid grass-bottomed vernal pools in grasslands and clear-water pools in sandstone depressions. This species is known only from four disjunct populations along the eastern margin of the central coast range from Concord, Contra Costa County south to Soda Lake in San Luis Obispo County: the Kellogg Creek watershed, the Altamont Pass area, the western and northern boundaries of Soda Lake on the Carrizo Plain, and Kesterson National Wildlife Refuge in the San Joaquin Valley.

Riverside Fairy Shrimp (Endangered): The Riverside fairy shrimp has a restricted distribution and is known only from vernal pools in the Santa Rosa Plateau, Skunk Hollow, and several small scattered pools in Riverside County; from El Toro Marine Cavalry Air Station and Saddleback Meadows in Orange County; from Otay Mesa, Camp Pendleton, and Miramar Naval Air Station in San Diego County; from the Moorpark area of Ventura County; and the Canyon Country/Santa Clarita area of Los Angeles County.

San Diego Fairy Shrimp (Endangered): The San Diego fairy shrimp belongs to the Family Branchinectidae. These fairy shrimp have a very restricted distribution and are only known from vernal pools in southwestern coastal California and extreme northwestern Baja California, Mexico. Less than 81 hectares (ha) (200 acres (ac)) of habitat likely remains.

No individuals have been found in riverine waters, marine waters, or other permanent bodies of water. All known localities are below 700 meters (m) (2,300 feet (ft)) and within 65 kilometers (km) (40 miles (mi)) of the Pacific Ocean, from Santa Barbara County south to northwestern Baja California. The majority of the vernal pools in this region, including many which likely served as habitat for the species, were destroyed prior to 1990. Between 1979 and 1986, approximately 68 percent of the privately owned vernal pools under the City of San Diego's jurisdiction were destroyed (Wier and Bauder 1991).

Vernal Pool Fairy Shrimp (Threatened): The vernal pool fairy shrimp inhabits vernal pools with clear to tea-colored water, most commonly in grass or mud bottomed swales, or basalt flow depression pools in unplowed grasslands. The vernal pool fairy shrimp has been collected from early December to early May. The vernal pool fairy shrimp is known from 34 populations extending from Stillwater Plain in Shasta County through most of the length of the Central Valley to Pixley in Tulare County, and along the central coast range from northern Solano County to Pinnacles in San Benito County (Eng *et al.* 1990, Fugate 1992, Sugnet and Associates 1993). In wet years, Fort Hunter Liggett, in southern Monterey County, supports hundreds of pools containing this species. Camp Roberts, which straddles the Monterey-San Luis Obispo county line, also contains pools with vernal pool fairy shrimp. Four additional, disjunct populations exist: one near Soda Lake in San Luis Obispo County; one in the mountain grasslands of northern Santa Barbara County; one on the Santa Rosa Plateau in Riverside County, and one near Rancho California in Riverside County. Three of these four isolated populations each contain only a single pool known to be occupied by the vernal pool fairy shrimp.
Conservancy Fairy Shrimp (Endangered): The Conservancy fairy shrimp inhabits vernal pools with highly turbid water. The species is known from six disjunct populations: Vina Plains, north of Chico, Tehama County; south of Chico, Butte County; Jepson Prairie, Solano County; Sacramento National Wildlife Refuge, Glenn County; near Haystack Mountain northeast of Merced in Merced County; and the Lockewood Valley of northern Ventura County.

<u>Reasons for Decline and Threats to Survival</u>: Fairy shrimp are imperiled by a variety of human-caused activities, primarily urban development, water supply/flood control projects, and land conversion for agricultural use. Habitat loss occurs from direct destruction and modification

of pools due to filling, grading, discing, leveling, and other activities, as well as modification of surrounding uplands which alters vernal pool watersheds. Other activities which adversely affect these species include off-road vehicle use, certain mosquito abatement measures, and pesticide/herbicide use, alterations of vernal pool hydrology, fertilizer and pesticide contamination, activity, invasions of aggressive non-native plants, gravel mining, and contaminated stormwater runoff.

In addition to direct habitat loss, the vernal pool habitat for the vernal pool fairy shrimp also has been and continues to be highly fragmented throughout their ranges due to conversion of natural habitat for urban and agricultural uses. This fragmentation results in small isolated vernal pool fairy shrimp populations. Ecological theory predicts that such populations will be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance (Gilpin and Soule 1986, Goodman 1987a,b). Should an extirpation event occur in a population that has been fragmented, the opportunities for recolonization would be greatly reduced due to physical (geographical) isolation from other (source) populations.

Only a small proportion of the habitat of these species is protected from these threats. State and local laws and regulations have not been passed to protect these species, and other regulatory mechanisms necessary for the conservation of the habitat of these species have proven ineffective.

Shasta Crayfish (*Pacifastacus fortis*)

Species Description and Life History: The Shasta crayfish was federally listed as endangered in 1988 (53 **FR** 190). A detailed account of the taxonomy, ecology, and biology of the Shasta crayfish is presented in the Draft Recovery Plan for this species (USDI-FWS 1997). Supplemental information is provided below.

The Shasta crayfish occurs in cool, clear, spring-fed lakes, rivers and streams, usually at or near a spring inflow source, where waters show relatively little annual fluctuation in temperature and remain cool during the summer. Most Shasta crayfish are found in still and slowly to moderately flowing waters. Although Shasta crayfish have been observed in groups under large rocks situated on clean, firm sand or gravel substrates (Bouchard, 1978; Eng and Daniels, 1982), they also have been observed on a fine, probably organic, material 1-3 centimeters thick on the bottom of Crystal Lake. Shasta crayfish is most abundant where plants are absent. The most important habitat requirement appears to be the presence of adequate volcanic rock rubble to provide escape cover from predators.

Foraging Ecology: Although the food habits of the Shasta crayfish are not well known, the morphology of the mouthparts suggests that the species relies primarily on predation, browsing on encrusting organisms, and grazing on detritus to obtain food. Aquatic invertebrates and dead fish probably provide food for the Shasta crayfish. Feeding and mating takes place at night.

Historic and Current Distribution: The Shasta crayfish is found only in Shasta County, California, in the Pit River drainage and two tributary systems, Fall River and Hat Creek subdrainages. In the Fall River subdrainage, populations occur in the Tule and Fall Rivers, Big Lake, Spring, Squaw and Lava Creeks, and in Crystal and Rainbow Springs. An additional population occurs in Sucker Spring Creek, a tributary of the Pit River just downstream from Powerhouse I, which lies between the two subdrainages (Bouchard, 1978; Eng and Daniels, 1982). In the Hat Creek subdrainage, historically, populations have been found in Lost Creek, Crystal, Baum, and Rising River Lakes. The populations in Lake Britton, Burney, Clark, Kosk, Goose, Lost, and Rock Creeks were extirpated prior to 1974 (Bouchard, 1977). Since 1978 the Shasta crayfish has been extirpated from Crystal Lake, Baum Lake and Spring Creek near its confluence with the Pit River, Rising River and Sucker Spring Creek near Pit Powerhouse I (McGriff, personal communication, 1986).

Reasons for Decline and Threats to Survival: The invasion of non-native crayfish species, in particular the signal crayfish, is the single largest threat to the continued existence of the Shasta crayfish. Human activities (such as levee repairs) in the historic range of the Shasta crayfish caused increased siltation, covering the volcanic rubble and reducing the amount of suitable habitat for the species. Two entire populations have been extirpated since 1978.

Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

Species Description and Life History: The vernal pool tadpole shrimp was listed as endangered on September 19, 1994 (59 **FR** 48153). Further details on the life history and ecology of the fairy shrimp are provided by Eng *et al.* (1990) and Simovich *et al.* (1992).

The vernal pool tadpole shrimp has dorsal compound eyes, a large shield-like carapace that covers most of the body, and a pair of long cercopods at the end of the last abdominal segment (Linder 1952, Longhurst 1955, Pennak 1989). It is primarily a benthic animal that swims with its legs down. Tadpole shrimp climb or scramble over objects, as well as move along or in bottom sediments. The females deposit their eggs on vegetation and other objects on the pool bottom. Tadpole shrimp populations pass the dry summer months as diapaused eggs in pool sediments. Some of the eggs hatch as the vernal pools are filled with rainwater in the fall and winter of subsequent seasons.

The life history of the vernal pool tadpole shrimp is linked to the phenology of its vernal pool habitat. After winter rainwater fills the pools, the populations are reestablished from diapaused eggs which lie dormant in the dry pool sediments (Lanaway 1974, Ahl 1991). Ahl (1991) found that eggs in one pool hatched within three weeks of inundation and sexual maturation was reached in another three to four weeks. The eggs are sticky and readily adhere to plant matter and sediment particles (Simovich *et al.* 1992). A portion of the eggs hatch immediately and the rest enter diapause and remain in the soil to hatch during later rainy seasons (Ahl 1991). The vernal pool tadpole shrimp matures slowly and is a long-lived species (Ahl 1991). Adults are often present and reproductive until the pools dry up in the spring (Ahl 1991, Simovich *et al.* 1992).

The genetic characteristics of this species, as well as ecological conditions, such as watershed continuity, indicate that populations of these animals are defined by pool complexes rather than by individual vernal pools (Fugate 1992; J. King, pers. comm., 1995). Therefore, the most accurate indication of the distribution and abundance of the species is the number of inhabited vernal pool complexes. Individual vernal pools occupied by the species are most appropriately referred to as subpopulations. The pools and, in some cases, pool complexes supporting these species are usually small.

The primary historic dispersal method for the vernal pool tadpole shrimp and likely was large scale flooding resulting from winter and spring rains which allowed the animals to colonize different individual vernal pools and other vernal pool complexes (J. King, pers. comm., 1995). This dispersal currently is non-functional due to the construction of dams, levees, and other flood control measures, and widespread urbanization within significant portions of the range of this species. Waterfowl and shorebirds likely are now the primary dispersal agents for vernal pool tadpole shrimp (Brusca, in. litt., 1992, King, in. litt., 1992, Simovich, in. litt., 1992). The eggs of these crustaceans are either ingested (Krapu 1974, Swanson *et al.* 1974, Driver 1981, Ahl 1991) and/or adhere to the legs and feathers where they are transported to new habitats.

Vernal pool tadpole shrimp are restricted to vernal pools/swales, an ephemeral freshwater habitat in California that forms in areas with Mediterranean climates where slight depressions become seasonally saturated or inundated following fall and winter rains. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland and Jain 1988). Tadpole shrimp are not known to occur in permanent bodies of water, riverine waters, or marine waters. Water remains in these pools/swales for a few months at a time, due to an impervious layer such as hardpan, claypan, or basalt beneath the soil surface.

Foraging Ecology: The diet of tadpole shrimp consists of organic detritus and living organisms, such as fairy shrimp and other invertebrates (Pennak 1989).

<u>Historic and Current Distribution</u>: Holland (1978) estimated that between 67 and 88 percent of the area within the Central Valley of California which once supported vernal pools had been destroyed by 1973. However, an analysis of this report by the Service revealed apparent arithmetic errors which resulted in a determination that a historic loss between 60 and 85 percent may be more accurate. Regardless, in the ensuing 23 years, threats to this habitat type have continued and resulted in a substantial amount of vernal pool habitat being converted for human uses in spite of Federal regulations implemented to protect wetlands. For example, the Corps' Sacramento District has authorized the filling of 189 hectares (467 acres) of wetlands between 1987 and 1992 pursuant to Nationwide Permit 26 (USDI-FWS 1992). The Service estimates that a majority of these wetland losses within the Central Valley involved vernal pools, the endemic habitat of the vernal pool tadpole shrimp and vernal pool fairy shrimp. Current rapid urbanization and agricultural conversion throughout the ranges of these two species continue to pose the most severe threats to the continued existence of the vernal pool tadpole shrimp and vernal pool fairy shrimp. The Corps' Sacramento District has several thousand vernal pools under

its jurisdiction (Coe 1988), which includes most of the known populations of these listed species. It is estimated that within 20 years 60 to 70 percent of these pools will be destroyed by human activities (Coe 1988).

The vernal pool tadpole shrimp is known from 19 populations in the Central Valley, ranging from east of Redding in Shasta County south to Fresno County, and from a single vernal pool complex located on the San Francisco Bay National Wildlife Refuge in Alameda County. It inhabits vernal pools containing clear to highly turbid water, ranging in size from 5 square meters (54 square feet) in the Mather Air Force Base area of Sacramento County, to the 36-hectare (89-acre) Olcott Lake at Jepson Prairie in Solano County. Vernal pools at Jepson Prairie and Vina Plains (Tehama Co.) have a neutral pH, and very low conductivity, total dissolved solids, and alkalinity (Barclay and Knight 1984, Eng *et al.* 1990). These pools are located most commonly in grass-bottomed swales of grasslands in old alluvial soils underlain by hardpan or in mud-bottomed claypan pools containing highly turbid water.

<u>Reasons for Decline and Threats to Survival</u>: Fairy shrimp are imperiled by a variety of human-caused activities, primarily urban development, water supply/flood control projects, and land conversion for agricultural use. Habitat loss occurs from direct destruction and modification of pools due to filling, grading, discing, leveling, and other activities, as well as modification of surrounding uplands which alters vernal pool watersheds. Other activities which adversely affect these species include off-road vehicle use, certain mosquito abatement measures, and pesticide/herbicide use, alterations of vernal pool hydrology, fertilizer and pesticide contamination, activity, invasions of aggressive non-native plants, gravel mining, and contaminated stormwater runoff.

In addition to direct habitat loss, the vernal pool habitat for the vernal pool fairy shrimp also has been and continues to be highly fragmented throughout their ranges due to conversion of natural habitat for urban and agricultural uses. This fragmentation results in small isolated vernal pool fairy shrimp populations. Ecological theory predicts that such populations will be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance (Gilpin and Soule 1986, Goodman 1987a,b). Should an extirpation event occur in a population that has been fragmented, the opportunities for recolonization would be greatly reduced due to physical (geographical) isolation from other (source) populations.

Only a small proportion of the habitat of these species is protected from these threats. State and local laws and regulations have not been passed to protect these species, and other regulatory mechanisms necessary for the conservation of the habitat of these species have proven ineffective.

Southern Sea Otter (*Enhydra lutris nereis*)

<u>Species Description and Life History</u>: The southern sea otter was listed as threatened in 1977 (42 **FR** 2968). Sea otters are one of the largest members of the family Mustelidae. Adult males are larger than adult females. Standard lengths of adult males and females average 51 inches and 47

inches, respectively, with males averaging 64 pounds and females averaging 44 pounds. Pups weigh between 3 to 5 pounds at birth. This account is based on information in Bonnell *et al.* 1983, and Costa & Kooyman 1980, 1982.

Unlike most other marine mammals, sea otters have very little subcutaneous fat, depending instead on their clean, dense, water-resistant fur for insulation against the cold. Contamination of the fur by oily substances can destroy the insulating properties of the fur and lead to hypothermia and death.

Although mating and pupping take place throughout the year, a peak period of pupping occurs from January to March. The general yearly reproductive pattern consists of a winter-spring pupping season and a summer-fall breeding season. Males may reach sexual maturity at about 5 years of age; however males probably do not establish territories or actively participate in breeding for some time after reaching puberty. Preliminary observations indicate that female southern sea otters may also reach sexual maturity between 4 and 5 years of age. Current estimates indicate that most adult females give birth to one pup each year, with a reproductive cycle ranging from 11-14 months in length. Gestation periods have been estimated at 4-6 months. Pup dependency periods in California range from 5-8 months. There appears to be a potential for considerable individual variation and plasticity with respect to the temporal phases of the reproductive cycle.

Foraging Ecology: Otters forage in both rocky and soft-sediment communities as well as in the kelp understory and canopy. Foraging occurs in both the intertidal and subtidal zones, but seldom deeper than 25 meters. The diet of sea otters is almost exclusively of a variety of nearshore macroinvertebrates. Prey items include abalones, rock crabs, sea urchins, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, and chitons. Sea otter teeth are adapted for crushing hard-shelled macro-invertebrates.

<u>Historic</u> <u>and</u> <u>Current</u> <u>Distribution</u>: Southern sea otters inhabit a narrow zone of shallow, littoral waters along the counties of Santa Cruz, Monterey, San Luis Obispo, and Santa Barbara. A reintroduced colony is located on San Nicolas Island, Ventura county. The majority of otters remain within 1.2 miles of shore, inshore of the outer kelp bed edge, which generally corresponds to the 60-foot (10 fathom) depth curve. However, some individuals may be found further off shore to the 30 fathom depth curve. Foraging activity is generally restricted to water depth of 90 feet (15 fathoms) or less. Southern sea otters are primarily associated with subtidal habitats characterized by rocky, creviced substrate, although they are also found in sandy substrate areas. Sea otter density within most of the range (with the exception of the north and south population fronts) is related to substrate type; rocky bottom habitats support an average density of 13 otters per square mile whereas sandy bottom areas support an average of 2 otters per square mile.

The number of southern sea otters increased to 2,377 in 1995, but has since declined to 2,229 in 1997. The Service is currently assessing whether this lower count represents an actual decline or an artifact of survey technique and a redistribution of southern sea otters.

Reasons for Decline and Threats to Survival: Threats to the survival of the southern sea otter include reduced population size, increased tanker traffic, oil spills, drowning in commercial fishing nets, municipal pollution, and increased harassment caused by increased use of near-shore areas. Some evidence suggests that the decline in population growth rate is due to infectious disease.

Elevated levels of heavy metals, chlorinated hydrocarbons, PCB's, and petroleum hydrocarbons were found in sea otters in the past. Chemical contamination may also reduce suitable foraging areas (USDI-FWS 1981).

Elevated levels of mercury are known to occur in Elkhorn Slough, a tributary to Monterey Bay. Elkhorn Slough is impacted by upstream discharges of mercury. Livers collected from sea otters found dead at this location had a maximum mercury concentration of (60mg/kg) (Mark Stephenson pers comm 1998). Wren, 1986 suggested normal mercury concentrations in river otter livers were 4 mg/kg (ppm). O'Conner and Nielsen (1981) found that length of exposure was a better predictor of tissue residue level than dose in otters but higher doses produced an earlier onset of clinical signs. Acute mercury poisoning in mammals is primarily manifested in Central Nervous System damage, sensory and motor deficits, and behavioral impairment. Animals initially become anorexic and lethargic. A dose of 0.09 mg/kg body weight (2 ppm in diet) for 181 days was enough to produce anorexia and ataxia in two of three otters (Lutra candensis. Associated liver residues were 32.6 mg/kg (O'Conner and Nielsen 1981). Muscle ataxia, motor control deficits, and visual impairment develop as toxicity progresses with convulsions preceding death. River otters fed 8 ppm died within a mean time of 54 days. Associated liver concentrations were 32.3 mg/kg (ppm) (O'Conner and Nielsen 1981). Smaller carnivores are more sensitive to methylmercury toxicity than larger species as reflected in shorter times of onset of toxic signs and time to death.

DIRECT AND INDIRECT EFFECTS OF THE PROPOSED ACTION

For the purposes of this opinion the Services have conducted their effects analysis based on the potential for the numeric criteria to result in effects to the aquatic ecosystem and the species that are dependent on its function for their survival and recovery. While 126 priority pollutants are addressed within the CTR, the Services have focused upon the numeric criteria for selenium, mercury, pentachlorophenol, cadmium and formula based criteria for metals on a dissolved basis as the most problematic for listed species and critical habitat. The Services have prepared this analysis of criteria for priority pollutants based on: (1) the adequacy of the proposed aquatic life criteria, including the necessity of wildlife criteria where aquatic life criteria are not sufficiently protective of wildlife; (2) the toxic effects to listed species or surrogates which may occur at proposed criteria concentrations; (3) the bioaccumulative nature of the priority pollutants at issue; and (4) the potential for interactive effects of pollutants at the proposed criteria concentrations. In some cases, such as mercury, if the aquatic life criteria were not protective and the human health criteria were lower, the adequacy of the human health numeric criteria to protect aquatic life was also considered.