ATTACHMENT III:

BASELINE STATUS AND CUMULATIVE EFFECTS FOR THE SAN FRANCISCO BAY LISTED SPECIES

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1: ALAMEDAWHIPSNAKE

1.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the Alameda whipsnake (*Masticophis lateralis euryxanthus*) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of urban and suburban development, construction and recreational use of paved and unpaved roads, fire suppression, inadequacy of regulatory mechanisms, and inappropriate grazing practices. Many of these factors are linked or act synergistically and create complex consequences for the Alameda whipsnake. For example, suburban developments often impede dispersal and movement of species, as well as contribute to habitat loss and degradation. Housing developments can increase public presence in local parks and trails, as well as increase the number of domestic and feral cats. These potential impacts further degrade Alameda whipsnake suitable habitat by promoting human interference and increased predatory pressures. Compounded with the other non-federal actions, populations become more susceptible to the accumulation of adverse effects.

1.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as the combined effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the Alameda whipsnake's status at this time. However, the baseline condition of the assessed whipsnake's habitat varies across locations and within each critical habitat unit. Details of the Alameda whipsnake's habitat description and known locations are included in Attachment III. Given the large amount of habitat and extent of the action area included in this assessment, the discussion of environmental baseline includes a general discussion of factors that may affect the Alameda whipsnake within the action area. This information is presented in Section 1.2.1. The baseline status and population dynamics of the species is provided in Section 1.2.2.

1.2.1 Factors affecting species within the action area

At the time of the Alameda whipsnake's listing in 1997, failure to implement appropriate fire management practices on public lands and rate of habitat loss on private lands was determined to place the Alameda whipsnake in most endangerment (USFWS, 1997). By the time critical habitat was finalized in 2006, continued urban development resulting in fragmentation and removal of essential features had been determined to be the greatest threat to all six critical habitat units (USFWS, 2006). Additional anthropogenic and natural factors listed as directly and indirectly affecting the species included the effects of fire suppression, inappropriate grazing practices, predation, and non-native species invasions (USFWS, 2006). Sections 1.2.1.1 through 1.2.1.5 provide details on each of these major issues affecting the Alameda whipsnake in the action area.

1.2.1.1 Urban development

Urban development is determined to be the greatest threat to all six critical habitat units (USFWS, 2006). Urban development and associated highway and road construction cause rapid habitat loss, degradation, and fragmentation, the primary reasons for the Federal and State listing of the species (USFWS, 2003). Urbanization removes features essential to the conservation of the species and is heaviest in the central and western portions of Alameda and Contra Costa Counties (USFWS, 1997; USFWS, 2006). Urbanization and road construction also result in habitat fragmentation and has led to the almost complete disseverment of the five remaining subpopulations by preventing movement and migration (USFWS, 1997).

McGinnis (1992) and the United States Fish and Wildlife Service (USFWS) have documented colonies throughout the species' range that are subject to the pressures of residential developments (USFWS, 1997). In the Hayward-Pleasanton Ridge population, there are a number of housing developments that could potentially cause destruction, modification and/or fragmentation to the species' suitable habitat range (USFWS, 1997). These developments include the proposed 500 acre (ac) (200 hectare (ha)) Schaefer Ranch Project with approximately 474 homes, and the proposed Hansen Ranch Project of 146 ac (58 ha) (USFWS, 1997). The Schafer Ranch development is located on Alameda whipsnake suitable habitat, while the Hansen Ranch Project is located nearby a sighting of an Alameda whipsnake (USFWS, 1997). Due to 161 ac (64 ha) of Schaefer Ranch Project land being dedicated to the East Bay Regional Park District (EBRPD), increased public use and related activities will compound the potential effects on suitable habitat (USFWS, 1997). Towards the south of the Hayward-Pleasanton Ridge population are the sites of two proposed housing developments: the 1,580 ac (632 ha) Hayward 1900 project and the 391 ac (156 ha) Bailey Ranch project (USFWS, 1997). Both of the project sites were occupied by the Alameda whipsnake when the developments were proposed, and contiguous habitat exists between known occupied habitat to the west and east of these future developed areas (USFWS, 1997). Although the Bailey Ranch project includes mitigation measures to alleviate potential impacts toward the species, the development is still expected to result in habitat fragmentation of the Hayward-Pleasanton population (USFWS, 1997). Undeveloped areas such as trails and vineyards are proposed for the Hayward 1900 project, but these open areas could also result in habitat fragmentation for the Alameda whipsnake (USFWS, 1997).

Several developments are proposed for the Oakland-Las Trampas population that may impact Alameda whipsnakes and their habitat (USFWS, 1997). Several of these proposed projects abut the eastern margin of Las Trampas Regional Wilderness and contain habitat known to be occupied by the Alameda whipsnake (USFWS, 1997). Two of these projects include the Rossmoor Neighborhood Nine Project of 22 ac (9 ha) and the proposed expansion of the Oakland Zoo (USFWS, 1997). These and other projects in the Oakland-Las Trampas population may result in direct loss of or impacts to the species' suitable habitat (USFWS, 1997). Although some of these projects have or may reserve habitat suitable for the Alameda whipsnake, the undeveloped hillsides that support chaparral growth will be subject to increased fire suppression due to the close proximity of urban development (USFWS, 1997). This fire suppression would result in habitat degradation and an increased likelihood of severe wildfires (USFWS, 1997).

Urbanization indirectly impacts the Mount Diablo-Black Hills, Tilden-Briones, and Sunol-Cedar populations (USFWS, 1997). The expanding cities of Pittsburg, Oakley, Brentwood, and Antioch are expected to increase the degree of public visitation, and thus disturbance, to the nearby EBRPD parks and Mount Diablo State Park (USFWS, 1997). In particular, the eastern flank of the Mount Diablo-Black Hills population will be exposed to these indirect impacts associated with the 115-unit Clayton Ranch (1,030ac (412 ha)) and the 5,200-unit Cowell Ranch (4,272 ac (1,709)), among others (USFWS, 1997). The southern portion of the Mount Diablo-Black Hills population will see increased urbanization pressure and its associated impacts from the proposed Dougherty Valley (6,000 ac (2,400 ha)) and Tassajara Valley (4,000 ac (1,600 ha) projects (USFWS, 1997). Increased population pressure from the north will be caused by the approved 800-unit Franklin Canyon (980 ac (392 ha)) projects (USFWS, 1997). Additional developments are approved or proposed adjacent to the Sunol-Cedar population in the rapidly growing centers near Dublin and Pleasanton in Alameda County (USFWS, 1997).

Fragmentation has been determined to be of serious concern, including impacts from urban development and associated highway and road construction (USFWS, 2003). The ecological consequences of fragmentation include the loss of native plants and animals, invasion of exotic species, increased soil erosion, and decreased water quality (USFWS, 2003). Fragmentation leads to habitat loss and isolation, which increase the likelihood of genetic complications, such as genetic drift and imbreeding depression (USFWS, 2003). These consequences place species living in fragmented habitats at highest risk of extinction if they (1) depend on native vegetation; (2) require combinations of different habitat types; (3) require large territories; and (4) exist at low densities (USFWS, 1997). The Alameda whipsnake is confronted with particularly high risk because the species can be characterized by all of these attributes: the species is known to associate with native Diablan sage scrub (attribute 1), to forage in adjacent grasslands (attribute 2), and to migrate along riparian corridors (attribute 3) (USFWS, 1997). Although the home range of the Alameda whipsnake is typically between 5 and 20 ac (2 and 9 ha), not large in comparison to some animal ranges, the narrow habitats in some areas may impose significant constraints on the species (USFWS, 1997). Because the Alameda whipsnake is currently listed as threatened, low densities (attribute 4) are also expected (USFWS, 1997). Due to the disproportionate effects of urbanization, some populations of the species are more prone to low densities than others (USFWS, 1997).

1.2.1.2 Fire suppression

Increasing urbanization prompted the state of California to adopt an avid fire suppression policy over 50 years ago (USFWS, 2003). The policy has disrupted the natural disturbance regime of the chaparral/scrub community, and many chaparral habitats in the east Bay area have not burned for several decades (USFWS, 2003). To provide the necessary stimulus for new sprouting, seedling recruitment, and seed dispersal, a cyclical fire regime of 10 to 30 years is required (USFWS, 1997). In the absence of reoccurring disturbance, these chaparral/scrub communities are becoming decadent (in a state of decline) and senile (reduced to a featureless plain) (USFWS, 2003). Other natural disturbances affecting the chaparral/scrub communities include landslides, droughts, and herbivory, although to a lesser degree (USFWS, 2003).

Fire suppression directly and indirectly affects the Alameda whipsnake (USFWS, 1997). Fire suppression exacerbates the effects of wildfires through the buildup of fuel (underbrush and woody debris), creating conditions for slow-moving, hot fires (USFWS, 1997). Conditions for the most intense fires occur in summer and early fall when accumulated fuel is dry and abundant (USFWS, 1997). During this period, hatchling and adult Alameda whipsnakes are aboveground and populations are likely to sustain the heaviest losses (USFWS, 1997). Fire suppression indirectly affects the Alameda whipsnake by altering plant compositions and structure of suitable habitat (USFWS, 1997). Infrequent fires allow the establishment of closed canopies and thus cooler understory environments. Both the Alameda whipsnake and its lizard prey base are potentially affected by cooler habitats, with Alameda whipsnakes being particularly sensitive because of their higher mean active body temperature (33.4°C) and higher degree of body temperature stability (stenothermy) compared to other snake species (USFWS, 1997; USFWS, 2006). Alameda whipsnakes maintain a higher temperature by using partially open and low growing shrub communities that provide predator protection and allow sunlight penetration (USFWS, 1997). This preference is supported by survey studies showing that Alameda whipsnakes are less likely to be found in closed canopy environments (USFWS, 1997). Fire suppression has also led to the encroachment of non-indigenous and ornamental trees into grassland habitats, further increasing fuel loads in and around Alameda whipsnake habitat (USFWS, 2006).

1.2.1.3 Predation

A number of native and nonnative mammals and birds are known or likely to be predators of the Alameda whipsnake: the California kingsnake (*Lampropeltis getula califoniae*), northern raccoon (*ProcyonProcyon lotor*), striped skunk (*Mephitis mephitis*), Virginia opossum (*Dideplhpis virginianus*), coyote (*Canis latrans*), gray fox (*Vulpes cinereoargenteus*), red fox (*Vulpes vulpes*), and hawks (*Buteo* species) (USFWS, 2003).

Increased predatory pressure may become excessive in situations where the Alameda whipsnake habitat has become fragmented, isolated, and otherwise degraded (USFWS, 2003). Alameda whipsnakes living adjacent to urban developments are at heightened risk because of the associated loss of cover habitats and an increase in predators using these areas (USFWS, 2006). For example, predation and harassment from domestic and feral cats (*Felis domesticus*) increases as urban development infringes upon Alameda whipsnake habitat (USFWS, 2003). The growing movement to maintain feral cat populations in parklands intensifies this threat (USFWS, 2003). Although predatory impacts on whipsnakes from cats have not been studied, cats are known to prey on the yellow-bellied racer (*Coluber constrictor*), a closely related snake species, as well as have significant impacts on lizard populations (USFWS, 2003). Additional introduced predators include rats (*Rattus* species), feral pigs (*Sus scrofa*), and dogs (*Canis familiaris*) (USFWS, 2003). Impacts from these species may be especially acute if urban developments directly abut Alameda whipsnake habitat (USFWS, 2003). Special management of nonnative predators may be required within all six critical habitat units (USFWS, 2006).

1.2.1.4 Grazing practices

Livestock grazing and inappropriate grazing practices may threaten the Alameda whipsnake (USFWS, 1997; USFWS, 2006). Studies have suggested that grazing east of the Coast Range has impacted suitable habitat for the Alameda whipsnake (USFWS, 1997). Livestock grazing becomes detrimental to the species when it significantly reduces or eliminates shrub and grass cover (USFWS, 1997). According to these studies, the Alameda whipsnake along with other snake species avoid such open areas because of magnified predatory risk and lack of prey (USFWS, 1997). Inappropriate grazing practices include overgrazing and burning or bulldozing to remove scrub prior to grazing (USFWS, 2006). These practices affect the scrub component of the vegetation mosaic, and reduce grass height or density to such levels that expose Alameda whipsnakes to greater predation from hawks (USFWS, 2006).

1.2.1.5 Non-native species

In addition to the increased predatory pressure from introduced species described in section 1.2.1.2., the Alameda whipsnake is also threatened by nonnative flora and fauna altering California's native grassland and coastal prairie (USFWS, 2003). Introduced plant species include *Genista monspessulana* (French broom), *Carpobrotus* spp. (iceplant), *Eucalyptus* (Eucalyptus), and *Ulex europaeus* (gorse) (USFWS, 2003). These species often outcompete and supplant native vegetation to the point of potentially degrading the habitat, increasing fire risk, and reducing whipsnake prey base (USFWS, 2003). Hybridization, which could decrease genetic variability, is also a consequence of nonnative invasions (USFWS, 2003). For example, *Arctostaphylos pallida*, a native member of California's chaparral ecosystems, can easily hybridize with ornamental *Arctostaphylos* spp. planted by homeowners (USFWS, 2003). Non-native fauna species can also affect native vegetation (USFWS, 2003). Studies of impacts from feral pigs on natural habitat have shown effects to plant species diversity, total herbaceous cover, and soil chemistry (USFWS, 2003). These vegetation alterations incrementally affect the distribution and abundance of small animals and amphibian species (USFWS, 2003).

1.2.2 Baseline Status

At the time of the species' "threatened" listing in 1997, the USFWS concluded that the Alameda whipsnake was likely to become endangered within the foreseeable future (USFWS, 1997). The USFWS cited the failure to implement adequate fire management practices and sustain suitable habitat as key factors in the continued decline of the species (USFWS, 1997). Degradation of suitable habitat is a consequence of urbanization, fragmentation, grazing practices, and mining, each affecting the species' subpopulations to a different degree. The Population Dynamics section will cover these factors threatening the survival of each subpopulation and the potential alterations to the currently designated species' critical habitat. The following Habitat Conservation Plan section will provide detail on authorized take of the species.

Population Dynamics

Currently, the Alameda whipsnake inhabits the Inner Coast Ranges in western and central Contra Costa and Alameda Counties, with occurrences additionally recorded in San Joaquin and Santa Clara Counties (USFWS, 1997, 2003, and 2006). The current distribution of the subspecies has been reduced to five separated ranges within a fragmented regional metapopulation, all occurring on private or public, non-Federal land, with little or no interchange due to habitat loss, alteration, and fragmentation (USFWS, 2003). These ranges correspond to the designated recovery units in the draft recovery plan for chaparral community species (USFWS, 2003) that preceded the determination of critical habitat (USFWS, 2006).

1. Sobrante Ridge, Tilden/Wildcat Regional Parks area to the Briones Hills, in Contra Costa County (Tilden-Briones population)
2. Oakland Hills, Anthony Chabot area to Las Trampas Ridge, in Contra Costa County (Oakland-Las Trampas population)
3. Hayward Hills, Palomares area to Pleasanton Ridge, in Alameda County (Hayward-Pleasanton Ridge population)
4. Mount Diablo vicinity and the Black Hills, in Contra Costa County (Mount Diablo-Black Hills population)
5. Wauhab Ridge, Del Valle area to the Cedar Mountain Ridge, in Alameda County (Sunol-Cedar Mountain population) (USFWS, 1997 and 2003).

The Tilden-Briones population is currently threatened by habitat loss and fragmentation due to suburban/rural growth (USFWS, 2003). While the periphery of the range is impacted most from urbanization, developments within the interior are also promoting fragmentation (USFWS, 2003). The Tilden-Briones population is further at risk from catastrophic wildfire and from nonnative invasion (eg. *Eucalyptus*) into chaparral and scrub communities (USFWS, 2003). Due to partial overlap of the habitat with regional parklands and municipal watersheds, regional preservation and land management to the benefit of the Alameda whipsnake may be possible (USFWS, 2003).

The Oakland-Las Trampas population is imperiled by the decadence of chaparral/scrub stands, high risk of catastrophic wildfires, and habitat loss and fragmentation from urban development (USFWS, 2003).

The Hayward-Pleasanton Ridge population is most susceptible to extirpation. Similar to the other subpopulations, the Hayward-Pleasanton Ridge population is threatened by urbanization (USFWS, 2003). Unlike any of the other five areas, however, the open space and conservation land within this habitat range is the most disconnected (USFWS, 2003). The northern and southern parts of the range are at immediate risk of isolation, which would increase the susceptibility to genetic drift and chance events (USFWS, 2003).

The Mount Diablo-Black Hills population is threatened by a high potential for catastrophic wildfire, suburban/rural development, and land uses (eg. mining) that are at odds with Alameda whipsnake suitable habitat (USFWS, 2003). This population is a good candidate for recovery considering the location of public lands, the actions of private nonprofit organizations regarding wildlife corridors, and the potential for improved fire and grazing practices (USFWS, 2003).
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The Sunol-Cedar Mountain population is threatened by catastrophic wildfires and incompatible land uses, including mining and off-road vehicle use. Currently the pressure from urbanization is relatively low, although the demand for housing developments is increasing. Unauthorized collection may be most acute in the southern region of the population. The area attracts reptile enthusiasts and collectors because of its remote roads and diversity of native reptiles.

The Caldecott Tunnel corridor links the Tilden-Briones and Oakland-Las Trampas populations, and the Niles Canyon-Sunol corridor connects the Hayward-Pleasanton Ridge and Sunol-Cedar Mountain populations. The Caldecott Tunnel corridor is heavily developed and highly fragmented (USFWS, 2003). Much of the vegetation was eradicated from a catastrophic firestorm on October, 20 1991, and revegetating the area is still in progress (USFWS, 2003). Future plant communities may not promote movement of the Alameda whipsnake because vegetation with low volatility is being discouraged (eg. *Artemesia californica,* California sagebrush) (USFWS, 2003). The Niles Canyon-Sunol corridor contains various physical barriers that could impede movement of the Alameda whipsnake: Alameda Creek; a 0.3- to 0.6-meter (12- to 24-inch) high concrete barrier; railroad tracks; and heavy vehicular traffic along Niles Canyon Road (USFWS, 2003). Much of the land is also under cultivation or mining (ie. gravel) and suitable vegetation for the Alameda whipsnake is limited (USFWS, 2003).

Habitat Conservation Plans

Habitat Conservation Plans are defined in Section 10 of the Endangered Species Act and target the actions of non-Federal entities in relation to listed species (USFWS, 2005). These plans authorize the take of listed species through Incidental Take Permits and help minimize or mitigate the effects of the action (USFWS, 2005). Two Habitat Conservations Plans are listed on the USFWS species profile webpage for the Alameda whipsnake (Species Profile). The first Plan is the East Bay Municipal Utility District and covers 28,000 ac between Alameda and Contra Coast Counties (Species Profile). The second plan is the East Contra Costa County HCP/NCCP (Species Profile). A permit was issued for this Plan in 07/25/07 for 175,435 ac of Contra Costa County for the duration of 30 years (Species Profile).

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2: BAY CHECKERSPOT BUTTERFLY

2.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the bay checkerspot butterfly (*Euphydryas editha bayensis*) (BCB) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of urban and suburban development, infrastructure expansions, commercial and private construction, agriculture, research, pest (gopher) control, and activities related to climate change. Many of these factors are linked or act synergistically and create complex consequences for the BCB. For example, urban and suburban developments encourage the expansion of infrastructure and intensify the effects of climate change. These developments contribute to habitat loss and facilitate habitat fragmentation, weakening the strength and stability of populations. Compounded with the other non-federal actions, populations become even more susceptible to the accumulation of adverse effects.

2.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as the effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the bay checkerspot butterfly's status at this time. However, the baseline condition of the assessed butterfly's habitat varies across locations and within each critical habitat unit. Details of the bay checkerspot butterfly's habitat description and known locations are included in Attachment III. Given the large amount of habitat and extent of the action area included of this assessment, the discussion of environmental baseline includes a general discussion of factors and a few listed projects that may affect the BCB within the action area. This information is presented in Section 2.2.1. The baseline status and population dynamics of the species is provided in Section 2.2.2. The majority of information included in the environmental baseline section is a summary of information on the bay checkerspot butterfly presented in the Recovery Plan for Serpentine Soil Species of the San Francisco Bay Area (USFWS, 1998), and the Designation of Critical Habitat for the Bay Checkerspot Butterfly (*Euphydryas editha bayensis*); proposed rule (USFWS, 2007).

2.2.1 Factors affecting species environment within the action area

A number of factors affect the species environment, defined by a habitat comprised of serpentine, or serpentine-like or -derived soil. This soil is formed from weathered ultramafic rocks, serpentinite, dunite, and peridotite, and presents harsh environments for plant growth: 1) a low calcium/magnesium ratio; 2)lack of essential nutrients such as nitrogen, potassium, and phosphorus; and 3) high concentrations of heavy metals (mineral toxicity) (Kruckenberg, 1984a). This native environment is primarily affected by destruction and degradation, and infestations of native species (accompanied by air pollution). Factors affecting the species include overutilization for research and private purposes, control of gophers, vehicular disturbance, fire and grazing practices, and climate change. Sections 2.2.1.1 through 2.2.1.6 provide details on the major issues facing bay checkerspot butterfly populations within the entire action area.

2.2.1.1 Habitat loss and degradation

Habitat loss and degradation is one of the two most influential factors affecting the BCB and is caused primarily by urban and suburban development (Murphy and Weiss. 1988a). Reducing suitable habitat for the BCB decreases the size and number of populations and increases vulnerability to stochastic environmental events. Elimination of serpentine habitats also destroys stepping-stone areas, small patches in between suitable habitat, and makes recolonization and persistence of metapopulations difficult (USFWS, 1998). Prior to BCB listing, two primary habitats thought to support persistent "core" populations were lost to housing and infrastructure developments. This habitat conversion occurred around the Hillsborough and San Mateo areas, and included losses to secondary and tertiary habitats (secondary habitat refers to small serpentine outcrops, while tertiary habitat refer to areas where host plants occur but not on serpentine soil). Extirpation was recorded to affect 29 of 32 probable secondary habitats, 5 of 8 known secondary habitats, at least 5 of 6 known marginal habitats, and over 9 other likely areas (USFWS, 1987). Over the past couple of decades, development in the city of San Francisco has occurred on top of several serpentine habitats (Hunters Point), although whether the areas supported the BCB with quality habitat is unknown. By the time of its listing in 1987, the BCB occupied as few as 2 out of its 16 known historic localities with persistent populations. Only two locations at present, the Edgewood Natural Preserve in San Mateo County and Coyote Ridge in Santa Clara County, support large populations of BCBs. Rates of loss in recent years have slowed considerably but have yet to fully subside (USFWS, 1998).

Habitat fragmentation is documented and is a concern for the BCB because of its limited dispersal ability and distribution within a metapopulation structure. Four patches of suitable habitat once found within San Mateo County have now been divided into 11 smaller units (Ehrlich, 1961, 1965). Some of these units are separated by a 6- or 8-lane freeway, which greatly hinders trans-patch movement and promotes populations to live in isolation. A BCB population on San Francisco Water Department land first declined, and then went extinct, after it was divided by the construction of I-280 from the Edgewood Natural Preserve population (A. Launer, pers. comm., 1997). The Edgewood Natural Preserve is the largest of the 11 fragmented habitats in San Mateo County, and the only one still supporting a significant population (Ehrlich and Murphy, 1987).

Urban and suburban developments continue to threaten BCB survival since the time of its original listing in 1987. The growth of the San Francisco Bay Area plus the expanding human population threatens the species into the near future. The U.S. Fish and Wildlife Service (USFWS) has been involved in a number of urbanization projects potentially impacting the BCB. Technical assistance was required for eight projects in 1995 and 1996, and 5 projects required informal consultations with the USFW. At the time the Recovery Plan for the Serpentine Soil Species (including the BCB) was released, four projects were proposed or currently proposed that would adversely affect serpentine grassland. These four projects were Cerro Plata, Metcalf Road widening, Richmond/Young Ranches, and Calero Lake Estates.

Cerro Plata

The Cerro Plata project in eastern San Jose was proposed on the Silver Creek Hills bay checkerspot population area. The project area incorporated 232 hectares (575 acres) of a housing development and golf course on mostly serpentine habitat. According to the USFWS, the Environmental Impact Report for the project underestimated the amount of bay checkerspot habitat at the site, and consequently underestimates the potential impacts. A Habitat Conservation Plan (HCP) (a conservation plan developed by non-Federal parties applying under section 10 of the Act for a permit for incidental taking of listed species) is being crafted by the site owner and consultants to revise the development proposal for the site (USFWS, 1998).

Metcalf Road

The proposed project in the City of San Jose would widen and straighten a portion of Metcalf Road. The impacted area would include serpentine habitats and affect the associated serpentine plant and animal species that are threatened or rare, and would increase road kills of the BCB. The Federally listed species, including the California red-legged frog (*Rana aurora draytonii*), would also be vulnerable to the effects of the project. The expanded road would facilitate human access to bay checkerspot core habitat: the Kirby, Metcalf, and San Felipe populations. The USFWS has recommended that the U.S. Army Corps of Engineers consult under section 7 of the Act (USFWS, 1998).

Richmond/Young Ranches

The proposed project would extend the City of San Jose Urban Growth Boundary to include portions of Young Ranch, Richmond Ranch, and smaller adjacent properties. The project is the result of combined interests of the YCS Investments of San Francisco and the owners of the Richmond Ranch, that together wish to create a planned residential community not exceeding 2,450 housing units, a golf course, resort, and other miscellaneous developments. The extension of the 735-hectare (1,817-acre) includes suitable serpentine habitat for the BCB Metcalf population (City of San Jose, 1997).

Calero Lake Estates

Not much information has been provided on the Calero lake Estates project. The available information states that the project is an incomplete housing subdivision in the southern Santa Teresa Hills, situated in contact with the Santa Teresa County Park. Bay checkerspot are known to occur in the serpentine habitat in this location and are vulnerable to the past and future developments (USFWS, 1998).

In addition to the well-described urbanization projects above, other developments in serpentine areas could potentially impact the BCB as well. Developments in Santa Clara County are predicted to increase into the near future. Developments in San Jose are continuing and are guided under a general plan until 2020. However, this is a separate measure from the proposed Santa Clara County HCP, and the Court of Appeals 6th district found that the City's zoning did not have to be consistent with the general plan (Juarez *et al.* v. City of San Jose *et al.* (6th District, Case no, CV736436 H014755)). This means that current habitat not within the urban growth boundary may still be proposed for development. Other projects include the Coyote Valley Specific Plan, residential and industrial developments, the Coyote Valley Research Park project, numerous proposals under the Santa Clara County HCP, and projects including single family residential units and road grading (USFWS, 2007).

2.2.1.2 Invasive plants and air pollution

Invasion of exotic species is the second of the two most influential factors affecting the BCB. Despite the inhospitable conditions of serpentine environments, non-natives are still capable of colonizing the area. Non-native grasses have been observed to choke out BCB host plants in the Edgewood Natural Preserve (USFWS, 1998) and serpentine soils in Silver Creek Hills (R. White, pers. comm.., 1997, A. Launer, pers. comm., 1997, S. Weiss, pers. comm., 1997, D. Murphy, pers. comm., 1997), and include, but are not limited to, the yellow star thistle (*Centaurea solstitialis*) and certain eucalyptus species (*Eucalyptus* spp.). Northern California continues to see new introductions of exotics through gardening, landscaping, and accidental means (USFWS, 1998).

Nitrogen deposition from air pollution facilitates the spread of non-native vegetation in serpentine habitats. The usually nutrient-poor serpentine soils are enriched with the addition of atmospheric nitrogen, a typical limiting factor for plant growth (Weiss 1999, p.1477). Nitrogen has been associated with thick patches of vegetative material (thatch) that inhibits growth of the native plant species (Huenneke *et al.* 1990, p.488). Thatch threatens the host plants of the bay checkerspot through competition and crowding (Weiss 1999, p. 1481).

2.2.1.3 Overutilization for research and private purposes

As part of a long-term research and monitoring program, Stanford University has investigated the biology of the BCB since 1960. The research has been carried out on the

Jasper Ridge Biological Reserve, and Harrison *et al.* (1991) has estimated the effects of collection on two BCB populations. From 1960 and 1983, between 0 and 386 (mean 57.3) females per year were removed from the populations for electrophoresis and other purposes. The number removed, on average, was 6.1 percent of the populations (Harrison *et al.* 1991). The study concluded that the effects of individual removal were much smaller than the effect of natural environmental variability. The study indicated, however, that the removal increased the risk of population extinction, from a negligible risk to a 15 percent probability over 30 years (depending on model used) (USFWS, 1998).

Foot-traffic connected with the study of one Jasper Ridge population was found to have a significant impact on the area's vegetation (Ehrlich and Murphy,1987). According to Ehrlich and Murphy (1987), trampling could have destroyed the BCB eggs, larvae and pupae. Another study by Orive and Baughman (1989) studied the impacts of a mark-and-recapture study on the BCB at Jasper Ridge and found that handling did not observably increase wing-wear.

Illegal collecting has been documented to affect the BCB, and the USFWS has prosecuted known cases. Illegal take will most likely affect those BCB populations that are small or have been reduced by natural or artificial factors, or that are easily accessible by people (USFWS, 1998).

2.2.1.4 Gopher control and vehicular disturbance

A positive correlation exists between the presence of gophers and BCB survival. A study by Singer (1972) noted that BCB larvae can survive later into the spring on *Plantago erecta* growing in soil disturbed by gophers (*Thomomus bottae*). Singer suggested that the gophers loosened up the soil and allowed for more root growth and water retention (Singer, 1972). A more recent study found BCB larval host plants to stay green and edible longer when located on or nearby soils recently tilled by gophers (Singer, 1972, p. 75; Murphy *et al.* 2004, p. 26). Another suggestion was made that gopher disturbance limits the performance of grasses and allows for the persistence of smaller forbs. The prediapause larva may be able to reach the fourth instar since the larval host plants stay greener for longer (Huenneke *et al.*, 1990, p. 490).

A negative correlation exists between vehicles and BCB individuals. Bay checkerspot mortality results from direct vehicular strikes and associated turbulence. The degree of annual mortality or injury, however, is uncertain (USFWS, 1998).

2.2.1.5 Fire and grazing practices

After a wildfire spread across portions of San Bruno Mountain in 1986, no bay checkerspot butterflies were subsequently observed (USFWS, 2007). Although only 50 individuals had been observed there in 1984 (CNDDB, 1984), and other factors besides fire were implicated in their extirpation, wildfires are still considered a threat to the BCB. Prescribed burns, however, are regarded as potentially effective tools to combat invasion of non-natives and stimulate growth of species adapted to the disturbance. However, benefits of burn regimes on serpentine grasslands are not well documented and have been provided by monitoring studies made possible after wildfires. An experimental prescribed burn was conducted on a small segment of Coyote Ridge in 2006, but the benefits to the native species, including the BCB, are not yet known (USWFS, 2007).

A limited amount of grazing is considered beneficial to BCB habitat, while over- or under- grazing threatens the survival of native host plants. If not appropriately managed, overgrazing threatens to reduce the density of native vegetative mass. A study by Huenneke *et al.* (1990, p.489) and Weiss *et al.* (1999, p. 1480) found that undergrazing resulted in an increase of annual grasses, which crowd out native forbs (USFWS, 2007). Sustainable grazing practices are generally recognized as non-threatening to the BCB (Thomas Reid Associates and Murphy 1987; Murphy, 1988; Weiss, 1996).

2.2.1.6 Climate change and climate extremes

The expected climate change in the San Francisco Bay Area is currently disputed, but any deviation from current conditions would most likely affect the bay checkerspot (Murphy and Weiss, 2002). Forecasted change scenarios impacting the BCB include wetter and warmer, wetter and cooler, drier and colder, and changes in the timing of rainfall. The BCB would likely be sensitive to weather extremes because its developmental requirements and host plants' survival depends on the climate. The largest and potentially most viable population, the Kirby Bay population, is not well-buffered against changes in temperature or precipitation. The drought in 1975 and 1977 caused the disappearance of many small populations in this normally dry area, compared to populations in normally wetter areas (Murphy and Weiss, 2002). Climate change is also predicted to change the dominance of native vs. non-native serpentine species (USFWS, 1998).

As with the aforementioned drought in 1975 and 1977, other dry periods and weather extremes have been linked to declines in BCB populations; however, clearcut BCB weather responses are hard to define (Ehrlich *et al.* 1980; Weiss, 1996). There is evidence that northern California has experienced more severe and long-lasting droughts in the past millennium than were ever recorded in history (Stine, 1994). The BCB high reproduction potential indicates the species is well-adapted to the area, as the bay region has been its home for a long time, but its poor resilience after recent weather extremes may be due to other, probably human-induced, factors (USFWS, 1998).

2.2.2 Baseline Status

The current bay checkerspot butterfly range is much reduced from its historic distribution (USFWS, 2007). The BCB previously occupied the area around the San Francisco Bay, spanning from San Bruno Mountain and Twin Peaks in the west, to Contra Coast County in the east, and south through Santa Clara County (Murphy and Ehrlich 1980; Opler *et al*., 1985; California Natural Diversity Data Base, 1996). Currently, the BCB has gone extinct in Contra Costa County (Franklin Canyon and Morgan Territory areas), Alameda County (Oakland Hills), San Francisco County (Twin Peaks and Mount Davidson), San Bruno Mountain, Buri Buri Ridge (Hillsborough), Pulgas Ridge (or "San Mateo"), and Redwood City in San Mateo County (historic site referred to as Woodside) (Murphy and Weiss, 1988a; California Natural Diversity Data Base, 1996).

The current distribution is an example of a metapopulation, and because of its interdependent extinction and colonization processes, the exact distribution of the butterfly varies throughout time (Wilcox and Murphy, 1985; Harrison *et al*. 1988). Any site within the proximity of the historic range could potentially be occupied by the BCB. There are reports of checkerspot ecotypes (*Euphydryas editha*) that occur in the south and east of the bay checkerspot distribution, but status of these subspecies is unknown at this time. Even though the BCB is patchily distributed, two large metapopulations have been identified. These occur in Santa Clara and San Mateo Counties (Murphy, 1988).

Within the two metapopulations, five core areas have also been identified for the BCB: one on the San Francisco Peninsula in San Mateo County (Edgewood County Park), and four in Santa Clara County. Core areas roughly correspond to primary habitat, defined as moderate to large areas of suitable habitat that support persistent bay checkerspot populations. These areas supply migrants to unoccupied sites, continuing the repopulation/extinction cycles. The Santa Clara core areas are located along a ridge immediately east of Santa Clara Valley, designated Coyote Ridge (one of its many common names) by the Recovery Plan (USFWS, 1998). The four core areas are entitled Kirby, Metcalf, San Felipe, and Silver Creek Hills, and are discontinuous serpentine areas that are all within butterfly flight distance of each other. The last of the four core areas, Silver Creek Hills, is not considered by the USFWS to be a significant population, although it has a status of a "core" population. These four core areas, comprising the Santa Clara County metapopulation, are thought to have the best chance of long-term persistence (Murphy, 1988; Murphy and Weiss 1988a). The Edgewood Park metapopulation is smaller, probably due to being located on a smaller area, with smaller and poorer quality secondary habitats surrounding it. Natural migration between the two metapopulations is uncommon because they are currently separated by 40 kilometers (25 miles) (USFWS, 1998).

Conservation measures emphasize the value of secondary or satellite populations that generally occur on smaller or of lesser quality land than primary habitats. Both core areas and secondary areas are necessary for the natural metapopulation dynamics of the BCB. Specific dynamics of the BCB are particularly important to fields of population and conservation biology and ongoing studies from the 1960s have generated significant data. Further, satellite populations act as stepping stones for dispersal. Tulare Hill appears to be a stepping stone habitat for BCB dispersal across Santa Clara Valley. Finally, secondary populations help protect populations against complete extirpation after unforeseen environmental disasters: fire, extreme weather, air pollution, or climate change. While core areas may be the targets of catastrophic events, secondary populations may survive to recolonize the formerly inhabited areas (USFWS, 1998).

Population Dynamics

Since its listing in 1987, the distribution and population size of the bay checkerspot has changed dramatically. Since the critical habitat designation in 2001, the number of known occurrences has continued to decline. For example, the USFWS designated 15 critical habitats in 2001. A revised critical habitat designation was proposed in 2007 and suggested redefining and reducing the number of units to 12. Most of the proposed units correspond to those from 2001. The eliminated areas were unlikely to support the PCEs for the BCB, or had since been developed. For example, the area designated unit six (Communications Hill) was excluded from the revised version due to developments and degradation from non-native invasions. Unit two (Pulgas Ridge) is a newly proposed area and is currently undeveloped. The unit resides within the historic range of the species and could potentially serve as a stepping-stone between proposed units four and five, the two southern most units in San Mateo County, and San Bruno Mountain. The proposal awaits final ruling (USFWS, 2007).

The San Mateo County subspecies have significantly declined, and populations at San Bruno Mountain, Pulgas Ridge, and Jasper Ridge have yet to be detected in limited surveys. No BCB individual was detected on a small southeastern portion of Pulgas Ridge where surveys took place from 1989 to1993 and 1994. However, these surveys only covered a section of the much larger area that was historically occupied by the butterfly. The Edgewood County Park population, deemed the only core population in San Mateo County, was last observed in 2002 (CNNDB, 2006). Restoration efforts in February and March of 2007 reintroduced approximately 1,000 post-diapause larvae to the Park to maintain a viable San Mateo County population (CNDDB, 2007). The successful fate of this population has yet to be determined.

Population trends have been discerned for limited portions of Coyote Ridge in Santa Clara County. These portions correspond to units 4, 8, 10, 11, 12, and 15 in the 2001 determination (66 FR 21450) (USFWS, 2007). The population at unit eight (Kirby) increased from 20,000 individuals in 1997 to 700,000 in 2004. In 2005, the population fell to about 100,000 individuals (Weiss, 2006, p.1). The population at unit 10 (Metcalf) increased from 200,000 individuals in 2000 to 400,000 in 2004, but then declined to 45,000 individuals in 2006 (Weiss, 2006, p.1). The population from unit 12 (Silver Creek) was estimated in larval units, and increased from 75,000 in 1992 to 128,000 in 1993, but dropped to an estimated 58,000 in 1994 after grazing was removed from sections of the area (Weiss, 1996, p. 93; Weiss, 1999, p. 1480). No larvae or adults were observed in 1998 (Weiss, 1999, p. 1480). In 2000 and 2001, a residential subdivision was constructed and grazing was reintroduced in unit 12. Annual surveys since this period have shown 11 adults in 2001, up to a maximum of 51 in 2005. Only 40 adults were detected in 2006, but no larva was observed (WRA 2006, p. 10). The population in unit 15 (Tulare Hill) contained approximately 2,000 individuals in 2002. The population significantly declined in 2003 with only one sighting of a post-diapause larvae (CH2M Hill, 2004, p. 8-6), and five sightings of adults in 2004 (CH2M Hill, 2005, p. 8-2). Weiss

(2007, p.1) estimated the Tulare Hill population to total 100 individuals from the 2004 observation. In 2005, seven adults were observed, with no detection of post-diapause larvae (CH2M Hill 2006, p. 8-2). Thousands of BCB were observed at unit 4 (Bear Ranch) in 1994, six adults were observed in 1997, and one adult in 1999. The USFWS does not possess any additional studies on the Bear Ranch population after 1999 (USFWS, 2007).

Habitat Conservation Plans
The San Bruno Habitat Conservation Plan was the first Habitat Conservation Plan for the bay checkerspot butterfly and was adopted in 1983. Set to expire in 30 years, the Plan will discontinue unless renewed by March 31, 2013. The Plan originally covered 1,400 hectares (3,400 acres) in northern San Mateo County and identified seven animal species and 44 plant species. At the time of its publication, the plan did not specifically focus on the BCB, but rather on the mission blue and San Bruno elfin butterflies (listed species at the time), the callippe silverspot butterfly (unlisted at the time), and their host plants (USFWS, 1998). The plan is currently undergoing its fifth amendment, and would include conservation actions that would benefit both the bay checkerspot and callipe silverspot butterflies. The conservation actions include: (1) vegetation management (e.g., prescribed fire, herbicide application, mowing, and grazing); (2) replanting and restoration; and (3) monitoring. The plan will also substantially protect all bay checkerspot PCEs. A finalized amendment is expected by the USFWS in 2008 (USFWS, 2007).

Two other Habitat Conservation Plans, accessed online though the Fish and Wildlife Service webpage, have been issued on behalf of the BCB. The first was entitled Metcalf-Evendale/Monta-Vista (PG&E) HCP and was issued in 1997. PG&E Metcalf - El Patio, Metcalf -Hicks/Vasona LE HCP was the second and was issued in 2007. Both extend three years, and the first Plan has since expired in 2000.

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3: CALIFORNIA CLAPPER RAIL

3.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the California Clapper Rail (CCR) (*Rallus longirostris obsoletus*) that are reasonably certain to occur in the action area. Future federal actions unrelated to the proposed action are not considered because they are subject to consultation pursuant to Section 7 of the Endangered Species Act (Act). Numerous non-federal actions that could affect the California Clapper Rail are reasonably certain to occur within the action area. These activities are associated with modifications to salt and brackish marsh habitats and typically include urban and industrial developments, filling and diking activities, maintenance of levees, and conversions of salt marsh to brackish water habitats. These actions could compound to create complex consequences for the CCR. For instance, urban development could lead to habitat loss, degradation, and fragmentation. These factors reduce the quality and suitability of CCR habitat by influencing the presence of tidal channel systems and the inland extent of high and low tides. Encroaching developments change CCR predation zones and affect second order predators that would otherwise benefit the CCR. Predatory impacts on the CCR are further intensified by high marsh and natural high tide cover.

3.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as the effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the California Clapper Rail's status at this time. Details of the California Clapper Rail's habitat description and known locations are included in Attachment III. However, the baseline condition of each assessed CCR species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline presents a general discussion of factors that may affect CCR within the action area. This information includes both detrimental and potentially beneficial factors affecting the species and is presented in Section 3.2.1. Additional information on the current distribution and population dynamics of the CCR is presented as part of the baseline status in Section 3.2.2.

3.2.1 Factors affecting species within the action area

Habitat loss and fragmentation present the most significant threat towards the CCR and its environment. Marsh size and habitat quality dictate the maximum number of rails in any particular marsh and are impacted by habitat alteration. The primary causes of this loss and fragmentation include: erosion, introduction of freshwater, change in landscape, subsidence, lack of tidal channels, invasive species, and land management practices. Hunting, predation, disease, and climate extremes are additional detrimental factors currently affecting the CCR and its habitat. Beneficial factors include restoration and protection projects that have been, or are currently being completed, to reconstruct and enhance the salt marshes of the San Francisco Bay Area. Sections 3.2.1.1 through 3.2.1.6 provide details on each of the major issues affecting the SFGS within the action area.

3.2.1.1 Habitat Loss and Fragmentation

Fragmentation and loss of habitat are the main factors affecting CCR populations (Albertson, 1996). An 84% reduction in the tidal marsh bordering San Francisco Bay has occurred since 1850 due to urbanization, filling, and diking of wetland areas (USFWS, 2007; Environmental Impact; Albertson, 1996; and Save the Bay, 2005). Marshes that have been filled have had the upland vegetation and most of the high marsh zones eliminated which leaves no refuge for the CCRs during high tide (USFWS, 1984). Vegetation high in the marsh has also been eliminated in Suisun Marsh due to diking and livestock grazing (USFWS, 1984). Since the mid 1960s, there has been little large scale habitat loss; however, the loss of many small habitats has resulted in significant overall wetland loss and degradation (Albertson, 1996).

Physical factors that affect the marsh habitat include erosion, freshwater introduction to the marsh, small size, fragmentation, subsidence, and a lack of tidal channels (USFWS, 2007). The shoreline of the East Bay is eroding, from San Leandro to Calaveras Point, and could lead to the loss of several CCR populations in the area (USFWS, 2007). The vegetation in about 600 acres of marsh that was previously salt marsh in South San Francisco Bay along Coyote Creek, Alviso Slough, and Guadalupe Slough, has been degraded and transformed to freshwater and brackish water vegetation as a result of freshwater input from South Bay wastewater facilities (USFWS, 2007). The vegetation in this area is dominated by alkali bulrush, which does not provide suitable nesting habitat, and CCR populations in this area have declined (USFWS, 1984). Land subsidence has also reduced the available CCR nesting habitat (USFWS, 1984). In South Bay, the difference between the high and low tides is greater than in San Pablo or Suisun Bays, resulting in marshes that become completely submerged during high tides (USFWS, 2007). Marshes that become completely submerged during high tide usually do not have sufficient escape habitat for the CCR which results in high predation rates and also nesting failures (USFWS, 2007).

CCR habitat is lost due to an increase in the presence of steep sided channels which have replaced "gently sloping channel flats" which are used by CCRs when they forage and when needed as escape routes from predators (SFEISP online). Many of the marshes in South San Francisco Bay are bordered by steep levees resulting in a loss of the upper marsh vegetation and less available cover for the CCR during winter flood tides (USFWS, 1984). The lack of high marsh habitat and steep levees limits the ability of the CCR population to expand (USFWS, 1984). The destruction of CCR habitat has allowed the upland predators to more easily reach the vulnerable and relatively unprotected CCR and their eggs (Save the Bay, 2005).

Invasive Species

CCR habitat is threatened by invasion of non-native cordgrass (CDFG, 2000). CCR depend on pickleweed and gum plant for nesting. Both of these plant species can be displaced by the non-native *Spartina* which invades channels and accretes sediment (SFEISP online). Excessive sedimentation can clog tidal sloughs which are used by the CCR for foraging (Goals Project, 2000). However, it appears that CCR have colonized marshes with this invasive species and used habitat areas that would otherwise not be suitable for this species (Spautz, 2007).

Land Management

The land within Suisun Marsh is managed primarily for waterfowl (USFWS, 1984). Although waterfowl management is very important, all native species that depend on this marsh should be protected and managed appropriately (USFWS, 1984). The recovery plan suggests that to achieve this goal "optimal freshwater and tidal flows" should be maintained to simulate historic flow regimes (USFWS, 1984). In addition, the CDFG and the USFWS should educate "property owners and hunters in waterfowl management areas" including Suisun Marsh about the presence and importance of CCRs (USFWS, 1984).

3.2.1.2 Hunting

Commercial and sport hunting of CCRs from 1850 to 1913 initially contributed to the depletion of the CCR (SFWS, 1984 and U.C. Berkeley, 2004), although there is no indication that hunting still impacts the CCR. The Migratory Bird Treaty Act of 1913, which restricted the harvest of game species, including CCR, allowed several rail populations to recover in the San Francisco Bay marshes; however, the CCR never fully recovered (USFWS, 1984; Albertson, 1996; and Goals Project, 2000).

3.2.1.3 Predation

Throughout their range, CCRs and their eggs are susceptible to mammalian and avian predators (USFWS, 2007). Non-native mammalian predators of the CCR and their eggs include Norway rats, red foxes, and feral cats and dogs (Environmental Impact and Goals Project, 2000). A study published in 1980 showed that rats were responsible for the loss of 24% of eggs in 50 nests, and a study of CCRs in three marshes in south San Francisco Bay, published in 1988 also suggested that Norway rats were responsible for 24% of nest failures (Goals Project, 2000). Another study in 1992 in the South Bay showed that rats took 31% of eggs in 54 nests (Goals Project, 2000). Raccoons (*Procyon lotor*) and skunks also prey on CCR adults, juveniles, and eggs (Berkeley, 2004). Gopher snakes (*Pituophis melanoleuces*) have been observed preying on CCR eggs (Schwarzbach et al., 2006). River otters (*Lutra Canadensis*) are found in Suisun Marsh and should also be considered to be a predator of the CCR (Goals Project, 2000).

Development in the areas surrounding the marsh displaces predators and also negatively affects higher order predators, including coyotes, which may limit the numbers of CCR predators (USFWS, 2007 and USFWS, 2003). An increase in the red fox population in the tidal marshes of the South Bay since 1986 has had a negative impact on CCR populations in this area and red foxes may pose the most serious threat to adult CCR (USFWS, 2007 and Goals Project, 2000). The population decline in several South Bay marshes (Greco Island, Bay Farm Island, Mowry Marsh, and Faber-Laumeister Marsh) after 1985 is likely due to predation by foxes and harriers (Foin et al., 1997).

The San Francisco Bay National Wildlife Refuge (SFBNWR) implemented a predator management program in 1991 (USFWS, 2003), which has reduced the impact of predators on some populations of the CCR (Albertson, 1996 and Steiner, 1997; Steiner, 1997). Other efforts are also underway to reduce non-native red fox populations (CDFG, 2000).

Avian predators of adult CCRs include northern harriers (*Circus cyaneus*), red-tailed hawks (*Buteo jamaicensis*), peregrine falcons (*Falco peregrinus*), and owls (USFWS, 1984; Foin et al., 1997). Predation of nests and eggs has also been attributed to ravens (Goals Project, 2000). Electric power transmission lines which cross tidal marshes provide hunting perches for avian predators allowing increased hunting efficiency and intensity of CCRs (USFWS, 2007 and USFWS, 2003). Remains of CCRs have been found under poles that have been used as perches (Foin et al., 1997). One study looking at predation showed that “64% of the rails killed were taken by raptors, primarily during the winter season” (Goals Project, 2000). Another study showed that 25% of the rail population in one marsh was taken by raptors during the period from April through November (Goals Project, 2000). The primary predator during the latter study was the barn owl (*Tyto alba*) (Goals Project, 2000).

3.2.1.4 Disease

It is currently unclear if the West Nile virus or other avian diseases will have an impact on the CCR (USFWS, 2007).

3.2.1.5 Climate Extremes

Fluctuations in rainfall and flooding, particularly during El Niño years, have the potential to affect the reproductive success of CCRs. However, while not all flooding events occur during El Niño years (and not all El Niño years cause widespread flooding), the potential for flooding during these years is indeed greater (Monteverdi, 1998). Populations of CCR may cyclically fluctuate with rainfall patterns, as do populations of the northern clapper rail in New Jersey (USFWS, 1984). In a study by Schwarzbach, *et al*. (2006), flooding was a minor factor in reproductive success, as opposed to major factors such as predation or contaminants. Flooding affected the number of eggs available to hatch by 2.3% (Schwarzbach, *et al*. 2006).

3.2.1.6 Habitat Protection and Restoration

A number of restoration, remediation, and habitat protection programs are ongoing. These are described in USFWS, 1984; Albertson, 1996; Goals Project, 2000; Berkely, 2004. Significant portions of CCR habitat in South San Francisco Bay have been acquired by the National Audubon Society, CDFG, and the USFWS (USFWS, 1984).

3.2.2 Baseline Status

Southern San Francisco Bay Area has historically supported the largest populations of CCRs. Salt marshes of the southern Bay Area included portions of San Mateo, Santa Clara, and Alameda Counties. Smaller populations occurred in San Francisco County and western Contra Costa County. Records indicated populations around Southampton Bay in Solano County, Napa Marsh in western Napa County, next to Petaluma in Sonoma County, and Tomales Bay in Marin County (counties all bordering the Bay). CCRs occurred further south of the San Francisco Bay Area in Monterey County (Elkhorn Slough). Reports from the 1930s and 1940s indicated populations in Humboldt County (Humboldt Bay) and San Luis Obispo County (Morro Bay), respectively (USFWS, 1984).

Currently, known CCR breeding populations are found only in tidal marshes in the San Francisco estuary (USFWS, 2007; Berkeley, 2004; and SFEISP online). The CCR only occurs in coastal wetlands in Alameda, Contra Costa, Marin, Napa, San Francisco, San Mateo, Santa Clara, Solano, and Sonoma counties, which form the San Francisco-Suisun Bay complex (CDPR online). Populations of CCRs can be found in all of the larger tidal marshes in south San Francisco Bay; however, the distribution of CCRs in the north Bay is fragmented (USFWS, 2007). Small populations of CCRs are found throughout San Pablo Bay (USFWS, 2007) and in the Suisun Marsh and surrounding areas (USFWS, 2007). Attachment III provides more details on the current range of CCR in the Suisan Marsh and Carquinez Strait, North Bay, Central Bay, and South Bay.

Population Dynamics
Historically the CCR population may have once been in the tens of thousands (Foin et al., 1997). Estimations of this number are between 4,000 and 48,000 (Foin et al., 1997). Based on surveys from 1971 to 1975, the entire CCR population was estimated at 4,200 to 6,000 individuals (USFWS, 1984 and Goals Project, 2000). This population was supposedly split with 55% occurring in South San Francisco Bay and 45% occurring in San Pablo Bay (USFWS, 1984). A breeding density of 1.6 CCR/ha in the South Bay was also described during this period (USFWS, 1984). A census reported in 1980 of two South Bay marshes supporting major CCR populations described a similar density of

CCR (1.5 CCR/ha) (USFWS, 1984). This consistency suggests that CCR populations have been stable since the 1970's (USFWS, 1984). However, CCR counts from the early to mid-1980's indicated that the CCR numbers were lower than those obtained in 1979 (USFWS, 1984). Greco Island and other large South Bay marshes that historically supported major CCR populations appeared to no longer provide optimal habitat and yielded lower numbers than would be expected by the large area of land (USFWS, 1984).

Censuses performed in the early to mid-1980s in San Pablo Bay established that these marshes do not support significant CCR populations even though this area represents 30% to 40% of the total CCR habitat (USFWS, 1984). This decline could be due to the growth of alkali bulrush over cordgrass attributed to above average winter precipitation (USFWS, 1984). The CCR population in the early to mid-1980s was approximately 50% lower than estimates from the early 1970s (USFWS, 1984). This decrease was attributed to the introduction of the red fox (*Vulpes vulpes regalis*) (Goals Project, 2000).

In 1990-1991, the CCR population was estimated at around 300 to 500 individuals (Save the Bay, 2005; and Goals Project, 2000). In 1993, the population was measured at around 800 individuals (Goals Project, 2000). The population increase in the South Bay was attributed to predator management that was implemented in 1991 (Goals Project, 2000). Surveys of the northern parts of the bay (Suisun and San Pablo bays) performed in 1992-1993 indicated a population of 390-564 individuals or 195-282 breeding pairs (Goals Project, 2000). CCR counts in the winter of 1997-1998 indicated that the South Bay population was estimated at 650 to 700 individuals (Goals Project, 2000). The estimated population today is about 1,200 individual animals and still growing (Save the Bay, 2005). It is estimated that the South Bay population is 500-600 individuals and that the population in the North Bay may be similar in size (CDFG, 2000). Another source estimates that the CCR population consists of 500-600 individuals in the southern portion of San Francisco Bay (USFWS, 2003).

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4: CALIFORNIA FRESHWATER SHRIMP

4.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the California freshwater shrimp (CA Shrimp) (*Rallus longirostris obsoletus*) that are reasonably certain to occur in the action area. Future federal actions unrelated to the proposed action are not considered because they are subject to consultation pursuant to Section 7 of the Endangered Species Act (Act). Numerous non-federal actions that could affect the CA shrimp are reasonably certain to occur within the action area. These activities are often associated with modifications to stream morphology and operate synergistically and/or cumulatively with each other in their effects to the CA shrimp. These activities include urban development, instream gravel mining, grazing, agricultural development, impoundments, and water diversions. These activities may simultaneously reduce the quality and availability of water, reduce the quality and availability of shelter, and contaminate suitable habitat. For instance, urban development may result in the loss of riparian vegetation. This could alter temperature regimes and dissolved oxygen levels in streams; increase bank erosions; and change the invertebrate species composition, impacting aquatic food source availability. The effects of urban development might then be compounded by activities that reduce water flow such as diversions and impoundments, which increase sediment and pollutant concentrations, and further decrease oxygen availability. As the CA shrimp does not have life history characteristics that favor quick recovery following disturbances, the CA shrimp is vulnerable to the cumulative effects of habitat degradation (USFWS, 1998).

4.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as the effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the California freshwater shrimp's status at this time. Details of the California freshwater shrimp's habitat description and known locations are included in Attachment III. However, the baseline condition of each assessed CA shrimp species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline presents a general discussion of factors that may affect CA shrimp within the action area. This information includes both detrimental and potentially beneficial factors affecting the species and is presented in Section 4.2.1. Additional information on the current distribution and population dynamics of the CA shrimp is presented as part of the baseline status in Section 4.2.2.

4.2.1 Factors affecting species environment within the action area

Since the species' listing in 1988, there continue to be numerous human-related and natural factors affecting the species' habitat (USFWS, 2007). Although there have not been any new threats documented since the listing, some of the factors have since intensified (USFWS, 1998). Furthermore, the CA shrimp is most likely affected by multiple factors in a majority of locations, rather than just one (USFWS, 1998). These factors include urbanization, stream maintenance, gravel mining, agriculture, silviculture, impoundments, water diversions, native and introduced predators, and water pollution (USFWS, 1998; USFWS, 2007). Environmental extremes, compounded by human disturbance, also present a threat to the CA shrimp (USFWS, 1998; USFWS, 2007). According to the species' 5-Year Review, overutilization for commercial, recreational, scientific, or educational purposes was not historically a factor affecting the CA shrimp at the time of its listing and no new information has come forward that would indicate a change in the imminence of this factor (USFWS, 2007). No information is available concerning the types of pathogens, parasites, or types of coexisting species that may associate with the CA shrimp (USFWS, 1998). Sections 4.2.1.1 through 4.2.1.8 provide details on each of the major issues affecting the CA shrimp within the action area.

4.2.1.1 Urbanization

On the heels of already much urban development since the CA shrimp's listing, urbanization has been increasing in rural areas within the San Francisco Bay region (USFWS, 1998). Development outside traditional populated centers has led, and will lead, to a larger portion of impacted watershed ultimately draining into CA shrimp habitat (USFWS, 1998). Urban development indirectly affects the CA shrimp due to the direct effects it has on stream quality (USFWS, 2007). The health of stream water is imperiled by the amount of non-point source pollutants in runoff, as well as the volume, temperature, and speed of runoff (USFWS, 2007). Urban runoff results from precipitation and from dry weather flows from irrigation (USFWS, 1998). Pollutants vary by locations, and can include sediment, petroleum, paint, household cleaning agents, and chlorine from pool water (USFWS, 2007). The acute and sublethal effects these pollutants may have on the species is unknown (USFWS, 2007).

Wastewater is another type of urban by-product marring stream health (USFWS, 1998). Wastewater discharges and releases from septic systems contribute to summer algal growth from nutrient loadings (USFWS, 1998). This growth and ensuing eutrophic conditions threaten the CA shrimp from reduced dissolved oxygen levels (USFWS, 1998). Failures in wasterwater treatment facilities may also result in discharges of partially treated effluent or chlorinated effluent (USFWS, 1998). A discharge of this nature, with ammonia and chlorine, occurred in 1993 in the Santa Rosa Creek and killed hundreds of small fish (USFWS, 1998).

4.2.1.2 Stream Maintenance

Stream maintenance activities that affect the species environment include flood control, bank protection, culvert, and grade control structures (USFWS, 1998). Flood control involves straightening, channelizing, and lining stream banks with concrete, effectively eliminating the flood plains, or riparian areas (USFWS, 1998). With such removal, CA shrimp habitat of undercut banks and riparian vegetation is removed, or at a minimum impaired, by increased water velocities and temperature fluctuations (USFWS, 1998). Bank protection activities result in similar consequences for CA shrimp habitat (USFWS, 1998). Bank protection refers to the practice of reinforcing stream banks to prevent natural erosion, and can be in the form of gabions or riprap providing stability (USFWS, 1998). Gabions, however, displace the herbaceous vegetation growing in the alluvial zones that provide warm season shelter for the species (USFWS, 1998). Loss of natural banks has occurred and can be expected to continue due to an increase in urban development (USFWS, 1998). Bank protection is also occurring without the requisite Army Corps of Engineers authorization of a section 404 permit that would include recommended measures to preserve CA shrimp habitat (USFWS, 1998). Other flood control methods degrade CA shrimp habitat though herbicide use, dredging, altering channel and bank configuration, and removing instream and riparian woody debris (USFWS, 1998).

Direct elimination of shrimp from certain streams could occur with barriers impeding the movement of CA shrimp upstream (USFWS, 1998). Sills, designed to protect bridge footing, have scoured downstream areas and formed ledges which prevent upstream movement (USFWS, 1998). Barriers from culvert or grade control structures also result in fragmented populations, possibly restricting gene flow and increasing the likelihood of extirpation (USFWS, 1998).

4.2.1.3 Water Resource Needs: Impoundments and Water Diversions

Impoundments are prevalent in stream drainage areas that support CA shrimp (USFWS, 1998). Impoundments, whether permanent or temporary, are constructed to reduce hazards, provide recreational benefits, and provide a water supply (USFWS, 1998). In damning a steam, however, impoundments directly and indirectly affect the CA shrimp by acting as migration barriers, by eliminating upstream habitat, supporting introduced predators, altering hydrology and sediment transport, reducing and altering downsteam flows, and by ushering individuals away from the population and into diverted waterways (USFWS, 1998). Impoundments raise the elevation zone, drown roots of riparian vegetation not adapted to higher inundation, and likely reduce riparian vegetation along the shore (USFWS, 2007). Lack of riparian vegetation harms the species by reducing habitat complexity, reducing detritus production, elimination of high flow refugia, and increasing bank scouring (USFWS, 2007). The natural winter flood cycle, needed to maintain undercut banks and pools for the CA shrimp, is also impacted by such water storage facilities (USFWS, 1998).

Water diversions imperil CA shrimp by directly reducing water flow, as well as by intensifying the concentrations, and thus effects, from pollutants (USFWS, 1998). At the time the recovery plan was drafted, there were substantial demands to appropriate water from many streams containing the CA shrimp (USFWS, 1998). In the Huichica Creek watershed, as an example, estimates of total water yield range between 1,759 acre-feet to 3,097 acre-feet (USFWS, 1998). If full permission were given to appropriate the 2,019 acre-feet of water that was requested by local landowners, the yearly volume of water in the creek would be reduced by two-thirds (USFWS, 1998). Without instream flow requirements, existing pools could become dry (USFWS, 1998). Such high demands of groundwater are also a concern, although for the fear of saltwater intrusion and the corollary stress of competing with shrimp found normally in brackish water (USFWS, 1998).

Since 1998, the USFWS has issued 13 biological opinions under section 7 of the Endangered Species Act authorizing incidental take of the shrimp (USFWS, 2007). However, not all projects are consulted on under section 7 (USFWS, 2007). It was discovered that at least one project since 1998, an illegally constructed water diversion (not consulted on), may have impacted the CA shrimp (USFWS, 2007). The water diversion channel included the removal of riparian vegetation along the bank of Olema Creek, as well as foraging and sheltering habitat for the CA shrimp (USFWS, 2007).

4.2.1.4 Gravel Mining

Gravel mining practices interrupt the supply of gravel downstream and can disrupt the natural channel geomorphology (USFWS, 1998). Long-term gravel mining retards the development of appropriate soil conditions for riparian vegetation, which under normal conditions would be the accumulation of fine sediments and organic materials from overbank flooding (USFWS, 1998). Without necessary safeguards, instream gravel mining activities along the historic CA shrimp habitat in Austin and Sonoma Creeks will prevent the reestablishment of suitable habitat conditions for the species (USFWS, 1998). To date, no specific studies have been conducted to determine the extent of effects of gravel mining on the CA shrimp (USFWS, 2007).

4.2.1.5 Agriculture Developments

Recent vineyard establishments have caused concern for the existence of the CA shrimp (USFWS, 1998). Vineyard coverage in Napa County is anticipated to reach 52,000 acres by 2010, approximately 19,000 acres more than the area covered in 1989 (USFWS, 1998). Land and surface water resources in not only Napa, but also Sonoma, counties are being intensively developed, and threats to CA shrimp populations include loss of riparian vegetation, inadvertent introduction of herbicides and pesticides into creek water, diversion of water, and increase soil erosion (USFWS, 1998).

Habitat quality for the CA shrimp is imperiled by livestock grazing operations and dairy farming (USFWS, 1998). Effects of these operations include riparian vegetation removal, stream bank and channel changes, decreased water quality, and increased water temperature fluctuations (USFWS, 1998). Stemple Creek watershed, as an example, supports approximately 30 dairy operations and grazing on 50 percent of its land (USFWS, 1998). Grazing activities typically occurs in the vicinity of steams and riparian areas where livestick are offered fresh water and palatable forage (USFWS, 1998). Because of the high concentration of livestock, Stemple Creek and its tributaries have lost much of their riparian vegetation that supported CA shrimp habitat (USFWS, 1998). In addition to affecting habitat suitability, loss of riparian vegetation impacts the availability of food sources (USFWS, 1998). Changes in the natural riparian zones alter invertebrate species composition and production in steams (USFWS, 1998). Grazed streams are characterized by having a high production of algae due to high insulation and increased nutrient input, as opposed to the limited instream primary productivity seen in streams with dense riparian vegetation (USFWS, 1998).

Poor water quality results from reduced forage cover and increased soil compaction (USFWS, 1998). Groundwater recharge is diminished from the compaction, as well as base flows during the summer and fall (USFWS, 1998). Reduced forage cover and compaction also result in higher peak flows following winter storms and high sediment-laden flood flows (USFWS, 1998). Runoff from manure lots contribute to raising nutrient levels and increase the concentration of ammonia (USFWS, 1998). Water samples from Americano and Stemple Creeks exceeded the Environmental Protection Agency's (EPA) ammonia criteria for aquatic life protection for the years 1991-1994 (recovery). The ammonia present is in an un-ionized form (NH_3) and an ionized form (NH_4^+), both of which cause mortality to aquatic organisms (USFWS, 2007). The elevated copper concentrations in these two creeks have been linked to dairy practices, and levels have been detected that also exceed the EPA's criterion for aquatic life protection (USFWS, 1998).

Timber harvesting, or silviculture, has and may continue to impact CA shrimp populations, although it is reported to be less of a threat today than it has been historically (USFWS, 1998; USFWS, 2007). Particularly those practices that remove stream side vegetation, timber harvesting reduces channel stability, decreases canopy cover (increasing water temperature), and increasing instream debris (USFWS, 1998). Timber harvesting also increases peak flows, decreases base flows, and increases sediment transport and deposition (USFWS, 1998). Austin creek is noted as having timber harvesting activity near its drainage, adding to channel degradation near the confluence with the Russian River (USFWS, 1998).

4.2.1.6 Predation

It can be inferred that the CA shrimp evolved in the presence of predators due to its cryptic coloration and behavioral characteristics (USFWS, 1998). Some native fish that prey on CA shrimp include the California roach (*Hesperoleucus symmetricus*), threespine stickleback (*Gasterostreus aculeatus*), riffle sculpin (*Cottus gulosus*) (USFWS, 1998). It is suspected that the following fish generally prey on shrimp: young coho samlmon (*Oncorhynchus kisutch*), steelhead trout *(Oncorhynchus mykiss*), Sacramento squawfish (*Ptchoncheilus grandis*), and northern squawfish (*Ptychocheilus oregonensis*) (USFWS, 1998). Other native aquatic predators may include western pond turtles, salamanders, newts, water scorpions, diving beetles, and dragonfly and damselfly nymphs (USFWS, 1998).

Nonnative fish have been introduced to the habitat of the CA shrimp and are implicated in the decline of native fish (USFWS, 1998). Nonnative species presence is correlated with the levels of human disturbance to a stream (USFWS, 1998). Introduced fish that potentially prey on the CA shrimp include mosquito fish (*Gambusia affinis*), green sunfish (*Lepomis cyanellus*), smallmouth bass (*Micropterus dolomieui*), largemouth bass (*Micropterus salmoides*), common carp (*Cyprinus carpio*), and certain species of minnows (USFWS, 1998). The introduced Chinese mitten crab (*Eriocheir sinensis*) is an invertebrate predator that is established in tributaries flowing into San Pablo Bay and may potentially prey on the CA shrimp (USFWS, 2007). Predation from introduced species can be more disruptive to a prey population than from native predators because the prey species have not developed mechanisms for defense or avoidance (USFWS, 1998). Some introduced species are also tolerant of poor water quality conditions, increasing the predation stress on their prey items (recovery). Both the mosquitofish and green sunfish persist in conditions that would otherwise permit CA shrimp to live in isolation of natural predators (USFWS, 1998). For example, the green sunfish is capable of surviving water temperatures up to 97 degrees Fahrenheit, low oxygen levels down to 3 ppm, and high alkalinities (USFWS, 1998).

4.2.1.7 Environmental Extremes

Environmental extremes, particularly droughts, spring flooding, and the consequences of climate change, threaten the persistence and stability of CA shrimp populations (USFWS, 2007). Reduced precipitation could result in lower stream flows and thus increase the chance that stream segments dry out during summer months. Natural events can devastate CA shrimp populations because they can cause local extinctions upon which recolonization is difficult (USFWS, 2007). A second, compounding factor of droughts is that with reduced precipitation comes an increased demand for water resources, further intensifying the fatal effects of stream dry-up (USFWS, 2007).

While not mentioned explicitly in the recovery plan, climate change induced sea level rise could impact the CA shrimp (USFWS, 2007). Sea level rise may result in increasing salinities in reaches supporting the CA Shrimp (USFWS, 2007). Hodgpeth (1968) observed shrimp surviving at 16 – 17 parts per thousand for up to 13 days, and mortality occurring after 7 hours in higher salinities (Hedgpeth, 1968).

4.2.1.8 Restoration Efforts

Since 1998, the USFWS has issued 13 biological opinions under section 7 of the Endangered Species Act authorizing take of the shrimp in the form of 1,048 linear feet and 3.05 acres of shrimp habitat (USFWS, 2007). The biological opinions for these projects have included a total of 548 linear feet combined with 5.49 acres of habitat restoration and/or protection in compensation for adverse effect (USFWS, 2007). Watershed plans for many streams sustaining CA shrimp population also strive to protect and restore habitat for the species, but many restoration activities for these plans have yet to be implemented (USFWS, 2007). Stemple Creek has received lots of restoration attention through activities supported by the Bay Institute and executed by the Students and Teachers Restoring Watershed (STRAW) program (USFWS, 2007). The projects aim to remove nonnative vegetation, plant native species, erect livestock exclusion fencing, and install cattle bridges (USFWS, 2007). To date, approximately 50,000 linear feet of stream bank has been restored through STRAW's 185 completed projects (USFWS, 2007).

4.2.2 Baseline Status

Long-term population estimates are available for the Lagunitas Creek population, known from an eight mile reach beginning from Shafter Bridge to just downstream of the Balleager Bridge U.S.Geological Survey gage (USFWS, 2007). The number of individuals collected at six sites increased from approximately 1,878 in 1991 to approximately 4,407 in 2000, the same period when (USFWS, 2007) the stream's morphology changed allowing increased water flows and improving and/or increasing habitat conditions (USFWS, 2007). However, in some areas these morphological changes worsened and/or decreased habitat conditions (USFWS, 2007).

Population Dynamics

The recovery plan listed populations of the California freshwater shrimp remaining in reaches of 17 streams (USFWS, 1998). These included five streams in Marin County (Lagunitas, Olema, Walker, Keys, and Stemple), and twelve streams in Sonoma County (Blucher, Jonive, Redwood, Green Valley, Salmon, East Austin, Big Austin, Sonoma, Yulupa, Garnett, Huichica, and Napa (USFWS, 1998). As of the 5-year Review, more recent surveys place the number of known locations of the CA shrimp up to 23. In addition to those already mentioned, the five reaches of streams known to support the CA shrimp are "Bud Creek" (Sonoma County), Franz Creek (Sonoma County), Ebabias Creek (Sonoma County), Cheda Creek (Marin County, an unnamed tributary of Huichica Creek (Napa County), and a second location on the Napa River (Napa County) (USFWS, 2007). The majority of new information documenting the existence of the species in these additional streams is a result of recent independent surveys. It should be noted, these findings do not represent a systematic canvassing of habitats for the CA shrimp that examines their full spatial distribution and extent (USFWS, 2007).

It is unknown if shrimp populations still persist in Laguna de Santa Rosa or Atascadero Creeks. The Yulupa Creek shrimp population is probably under the greatest threat of extirpation (USFWS, 1998). Lagunitas is the only shrimp stream on federal and state land, all others are in private ownership (Serpa, 1996 and USFWS, 1998). A substantial portion of Lagunitas Creek flows through the Samuel P. Taylor State Park, managed by the California Department of Parks and Recreation, and the Golden Gate National Recreation Area, managed by the National Park Service (USFWS, 1998). A small segment of Salmon Creek flows through the Watson School historic site, managed by the Sonoma County Department of Parks and Recreation (USFWS, 1998). On East Austin Creek, the Austin Creek State Recreation Area lies immediately upstream of shrimp populations (USFWS, 1998).

Preliminary genetic analysis on approximately 12 individuals from eight streams indicates populations can be divided into distinct groups based on genetic similarities of mitochondrial DNA (USFWS, 2007). These tentative data suggest genetic variation between populations may not correspond to the drainage units identified in the recovery plan (USFWS, 2007). The drainage units identified in the recovery plan were originally developed in an effort to maximize genetic diversity (USFWS, 2007).

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5: CALIFORNIA TIGER SALAMANDER: CENTRAL CALIFORNIA DISTINCT POPULATION SEGMENT

5.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the central California distinct population segment (DPS) of the California tiger salamander (*Ambystoma californiense*) (central CTS) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of intensive agriculture (*i.e.*, vineyards, row crops, and, orchards), urban development, grazing operations, oil production, and contamination from runoff and spills. Many of these factors are linked or act synergistically and create complex consequences for the central CTS. For example, conversions to intensive agriculture and urban developments have resulted in the loss of breeding habitat from the destruction or alteration of natural vernal pools and seasonal ponds. If these water bodies are destroyed by deep ripping, a technique that uses a 4- to 7-foot deep plow to break up the hardpan layer, the once compacted soil will no longer retain water and will inhibit ponding. Repeated deep-ripping, in addition to plowing and discing in upland habitats, can alter hydrology of the pools and ultimately destroy them. In contrast to destroying water bodies, some seasonal ponds are being converted into permanent ponds for irrigation. However, these ponds are then frequently stocked with fish to control mosquitoes and other problematic insects, and can limit central CTS populating the water edges. Both destruction and reduced suitability of breeding habitat, compounded with increased urbanization, fragment and isolate subpopulations from each other and inhibit potentially beneficial gene flow (USFWS, 2004).

5.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the central CTS's status at this time. However, the baseline condition of the central CTS population and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included in this assessment, the environmental baseline includes a general discussion of factors that may affect central CTS within the action area. This information is presented in Section 5.2.1. Additional information on the current distribution and population dynamics of the central CTS is presented as part of the baseline status in Section 5.2.2 (USFWS, 2004).

5.2.1 Factors affecting species continued existence within the action area

The central CTS's continued existence is primarily affected by the destruction, modification, or curtailment of suitable habitat due to rangeland conversion for urban and agricultural uses. Other influential factors affecting central CTS include: disease; predation, competition, introduced species; contaminants; pest control; livestock grazing; and vehicular strikes. As opposed to many of these factors that negatively affect the CTS, grazing can, in moderation, have a neutral to positive impact. While some of these factors are known to specifically affect the California Tiger Salamander population in central California (central CTS), others are known to generally affect the California Tiger Salamander (CTS) population throughout its entire range. The factors known to affect the larger population are still relevant to the central CTS, but are referred to as CTS. Sections 5.2.1.1- 5.2.1.4 provide details on each of the major issues affecting central CTS environment within the action area (USFWS, 2004).

5.2.1.1 Rangeland Conversion and Habitat Fragmentation

A variety of urban and agricultural land uses are implicated in the destruction, modification, and curtailment of central CTS habitat (USFWS, 2004). The United States Fish and Wildlife Service defines urban uses as including nonagricultural development activities such as building and maintenance of housing, commercial, and industrial developments; construction and widening of roads and highways; golf course construction and maintenance; landfill operation and expansion; operation of gravel mines and quarries; dam building and inundation of habitat by reservoirs; and other infrastructure activities (USFWS, 2004). Agricultural land uses are defined as including native habitat conversion by discing and deep-ripping; and cultivation, planting, irrigation, and maintenance of row crops, orchards, and vineyards (USFWS, 2004). Both urban and agricultural conversions have reduced or modified natural vernal pools and seasonal ponds used for breeding habitats, as well as reduced upland habitat used for aestivation and migration (USFWS, 2000).

Urban development in central CTS habitat proceeds at an elevated rate. Between the years of 1980 and 1997, 2.8 million new housing units were constructed in California, and 220,000 units are required each year for the next 20 years to accommodate the growing population. Most of this development will be outside of the major metropolitan areas, referring to San Francisco, Los Angeles, and San Diego, and will be occurring in the Sacramento, San Joaquin, and Imperial Valleys. Two of these valleys, the Sacramento and San Joaquin Valleys, are inhabited by the central CTS. Low Density residential housing, found outside major metropolitan areas, is of great threat to the central CTS. The housing areas and rural developments may result in extirpation of the CTS due to the destruction or degradation of breeding sites, upland dispersal habitat, and migration corridors (USFWS, 2004).

Conversion of rangeland and vernal pool grasslands to intensive agriculture have been well documented in counties inhabited by the central CTS. The cumulative loss of vernal pool grasslands has been estimated at 78 percent by the later 1990s. Historically, approximately 2.7 million hectares (ha) of valley and coastal grasslands existed within the range of the central CTS. Today, virtually all valley grassland and oak savanna habitat has been eliminated from the Central Valley floor due to urbanization and intensive agriculture. It is estimated that less than 10 percent of California's Central Valley grasslands remain. Future conversion is difficult to estimate, but it is predicted to be a continuing threat to the central CTS because of the projected increase in human population. Furthermore, there is a financial incentive to convert rangeland into irrigated crops as rangeland in central CTS habitat is valued (value/acre) at a lower cost. Other factors influencing rangeland conversion include a decline in farm rancher income, the aging of ranchers, tax implications of intergenerational transfers of ranches, and the difficulty of beginning a ranching operation (*e.g*., cost and knowledge). However, the rate of conversion may soon decrease as areas with suitable soils and water availability necessary for intensive agriculture become scarce (USFWS, 2004).

Changes to vernal pool or pond inundation, duration, or depth can detrimentally affect the reproductive success of the CTS. These changes are caused by, among others, digging of drainage or irrigation ditches, construction of permanent ponds or reservoirs, deepening or berming of seasonal wetlands, and redirection or runoff from developments (USFWS, 2000). The repercussions of these activities on the reproductive success of the CTS include: (1) prematurely drying wetlands and desiccating CTS larvae; (2) extending the inundation period and facilitation invasion of non-native predators; (3) creating conditions that foster the hybridization with the non-native tiger salamander; and (4) increasing vulnerability to disease from isolation and fragmentation. Wetland habitats are utilized for reproduction; thus destructive activities such as filling, discing, or excavating, are injurious or lethal to the eggs, larvae, or breeding adults. Earthmoving operations and cultivation in upland habitat also threaten the CTS, which use the habitat to breed, feed, and seek shelter. CTS are at risk in burrows or on the surface from discing, deep-ripping, or grading of upland habitat. Indirect impacts to the CTS include increased predation from over-exposure, high temperatures, low humidity, and diminished food sources. Earth moving operations may further disturb California ground squirrel burrows and crevices, protected habitat utilized by the CTS, and may render upland locations untenable or unavailable (USFWS, 2004).

Fragmentation

The widespread conversion of land to residential and agricultural uses has led to the fragmentation and isolation of habitat throughout the range of the central CTS. Human-caused fragmentation places the CTS at particular risk to local extinctions because its metapopulation structure is distributed across the landscape (USFWS, 2000).

> "A metapopulation is a group of spatially distinct populations that can occasionally exchange dispersing individuals. The populations in a metapopulation are usually thought of as having interdependent extinction and colonization processes, where individual populations may be extirpated from a local area and later be recolonized from another population that is still extant" (FR, 72:48177, p.6).

A reduction in CTS dispersal greatly inhibits the dynamics of a metapopulation and reduces the abilities of the CTS to persist over time. Of all factors associated with urbanization and land conversion, roads accelerate fragmentation by increasing mortality and preventing recolonization of sites that would otherwise be only temporarily extirpated (USFWS, 2000).

5.2.1.2 Disease

The direct effect of disease on the CTS is unknown and risks to the central California DPS cannot be precluded (USFWS, 2000). While no information suggests disease is prolific in existing populations in California, several pathogenic (disease-causing) agents have been linked to deaths of closely-related tiger salamanders and other amphibian species. These pathogens include fungi, viruses, and at least one bacterium (*Acinetobacter* spp.) (USFWS, 2000; USFWS, 2004). Chytrid fungus infections (chytridiomycosis) have been detected specifically in Central California tiger salamanders, although the prevalence and effect on the species is unknown (USFWS, 2004).

The bacterium *Acinetobacter* is common in soil and animal feces (USFWS, 2000). Affected tiger salamanders showed symptoms of red, swollen hind legs and vents, and hemorrhaging of skin and internal organs. Iridoviruses, (a family, containing the genus *Ranavirus*) are viruses with DNA as the genetic material that occur in insects, fish, and amphibians, including tiger salamanders (Harp, 2006; USFWS, 2000). These viruses can cause a range of symptoms, including skin lesions and mortality, or they can be asymptomatic (USFWS, 2000). Ranaviruses are of particular concern because these viruses can be spread through fishing gear, artificial stock ponds, and water birds (USFWS, 2003). Outbreaks of ranavirus in tiger salamanders have been directly associated with altered habitats and artificial ponds (USFWS, 2003). Ranaviruses, in addition to the fungus *Batrachochytrium dendrobatidis* that causes Chytridiomycosis, have been identified as potential threats to the CTS because these pathogens have adversely impacted other amphibians, including tiger salamanders (USFWS, 2004). Non-native species of bullfrogs and tiger salamanders are potential carriers of these diseases (USFWS, 2004).

A study conducted in 1997 showed that tiger salamanders in Georgia reared in ponds contaminated with silt were susceptible to infection by the fungus *Saprolegnia parasitica* (USFWS, 2003). Die offs associated with western toads and Pacific tree frogs, known prey of the CTS, have been associated with *Saprolegnia* infections (USFWS, 2000). High nitrogen and silt levels from overgrazing or agricultural or urban runoff may also increase susceptibility to disease (USFWS, 2000). High levels of nitrogen have been associated with bacterial blooms and increased silt has been associated with fungal infections (USFWS, 2000).

5.2.1.3 Predation, Competition, and Hybridization

Native predators of the CTS include great blue herons (*Ardea herodias*), egrets (*Casmerodius*), western pond turtles (*Clemmys marmorata*), garter snakes (*Thamnophis* spp.), larger CTS larvae, larger spadefoot toads (*Scaphiopus hammondii*), and California red-legged frogs (*Rana draytonii*) (USFWS, 2004). Native as well as non-native crayfish also prey on CTS (USFWS, 2003). While native predation and hybridization is probably a discountable threat in healthy populations, these native predators may have significant impacts on population viability when populations are subjected to the cumulative effects of contaminants, migration barriers, and/or habitat alteration (USFWS, 2003).

Predation by and competition with non-native and introduced species contributes to the decline of the CTS (USFWS, 2003). Non-native predators of both adult and larval CTS include Louisiana red swamp crayfish (*Procambarus Clarkii*), mosquitofish (*Gambusia affinis*), sunfish spp. (*e.g.*, large mouth bass (*Micropterus salmoides*)) bluegill (*Lepomis macrochirus*), catfish spp. (*Ictalurus* spp.), fathead minnow (*Pimephales promelas*), carp (*Cyprinus carpio*), and American bullfrog (*Rana catesbeiana*) (USFWS, 2004; USFWS, 2007). Bullfrogs are known to prey on CTS larvae and studies have shown that the number of CTS decreases as bullfrog numbers increase (USFWS, 2000). A study observing the effect of bullfrogs on the CRLF, a prey item of CTS, showed that less than 5% of CRLF tadpoles survived to metamorphosis when they were raised with bullfrog tadpoles (USFWS, 2000). Because bullfrogs are still used for food and sport in California, the threat of transport for intentional establishment of bullfrogs in central CTS suitable habitat is significant (USFWS, 2004).

Western mosquito fish, which are used to control mosquitoes because they eat mosquito larvae, are also known to prey on amphibian species (USFWS, 2003). Incidentally, field and laboratory experiments have shown that mosquito fish will preferentially prey on amphibians even in the presence of mosquito larvae (USFWS, 2003). It is not known if mosquito fish target CTS specifically; however, larval CTS may be particularly at risk due to their external gills which are attractive to mosquito fish (USFWS, 2003). A study performed in 1994 found that no CTS had been found in ponds where mosquito fish were located (USFWS, 2003). However, another study performed in 2000 found that CTS numbers were reduced in ponds stocked with mosquito fish at densities similar to those found in many stock ponds (USFWS, 2004a). CTS larvae found in ponds containing mosquito fish were smaller, took longer to metamorphose, and typically suffered injuries, such as shortened tails (USFWS, 2004a). Another study performed in 2003 showed that at low densities mosquito fish did not have a significant impact on larval CTS growth and survival, but that growth and size at metamorphosis was reduced at high fish densities (USFWS, 2004a). It is thought that large numbers of mosquito fish may also out-compete CTS larvae for food (USFWS, 2000).

Other species of non-native fish, including largemouth bass, bluegill, catfish, and fathead minnows may also either be directly responsible for or have the potential to cause the decline of the CTS (USFWS, 2003). In eastern Merced County, inhabited by the central CTS, CTS were absent from stock ponds that contained non-native fish, but stock ponds without non-native fish supported populations of CTS. These non-native fish may affect the prey base of the CTS or prey on larval CTS, either of which is capable of reducing or eliminating CTS populations (USFWS, 2000). Several of these species, including largemouth bass, bluegill, catfish, and bullheads, have been and still are stocked in ponds throughout California for fishing (USWFWS, 2003).

Hybridization

Hybridization is a serious threat in the Central Coast region of the CTS. Virtually all CTS populations within this region have hybrid genes; introduced genes have been found from southern Santa Clara County through most of Monterey County down to Fort Hunter Liggett on the San Luis Obispo County Line. Hybridization was studied at a site in Monterey County between the central CTS and non-native tiger salamanders. Researchers found clear evidence that the CTS was interbreeding in the wild and were producing viable, hybrid offspring. The extent of interbreeding, though, was found to depend on habitat quality. Pure central CTS were found to be most prevalent in natural habitats, rather than artificial or disturbed ones. According to the study, vernal pools contained a larger percentage of pure CTS larvae. The authors suggest that more hybrids are found in artificial ponds because of the lack of barriers preventing gene exchange. In the last decade, hybrid tissue has been documented for the first time in Fort Ord, the upper Carmel Valley, and Merced County, where pure CTS individuals had been originally found. Hybridization and introgression is an important evolutionary component, but it becomes a concern when it is fostered by human disturbance activities (USFWS, 2004).

5.2.1.4 Other Factors affecting the species continued existence

Chemical Contaminants

Contaminants from oil production and road runoff (*e.g*., hydrocarbons) directly and indirectly affect the CTS. The runoff from roads (*e.g*., oil and other contaminants) is found in adjacent ponds and is implicated in amphibian deformities, as well as amphibian and invertebrate mortality (USFWS 2000). CTS are particularly vulnerable because they inhabit both aquatic and terrestrial habitats which expose them to a variety of toxins throughout their life cycle. Their permeable skin further increases their vulnerability (USFWS, 2004), and studies have found developmental effects in marbled (*Ambystoma opacum*) and eastern tiger salamanders (*A. tigrinum*), and limited direct effects in five week old salamanders. Study results indicated that salamanders from oil-contaminated natural ponds metamorphosed earlier at smaller sizes and that those from oil-contaminated artificial ponds had slower growth rates than larvae raised in non-contaminated ponds. However, affects to eggs and early life stages were not addressed where effects could be more severe. Other studies have examined the toxicity of fluoranthene, a polycyclic aromatic hydrocarbon present in petroleum products and urban runoff, on spotted salamanders (*A. maculatum*), northern leopard frogs (*Rana pipiens*), and African clawed frogs (*Xenopus laevis*). Researchers evaluated concentrations at levels commensurate to those measured in service stations and other urban runoff, and found reduced survival and growth abnormalities in all species under field and laboratory conditions. Effects were worsened when the larvae were exposed to fluoranthene in natural sunlight rather than in artificial light (USFWS, 2000).

Sedimentation from road construction, maintenance, and runoff is a second type of contaminant threatening the CTS. Breeding ponds are affected by the altered hydrology near roads, which leads to erosion, gullies, and increased sediment deposits in wetland systems. Increased dust from traffic affects aquatic and emergent vegetation and may asphyxiate CTS eggs. Excessive sedimentation ultimately fills pools otherwise usable by the CTS. Sedimentation may also impair the CTS from detecting food items (USFWS, 2000). The risk factor associated with contaminants in runoff is of concern and will likely increase in both roadside ditches and the rest of CTS suitable habitat (USFWS, 2003).

Rodent Control
CTS utilize ground squirrel and pocket gopher burrows for aestivation (USFWS, 2003). While the CTS is probably not at risk from ingesting the rodenticides used to control these two species, the CTS may potentially be at risk from indirect exposure to them inside the burrows or concentrated in breeding ponds (USFWS, 2003). Some of these rodenticides used in central CTS habitat include chlorophacinone, diphacinone, and strychnine (USFWS, 2003). Fumigants are other pesticides used for rodent control, such as aluminum phosphide, carbon monoxide, and methyl bromide, which are injected into burrows by using a cartridge or by pumping (USFWS, 2003). These fumigants can be both directly and indirectly lethal to the CTS, although the effects of these poisons on CTS have not been assessed (USFWS, 2003). Besides the use of rodenticides, control of rodents by means of eliminating burrows results is a loss of tenable habitat for the CTS (USFWS, 2003). Cattle owners, for example, destroy ground squirrel burrows and thus eliminate CTS habitat because the burrows put cattle at risk of breaking their legs (USFWS, 2003).

Livestock Grazing
As opposed to many factors that negatively affect the CTS, grazing can, in moderation, have a neutral to positive impact on the CTS. Grazing maintains shorter vegetation and promotes the persistence of ground squirrel burrowing, which creates habitat essential for the CTS. In some cases rangelands present the only undeveloped habitat in the County, and thus provided the only opportunity for the salamanders to maintain sustainable populations (USFWS, 2000). Rangelands not only provide natural vernal pools, but artificial stock ponds which have likely saved many populations from extirpation (USFWS, 2004). Although trampling from cattle affect water levels and soil integrity of ponds and banks, grazing is still compatible with CTS as long as burrowing rodents are not completely eradicated (USFWS, 2004).

Road-Crossing Mortality
It is well documented that numerous CTS are killed by vehicular traffic while crossing roads (USFWS, 2003). In one study, there were 45 CTS collected during one hour period on a road bordering Lake Lagunita on the Stanford University Campus. Of the 45 collected, 28 had been killed by cars. Estimations of CTS mortality from vehicular strikes range from 25 to 72 percent. Mortality may be increased by the presence of curbs and berms, which allow salamanders to climb onto the road but can restrict their

movement off of the road (USFWS, 2003). Vehicle usage in California is also increasing as the human population continues to grow. From 1972 to 2001, the State highway system total vehicular usage rose from 108.6 kilometers (km) to 270,000,000,000 (270 billion) km. It is believed that the relatively high road-use and road-density values make road-kill mortality a threat to the species (USFWS, 2004).

5.2.2 Baseline Status

The CTS is found only in California. The historic range of the CTS included large portions of the Central Valley of California from the southern Sacramento Valley (north of the Sacramento River delta) to the southern San Joaquin Valley (USFWS, 2002). The CTS was also found in the lower foothills along the eastern side of the Central Valley and in the foothills of the Coast Ranges (USFWS, 2002). CTS have been historically documented in 27 counties but are no longer found in three (USFWS, 2005). The USFWS believes that the CTS is still found in the remaining 24 counties (USFWS, 2005). CTS have been found in 10 of the 17 California vernal pool designations defined by Keeler-Wolf in 1998 (USFWS, 2004).

Although the CTS has been eliminated from much of its former range in the Central Valley, it still occurs throughout most of its overall historical range and can be locally common (Hammerson, 2004). In addition, it has been recently rediscovered on the San Francisco Peninsula (Hammerson, 2004). The quality, connectivity, and distribution of the habitat within the current range has been degraded, even though the current range of the CTS is close in size to its historic range (USFWS, 2004).

Population Dynamics

Studies suggest that the present patchy distribution pattern of CTS inhabited areas was caused by a combination of extreme anthropogenic changes in and around the Central Valley and the restrictive breeding requirements of the species. Geographic Information Systems (GIS) indicate that there are approximately 378,882 ha of Central CTS upland and aquatic habitat remaining. Of the total, 28,526 ha fall within planned areas for high-density residential, medium density residential, industrial, or commercial developments. An additional 24,240 ha are slated for low-density residential development, and 45,880 intended for very-low-density residential development (20 to 160 acre parcels). These projected developments may destroy and fragment central CTS upland and aquatic breeding habitat and ultimately reduce the long-term persistence at the affected localities. In contrast to the detrimental development, part of the central CTS habitat is being protected to some degree. There is an estimated 76,501 ha, 20 percent of the total inhabited area, which is protected from further development or land conversion. However, these sites vary in their degree of protection due to land use designation, non-native predators, agriculture and landscaping contaminants, rodent control, roads, and hybridization. Thus, the determined “protected” habitat is considered a liberal estimation (USFWS, 2004).

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6: CALIFORNIA TIGER SALAMANDER: SANTA BARBARA COUNTY DISTINCT POPULATION SEGMENT

6.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the Santa Barbara County (SB) distinct population segment (DPS) of the California tiger salamander (*Ambystoma californiense*) (SB CTS) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of intensive agriculture (*i.e*., vineyards, row crops, and flower), urban development, grazing operations, oil production, and contamination from runoff and spills. Many of these factors are linked or act synergistically and create complex consequences for the SB CTS. For example, conversions to intensive agriculture and urban developments have resulted in the loss of breeding habitat from the destruction or alteration of natural vernal pools and seasonal ponds. If these water bodies are destroyed by deep ripping, a technique that uses a 4- to 7-foot deep plow to break up the hardpan layer, the once compacted soil will no longer retain water and will inhibit ponding. Repeated deep-ripping, in addition to plowing and discing in upland habitats, can alter hydrology of the pools and ultimately destroy them. In contrast to destroying water bodies, some seasonal ponds are being converted into permanent ponds for irrigation. However, these ponds are then frequently stocked with fish to control mosquitoes and other problematic insects, and can limit SB CTS populating the water edges. Both destruction and reduced suitability of breeding habitat, compounded with increased urbanization, fragments and isolates subpopulations from each other and inhibits beneficial gene flow.

6.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the SB CTS's status at this time. However, the baseline condition of each assessed SB CTS species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included in this assessment, the environmental baseline includes a general discussion of factors that may affect SB CTS within the action area. This information is presented in Section 6.2.1. Additional information on the current distribution and population dynamics of the SB CTS is presented as part of the baseline status in Section 6.2.2.

6.2.1 Factors affecting species continued existence within the action area

SB CTS's continued existence is primarily affected by the destruction, modification, or curtailment of suitable habitat due to rangeland conversion for urban and agricultural uses, with road construction, intensive agriculture, and urbanization constituting the major three uses. Other influential factors affecting SB CTS include contaminants; pest control; grazing practices and water drawdown; predation and introduced species; and disease. As opposed to many of these factors that negatively affect the SB CTS, grazing can, in moderation, have a neutral to positive impact on the SB CTS. Some of these factors are known to affect the SB CTS, while others are known to generally affect the CTS population throughout its entire range. The factors known to affect the larger population are still relevant to the SB CTS, but are referred to as CTS. Sections 6.2.1.1-6.2.1.6 provide details on each of the major issues affecting SB CTS environment within the action area.

6.2.1.1 Conversion of rangeland for urban and agricultural uses

A variety of urban and agricultural land uses are implicated in the destruction, modification, and curtailment of SB CTS habitat. The USFWS (2004a) defines urban uses as including nonagricultural development activities such as building and maintenance of housing, commercial, and industrial developments; construction and widening of roads and highways; golf course construction and maintenance; landfill operation and expansion; operation of gravel mines and quarries; dam building and inundation of habitat by reservoirs; and other infrastructure activities. Agricultural land uses are defined as including native habitat conversion by discing and deep-ripping; and cultivation, planting, irrigation, and maintenance of row crops, orchards, and vineyards. Both urban and agricultural conversions have reduced or modified natural vernal pools and seasonal ponds used for breeding habitats, as well as reduced upland habitat used for aestivation and migration (USFWS, 2000).

All known locations of the SB CTS are affected by railroads, highways, or other roads causing extensive habitat fragmentation. SB CTS require large tracts of unobstructed landscape for dispersal and migration. Larger roads, highways, and railroads act as permanent physical barriers and can prevent genetic interchange and movement to and from breeding and aestivation sites. The temporary construction of road and rail is also detrimental to the SB CTS. Slow-moving animals are unable to escape the initial intrusion and disruption of construction and any SB CTS in underground burrows are likely to be crushed. Once the road is open, the salamanders are at risk for being run over on their first dispersal runs to the pond. Studies indicate that up to 9 to 12 SB CTS are killed as they try to cross roads during migrations between breeding and upland sites (USFWS, 2000). Soil compaction and alteration, including soil density, soil water content, dust, surface-water flow, patterns of runoff, and sedimentation, are other issues affected the SB CTS in areas underneath and adjacent to the road bed. Harmful ecological effects of roads can extend outward on average of 0.6 km from the road bed and can make burrowing and migrating near the road difficult for the SB CTS. A study on road distances to wetlands indicated that the number of amphibian species in a wetland was adversely affected by roads being located within a 2 km (1.2mi) range of that wetland (USFWS, 2000). Berms and curbs as low as 9 to 12 cm also are of particular concern, as they allow salamanders to climb onto the road but can restrict their movement off of the road (USFWS, 2003). Such berms exist on the State highway and the secondary road in proximity to three ponds in Santa Barbara County (USFWS, 2003)

Intensive agricultural practices began in Santa Barbara County more than 30 years ago, in the Santa Maria River and San Antonio Creek Valleys, and continue today. Row crop acreage in the Los Alamos and Santa Rita Valleys has increased by more than 9,900 hectares (ha) (more than 25,000 acres (ac)) between the years of 1986-1997. An installation of 4,000 ha of vineyards from 1996-1999 has more than doubled the land cultivated for grapes. Between the nine years from 1992 to 2000, irrigated cropland in Santa Barbara County has increased by 49%, or by 15,700 ha (38,850 ac), and 36% of this growth occurred in between 1997 and 1999 (5,670 ha or 14,000 ac) (USFWS, 2000).

Vernal pools, seasonal ponds, and upland habitat continue to be lost as suitable land is converted for agricultural and urban uses. Changes to vernal pool or pond inundation, duration, or depth can detrimentally impact the reproductive success of the SB CTS. These changes are caused by, among others, digging of drainage or irrigation ditches, construction of permanent ponds or reservoirs, deepening or berming of seasonal wetlands, and redirection or runoff from developments (USFWS, 2000). The repercussions of these activities on the reproductive success of the SB CTS include: (1) prematurely drying wetlands and desiccating SB CTS larvae; (2) extending the inundation period and facilitation invasion of non-native predators; (3) creating conditions that foster the hybridization between the non-native tiger salamander; and (4) increasing vulnerability to disease from isolation and fragmentation. Wetland habitats are utilized for reproduction; thus destructive activities such as filling, discing, or excavating, are injurious or lethal to the eggs, larvae, or breeding adults. Earthmoving operations and cultivation in upland habitat also threaten the SB CTS, which use the habitat to breed, feed, and seek shelter. SB CTS are at risk in burrows or on the surface from discing, deep-ripping, or grading of upland habitat. Indirect impacts to the SB CTS include increased predation from over exposure, high temperatures, low humidity, and diminished food sources. Earth moving operations may further disturb California ground squirrel burrows and crevices, protection utilized by the SB CTS, and may render upland locations untenable or unavailable (USFWS, 2004 a).

Urbanization and rural residential development threaten the SB CTS rangeland areas that may contain vernal pool and grassland habitats. Between 1982 and 1997, privately owned rangeland decreased by 252,524 ha (624,000ac), and between 1998 and 2000 the state of California lost an additional 21,555 ha (53,263 ac). This loss of rangeland results from a myriad of factors, including a decline in farm rancher income, the aging of ranchers, tax implications of intergenerational transfers of ranches, and the difficulty of beginning a ranching operation (*e.g.*, cost and knowledge) (USFWS, 2004 a).

Fragmentation

The primary factors causing SB CTS fragmentation are discussed above: road construction, urbanization, and intensive agriculture. Human-caused fragmentation places the SB CTS at particular risk to local extinctions because its metapopulation structure is distributed across the landscape (USFWS, 2000).

> "A metapopulation is a group of spatially distinct populations that can occasionally exchange dispersing individuals. The populations in a metapopulation are usually thought of as having interdependent extinction and colonization processes, where individual populations may be extirpated from a local area and later be recolonized from another population that is still extant" (FR, 72:48177, p.6).

A reduction in SB CTS dispersal greatly inhibits the dynamics of a metapopulation and reduces the abilities of the SB CTS to persist over time. Of the three factors, roads accelerate fragmentation by increasing mortality and preventing recolonization of sites that would otherwise be only temporarily extirpated (USFWS, 2000).

6.2.1.2 Contaminants

Contaminants from oil production and road runoff (*e.g*., hydrocarbons) directly and indirectly affect the SB CTS. The runoff from roads (*e.g*., oil and other contaminants) is found in adjacent ponds and is implicated in amphibian deformities, as well as amphibian and invertebrate mortality (USFWS 2000). SB CTS are particularly vulnerable because they inhabit both aquatic and terrestrial habitats which expose them to a variety of toxins throughout their life cycle. Their permeable skin further increases their vulnerability (USFWS, 2004a), and studies have found developmental effects in marbled (*Ambystoma opacum*) and eastern tiger salamanders (*A. tigrinum*), and limited direct effects in five week old salamanders. Study results indicated that salamanders from oil-contaminated natural ponds metamorphosed earlier at smaller sizes and that those from oil-contaminated artificial ponds had slower growth rates than larvae raised in non-contaminated ponds. However, effects to eggs and early life stages, which could be more severe, were not addressed. Other studies have examined the toxicity of Fluoranthene, a polycyclic aromatic hydrocarbon present in petroleum products and urban runoff, on spotted salamanders (*A. maculatum*), northern leopard frogs (*Rana pipiens*), and African clawed frogs (*Xenopus laevis*). The researchers evaluated concentrations at levels commensurate to those measured in service stations and other urban runoff, and found reduced survival and growth abnormalities in all species under field and laboratory conditions. Effects were worsened when the larvae were exposed to Fluoranthene in natural sunlight rather than in artificial light (USFWS, 2000).

Sedimentation from road construction, maintenance, and runoff is a second type of contamination threatening the SB CTS. Breeding ponds are affected by the altered hydrology near roads, which leads to erosion, gullies, and increased sediment deposits in wetland systems. Increased dust from traffic affects aquatic and emergent vegetation and may asphyxiate SB CTS eggs. Excessive sedimentation ultimately fills pools otherwise usable by the SB CTS. Sedimentation may also impair the SB CTS from detecting food items (USFWS, 2000).

6.2.1.3 Pest control

Pesticides and rodenticides are heavily used in the agricultural areas associated with CTS aquatic and terrestrial habitats. Therefore, CTS are likely exposed to a variety of pesticides in their natural habitat. CTS are particularly sensitive to these chemicals because of the permeability of their skin (USFWS, 2007). Furthermore, the use of pesticides can negatively impact CTS by reducing their prey availability (USFWS, 2007). For example, some studies have shown that methoprene, used for mosquito control, is associated with reduced survival rates and increased malformations of some anuran species (USFWS, 2003). The USFWS reported that other insecticides besides methoprene have adversely impacted growth rates in gray treefrog (*Rana clamitans*) tadpoles and increased mortality rates in southern leopard frogs (*R. sphenocephala*) (USFWS, 2000). Temephos was identified by the USFWS as an example of these additional insecticides (USFWS, 2000). *Bacillus thuringensis israeli*, a bacterium, is also used in Santa Barbara County for mosquito control (USFWS, 2000). The effects of this bacterium on the CTS prey base are currently unknown (USFWS, 2000).

CTS are also negatively affected by rodenticides because they are smaller than the target species (USFWS, 2007). Ground squirrel populations were controlled in Santa Barbara County using compound 1080 (sodium fluoroacetate) until about 1990 (USFWS, 2000). This chemical is highly toxic to non-target species, including fish, birds, and mammals and it is thought that the use of compound 1080 in some areas may have contributed to the observed reduction in CTS populations (USFWS, 2000). Grain poisoned with anticoagulant substances is the preferred method to control ground squirrels where children, pets, or poultry might be present, although grains poisoned with strychnine are also used (USFWS, 2003). The rodenticide, zinc phosphide, can indirectly impact CTS if it is washed into burrows or ponds used by the salamanders (USFWS, 2003). Two commonly used rodenticides, chlorophacinone and diphacinone, both anticoagulants, can be absorbed through the skin and are therefore very toxic to CTS (USFWS, 2003). Several gasses also used to control rodents, identified as aluminum phosphide, carbon monoxide, and methyl bromide, are introduced into the small mammal burrows where CTS may live (USFWS, 2003).

Another threat to CTS includes habitat loss due to elimination of small mammal burrows. Small mammal control programs have a profound negative effect on CTS because they reduce the number of burrows available for CTS use. Studies have found that CTS use both occupied and unoccupied burrows; however, burrows typically collapse within 18 months after the burrow has been vacated by the small mammal. It has not been determined whether CTS are able to use collapsed burrows (USFWS, 2000 and 2003).

6.2.1.4 Grazing and Water drawdown

As opposed to many factors that negatively affect the SB CTS, grazing can, in moderation, have a neutral to positive impact on the SBTS. Grazing maintains shorter vegetation and promotes the persistence of ground squirrel burrowing, which creates habitat essential for the SB CTS. As of the Federal Register final rule (2000), grazing land in Santa Barbara County offered the only large amounts of suitable habitat for the SB CTS. Because rangelands presented the only undeveloped habitat in the County, they provided the only opportunity for the salamanders to maintain sustainable populations (USFWS, 2000). Rangelands not only provide natural vernal pools, but artificial stock ponds, which have likely saved many populations from extirpation (USFWS, 2004 a). Although trampling from cattle may impact water levels and soil integrity of ponds and banks, grazing is considered to be compatible with SB CTS, as long as burrowing rodents are not completely eradicated (USFWS, 2004 a).

Water drawdown affects many of the ponds in Northern Santa Barbara County from agricultural practices such as irrigation and frost control. Submersible pumps are used to remove the water during frost control; thus, SB CTS larvae and adults may be killed by being sucked into pump mechanisms. Water drawdown also puts the pond at risk of prematurely drying in the spring or summer, thus desiccating the un-metamorphosed larvae (USFWS, 2000).

6.2.1.5 Predation, Competition, and Hybridization

At least four of the six SB CTS metapopulations are subjected to predation and competition by native and non-native species. Native predators of the CTS include great blue herons (*Ardea herodias*), egrets (*Casmerodius*), western pond turtles (*Clemmys marmorata*), garter snakes (*Thamnophis* spp.), larger CTS larvae, larger spadefoot toads (*Scaphiopus hammondii*), and California red-legged frogs (*Rana draytonii*) (USFWS, 2004a). Native as well as non-native crayfish also prey on CTS (USFWS, 2003). Predation of CTS by sticklebacks, fish which are native to California, is unknown (USFWS, 2003). While native predation and hybridization is probably a discountable threat in healthy populations, these native predators may have significant impacts on population viability when populations are subjected to the cumulative effects of contaminants, migration barriers, and/or habitat alteration (USFWS, 2003).

Predation by and competition with non-native and introduced species contributes to the decline of the CTS (USFWS, 2003). Non-native predators of both adult and larval CTS include Louisiana red swamp crayfish, mosquitofish, sunfish spp. (*e.g*,. large mouth bass (*Micropterus salmoides*)) bluegill (*Lepomis macrochirus*), catfish spp. (*Ictalurus* spp.), fathead minnow (*Pimephales promelas*), carp (*Cyprinus carpio*), and American bullfrog (*Rana catesbeiana*) (USFWS, 2004 a; USFWS, 2007). Louisiana crayfish are found in two known CTS breeding locations in Santa Barbara County (USFWS, 2000). Bullfrogs are known to prey on CTS larvae and studies have shown that the number of CTS decreases as bullfrog numbers increase (USFWS, 2000). A study observing the effect of bullfrogs on the CRLF, a prey item of CTS, showed that less than 5% of CRLF tadpoles survived to metamorphosis when they were raised with bullfrog tadpoles (USFWS, 2000).

Western mosquito fish, which are used to control mosquitoes because they eat mosquito larvae, are also known to prey on amphibian species (USFWS, 2003). Incidentally, field and laboratory experiments have shown that mosquito fish will preferentially prey on amphibians even in the presence of mosquito larvae (USFWS, 2003). It is not known if mosquito fish target CTS specifically; however, larval CTS may be particularly at risk due to their external gills which are attractive to mosquito fish (USFWS, 2003). A study performed in 1994 found that no CTS had been found in ponds where mosquito fish were located (USFWS, 2003). However, another study performed in 2000 found that CTS numbers were reduced in ponds stocked with mosquito fish at densities similar to those found in many stock ponds (USFWS, 2004a). CTS larvae found in ponds containing mosquito fish were smaller, took longer to metamorphose, and typically suffered injuries, such as shortened tails (USFWS, 2004a). Another study performed in 2003 showed that at low densities mosquito fish did not have a significant impact on larval CTS growth and survival, but that growth and size at metamorphosis was reduced at high fish densities (USFWS, 2004a). It is thought that large numbers of mosquito fish may also out-compete CTS larvae for food (USFWS, 2000).

Other species of non-native fish, including largemouth bass, bluegill, catfish, and fathead minnows may also either be directly responsible for or have the potential to cause the decline of the CTS (USFWS, 2003). The Los Alamos and West Orcutt metapopulations, in particular, are limited by these non-native fish species and CTS suitable habitat has supported non-native fish populations for a number of years. These fish may affect the prey base of the CTS or prey on larval CTS, either of which is capable of reducing or eliminating CTS populations (USFWS, 2000). Several of these species including largemouth bass, bluegill, catfish, and bullheads have been, and still are, stocked in ponds throughout California for fishing (USWFWS, 2003).

Hybridization

Non-native subspecies of tiger salamanders have been brought to California and used for fishing bait (USFWS, 2004 a). Whether by bait-bucket release or intentional introduction, non-native tiger salamanders are now established in California (CDFG, 2010). The practice of importing barred tiger salamanders (*A. tigrinum mavortium*) and Arizona tiger salamanders (*A. tigrinum nebulosum*) was legal in California until December of 2000 (CDFG, 2010). At this time, the California Department of Fish and Game (Department) ruled that using *A. tigrinum* as bait was illegal, as well as possessing any individual in the genus *Ambystoma* in California without first obtaining a Department-issued permit (CDFG, 2010). This regulation was intended to mitigate CTS hybridization with non-native tiger salamanders (CDFG, 2010). As reported by the USFWS, introduced tiger salamanders have been observed in Santa Barbara County at one location west of Santa Rita Valley (USFWS, 2000). The non-native species have been known to escape in areas where CTS are found, and can compete and breed with CTS (USFWS, 2003). The hybrids that are created when non-native tiger salamanders and CTS breed were originally thought to be poorly-adapted to survive or be sterile past the first few generations (USFWS, 2007). However, more recent data suggest that the hybrids are viable and can breed with the CTS (USFWS, 2003). Pure CTS have been found more frequently in natural habitats compared with artificial or disturbed ones (USFWS, 2004 a). It seems that hybrids are less likely to be found in vernal pools, stressing the importance of protecting existing natural vernal pools (USFWS, 2004 a).

6.2.1.6 Disease

The direct effect of disease on the SB CTS is not known, and risks to the distinct population segment (DPS) cannot be precluded (USFWS, 2000). Disease potentially threatens the SB CTS because it is found in relatively few sites and its metapopulation structure is distributed across a small portion of the County (USFWS, 2000). Several pathogenic (disease-causing) agents have been linked to deaths of closely-related tiger salamanders and other amphibian species, and include fungi, viruses, and at least one bacterium (*Acinetobacter* spp.) (USFWS, 2000; USFWS, 2004 a). These pathogens could devastate current SB CTS populations if introduced into the county (USFWS, 2000).

The bacterium *Acinetobacter* is common in soil and animal feces (USFWS, 2000). In one study, this bacterium was found to increase with increasing nitrogen levels from atmospheric deposition as the study lake in Utah (Desolation Lake) dried out (USFWS, 2000). Affected tiger salamanders showed symptoms of red, swollen hind legs and vents, and hemorrhaging of skin and internal organs. Iridoviruses, (a family, containing the genus *Ranavirus*) are viruses with DNA as the genetic material that occur in insects, fish, and amphibians, including tiger salamanders (Harp, 2006; USFWS, 2000). These viruses can cause a range of symptoms, including skin lesions and mortality, or they can be asymptomatic (USFWS, 2000). Ranaviruses are of particular concern because these viruses can be spread on fishing gear, artificial stocking of ponds for fishing, and water birds (USFWS, 2003). Outbreaks of ranavirus in tiger salamanders have been associated with altered habitats and artificial ponds (USFWS, 2003). Chytridiomycosis, caused by the fungus *Batrachochytrium dendrobatidis*, and ranaviruses have been identified as potential threats to the CTS because these diseases have adversely impacted other amphibians, including tiger salamanders (USFWS, 2004a). Non-native species of bullfrogs and tiger salamanders are potential carriers of these diseases (USFWS, 2004 a).

A study conducted in 1997 showed that tiger salamanders in Georgia reared in ponds contaminated with silt were susceptible to infection by the fungus *Saprolegnia parasitica* (USFWS, 2003). Die offs associated with western toads and Pacific tree frogs, known prey of the CTS, have been associated with *Saprolegnia* infections (USFWS, 2000). High nitrogen and silt levels from overgrazing or agricultural or urban runoff may also increase susceptibility to disease (USFWS, 2000). Two of the three ponds in the West Orcutt metapopulation area are in overgrazed grasslands and are at risk of receiving runoff that has both high nitrogen and high silt levels (USFWS, 2000). Four ponds in the Los Alamos metapopulation and the two ponds in the Santa Rita metapopulation are on grazing lands (USFWS, 2000). High levels of nitrogen have been associated with bacterial blooms and increased silt has been associated with fungal infections (USFWS, 2000).

6.2.2 Baseline Status

The CTS is found only in California. The historic range of the CTS included, “large portions of the Central Valley of California, from the southern Sacramento Valley north of the Sacramento River delta to the southern San Joaquin Valley” (USFWS, 2002). The CTS was also found, “in the lower foothills along the eastern side of the Central Valley and in the foothills of the Coast Ranges” (USFWS, 2002). CTS have been historically documented in 27 counties but are no longer found in three of these counties (USFWS, 2005). The USFWS believes that the CTS is still found in the remaining 24 counties (USFWS, 2005). CTS have been found in 10 of the 17 California vernal pool designations defined by Keeler-Wolf in 1998 (USFWS, 2004 a).

The historical distribution and numbers of the CTS are unknown and the challenges to estimate of the total number of CTS in Santa Barbara County have been noted by a number of biologists (USFWS, 2004b). SB CTS now occur in six scattered metapopulations across the species’ historic range in northern Santa Barbara County (USFWS, 2000; USFWS, 2004b). These metapopulations are listed by region as follows, with the critical habitat unit identified in parentheses: (1) West Orcutt (Western Santa Maria/Orcutt); (2) Bradley-Dominion (Eastern Santa Maria); (3) North Los Alamos (Western Los Alamos/Careaga); (4) East Los Alamos (Eastern Los Alamos); (5) Purisima Hills (Purisima Hills); (6) Santa Rita (Santa Rita Valley) (USFWS, 2000; USFWS 2004b). The threats from agriculture, urbanization, overgrazing, fragmentation, and roadkill are severe in four of these metapopulations, moderate in one, and nominal in the sixth (USFWS, 2000). Due to limited information on population dynamics, the most significant factors affecting the species continued existence are used as a representation of the metapopulation’s current baseline status. This information on each metapopulation is discussed by region as follows:

(1) <u>West Orcutt (Western Santa Maria/Orcutt)</u>
This critical habitat unit comprises 26 percent of the total area identified in Santa Barbara County containing SB CTS primary constituent elements (PCEs) and containing 12 of the 46 known breeding ponds in Santa Barbara County (USFWS, 2004b). These breeding habitats include seven vernal pools, and a handful of smaller water bodies that make suitable breeding grounds for the SBTS but have never been surveyed (USFWS, 2004b). These 46 known breeding habitats of the SB CTS are comprised of 23 artificial ponds, four human-altered ponds, and 19 natural ponds (vernal pools) (USFWS, 2004b). The greatest threat to this metapopulation is human population growth coupled with residential and commercial development (USFWS, 2004b). The city of Santa Maria is the fastest growing city in the county and urbanization pressure on the SB CTS will increase concomitantly as the population (as projected) grows by at least 160,000 inhabitants in the next 30 years (USFWS, 2004b). The county may need to develop over 15,000 ac (6,070 ha) to support this growth (USFWS, 2004b).

(2) Bradley-Dominion (Eastern Santa Maria)
This unit also covers 26 percent of the total area identified containing SB CTS PCEs, as well as containing four known breeding ponds (natural vernal pools) (USFWS, 2004b). This unit faces increased development pressure similar to unit 1, although unit 2 also experiences losses to SB CTS habitat from illegally-conducted ground disturbing activities (USFWS, 2004b). This metapopulation is considered to be at highest risk of perishing due to agriculture intensification (USFWS, 2000).

(3) North Los Alamos (Western Los Alamos/Careaga)
This unit supports four known breeding ponds, as well as human-made ponds within dispersal distance of the SB CTS breeding ponds in the western Los Alamos Valley that may be used for breeding (USFWS, 2004b). The Careaga Divide pond, determined to be one of the most unique and pristine vernal ponds where SB CTS breed, is found in this regions (USFWS, 2004b). In contrast to many other breeding locations, the Careaga Divide pond is encased on two sides by extensive, dense coast live oak woodland, in addition to coastal sage scrub and grasslands on the others (USFWS, 2004b). This unit requires special management to preserve habitat in the form of fish removal, sediment control, and reduction of threats associated with berm failure and vineyard development (USFWS, 2004b).

(4) East Los Alamos (Eastern Los Alamos)
This region is the site of four known breeding locations for the SB CTS (USFWS, 2004b). However, the property on which the four ponds sit has been excluded from critical habitat determination due to a conservation strategy created by landowners to enhance or create SB CTS aquatic habitat (USFWS, 2004b). Even without these breeding ponds, this site provides essential upland habitat and acreage needed to foster a self-sustaining population (USFWS, 2004b). This unit requires special management to address the threats of road mortality and upland habitat loss (USFWS, 2004b).

(5) Purisima Hills (Purisima Hills)
This unit is threatened with habitat loss and requires special management (USFWS, 2004b). This habitat is unique in that it has steeper terrain and is more densely vegetated than all other units (USFWS, 2004b). It is the only unit containing breeding ponds completely surrounded by coastal sage chaparral (USFWS, 2004b). This metapopulation is the least threatened of the six, and one landowner expressed interest in working with the Land Trust of Santa Barbara County to establish conservation easements protecting the species (USFWS, 2000).

(6) Santa Rita (Santa Rita Valley)
This unit is ‘severely affected’ by agricultural grading, conversion to row crops, and livestock facilities in the west (USFWS, 2000). The eastern part, and population of SB CTS, is affected by alterations to breeding ponds for use by cattle and threats of road strikes. This region is also used for oil production (USFWS, 2000). SB CTS in the Santa Rita Valley are at risk of contamination from toxic water bodies, potentially causing developmental deformities or death (USFWS, 2000).

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7: CALIFORNIA TIGER SALAMANDER: SONOMA COUNTY DISTINCT POPULATION SEGMENT

7.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the Sonoma County distinct population segment (DPS) of the California tiger salamander (*Ambystoma californiense*) (STS) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of urban development, intensive agriculture (*e.g.*, vineyards), grazing operations, oil production, contamination from runoff and spills, and non-native species introductions. Many of these factors are linked or act synergistically and create complex consequences for the STS. For example, conversions to intensive agriculture and urban developments have resulted in a reduction in the sizes and connectivity between patches of suitable and occupied habitat on the Santa Rosa Plain. The extent of this habitat reduction makes recolonization of some sites more difficult following local extinction. As the STS is found in a metapopulation structure, the persistence depends on the combined dynamic of these local extinctions and subsequent recolonization. Both destruction and reduced suitability of breeding habitat, compounded with increased urbanization, fragments and isolates subpopulations from each other and inhibits potentially beneficial gene flow. Further, habitat loss magnifies the effects of other factors, such as amount of food, availability of rodent burrows, pesticide use, mortality from vehicles, and predators, more pronounced given the smaller area now exposed to such impacts (USFWS, 2003).

7.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the STS's status at this time. However, the baseline condition of each assessed STS species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included in this assessment, the environmental baseline includes a general discussion of factors that may affect STS within the action area. This information is presented in Section 7.2.1. Additional information on the current distribution and population dynamics of the STS is presented as part of the baseline status in Section 7.2.2.

7.2.1 Factors affecting species continued existence within the action area

STS's continued existence is primarily affected by continued destruction, degradation, and fragmentation of suitable habitat. Secondary threats to the DPS include predation and competition from introduced exotic species; possible commercialization with non-native salamanders; various chemical contaminants; road-crossing mortality; rodent control operations; and a small remaining population. Livestock grazing is another factor affecting STSs, although it can in moderation be a positive rather than negative influential factor (USFWS, 2002; USFWS, 2003; USFWS, 2004). Some of these factors are known to affect the California Tiger Salamander population in Sonoma County (STS), while others are known to generally affect the California Tiger Salamander (CTS) population throughout its entire range. The factors known to affect the larger population are still relevant to the STS, but are referred to as CTS. Sections 7.2.1.1-7.2.1.4 provide details on the major issues affecting STS environment within the action area.

7.2.1.1 Habitat Destruction, Degradation, and Fragmentation

As of 2003 when the Sonoma County DPS was listed as endangered, there were eight remaining known breeding sites on the Santa Rosa Plain. These subpopulations are threatened by construction of high-density housing, office buildings, roads, and other developments. A study conducted in 1990 found that 25 percent of a study area of 11,300-hectares (ha) was being used for subdivisions, "ranchettes", golf courses, and commercial buildings. Seventeen percent of the study area had been converted for agricultural uses. Since 1990, land conversion to urban sites and intensive agriculture continue, particularly to vineyard operations. Other relatively minor land conversions to roads, storm drains, and road curbs have the potential to adversely impact the STS by fragmenting habitat or impeding migration. Of the eight known breeding sites at the time of listing, five are within 100 meters (m) of major development activities, and all breeding pools are within 450 m of roads and residential development (USFWS, 2003).

Urban Development
The Santa Rosa County is rapidly urbanizing. Since 1980, the city of Santa Rosa has increased 53% in population size. Between 1980 and 1997, housing units grew by 66 percent, an increase from 35,403 units to 53,558 units. More recently, five known breeding locations were lost due to commercial and residential development between 2001 and 2003. Six of the eight known remaining breeding sites are located around the former Air Center, in Southwest Santa Rosa. This area contains one of the largest blocks of undeveloped lands within the city limits. In the mid-1980s, the airport closed and the property was sold to the City. The Southwest Area Plan proposes to development the majority of this area and places STS breeding grounds in peril (USFWS, 2003).

Roads and Highways
Roads and highways act as physical barriers that inhibit successful migration of the STS. The CTS needs large tracts of barrier-free landscape and roads present an obstacle in the way of moving to new breeding habitat, or returning to aestivation sites. In worst case scenarios, roads can completely isolate a breeding site or larger portions of a metapopulation and render them untenable for the species (USFWS, 2003).

As mentioned previously, all breeding pools are within 450 m of a type of road. A study on road distances to wetlands indicated that the number of amphibian species in a wetland was adversely affected by roads being located within a 2 km (1.2 mi) range of that wetland. In addition to habitat fragmentation, direct vehicular strikes are of huge concern for the STS caused by roads. In the Central Valley, large numbers of CTS ranging from 15-20 per mile are killed by vehicles. Some estimates place the percentage of breeding population lost to vehicular strikes at 35-72 percent. Between November 21, 2001 and December 5, 2001, 26 STS individuals were found dead on Stony Point Road running between Santa Rosa and Cotati. This road in western Santa Rosa has seen an increase in vehicular traffic by about four fold due to its recent support of heavy commuter traffic. The combination of roads with curbs and berms places even greater threat on the CTS; Curbs and berms as low as 9 to 13 cm allow movement onto a road but can restrict movement off of a road (USFWS, 2003).

Breeding locations

Except for the Hall Road Preserve, all of the known breeding sites of the STS are found in sites being rapidly converted to high-density housing and office buildings. The STS in each breeding location continues to be impacted from isolation and fragmentation. Only three breeding sites have hydrologic regimes necessary to support the species (Hall Road Preserve, FEMA/Broadmore North Preserve, and Engel Preserve). Five of the breeding locations are on private property, two of which do not allow surveying and the presence of STS there are unknown. Furthermore, the plans for the development of the old airfield, if completed, will eliminate all existing migration corridors between extant populations (USFWS, 2003).

Fragmentation

The primary factors causing CTS fragmentation are discussed above: road construction, urbanization, and intensive agriculture. Human-caused fragmentation places the STS at particular risk to local extinctions because its metapopulation structure is distributed across the landscape (USFWS, 2000).

> "A metapopulation is a group of spatially distinct populations that can occasionally exchange dispersing individuals. The populations in a metapopulation are usually thought of as having interdependent extinction and colonization processes, where individual populations may be extirpated from a local area and later be recolonized from another population that is still extant" (USFWS, 2007 a).

A reduction in CTS dispersal greatly inhibits the dynamics of a metapopulation and reduces the abilities of the CTS to persist over time. Of the three factors, roads accelerate fragmentation by increasing mortality and preventing recolonization of sites that would otherwise be only temporarily extirpated (USFWS, 2000).

7.2.1.2 Disease

The direct effect of disease on the CTS is not known and risks to the distinct population segment (DPS) cannot be precluded (USFWS, 2000). Several pathogenic (disease-causing) agents have been linked to deaths of closely-related tiger salamanders and other amphibian species, and include fungi, viruses, and at least one bacterium (*Acinetobacter* spp.) (USFWS, 2000; USFWS, 2004). These pathogens could devastate current STS populations if introduced into the county (USFWS, 2003).

The bacterium *Acinetobacter* is common in soil and animal feces (USFWS, 2000). Affected tiger salamanders showed symptoms of red, swollen hind legs and vents, and hemorrhaging of skin and internal organs. Iridoviruses, (a family, containing the genus *Ranavirus*) are viruses with DNA as the genetic material that occur in insects, fish, and amphibians, including tiger salamanders (Harp, 2006; USFWS, 2000). These viruses can cause a range of symptoms, including skin lesions and mortality, or they can be asymptomatic (USFWS, 2000). Ranaviruses are of particular concern because these viruses can be spread through fishing gear, artificial stock ponds, and water birds (USFWS, 2003). Outbreaks of ranavirus in tiger salamanders have been directly associated with altered habitats and artificial ponds (USFWS, 2003). Ranaviruses, in addition to the fungus *Batrachochytrium dendrobatidis* that causes Chytridiomycosis, have been identified as potential threats to the CTS because these pathogens have adversely impacted other amphibians, including tiger salamanders (USFWS, 2004). Non-native species of bullfrogs and tiger salamanders are potential carriers of these diseases (USFWS, 2004).

A study conducted in 1997 showed that tiger salamanders in Georgia reared in ponds contaminated with silt were susceptible to infection by the fungus *Saprolegnia parasitica* (USFWS, 2003). Die offs associated with Western Toads (*Bufo boreas*), Cascades frogs (*Rana Cascadae*), and Pacific Tree frogs (*Psedocris regilla*), known prey of the CTS, have been associated with *Saprolegnia* infections (USFWS, 2000). *Saprolegnia ferax* outbreaks have caused high amphibian embryo mortalities in the Pacific Northwest (USFWS, 2003). High nitrogen and silt levels from overgrazing or agricultural or urban runoff may also increase susceptibility to disease (USFWS, 2000). High levels of nitrogen have been associated with bacterial blooms and increased silt has been associated with fungal infections (USFWS, 2000).

7.2.1.3 Predation, Competition, and Hybridization

All of the known STS metapopulations are potentially affected by native and non-native species predation and competition (USFWS, 2003). Native predators of the CTS include great blue herons (*Ardea herodias*), egrets (*Casmerodius*), western pond turtles (*Clemmys marmorata*), garter snakes (*Thamnophis* spp.), larger CTS larvae, larger spadefoot toads (*Scaphiopus hammondii*), and California red-legged frogs (*Rana draytonii*) (USFWS, 2004). Native as well as non-native crayfish also prey on CTS (USFWS, 2003). Predation of CTS by sticklebacks, fish which are native to California, is unknown; however, sticklebacks have been present in California for at least 16 million years and STS establishment in stickleback habitat is thought to be restricted (USFWS, 2003).

While native predation and hybridization is probably a discountable threat in healthy populations, these native predators may have significant impacts on population viability when populations are subjected to the cumulative effects of contaminants, migration barriers, and/or habitat alteration (USFWS, 2003).

Predation by and competition with non-native and introduced species contributes to the decline of the CTS (USFWS, 2003). Non-native predators of both adult and larval CTS include Louisiana red swamp crayfish (*Procambarus Clarkii*), mosquitofish (*Gambusia affinis*), sunfish spp. (*e.g*., large mouth bass (*Micropterus salmoides*)) bluegill (*Lepomis macrochirus*), catfish spp. (*Ictalurus* spp.), fathead minnow (*Pimephales promelas*), carp (*Cyprinus carpio*), and American bullfrog (*Rana catesbeiana*) (USFWS, 2004; USFWS, 2007 b). Introductions of sunfish, largemouth bass, bluegill, catfish, and fathead minnows are thought to be responsible for the elimination of the STS from several breeding locations in Sonoma County (USFWS, 2003). These fish may affect the prey base of the CTS or prey on larval CTS, either of which is capable of reducing or eliminating populations of CTS (USFWS, 2000). Bullfrogs are known to prey on CTS larvae and studies have shown that the number of CTS decreases as bullfrog numbers increase (USFWS, 2000). A study observing the effect of bullfrogs on the CRLF, a prey item of CTS, showed that less than 5% of CRLF tadpoles survived to metamorphosis when they were raised with bullfrog tadpoles (USFWS, 2000). One of the pools at the Hall Road preserve and two of the pools at the FEMA/Broadmore North preserve are located within 46 m of ditches or creeks known to contain bullfrogs or crayfish. Bullfrogs also likely occur in Roseland Creek, near the FEMA/Broadmore North preserve (USFWS, 2003).

Western mosquito fish, which are used to control mosquitoes because they eat mosquito larvae, are also known to prey on amphibian species (USFWS, 2003). Incidentally, field and laboratory experiments have shown that mosquito fish will preferentially prey on amphibians even in the presence of mosquito larvae (USFWS, 2003). It is not known if mosquito fish target CTS specifically; however, larval CTS may be particularly at risk due to their external gills which are attractive to mosquito fish (USFWS, 2003). A study performed in 1994 found that no CTS had been found in ponds where mosquito fish were located (USFWS, 2003). However, another study performed in 2000 found that CTS numbers were reduced in ponds stocked with mosquito fish at densities similar to those found in many stock ponds (USFWS, 2004). CTS larvae found in ponds containing mosquito fish were smaller, took longer to metamorphose, and typically suffered injuries, such as shortened tails (USFWS, 2004). Another study performed in 2003 demonstrated that at low densities mosquito fish did not have a significant impact on larval CTS growth and survival, but that growth and size at metamorphosis was reduced at high fish densities (USFWS, 2004). It is thought that large numbers of mosquito fish may also out-compete CTS larvae for food (USFWS, 2000).

Another confounding impact of mosquito control is the effect of methroprene, an insecticide hormone mimic. This pesticide has been documented to delay the onset of molting in crustacea that had identical molting hormones as insects. The use of methropene may decrease the availability of prey of aquatic invertebrates and thus may indirectly affect the STS (USFWS, 2003).

Other species of non-native fish, including largemouth bass, bluegill, catfish, and fathead minnows may also either be directly responsible for or have the potential to cause the decline of the CTS (USFWS, 2003). These fish may affect the prey base of the CTS or prey on larval CTS, either of which is capable of reducing or eliminating CTS populations (USFWS, 2000). Several of these species, including largemouth bass, bluegill, catfish, and bullheads, have been and still are stocked in ponds throughout California for fishing (USWFWS, 2003).

Hybridization

Non-native subspecies of tiger salamanders have been brought to California and used for fishing bait (USFWS, 2004). This is still legal in California but is restricted to fewer counties and is regulated by the California Department of Fish and Game (USFWS, 2000). Currently, non-native CTS are not known to occur in the range of the Sonoma County CTS population (USFWS, 2004 a); however, the non-natives have been known to escape in areas where CTS are found, and can compete and breed with CTS (USFWS, 2003). The hybrids that are created when non-native tiger salamanders and CTS breed were originally thought to be poorly-adapted to survive or be sterile past the first few generations (USFWS, 2007 b). However, more recent data suggest that the hybrids are viable and can breed with the CTS (USFWS, 2003). Pure CTS have been found more frequently in natural habitats compared with artificial or disturbed ones (USFWS, 2004). It seems that hybrids are less likely to be found in vernal pools making it all that much more important to protect existing natural vernal pools (USFWS, 2004). The loss of any breeding sites of the STS due to hybridization is of serious concern (USFWS, 2003).

7.2.1.4 Other Factors affecting the species continued existence

Chemical Contaminants

Contaminants from oil production and road runoff (*e.g*., hydrocarbons) directly and indirectly affect the STS. The runoff from roads (*e.g*., oil and other contaminants) is found in adjacent ponds and is implicated in amphibian deformities, as well as amphibian and invertebrate mortality (USFWS 2000). STS are particularly vulnerable because they inhabit both aquatic and terrestrial habitats which expose them to a variety of toxins throughout their life cycle. Their permeable skin further increases their vulnerability (USFWS, 2004), and studies have found developmental effects in marbled (*Ambystoma opacum*) and eastern tiger salamanders (*A. tigrinum*), and limited direct effects in five week old salamanders. Study results indicated that salamanders from oil-contaminated natural ponds metamorphosed earlier at smaller sizes and that those from oil-contaminated artificial ponds had slower growth rates than larvae raised in non-contaminated ponds. However, affects to eggs and early life stages were not addressed where effects could be more severe. Other studies have examined the toxicity of fluoranthene, a polycyclic aromatic hydrocarbon present in petroleum products and urban runoff, on spotted salamanders (*A. maculatum*), northern leopard frogs (*Rana pipiens*), and African clawed frogs (*Xenopus laevis*). Researchers evaluated concentrations at levels commensurate to those measured in service stations and other urban runoff, and found reduced survival and growth abnormalities in all species under field and laboratory conditions. Effects were worsened when the larvae were exposed to fluoranthene in natural sunlight rather than in artificial light (USFWS, 2000).

Sedimentation from road construction, maintenance, and runoff is a second type of contaminant threatening the CTS. Breeding ponds are affected by the altered hydrology near roads, which lead to erosion, gullies, and increased sediment deposits in wetland systems. Increased dust from traffic affects aquatic and emergent vegetation and may asphyxiate CTS eggs. Excessive sedimentation ultimately fills pools otherwise usable by the STS. Sedimentation may also impair the CTS from detecting food items (USFWS, 2000). Many STS breeding populations inhabit roadside ditches in place of pools or ponds because these ditches are the last remaining habitats available (USFWS, 2003). The risk factor associated with contaminants in runoff is of concern and will likely increase in both roadside ditches and the rest of CTS suitable habitat (USFWS, 2003).

Rodent Control

CTS utilize ground squirrel and pocket gopher burrows for aestivation, with the STS most often utilizing the burrows of gophers (USFWS, 2003). It has been found that the presence of STS is significantly correlated with the presence of gophers (USFWS, 2003). While the CTS is probably not at risk from ingesting the rodenticides used to control these two species, the CTS may potentially be at risk from indirect exposure to them inside the burrows or concentrated in breeding ponds (USFWS, 2003). Some of these rodenticides used in Sonoma County include chlorophacinone, diphacinone, and strychnine (USFWS, 2003). Fumigants are other pesticides used for rodent control, such as aluminum phosphide, carbon monoxide, and methyl bromide, which are injected into burrows by using a cartridge or by pumping (USFWS, 2003). These fumigants can be both directly and indirectly lethal to the CTS, although the effects of these poisons on CTS have not been assessed (USFWS, 2003). Besides the use of rodenticides, control of rodents by means of eliminating burrows results is a loss of tenable habitat for the CTS (USFWS, 2003). This is especially true for the Sonoma County DPS where most breeding locations are likely to experience a heightened degree of rodent control due to the landscaping concerns of developed areas (USFWS, 2003). Cattle owners also destroy ground squirrel burrows, and, thus, eliminate CTS habitat because the burrows put cattle at risk of breaking their legs (USFWS, 2003).

Road-Crossing Mortality

Although no 'systematic' studies of road-crossing mortality of CTS in Sonoma County have been conducted, CTS killed by vehicular traffic have been documented both in Sonoma County and in other portions of the species ranges. In addition to the road kills mentioned previously, in one study there were 45 CTS collected during one hour period on a road bordering Lake Lagunita on the Stanford University Campus. Of the 45 collected, 28 had been killed by cars. Estimations of CTS mortality from vehicular strikes range from 25 to 72 percent. Mortality may be increased by the presence of curbs and berms, which allow salamanders to climb onto the road but can restrict their movement off of the road (USFWS, 2003).

Livestock Grazing
As opposed to many factors that negatively affect the CTS, grazing can, in moderation, have a neutral to positive impact on the CTS. Grazing maintains shorter vegetation and promotes the persistence of ground squirrel burrowing, which creates habitats essential for the CTS. Rangelands not only provide natural vernal pools, but artificial stock ponds, which have likely saved many populations from extirpation (USFWS, 2004). Although trampling from cattle affect water levels and soil integrity of ponds and banks, grazing is still compatible with CTS as long as burrowing rodents are not completely eradicated (USFWS, 2004).

7.2.2 Baseline Status

The CTS is found only in California. The historic range of the CTS included large portions of the Central Valley of California from the southern Sacramento Valley (north of the Sacramento River delta) to the southern San Joaquin Valley (USFWS, 2002). The CTS was also found in the lower foothills along the eastern side of the Central Valley and in the foothills of the Coast Ranges (USFWS, 2002). CTS have been historically documented in 27 counties but are no longer found in three (USFWS, 2005). The USFWS believes that the CTS is still found in the remaining 24 counties (USFWS, 2005). CTS have been found in 10 of the 17 California vernal pool designations defined by Keeler-Wolf in 1998 (USFWS, 2004).

The historical distribution and numbers of the CTS in Sonoma County on the Santa Rosa Plain are uncertain due to limited information collected on this population prior to the 1990s. CTSs are believed to have occupied a much larger region based on habitat requirements of the species for low elevation, seasonally filled breeding ponds and small rodent burrows, the ecology of the taxon, the general trend of urban development, and other adverse factors affecting the species. The closest CTS populations to Sonoma County are located in Contra Costa, Yolo, and Solano Counties, which are separated from the Sonoma County population by the Coast Range, Napa River, and the Carquinez Straits, and distance of about 72 km (USFWS, 2003).

Population Dynamics
Currently, the Sonoma County CTS and the Santa Barbara CTS are considered the most vulnerable populations of California tiger salamanders. Urban development causing habitat loss continues to be the primary factor affecting their continued existences, and both populations are particularly vulnerable to the risks associated with small, restricted populations. Low population numbers amplify the impact of high death and low birth rates, as well as the effects of genetic drift and inbreeding. A reduction in genetic variability can render a species less capable of adapting to future environmental changes, thus reducing overall fitness. Small populations are also threatened by the deterioration of environmental quality (USFWS, 2003)

Although the total number of individual CTS in Sonoma County is unknown, the STS population is scattered in isolated breeding sites on a small portion of its historic range on the Santa Rosa Plain. As of the final rule determining endangered status of the STS, there were eight breeding sites known to exist, distributed in and around the City of Santa Rosa. Other sightings of STS had been documented in locations south of the Cotati area in roadside ditches and low quality pools, although these locations were not considered viable breeding sites. The eight viable breeding sites were defined as follows: (1) Hall Road Preserve; (2) Federal Emergency Management Agency (FEMA)/Broadmore North Preserves; (3)Engel Preserve; (4) Northwest Air Center; (5) Southwest Air Center; (6) North Air Center; (7) Wright Avenue; (8) South Ludwig Avenue (USFWS, 2003). Specific elements affecting each population are outlined below:

1) Hall Road Preserve

This breeding site sits on 76-ha and is the largest preserved area where STS are currently known to inhabit. However recent studies have demonstrated the low productivity of breeding occurring. The land surrounding the preserve is slated for urban development. Exotic predators of the salamander possibly impacting this population are the Louisiana crayfish, stickleback fish, and bullfrogs (USFWS, 2003).

2) FEMA/Broadmore North Preserve

This breeding site consists of two contiguous properties on 30.5 acres (ac). The FEMA preserve on 24 ac is one of the most productive STS breeding sites, and the Broadmore North Preserve is a conservation area of the Bellvue School District. Urban development encroaches on the site from the north and east sides, and a new road and housing development on the west partially blocks the migration route to breeding pools at the Air Center. This construction, as of 2003, was only half completed, and it is expected that the final development will entirely block migration between the two sites (USFWS, 2003).

3) Engel Preserve

Not many STS individuals are suspected at this site due to low numbers of larvae observed. Lying on 16-ha of privately-owned preserve, salamanders were not found at this breeding habitat until the 2001/2002 rainy season. The presence of Todd Road along the southern boundary of the preserve threatens the population because it obstructs the migration routes to southern aestivation sites (USFWS, 2003).

4) Northwest Air Center

This breeding site consists of one pond on private land. The north and west borders are adjacent to roads supporting heavy traffic. To the east and south, housing developments have eliminated migration routes, thus rendering the breeding area isolated from the remaining seven (USFWS, 2003).

5) Southwest Air Center

This breeding site also consists of one pond on private land. The City of Santa Rosa has issued permits for development that will likely extirpate the population from this site. The destruction of the breeding area will further isolate this population and all STS populations, although individuals may still utilize the breeding areas in the FEMA/Broadmore Preserve (USFWS, 2003).

6) North Air Center
This breeding site consists of one pond on private land. Migration is restricted on three sides due to residential and commercial developments: houses to the west, road to the north, and corporate park to the east. More residential developments are approved by the City of Santa Rosa toward the south, and the Army Corps of Engineers has issued permits with the USFWS for the fill of this breeding site. Due to the emergency listing of the species, construction has temporarily halted (USFWS, 2003).

7) Wright Avenue
The condition or existence of this population cannot be confirmed. This breeding site is located on private land, to which access has been denied for many years. Construction on this property is not currently proposed; however, no protection exists to prevent breeding sites and upland habitat from being developed. The City of Santa Rosa approved residential development along Wright and Ludwig Avenues, which will concomitantly destroy habitat and increase vehicular traffic (USFWS, 2003).

8) South Ludwig
The condition or existence of this population also cannot be confirmed, as it sits on inaccessible, private land. Construction on this property is not currently proposed; however, no protection exists to prevent breeding sites and upland habitat from being developed. Threats from increased traffic on Ludwig Avenue threaten this population (USFWS, 2003).

Not long after the STS was listed as endangered, six additional breeding sites were recognized: (1) Gobbi; (2) Duer Road; (3) Haroutunian; (4) Alton Lane; (5) Southwest Community Park; (6) Yuba Drive. All breeding locations with the exception of Haroutunian and Alton Lane are distributed in the City of Santa Rosa and immediate associated unincorporated areas (USFWS, 2004).

Habitat Conservation
One habitat conservation plan was issued on September 12, 2008 for the Sonoma County DPS that will help mitigate impacts of a 4.13 acre loss of habitat (Sonoma). A second plan that will help mitigate future development impacts is the Santa Rosa Plain Conservation Strategy. This strategy was finalized in December of 2005 and aims to mitigate future impacts to listed species from development on the Santa Rosa Plain. The recovery of these species is a second component of the strategy; areas for species conservation are proposed and short- and long-term milestones are formulated to accomplish recovery goals. The result of this strategy is the adoption of eight conservation areas for the STS, along with one STS preserve system (Santa Rosa).

7.3 References

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8: DELTA SMELT

8.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the Delta smelt (*Hypomesus transpacificus*) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The non-federal actions that are reasonably certain to occur within the action area consist of urbanization, toxic substance contamination of Sacramento-San Joaquin Delta water, and water resource developments for agriculture and municipal demands. The water resource developments include, but are not limited to, reservoirs, water facilities, development that fosters increased pumping capacities, and hundreds of small water intake pipes and diversions that supply water for Delta farms. The majority of these actions divert water out of the Sacramento-San Francisco estuary. As the water resource developments increase in number, the volume of outflow in the Delta incrementally diminishes and causes complex consequences for the delta smelt. Coupled with seasonal fluctuations in precipitation, these water diversions act synergistically to hinder sufficient food resources, influence natural flow patterns, physically entrain individuals in pumping facilities, change non-native species abundance, and alter concentrations of toxic substances.

8.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the delta smelt status at this time. However, the baseline condition of each assessed delta smelt species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline includes a general discussion of factors that may affect delta smelt within the action area. This information is presented in Section 8.2.1. Additional information on the current distribution and population dynamics of the delta smelt is presented as part of the baseline status in Section 8.2.2.

8.2.1 Factors affecting species environment within the action area

The delta smelt is highly vulnerable to extinction because of its short life span, present small population size, and restricted distribution (USFWS, 1993). According to the USFWS Sacramento-San Joaquin Delta Native Fishes Recovery Plan (USFWS, 1996), delta smelt decline has multiple and synergistic causes (USFWS, 1996). The Plan listed the following causes in order of relative importance to the species decline: (1) Reduction of outflows; (2) Entrainment losses to water diversions; (3) High outflows; (4) Changes in food organisms; (5) Toxic substances; (6) Disease, competition, and predation; (7) Loss of genetic integrity. In approximately the same descending order of importance, these causes are listed below as sections 8.2.1.1 - 8.2.1.5 (USFWS, 1996).

8.2.1.1 Habitat alteration, curtailment, and degradation

The Sacramento-San Joaquin Delta has been significantly altered by human activity and restoration programs targeting Delta fishes, which have influenced spawning habitats, migration corridors, and rearing areas in upstream locations, the Delta, and Suisun Bay and Marsh (USFWS, 1996). While the trend of building large-scale water projects upstream that deplete Delta inflows has subsided, huge demands on already dwindling water resources will continue as the population of California increases to 50 million people by 2020 (USFWS, 1996). The delta smelt may be particularly sensitive because studies have shown the species to display a weak stock-recruitment relationship (ie., little evidence of the effect of parent population size on subsequent recruitment), which indicates that environmental or habitat factors play a significant role in limiting population abundance (USFWS, 1993).

Water Outflow

Delta smelt are sensitive to changes in the delta hydrology that result from water resource development in and upstream of the Delta (USFWS, 2004a). Moderately high spring outflows are critical to moving fish downstream to shallow water areas around Suisun Bay (USFWS, 1996). Production of zooplankton and phytoplankton are promoted by the well-mixed shallow water and offer rich feeding grounds for the delta smelt (USFWS, 1996). Both a reduction and an increase in the Delta outflow has the potential to disturb the delta smelt's access to this source of food (USFWS, 1996).

Reduced outflows caused by upstream storage and diversions are determined to be the primary cause of delta smelt decline, particularly when in coincidence with dryer years (USFWS, 1996). A reduction in Delta outflow maintains fish larvae and juveniles in the deep, narrow channels of the Delta where productivity of phytoplankton is lower due to lack of sunlight penetration (USFWS, 1996). Inadequate food resources in these habitats result in threats of poor nutrition and starvation for the delta smelt (USFWS, 1996). While no statistical relationship has been found between outflow rate and abundances of delta smelt, Kimmerer (2002) has shown a change in abundance as it relates to historical flow (USFWS, 2004a). Moyle and Herbolf (1989) have also found that the lowest delta smelt numbers occurred either in years of low or extremely high outflows: unusually dry years from 1987 to 1991 have coincided with declines in delta smelt abundance, as well as unusually wet years with exceptionally high outflows during the years of 1982-1983, 1986, and 1998 (USFWS, 2004a). However, there was no outflow-abundance relationship at intermediate outflows (USFWS, 1996). Years of high delta smelt abundance were strongly correlated with springtime location and duration of the 2 parts per thousand (ppt) bottom isohaline (X2) demarcation (USFWS, 2004a). This correlation is supported by the higher captures of delta smelt below 2 ppt and in shallow habitats (when waters of 2 ppt are near shallow habitats). The link between preferred habitat and fall abundance of the delta smelt also supports the notion that suitable habitat (eg. available nursery habitat) may limit population abundance (USFWS, 2004a).

High outflows also change the Delta hydrology, although the resulting consequences to the delta smelt are less severe than those resulting from reduced outflows (USFWS, 1996). High outflows may flush delta smelt and its spawn out of the system, along with its food source, zooplankton (USFWS, 1996). These depleted populations of invertebrates reduced the native fish populations and allow easier access to the delta by non-native copepods, clams, and fish (USFWS, 1996).

Water Diversions

Water is pumped out of the Delta system mainly by large diversions of the Federal Central Valley Project (CVP) and State Water Project (SWP) for agriculture and municipal demands in California (USFWS, 1996, 2004a). Over 1,800 smaller diversions also pump water out of the system for Delta farms and power plants west of the Delta (USFWS, 2004a). Besides reducing the outflow of water, water diversions cause huge entrainment losses to the delta smelt. Especially for younger fish that are planktonic and weak swimmers, large numbers of young delta smelt are entrained at the CVP and SWP plants. In addition, when these two large diversions are in operation, the young delta smelt are susceptible to entrainment by hundreds of siphons and pumps throughout the delta that irrigate Delta islands (USFWS, 1996, 2004a). Efforts are made to salvage the detained fish and truck them back to the Delta, but only those fish longer than 20 mm are rescued (USFWS, 2004a). Unfortunately, effectiveness of this rescue operation has not been well-assessed, and a majority of delta smelt are too fragile to survive through the process (USFWS, 2004a).

Larvae are more likely to be entrained during dry years due to their higher concentration in river channels and the changes in Delta hydraulics (USFWS, 2004a). High export pumping in dry years shifts small delta smelt upstream to Delta channels rather than in Suisun Bay where they are relatively free from entrainment (USFWS, 2004a). Studies have quantified the losses of entrained larvae and juvenile delta smelt and have estimated numbers to be around several million (USFWS, 2004a). More studies are being conducted, as the season, location, and size of the diversion are all major factors influencing entrainment (USFWS, 2004a). The California Department of Fish and Game (CDFG) has pinpointed the CVP and SWP as a major source of population impacts to the delta smelt under certain conditions (USFWS, 2004a). They have estimated the loss of juvenile smelts to the CVP and SWP operations to range between 11 to 46% annually (USFWS, 2004a). Other major diversions affecting delta smelt habitat are the power generation facilities near Antioch and Pittsburg owned by the Pacific Gas and Electric Company (USFWS, 1996).

Entrainment data has been collected for years at the CVP and SWP operations and reveals that a large number of delta smelt are annually subject to mortality (USFWS, 2004a). Furthermore, optimal rearing conditions within the Delta have been removed since the 1970s due to water exports: delta smelt were more abundant in the delta and less abundant in Suisun Marsh in pre-decline period, and delta smelt are now less abundant in the Delta and more abundant in Suisun Marsh[1] (USFWS, 2004a). The possible decision to switch temporary barriers to permanent barriers in the Delta that maintain water levels for in-delta diverters results in additional impacts to the delta smelt (USFWS, 2004a). The temporary barriers operate from April of each year to November, where upon they are removed, and prevent smelt movement and alter water hydraulics (USFWS, 2004a). Permanent barriers could increase the operating period, as well as introduce other influences on the delta smelt (USFWS, 2004a). Computer simulations by the California Department of Water Resources (2003) have shown that the permanent barriers change south delta hydrodynamics, increasing central delta flows toward the state and federal export facilities (USFWS, 2004a). These barriers are currently being proposed by the California Department of Water Resources (CDWR) and U.S. Bureau of Reclamation (USFWS, 2004a).

Proposed Modifications to the Water System

Increasing water resource demands go hand-in-hand with an increasing population. Specifically for the Central Valley in California, six new projects are proposed that would greatly increase the amount of water diverged and would likely result in lower Delta outflows and increased entrainment (USFWS, 2004a). These projects are listed below in Table 8.1. The CALFED Bay-Delta Program proposes to increase their surface water storage capacity through the following projects: (1) north of the delta off stream storage; (2) Shasta enlargement; (3) Los Vaqueros Expansion; (4) in-delta storage; and (5) additional storage in the Upper San Joaquin (Friant) (USFWS, 2004a). Management of the Delta's resources by State and Federal agencies, along with stakeholders, has helped alleviate some of the dangers of diverting too much water, although how effective these management tools will be in future years remains unclear (USFWS, 2004a).

Table 8.1: Proposed Projects for Central Valley, California (USFWS, 2004a)

Project	Capacity Increase[1]	Location
Freeport Regional Water Project	Project diverts up to 185,000 acre-feet(af)/year of water.	Located at a point of diversion north of the delta at Freeport.
Los Vaqueros Reservoir Expansion	Project entails additional 400,000 af of off-stream storage.	New and existing facilities used at Old River and/or Middle River.
Reclamation and California Department of Water Resources (CDWR)	Pumping capacity will be increased from 6,680 ft^3/s, to 8,500 ft^3/s, eventually reaching 10,300 ft^3/s.	State Water Projects (SWP) Banks pumping plant

[1] This "decline" period refers to the years of 1982-1983 when the delta smelt population took a precipitous decline, and populations numbers stayed consistently low though the years 1984-1992 (USFWS, 1995, 2004a, 2004b).

Reclamation and CDWR	Construction of a 400 ft^3/s intertie connecting their aqueducts, increasing pumping ability from 4,200 to 4,600 ft^3/s.	Increased pumping at Tracy Pumping Plant.
CALFED[2] Bay-Delta Program	Expansion of surface water storage capacity by 3.5 million af (including the 400,000 af project at Los Vaqueros.	Project located at existing reservoirs and off-stream sites.
City of Stockton Proposal	Construction of new intake with an ultimate diversion capacity of 371 ft^3/s.	Intake located at the southwestern tip of Empire Tract on the San Joaquin River.

[1]af = acres/feet.

8.2.1.2 Changes in food Organisms

A reduction of phytoplankton has occurred in the past few decades, in part due to the Asiatic clam invasion (USFWS, 2004a). While a study by Kimmer (2002) demonstrated that the delta smelt declined in abundance prior to any changes in phytoplankton, lower phytoplankton numbers suggests a limitation on the Delta system's capacity to support higher levels of the food web (USFWS, 2004a). In effect, key zooplankton abundance has been tied to the reduction in plankton numbers, although correlations to food reduction for fish are not as strong (USFWS, 2004a).

While a general reduction in plankton has been documented, the diatom *Melosira* has instead grown more abundant (USFWS, 2004a). This particular diatom grows in long chains and is difficult for zooplankton to graze on (USFWS, 2004a). Thus, the growth of this diatom may also be implicated in zooplankton declines (USFWS, 2004a). The cause of *Melosira* increase is unknown (USFWS, 2004a).

8.2.1.3 Toxic Substances

The waters of the estuary are contaminated with a variety of toxic substances, including agricultural pesticides, heavy metals, and other products of urbanized society (USFWS, 1996). Although the effects of these substances on the delta smelt is poorly known at this time, planktonic organisms serving as prey items may be affected during periodic flushes of certain chemicals (USFWS, 1996).

[2] The CALFED Bay-Delta Program is a collaboration among 25 State and Federal agencies to improve California's water supply and the ecological health of the San Francisco Bay/Sacramento-San Joaquin Delta (CALFED, 2007).

8.2.1.4 Disease, Predation, and Competition

Disease
The bacterium *Mycobacterium* ssp. and its correlated disease, mycobacteriosis, has been identified as potentially impacting the delta smelt (USFWS, 2004a). In a study by Antonio *et. al.* (2000), presence and infections from this bacterium were examined on wild and captive delta smelt (USFWS, 2004a). No detections of *Mycobacterium* spp. were made after fish were collected from the Sacramento-San Joaquin Estuary, nor as the fish were maintained as broodstock at water temperatures of 9-12°C (USFWS, 2004a). Abundance of *Mycobacterium* spp. was found among frequently handled broodstock, as well as those more stressed (USFWS, 2004a). The study concluded that *Mycobacterium* spp. may be lying dormant in the wild population, with infections breaking out under intensive culture conditions (USFWS, 2004a). The relevance and threat to wild delta smelt populations remain unknown at this time (USFWS, 2004a).

Predation and Competition
The non-native silverside (*Menidia beryllina*) population may be an important predator of larval delta smelt and a competitor for copepod prey (USFWS, 2004a). The silversides' population expanded rapidly in the 1990s after an accidental introduction in 1975, and estimates of silverside population abundance are inversely correlated with that of the delta smelt (USFWS, 2004a). Further, shallow water areas specifically restored for the delta smelt are being occupied by dense schools of silversides, detracting from the value of this habitat. Another potential planktivore competitor of delta smelt is threadfin shad (*Dorosoma petenense*) (USFWS, 1996). Effects on the delta smelt, however, have not been studied (USFWS, 1996).

Larval fish predators to the delta smelt that have seen population increases since the 1980s include coded-wire-tagged chinook salmon smolts (*Oncorhynchus tshawytscha*), Chameleon gobies (*Tridentiger trigonocephalus*), yellowfin goby (*Acanthogobius flavimanus)*, non-native centrarchids, and striped bass (*Morone saxatilis*) (USFWS, 1996, 2004a). Northern pike (*Esox lucius*) introduced into Lake Davis are also of concern; the delta smelt would surely be affected if this fish ever escaped or was introduced into the Sacramento River System (USFWS, 2004a).

8.2.1.5 Loss of genetic integrity

Interbreeding between the delta smelt and wakasagi (Japanese pond smelt) (*Hypomesus nipponensis*) is of potential concern because it may result in loss of valuable gametes of felta smelt and hinder the population from recovering (USFWS, 2004a). While hybridization is possible, the threat of introgression at the population level is believed to be low due to the sterility or lack of viable offspring (USFWS, 2004a). According to Swanson *et al*. (2000), temperature, salinity, and flow tolerances of the delta smelt place it at a disadvantage to the wakasagi against suboptimal conditions (USFWS, 2004a).

8.2.2 Baseline Status

The delta smelt currently occupies its historic range, although abundance is considerably lower in the south Delta (USFWS, 2004a). Delta smelt of all sizes are found in the main channels of the Delta and Suisun Marsh and the open waters of Suisun Bay where the waters are well oxygenated and temperatures are relatively cool, usually less than 20-22 degrees Celsius in the summer (68-72 degrees Fahrenheit) (USFWS, 1996 and 2004b). A substantial population size is necessary to compensate for a life history that is at a disadvantage in current Delta conditions: pelagic life style, short life span, spawning habits, weak swimming ability compared with other Delta fishes, and relatively low fecundity (USFWS, 2004a).

Population Dynamics

The Summer Tow Net Survey data show an almost complete disappearance of juvenile delta smelt in the south delta sampling stations by the mid-1970s (USFWS, 2004a). Additional studies have shown that the population has not rebounded to its original size, although a general rise in abundance has been observed in the aftermath of the long drought (USFWS, 2004a). The two-year running average of Delta Smelt Recovery index for 2003, as determined from the Fall Midwater Trawl (FMWT), is the second lowest it has been since the species was listed (USFWS, 2004a). Entrainment at the CVP and SWP continues to destabilize the population and the species will remain threatened in the foreseeable future due in part to continuing demands for water exports (USFWS, 2004a). According to Moyle (2003), delta smelt will be in danger of extinction until permanent and reliable changes are made to flow and temperature regimes that favor the smelt (USFWS, 2004a). Thus, the USFWS has concluded that the delta smelt abundance has not recovered to its pre-decline (prior to 1982) levels and that the overall population trend is negative (USFWS, 2004a).

Population estimates provided by the San Luis and Delta-Mendota Water Authority indicate an abundance of delta smelt individuals between 1 million and 12 million (USFWS, 2004a). The USGWS and others have identified flawed methods with this estimation, specifically in the reinterpretation of the Fall Midwater Trawl (FMWT) index (USFWS, 2004a). The USFWS acknowledges that this estimate is limited by uncertainties, and notes uncertainties with other estimates provided by the Summer Townet Survey and the Kodiak Trawl (USFWS, 2004a). Species, like delta smelt, whose distributional patterns are unknown but which are likely to demonstrate different abundances and distributional patterns in different parts of their range are unlikely to be estimated with any useful degree of accuracy (USFWS, 2004a). Efforts to continue future research to determine more reliable population size estimates are encouraged by the USFWS (USFWS, 2004a).

The San Luis and Delta-Mendota Water Authority (2002) used the population estimates discussed above to estimate the probability of extinction of delta smelt (USFWS, 2004a). They estimated that if the sub-adult population is 12 million, then the probability of extinction of delta smelt by 2050 is less than one percent (USFWS, 2004a). However, the USGS (2003) indicated that the use of their population estimates as a basis for estimating extinction probability would result in a severe underestimate (USFWS, 2004a). Bennett (2003) provided another type of analysis (USFWS, 2004a). He evaluated the likelihood of delta smelt populations falling below an "effective population size" (ie. a Fall Midwater Trawl index of less than 100 for two straight years) (USFWS, 2004a). Using this methodology, he determined there was a 13% chance that the delta smelt's population size could fall below an effective size within 10 years, and a 33% chance that the delta smelt's population size could fall below an effective population size by 2025 (USFWS, 2004a).

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9: SALT MARSH HARVEST MOUSE

9.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the Salt Marsh Harvest Mouse (SMHM) (*Reithrodontomys raviventris*) that are reasonably certain to occur in the action area. Future federal actions unrelated to the proposed action are not considered because they are subject to consultation pursuant to Section 7 of the Endangered Species Act (Act). Numerous non-federal actions that could affect the SMHM are reasonably certain to occur within the action area. These activities are associated with modifications to salt and brackish marsh habitats and typically include urban and industrial developments, filling and diking activities, maintenance of levees, and conversions of salt marsh to brackish water habitats. Many of these activities are linked and create complex consequences for the listed species in the action area. These factors reduce the quality and suitability of SMHM habitat by influencing the presence of tidal channel systems and the inland extent of high and low tides. Encroaching developments exacerbate the inland extent of tides by reducing SMHM escape cover from predators. Moreover, predatory impacts are intensified as urbanization displaces predators farther into marshland habitat.

9.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as the effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the SMHM's status at this time. Details of the SMHM's habitat description and known locations are included in Attachment III. However, the baseline condition of each assessed SMHM species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included in this assessment, the environmental baseline presents a general discussion of factors that may affect the SMHM within the action area. This information includes both detrimental and potentially beneficial factors affecting the species and is presented in Section 9.2.1. Additional information on the current distribution and population dynamics of the SMHM is presented as part of the baseline status in Section 9.2.2.

9.2.1 Factors affecting species environment within the action area

In the US Fish and Wildlife Service (USFWS) Recovery Plan (1984), SMHM decline is attributed to five primary factors: habitat loss, marsh fragmentation, loss of high marsh zone due to backfilling, land subsidence, and vegetational change. These factors continue to affect the species today, with habitat loss and fragmentation contributing most significantly to SMHM decline. These factors all relate to habitat alteration and are discussed together in Section 9.2.1.1. Predatory impacts also affect the SMHM and its environment. Predators include cats, non-native red foxes, and the Norway rat, among others. Predatory impacts are discussed in Section 9.2.1.2. Conservation actions include restoration and protection projects that have been, or are currently being completed, to reconstruct and enhance the salt marshes of the San Francisco Bay Area. These projects are aimed at the SMHM, as well as other San Francisco Bay species, and are presented in 9.2.1.3.

9.2.1.1 Habitat loss and alteration

Fragmentation and loss of habitat are the main factors affecting SMHM populations (USFWS, 1984). An 84% reduction in the tidal marsh bordering San Francisco Bay has occurred since 1850, with the greatest degree of destruction occurring in the southern Bay region (USFWS, 1984). This destruction is attributed to urbanization, and the filling and diking of wetland areas (USFWS, 2007a). Diked wetlands only provide marginal to inappropriate habitat for the SMHM, with most in the Bay region specifically managed as waterfowl habitats (USFWS, 1984 and Goals Project, 2000). Recently, however, diked wetlands adjacent to the Bay have become more important as the tidal marshes have decreased in size and quality (Goals Project, 2000), although they too are threatened by urban and industrial development (Goals Project, 2000).

Two physical factors that affect marsh habitat include erosion and freshwater introduction (USFWS, 2007a; and Goals Project, 2000). The shoreline of the East Bay is eroding, from San Leandro to Calaveras Point, and could lead to the loss of several SMHM populations in the area (USFWS, 2007a). Fresh water introduction along Coyote Creek, Alviso Slough, and Guadalupe Slough (South San Francisco Bay) has caused salt water vegetation to be degraded and converted to freshwater and brackish water vegetation. The South Bay wastewater facilities in the Alviso and Sunnyvale areas are responsible for the freshwater input into about 600 acres of previous salt marsh, and the converted marsh areas probably no longer support SMHM (USFWS, 1984 and 2007a). The San Jose sewage treatment plant, one of these facilities, pumps more than 120 million gallons a day of treated water into San Francisco Bay, while diluting the salt content in the marsh and changing the marsh ecology (Rendon, 1999). This freshwater intrusion kills the pickleweed on which the SMHM depends and benefits plant species like cattails and bulrush which are not used by the SMHM (Rendon, 1999 and Shellhammer, 1998).

Subsidence (up to 10 feet) is a third physical factor affecting SMHM habitat. Caused by groundwater pumping, subsidence has occurred from Palo Alto to Alviso during the last hundred years. Consequently, many marshes, including Palo Alto marsh, have changed from pickleweed to cordgrass, resulting in a decline in SMHM (USFWS,1984). Four species of non-native cordgrass (*Spartina spp.*) are rapidly establishing in the salt marshes and mud flats in the San Francisco Estuary and threaten the physical and biological ecology of the estuary (SFEISP website). Three of these species, "*S.*

densiflora, S. patens and *S. alterniflora*-hybrids can all displace native pickleweed (Salicornia spp.), which provides critical habitat for the salt marsh harvest mouse" (SFEISP website). Non-native Spartina populations were first introduced 25 years ago and have now spread to more than 1000 acres (SFEISP website).

Subsidence and diking throughout the SMHM's range, but especially in South San Francisco Bay, have eliminated the peripheral halophyte zone which is important to the SMHM (USFWS, 1984). The upper peripheral halophyte zone of most marshes has been filled in, covered over, or converted to salt ponds. Certain marshes are completely submerged during high tides, leaving insufficient escape habitat for the SMHM (USFWS, 2007a). SMHM disappear from marshes without escape cover because, during the highest tides, they either expose themselves to predators by moving into the open, or they drown (USFWS, 1984). Development and subsidence have reduced the upper marsh to strips that are "just a few feet wide on the steep sides of levees" (Wong, 2004). These strips are often separated from each other by distances too great for the SMHM to travel (USFWS, 1984). In addition, the southern subspecies of SMHM is also endangered due to commercial and residential development around San Francisco Bay which has caused a decrease in the available pickleweed habitat (Brylski, 1999). These habitat changes have led to the loss of SMHM from many marshes and the small numbers of animals seen in most other marshes (Goals Project, 2000).

9.2.1.2 Predation

In addition to its native predators, including snakes, owls, hawks, egrets, herons, clapper rails, and other raptors, the SMHM also has non-native predators such as cats (house and feral), dogs (feral) and red foxes (USFWS, 1984; Shellhammer, 1998; CDFG, 2000; Rendon, 1999; CDPR online; Goals Project, 2000; and eNature, 2007). Other predators may include gulls, weasels, and other mammals (Brylski, 1999). Predators typically strike when the SMHM is forced out into the open during high tide. However, very little is known about the effects of predators on the SMHM (Goals Project, 2000). Predator control, usually for red foxes, has been implemented in several marshes containing California clapper rails and SMHM; however, predator control is not typically found in marshes that may only contain SMHM (Goals Project, 2000). Non-native predators have gained access to the SMHM habitat because most remaining marshes share an upper side with a leveed salt pond, business park, or subdivision (Shellhammer, 1998). The amount of upland buffer necessary to protect the SMHM from predators (especially cats) is currently unknown (Goals Project, 2000). The USFWS recommends 100 feet; however, 100 feet of grassland may not be sufficient to protect the SMHM from feral cats and dogs (Goals Project, 2000).

Little is known about the interactions between rodent species within the diked marshes; however, one study in 1988 showed that SMHM were seasonally displaced from "optimal habitat by California voles" (Goals Project, 2000). In addition, little is known about the interaction between SMHM and non-native rats. The non-native Norway and roof rats are considered potential predators of the SMHM and are known to occur with the SMHM in several marshes (Goals Project, 2000). Rat control has not been implemented and is especially difficult because there is no poison specific to rats that is concurrently safe for the SMHM (Goals Project, 2000).

9.2.1.3 Conservation actions

The Recovery Plan (USFWS, 1984) proposed five actions required to secure the status of the SMHM: (1) manage existing marsh and potential habitat; (2) create new habitat; (3) restore upper portions of marshes; (4) conduct additional biological research; and (5) continue ongoing management. The Plan documented several recovery actions by the USFWS and the California Department of Fish and Game (DCFG) that had addressed these goals, involving the San Francisco Bay National Wildlife Refuge (SFBNWR), the San Pablo Bay State Wildlife Area (SPBWA), the Suisun and San Pablo Bay marsh areas, the Palo Alto City Nature Center Marsh, and other smaller marshes and bays along the San Francisco Bay coast.

Conservation plans have continued to be implemented in the 24 years since the Recovery Plan was publicly released. One Habitat Conservation Plan (HCP) was issued for the SMHM on February 23, 1999 for the duration of three years (HCP). The Zanker Road Resource Mgmt., Ltd. HCP is an 0.83 acre parcel of land located in Santa Clara County consisting of "ruderal grassland on levees of diked wetland" (HCP). Other plans have included restorations of diked and salt water-degraded marshes, acquisitions of private property for wetland restoration, the South Bay Restoration Project involving the Cargill Company, and conversions of agricultural land to marsh habitat. These conservation actions are described in Winton, 2001; Sunnyvale, 2005; Wong, 2004; USFWS 2001; USFWS 2006.

9.2.2 Baseline Status

SMHM evolved with the creation of San Francisco Bay about 8,000 to 25,000 years ago and were historically found in most of the marshes throughout San Francisco Bay (USFWS, 1984). SMHM are currently found in tidal and non-tidal salt marshes in San Francisco, San Pablo, and Suisun bays (USFWS, 2007a and CDFG website). It is thought that the wetlands and marshes of the Sacramento-San Joaquin Delta were probably too fresh to support the SMHM and that the Collinsville-Anitoch area was, and likely remains, the eastern limit of their distribution (USFWS, 1984 and 2007a). Populations of SMHM occur in the diked marshes near Collinsville and in both diked and tidal marshes along the Contra Costa County coast (USFWS, 1984). The western limit of the northern subspecies is the marshes bordering the mouth of Gallinas Creek on the upper Marin Peninsula (USFWS, 1984).

The southern subspecies of the SMHM historically had two populations in San Pablo Bay (USFWS, 1984). One population could be found on the CDFG Ecological Reserve at Corte Madera (USFWS, 1984). In 1980, trapping at this location was unable to confirm the presence of SMHM (USFWS, 1984). However, the other population is moderate in size and exists near the Richmond landfill (USFWS, 1984). Populations of southern subspecies can also be found south of the San Mateo Bridge (USFWS, 1984).

Population Dynamics

There are few accurate density figures for SMHM because their numbers are so low (hence errors of sampling are high) (USFWS, 1984). In 1980, SMHM populations in several of the larger marshes around the Bay, including Petaluma Marsh, Corte Madera Ecological Reserve, Benecia State Park (Southampton Bay), areas west of Pittsburg, Belmont, New Chicago Marsh near Alviso, along Mowry Slough, and along the Alameda Flood Control Channel were either extremely low or completely missing (USFWS, 1984). Major SMHM populations were found at the mouth of Tolay Creek, Lower Tubbs Island, Fagan Marsh, and the marshes near the San Francisco Bay National Wildlife Refuge headquarters in Newark, near the mouth of Old Alameda-Mt. Eden Creek, and the Collinsville marshes (USFWS, 1984). In the mid-1980s, there appeared to be a few thousand animals at the peak of their numbers each summer, distributed around the Bay in small, disjunct populations, often in marginal vegetation and almost always in marshes without an upper edge of upland vegetation (USFWS, 1984). A population estimate from 1993 put the wild SMHM population at 2000 (Massicot, 2005).

More recently, the highest consistent SMHM populations are found in "large marshes along the eastern edge of San Pablo Bay and in old dredge spoil disposal ponds on former Mare Island Shipyard property" (Goals Project, 2000). "Most of these marshes are in or will be included in the San Pablo Bay unit of the San Francisco Bay National Wildlife Refuge" (Goals Project, 2000). Some parts of the Contra Costa County coastline and some parts of the Petaluma Marshes as well as the Calaveras Point Marsh in South San Francisco Bay also support large populations of SMHM (Goals Project, 2000). However, the habitat at Calaveras Point Marsh is deteriorating due to the decreasing salinity and corresponding vegetation changes (Goals Project, 2000). The SMHM is thought to be sustaining itself in a few marsh areas in the SFBNWR, including Calaveras and Dumbarton Points, Greco Island, and New Chicago Marsh (Shellhammer, 1998).

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10: SAN FRANCISCO GARTER SNAKE

10.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the San Francisco garter snake (*Thampnophis sirtalis tetrataenia*) (SFGS) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of expansion of utility matrices, residential and commercial developments, recreational facilities, land management, agriculture, dredging, channelization, flood control, and overutilization. Many of these factors are linked or act synergistically and create complex consequences for the SFGS. For example, roads often impede dispersal and movement of species. Highways with heavy traffic can be a physical deterrent for snake mobility and cause direct mortality from vehicular strikes. Highways may also concurrently impede or inhibit SFGS anuran prey species from dispersal and movement. If close to wetlands, these highways might further reduce the water quality from chemical run-off and thus affect the recruitment of anuran prey species.

10.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the SFGS status at this time. However, the baseline condition of each assessed SFGS species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline includes a general discussion of factors that may affect SFGS within the action area. This information is presented in Section 10.2.1. Additional information on the current distribution and population dynamics of the SFGS is presented as part of the baseline status in Section 10.2.2. The majority of information included in the environmental baseline section is a summary of information presented in the 5-Year Review: Summary and Evaluation of the San Francisco Garter Snake (USFWS, 2006b).

10.2.1 Factors affecting species environment within the action area

SFGS are primarily affected by habitat loss and degradation. In the USFWS Recovery Plan for the SFGS, habitat alteration has been cited as the primary threat of species decline (USFWS, 1985). Habitat loss and degradation continues to be the primary cause today. Other influential factors include overutilization, disease, predation, and invasive species. Additional anthropogenic and natural factors considered to be of a lower threat include reservoir hydrology and topography, vehicular strikes, hybridization with the red-sided garter snake (RSGS), and interspecific congenors (other *Thamnophis* species and subspecies). An overall measure of these lower level impacts has yet to be determined due to a lack of accurate population estimates. Sections 10.2.1.1 through 10.2.1.4 provide details on each of the major issues affecting the SFGS within the action area.

10.2.1.1 Habitat Loss and Degradation

Habitat loss and degradation are caused by three main factors. In the order of most to least influential, these factors include expansion of infrastructure, management practices, and fluctuating water levels. The expansion of infrastructure causes habitat fragmentation by increasing residential and commercial developments such as new roads, utility matrices, and recreational activities and facilities. Management practices, such as seral succession, use of perch ponds (e.g., shallow man-made ponds for irrigation often used in San Mateo County), decreased use of stock ponds, and practice of dredging waterways may result in SFGS habitat loss and degradation. In addition, fluctuating water levels in reservoirs, channelization and flood control, and saline inundation events can also affect habitat degradation.

Infrastructure

Urban areas are currently expanding in San Mateo County. As of 2005, the human population in the San Francisco Peninsula has increased by over 500,000 people since the end of World War II (US Census Bureau *in litt.*, 2006). Large ranches within this area have been replaced by high density urban development as well as intense agriculture operations (San Mateo County Department of Agriculture, 2004). Table 10.1 shows a number of proposed urbanization projects within San Mateo County that potentially threaten known populations of the SFGS. These five projects are currently undergoing consultation with the USFWS (USFWS 1996b). Other urbanization projects have been completed in recent years with identifiable consequences on the SFGS and its habitat. These three finished projects are summarized in Table 10.2.

Table 10.1: Proposed urbanization projects within San Mateo County currently under consultation with the USFWS (USFWS, 2006)

Location	Party	Proposed Action Type	Description	Location
Oak Avenue Park, City of Half Moon Bay	City of Half Moon Bay	Recreational Development	Clean-up and restoration construction, and development of a bicycle and pedestrian trail along Pilarcitos Creek	Oak Avenue Park, City of Half Moon Bay

Pilarcitos Watershed	Pilarcitos Quarry	Operations Expansion	Expansion of quarry into SFGS habitat	Pilarcitos Watershed
City of Pacifica, upland area between Mori Point and Calera Creek	California Department of Transportation (Caltrans)	Housing and Infrastructure Development	Housing development, and Calera Parkway Project that will widen sections of Highway 1	City of Pacifica, upland area between Mori Point and Calera Creek
Dennison Reservoir	--	Continuation of Activity	Potentially continue dredging activities due to increased levels of siltation	Dennison Reservoir
San Francisco Public Utility Commission Property (SFPUC), San Mateo County	SFPUC	Development	Construction of wireless facilities on property	San Francisco Public Utility Commission Property (SFPUC), San Mateo County

Table 10.2: Completed urbanization projects in San Mateo County

Location	Party	Project	Description	Citation
Canada Rd. on San Andreas Ridge down to Brisbane	Pacific Gas and Electric (PG&E)	Jefferson-Martin transmission line project	Replacement and rerouting of 27 miles of transmission line	USFWS, 2005
Cupid's Row Canal, West of Bayshore	---	---	Temporary dredging activity	USFWS, 1996b
San Francisco International Airport (SFO), West of Bayshore	California Department of Transportation (CALTRANS)	Bay Area Regional Transport Station (BART)	Expansion of station, requiring realignment of creek and Cupid's Row Canal.	USFWS, 1996b; California Department of Fish and Game, 2005

The Jefferson-Martin transmission line project affected 0.38 acres of wetlands that provided foraging and dispersal area for the SFGS due to substation expansion. The construction also temporarily affected 7.55 acres of upland habitat suitable for species dispersal and aestivation. Mitigation for these effects is now being established on California Public Utilities Commission (CPUC) property. The dredging activity at Cupid's Row Canal may have been a temporary disturbance, but it indicates the continuing threat of habitat destruction for the SFGS (USFWS, 2006a). The expansion of a Bay Area Regional Transportation (BART) station at the San Francisco International Airport (SFO) bisected a West of Bayshore SFGS population (USFWS,1996b; California Department of Fish and Game, 2005). The BART expansion project required the realignment of a creek and Cupid's Row Canal, and human activity during the construction caused the mortality of six SFGS individuals (USFWS, 2006a). Although construction caused the take of SFGS individuals, the impact of the completed BART expansion project on the SFGS is thought to be low due to the elevated design of the structure. However, the long term impacts of the project are unknown. Mitigation for the detrimental impacts of the BART project included the purchase of Steele Ranch, a suitable habitat for the SFGS, as well as restoration activities at West of Bayshore (USFWS, 1996b). However, both mitigation efforts were never fully realized (McGinnis, pers.comm. 2006), and no further information on the efforts is available (USFWS, 2006b).

The increased growth in human population has increased the demand for recreational opportunities and supported the growth of golf courses. Recreational activities, such as running and jogging, are not threats to the SFGS; however, biking and off-road vehicle (OHV, or off-highway vehicle) usage at the West of Bayshore are known to degrade SFGS habitat and kill individuals (Larson, 1994). OHVs have led to the degradation and erosion of upland habitat at Mori Point (D. Fong, pers. comm. 2006). More enforcement and regulation are needed to prevent OHV trespass on private and protected public land. Chemical and pesticide use at golf courses contaminates nearby lakes and streams that are not usually associated with urban development. These waterways are important aquatic habitats and movement corridors for the SFGS and its primary anuran prey base. Chemical applications may potentially contribute to the reduction in habitat quality (Sparling *et al.* 2000; A. Willy, pers, comm. 2006). In one case, over-application of phosphorous to golf course ponds in Solano County resulted in the mortality of the California Red-Legged Frogs (*Rana aurora draytonii*) and their larvae (USFWS, 2002a). Similar events causing direct mortality of the SFGS are a possibility.

Management Practices

Changes in management practices have encouraged the presence of seral ecosystems in both protected grasslands and aquatic habitats. The word “seral” refers to the successive changes in flora and fauna during the process of ecological succession. These dynamic ecosystems threaten the SFGS in certain areas of the San Francisco Bay Peninsula (H. Mcquillen, pers. comm. 2006; S. Larson, pers. comm.). Grass-dominated uplands are important habitats for the SFGS because rodent burrows are utilized for hibernation during winter months (Larsen 1994; McGinnis *et. al.* 1987) and for migration corridors between aquatic habitats. The loss of fire suppression allows for the dominance of woody species over grasslands and potentially inhibits rodent burrowing (D. Hankins, *in litt*. 2006). The absence of domestic grazing, which results in thick brush canopies, renders grassland habitat unsuitable for the SFGS. Seral conditions in aquatic habitats also threaten the SFGS and its amphibian prey. The presence of cattails increase siltation rates and lead to a reduction in open water, a necessity for Pacific tree frog and California red-legged frog reproduction and survival (McGinnis *et. al.*, 1987).

Agriculture practices contribute to the loss of suitable SFGS habitat. Although data are available on the relationship between other garter snakes and row crops, information specific for the SFGS is not available. One study found the plains garter snake (*Thamnophis radix*), which has a habitat similar to the SFGS, absent from fields that had been plowed, but not from fields that had been grazed or burned (Conant *et. al.,* 1945*).* More recent studies have found garter snake numbers to be lower than average in suitable wildlife habitats surrounded by plowed agricultural areas (Keller and Heske, 2000). Scientists attribute these low numbers primarily to plowing activities and speculate that heavy machinery increases soil compaction, reduces overall soil tilth (physical condition of the soil in relation to its fitness for growth), and leads to soil erosion (A. Allen *in litt.*, 1995).

Fluctuating Water Levels
Stock ponds in ranges or pastures benefit the SFGS by providing aquatic habitats. However, these types of ponds are decreasing. Instead, perch ponds are in high demand in rural areas of San Mateo County and are of little value to the SFGS (J. Howard *in litt.*, 2006). Because perch ponds are primarily used for irrigation, water levels fluctuate between growing seasons. Rapid draw-down for summer crops, a high utilization period, removes much of the water and renders the ponds unsuitable for the SFGS and amphibian prey species (McGinnis, 1984; McGinnis, 1987; San Mateo County Department of Agriculture, 2004). Perch ponds further affect the SFGS because the water is acquired from nearby waterways. Extracting water from these creeks and streams results in unreliable water levels, potentially extirpating the SFGS and its prey species (McGinnis, 1984).

Shallow ponds are important habitats for SFGS prey items, including the California red-legged frog and the Pacific tree frog, because they can provide aquatic habitats through the spring and summer months. Frog egg masses may become exposed to predation and dry out when water levels drop. In areas where their prey species are unable to survive, SFGSs will likely disperse and potentially be subject to predation. Rapid draw down also increases the potential for bullfrogs to establish themselves where native ranids used to live. Bullfrogs are known to prey on other amphibians, including SFGS prey species, potentially resulting in a decline of native ranids and narrowing the feeding opportunities of the SFGS (P. Keel, pers. comm. 2006). Bullfrogs pose a direct threat to the SFGS because they consume juvenile individuals, although the SFGS may also feed on juvenile bullfrogs. While the predatory threat of bullfrogs to SFGS is thought to be low, the threat to amphibian prey is more consequential (Barry *in litt.*, 2005). Similar events related to unsuitable habitat and amphibian prey may occur at reservoir systems where rapidly fluctuating water levels are common (Freel and Giorni, 1994).

Degradation of riparian habitats, stream channelization, and increased levels of salinity in freshwater habitats also threaten the SFGS (USFWS, 1985; Larsen, 1994). Alterations in stream and riparian habitats reduce the connectivity between sites, limit these habitats from functioning as movement areas, and reduce the complexity of streams serving as migration corridors (USFWS, 1985). Both the SFGS and its amphibian prey are negatively affected by these alterations (California Department of Fish and Game, 2005). Salinity in freshwater is detrimental to anurans, as concentrations over 7.0 ppt are lethal to Pacific tree frogs and their larvae (Larson, 1994). Increased salinity has been documented in a number of areas used as habitat by the SFGS, and the presence of saline-tolerant plants indicates degradation in freshwater quality (S. O'Brien, pers. comm.

2006). Places affected by salinization include Cupid's Row Canal, West of Bayshore, Pescadero Marsh, and Mori Point.

10.2.1.2 Overutilization for Recreational, Scientific, Commercial, and Educational Purposes

Although unauthorized take of the SFGS for recreational, scientific, commercial, and educational purposes has decreased in recent years, it is still considered a threat to the species according to employees with the California State Parks (J. Kerbavaz, pers. comm. 2006; P. Keel, pers. comm. 2006). During the 1970s and 1980s, illegal take was a primary threat occurring at SFGS habitat in West of Bayshore; however illegal take has reportedly subsided in recent years. Several factors, such as speculation towards genetic purity, may have contributed to the decline in illegal take at West of Bayshore (S. Barry *in litt.* 2006b); however, the level of reduction and impact on the species remains unclear (P. Keel, pers. comm. 2006; J. Kerbavaz, pers. comm. 2006). Amateur herpetologists have collected the species for their beautiful coloration and rarity, and some amount of illegal collection still occurs. Illegal take has also been known to occur at the Pescadero Marsh since the early 1990s as well as at ANSR (J. Kerbavaz, pers. comm. 2006; P. Keel, pers. comm. 2006). Although unauthorized take has reportedly declined in recent years, it should be noted that staffing restrictions within the California State Parks Department prevent further resource allotment to SFGS take enforcement (P. Keel, pers. comm. 2006).

10.2.1.3 Disease and Predation

Disease, parasites, and predation all negatively affect the SFGS. Of the three, disease caused by the chytrid fungus is the most threatening. The chytrid fungus (*Batrachochytrium dendrobatridus*), a lethal parasite, poses an indirect threat to the SFGS by drastically affecting its anuran prey base. The fungus has caused an epidemic that has seen prolific growth throughout the world in recent years, to some degree attributed to altered weather patterns (Pounds *et. al.*, 2006). Lethal pandemics are a real possibility for certain frog species that now, due to global warming, inhabit locations where humidity is high and daily temperature extremes are minimized (P. Johnson *in litt.*, 2006). A chytrid fungus pandemic can spread rapidly and is capable of wiping out entire amphibian populations on the Peninsula. Combined with food shortages, such a situation would have dire consequences for the SFGS, in addition to all other garter snakes in the area (Jennings *et. al.*, 1992; AmphibiaWeb *in litt.*, 2006).

Although other parasitic infections of the SFGS are threats, they are not considered ones of catastrophic magnitude. The parasites known to affect the SFGS include tapeworms, flagellate protists, and nemotode worms (Larson, 1994). Several juvenile mortalities in the West of Bayshore are attributed to these three types of parasites. Parasitic tapeworms and thorny-headed worms may use mosquito fish as hosts in the northern Bay area.

Various ranid species and the SFGS that feed on the mosquito fish are at risk for infection (M. Kolipinski *in litt.*, 2006).

Predation by native avian species and bullfrogs are a threat to the SFGS, although the impact on SFGS individuals is unknown. Of the avian species, the American crow (*Corvus brachyrhynchos)* was found to be a significant predator of the SFGS (Shine *et al.*, 2001). Other probable avian predators include red-tailed hawks (*Buteo jamaicensis*), red-shouldered hawks (*Buteo lineatus*), great egrets (*Ardea alba*), snowy egrets (*Egretta thula*), black crowned night herons (*Nycticorax nicticorax*), northern harriers (*circus cyaneus*), and great blue herons (*Ardea herodias*) (Larson, 1994; Freel and Giorni, 1994). Long-tailed weasels (*Mustela frenata*), and large mouth bass (*Micropterus affinis)* are also listed as potential predators of the SFGS (Larsen 1994).

Bullfrogs are known to prey on SFGS, although the extent of predation is unclear. Bullfrogs have only been observed preying upon juveniles, and it has been suggested that they may not significantly affect the SFGS population and can be discounted as a threat. Alternately, the SFGS preys on bullfrogs as a secondary food source, suggesting that bullfrogs may serve beneficial purposes (Barry *in litt.*, 2005). As stated previously, bullfrogs prey on the California red-legged frog and the Pacific tree frog, two significant food sources of the SFGS (Lawler *et al.*, 1999; USFWS, 2002b). Bullfrogs may also compete with the California red-legged frog for food and habitat (Lawler *et al.*, 1999; USFWS, 2002b; K. Leyse, pers. comm. 2006; S. McGinnis, pers. comm. 2006; P. Keel, pers. comm. 2006). The degree of competition is debatable because the two frogs may have significantly different diets that preclude competition. Some scientists argue that the cooler annual climate on the Peninsula limits reproduction and prevent bullfrogs from out-competing native frogs that are better adapted to the cooler temperatures (Barry, 1994; P. Keel, pers.comm. 2006).

10.2.1.4 Other natural and anthropogenic factors affecting existence

Invasive species, such as the bullfrog (discussed in the preceding paragraph), threaten the continued existence of the SFGS. Other invasive species include exotic centrarchid fish, large mouth bass (*Micropterus salmoides)* and the sunfish (*Lempomis*). All three of these non-natives prey on tadpoles of the California red-legged frog and Pacific tree frog, prey of the SFGS. In addition, the effects of bullfrogs and non-native fish species may act synergistically (Simberloff and Van Holle, 1999). Research has demonstrated that non-native bluegills (*Lepomis macrochirus)* promote the survival of bullfrogs by consuming dragonfly nymphs that prey on bullfrog larvae. Fish also avoid bullfrogs because of their unpalatability (Kruse and Francis, 1977).

Artificial water impoundments and highways may adversely affect the SFGS. The presence of steep banks and absence of level areas next to dense vegetation prevents basking areas and hinders thermoregulation of the snakes and prey species. The absence of the vegetation also increases exposure to predators (Barry, 1994). Roads and highways affect the dispersal and movement of the SFGS, and contribute to direct mortality from vehicular strikes. Snakes utilize roads for thermoregulation, and the SFGS has been observed on dirt roads around the SFO, SFPUC, and Ano Nuevo areas. Amphibian prey is also affected by roads and highways. Due to nocturnal habits, roads with activity from 2200 hours to 0400 hours may serve as effective dispersal barriers. Amphibians are sensitive to water pollutants and are adversely affected by roads in proximity to wetlands. Storm water runoff contaminates aquatic habitats and reduces recruitment of anuran individuals, potentially contributing to the decline of the SFGS.

Hybridization and interspecific competition have been suggested as possible low level threats to the SFGS. A broader range of color variation in western populations lends support to the idea of hybridization (Wharton, 1989; Barry, 1994; Larsen, 1994; McGinnis *in litt.*, 2005). Stebbins, 1985, categorized salamanders into single phenotypic traits dictating color pattern; however, others suggest that the variations result from a range of environmental variables (McGinnis *in litt.* 2005). Barry (1994, *in litt.* 1996) argues that, because the SFGS and RSGS have remained distinct populations while living in close proximity in La Honda Upland, the possibility and threat of hybridization is low. Interspecific competition between the SFGS and other *Thamnophis* species, garter and ribbon snakes (Myers *et al.*, 2008), is low and does not pose a significant threat. Dietary differences between the SFGS and the Santa Cruz garter snake (*T. atratus atratus*, SCGS) result in very little habitat overlap between the two species. While the coast garter snake (*T. elegans terrestris*, CGS) does overlap in range with the SFGS, its diet of primarily slugs and rodents results in low levels of competition as well.

10.2.2 Baseline Status

The current distribution of the SFGS lies entirely within the limits of San Mateo County, California (USFWS, 1985). Within the County, SFGSs are mainly found in the coastal areas, in addition to other small inhabited locations (USFWS, 2005). A population in San Bruno Mountain once represented the most northeastern extension of the range, although today it may be extirpated. The San Bruno Mountain population may have been moved to that area for conservation purposes by amateur herpetologists. The species from the Half Moon Bay region may have had a similar origin from attempted conservation, although no exact records are available for confirmation (Barry, 1994). Regardless, the species remain extant at Half Moon Bay (McGinnis, 1988).

The current distribution is not precisely known because much of the range is comprised of private property; however, studies indicate that the SFGS can still be found within much of its historic range (USFWS, 2006 and 2007). The historic distribution of the SFGS extends just north of the San Francisco-San Mateo County line near Merced Lake south to Waddell Creek located within the Big Basin Redwoods State Park. The population at Waddell Creek is questionable because of the suspected presence of hybrid species (Goals Project, 2000). Records show that the species may have extended south into Stanford of Santa Clara County and the west coast of the Santa Cruz Mountains. However, SFGS have been extirpated from certain locales, and field studies in these areas are restricted due to private property (California Natural Diversity Data Base, 2006). The historic distribution extends along the western coast of the peninsula south to Ano Nuevo State Reserve (ANSR). On the eastern edge of their range, the SFGS occupied the Buri Buri Ridge along the San Andres Rift and then south, in an arc, from the San Gregorio-Pescadero State Park west to Tunitas Creek (Barry, 1994).

While the current distribution may extend throughout most of its historic range, the number of SFGS populations has declined in recent years (USFWS, 2007). Habitat destruction due to small projects constantly afflicts the species. Urbanization continues to expand and the prevalent agriculture practices of disking and planting cycles increases the fragmentation of suitable habitat (Barry, 1994). Sightings of the SFGS have occurred in the Upper Crystal Springs Reservoir and Mud Dam in 1998 (San Francisco Planning Department, 2001). Others sightings have been recorded from “San Andreas Reservoir and in a sag pond between San Andreas and Crystal Springs” (San Francisco Planning Department, 2001). Currently, wild SFGS populations are limited to coastal San Mateo County and other small pockets (USFWS, 2005).

Population Dynamics

Some sources state that 65 “permanent” reproductive populations ranging from two to over 500 adults have been found on the San Francisco Peninsula (Kaplan, 2002 and Goals Project, 2000). The total SFGS population is believed to be around 1500 snakes over one year old (Kaplan, 2002 and Goals Project, 2000). All young under one year old are not included in population counts because the population increases when the young are born and returns to around 1500 the following spring due to insufficient resources (Kaplan, 2002). It is thought that half of the populations are protected to some degree by refuges including preserves and state parks (Goals Project, 2000).

Six areas are known to contain significant populations of SFGS: 1) West of Bayshore, or Milbrae, 2) Laguna Salada (Sharp Park), 3) San Francisco State Fish and Game Refuge (including both Upper and Lower Crystal Springs Reservoirs), 4) Pescadero Marsh Natural Preserve, 5) Ano Nuevo State Reserve (ANSR), and 6) Cascade Ranch (USFWS, 1985). A discussion on baseline status for each of the major populations is provided below. With the exception of West of Bayshore, sufficient information for the other five locales does not exist to discern discrete population trends. Alternatively, these trends are inferred from current habitat dynamics and conditions.

West of Bayshore

Sufficient population data exists to reveal a trend for the West of Bayshore habitat, near the SFO. At one time, this area was considered to hold the largest SFGS population, and between 1983 and 1985, 695 individuals were captured (USFWS, 1985; Natureserve *in litt.*, 2006; Larsen, 1994). Based on another study completed in the mid-1990s, only 179 individuals were captured (Larsen, 1994). Although drawing comparisons between the studies is difficult due to different methods and sampling techniques, both studies indicate that the population has declined. A third trapping occurred in 1997 before the construction of the Bay Area Regional Transport (BART) station, and only 25 snakes were captured (Larsen, 1994; S. Larsen, pers. comm., 2006; USFWS, pers. comm.. 2006). While this trapping occurred in a limited region of the West of Bayshore habitat, the low number of captured individuals was unanticipated considering the much higher numbers observed and caught in prior years (S. Larsen, pers. comm. 2006).

Declines in the West of Bayshore population are expected due to ongoing construction and habitat degradation. The reduction in the open water component of the canal system in recent years has led to a decrease in suitable habitat for an amphibian prey base. Both increases in vegetation and siltation reduce the quantity and quality of habitat (S. Larsen, pers. comm. 2006). Other negative activities affecting the SFGS in the area include encroaching development, illegal collecting, and limited law enforcement (Larsen, 1994). The SFO and USFWS have recently agreed to improve the habitat quality for the SFGS at the West of Bayshore area.

Laguna Salada

The SFGS population at Laguna Salada is also thought to have declined. This population trend can be attributed to two influxes of salt water into the snake's habitat in the 1980s (Steiner and Hafernik, 1992). No salination events have occurred since a protective levee was built, and it has been suggested that the snakes never historically occupied the site (Barry, 1994). Other detrimental activities include off-road vehicle use and illegal trash dumping at the Mori Point area (Steiner and Hafernik, 1992; D. Fong, pers. comm. 2006). These habitat degrading actions have been mitigated by the National Park Service (D. Fong, pers. comm. 2006; National Park Service, pers. comm. 2006), although no information on these actions has been provided (USFWS, 2006b). Further relief has come from a partnership between the National Park Service and the USFWS via an agreement to construct two ponds for amphibian prey of the SFGS (S. Larsen, pers. comm. 2006; H. McQuillen, pers. comm. 2006). In 1996, volunteers of the National Park Service recorded the SFGS foraging for the California red-legged frog in both of the ponds, demonstrating the benefits of the constructed ponds (S. Gardner *in litt.*, 2006).

Mori Point is located within the greater Laguna Salada area. Although the SFGS historically occupied Laguna Salda at Mori Point, the species is not thought to currently exist in this area (McGinnis, 1990; McGinnis, pers. comm. 2006). The extirpation of the SFGS occurred after ponds in the area were filled and agricultural disking practices were implemented in 1990 along Calera Creek. Prior to these landscape changes, high occurrences of SFGS individuals were reported in Calera Creek Channel (McGinnis, 1990; McGinnis, pers. comm. 2006). Following these changes, no sightings have been reported at Calera Creek based on informal walk-through surveys (MsGinnis, pers. comm.; K. Swaim, pers. comm. 2006). Observations from 1978 and 1990 indicate that there may have been SFGS movement between the Calera Creek corridor and Mori Point, based on SFGS sightings along a ridgeline separating the corridor and Mori point (Barry *in litt.*, 2006). However, no occurrences of SFGS individuals have been reported between these two locations since the development in 1990 (K. Swaim pers. comm.. 2006). A proposal for road construction and residential development along Calera Creek further challenge the recovery of the species (S. Larsen, pers. comm. 2006; H. McQuillen, pers. comm. 2006).

San Francisco State Fish and Game Refuge

This population of SFGSs lives in proximity to the Crystal Springs and San Andreas Reservoirs. While this area has been designated a refuge, it is owned and managed by the San Francisco Public Utility Company (SFPUC) (J. Stolz, pers. comm. 2006). This public land provides quality habitat for the SFGS because of the restricted access designated near the reservoirs (J. Naras pers. comm. 2006). Large quantities and high densities of individuals have been observed (Barry, 1996). However, the population is threatened from human activity because the area is not specifically managed for the SFGS (J. Stolz, pers. comm. 2006). Current trail systems and proposals for additional trails along the waterways of the refuge increase human presence and possible disturbances to the SFGS. One SFGS individual was run over by a bicycle on a road next to the SFCUP property, and other human-related stressful or lethal activities are possible (A.M. McGraw *in litt.*, 2005). A habitat conservation plan has been discussed between the USFWS and the SFPUC, although no plan has been implemented to date (S. Larsen, pers. comm. 2006; J. Naras, pers. comm. 2006).

Pescadero Marsh Natural Preserve

The population at Pescadero Marsh is thought to be declining, despite recent improvements in certain habitat conditions (USFWS, 2006b). This population has been deemed genetically significant because the SFGS phenotype in the area more closely resembles the "holotype" (i.e., defined as "the single specimen chosen as a representative type when establishing the taxonomic group) than any other population (Larsen, pers.comm. 2006). Pescadero Marsh has regular influxes of salt water that reduce the habitat quality of the area for the SFGS. During the 1990s, several recovery actions improved the fresh water quality, although no information on those actions is provided (USFWS, 2006b). The SFGS now inhabits eastern portions of the marsh and a few artificial ponds that are adjacent to originally inhabited areas. Some California State Parks staff believe that the habitat for the SFGS and their anuran prey have markedly improved (McGinnis, 2002; J. Kerbavoz, pers. comm.). However, the high salinity levels continue in much of the marsh resulting in unsuitable habitat for amphibian prey (C. Atkinson, pers. comm. 2006; P. Keel, pers. comm. 2006; J. Smith, pers. comm. 2006). Recent restoration efforts to limit salt water influx into the marsh have resulted in little success. However, restoration work aimed at neighboring properties may be benefitting the population. Staff from Pescadero Marsh and Peninsula Open Space Trust (POST) are implementing prescribed burns on Cloverdale Ranch as part of an upland conservation management plan. Further information is needed to evaluate the effectiveness of the project; however, it is anticipated to help the population (A. Willy, pers.comm. 2006).

Ano Nuevo State Reserve (ANSR)

There have been sightings of the SFGS at the ANSF since 1975. In 1987, 13 SFGS individuals were trapped around the headquarters' pond (Barry, 1978; McGinnis *et al.*, 1987). In 1988, a more comprehensive survey of the ANSR documented 57 individuals (Keel *et al.*, 1991). The study authors believe that the ANSR may contain one of the largest known SFGS populations, and they attribute these large numbers, in part, to the available wetlands and suitable upland foraging sites for amphibian prey. Between 2004 and 2005, the California State Parks and the USFWS performed two burns on 45 acres of SFGS habitat to increase local populations (Halbert, 2005). According to trapping surveys before and after the burn, the number of SFGS individuals increased from 7 to 53. Given differences in survey methodology, it is difficult to draw conclusions from the results of the two studies. Therefore, additional information is needed to access the effectiveness of burn disturbance in improving habitat quality for the SFGS (P. Halbert, pers. comm. 2006). According to California State Parks, ANSR may acquire new land from a conveyance of property in the near future. The California State Parks is currently discussing acquisition of the land, which would provide an area suitable for California red-legged frog restoration ,with a private land owner. The land would also provide more protected land for the SFGS (A. Willy, pers. comm. 2006; V. Roth, pers. comm. 2006).

Cascade Ranch

Cascade Ranch is the only private property discussed in the Recovery Plan for the SFGS (USFWS, 1985). Little is known about the population distribution on the site. However, overgrazing, in the late 1980s, and a 1989 USFWS-approved construction of a resort lodge, general store, cabins, and camping area may have reduced habitat quality for the SFGS. The project included a recovery plan for the species and a monitoring program to continue for 5 years (Biosearch Associates, 2003). The final monitoring report in 2005 demonstrated that the recovery plan had succeeded in providing suitable habitat for the SFGS as well as its amphibian prey (Biosearch Associates, 2005). In 1994, Cascade Ranch was designated as a “high quality” SFGS habitat due to the presence of ANSR breeding ponds nearby. However, the presence of Highway 1 and the proximity to agriculture fields in the area may hinder the mobility and dispersal of the species (Freel and Giorni, 1994).

10.3 REFERENCES

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11: SAN JOAQUIN KIT FOX

11.1 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the San Joaquin kit fox (*Vulpes macrotis mutica*) (SJKF) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). The numerous non-federal actions that are reasonably certain to occur within the action area consist of industrial and petroleum field development, agriculture, surface mining, and urbanization projects related to infrastructure, irrigation canals, wind farms, power lines, and aqueducts. Many of these factors are linked or act synergistically and create complex consequences for the SJKF. More specifically, these habitat modifications compress and fragment the species' range. Combined with the high mortality of the SJKF during dispersal, movement between habitats is limited and the long-term viability of the population is threatened.

11.2 ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the SJKF status at this time. However, the baseline condition of the assessed SJKF's habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline includes a general discussion of factors that may affect the SJKF within the action area. This information is presented in Section 11.2.1. Additional information on the current distribution and population dynamics of the SJKF is presented as part of the baseline status in Section 11.2.2. The majority of information included in the environmental baseline section is a summary of information presented in the Recovery Plan for Upland Species of the San Joaquin Valley, California (USFWS, 1998).

11.2.1 Factors affecting species environment within the action area

The San Joaquin kit fox was listed as "endangered" by the U.S. Fish and Wildlife Service (USFWS) in 1971. More than 90% of its native habitat in the San Joaquin Valley has become fragmented or destroyed as a result of agricultural, industrial, and urban developments. As the human population in California continues to expand, suitable habitat for the kit fox will inevitably decrease. Such land conversions contribute to SJKF declines through displacement, reduction in prey abundance, and direct and indirect moralities. Since the 1950s, the primary factor of SJKF decline is attributed to loss, degradation, and fragmentation of habitats. Less severe threats to the species' survival are attributed to: (1) non-native invasion, predation, and competition; and (2) disease and climatic extremes. A detailed discussion of these three factors is presented in Sections 11.2.1.1 through 11.2.1.3.

11.2.1.1 Habitat Destruction, Degradation, and Fragmentation

The survival of the SJKF is threatened primarily by industrial, agricultural, and urban developments that readily reduce the size and quality of available habitat (USFWS, 1998). Grazing practices (agriculture), however, are not considered part of that threat, and may actually benefit the SJKF. Livestock may alter population densities of prey species, but may not harm the SJKF if grazing intensity is controlled (Morrell 1975; Orloff *et al.*, 1986; Laughrin, 1970; Balestreri, 1981). Excess grazing of shrub cover that affects prey numbers may be detrimental (O'farrell *et al.*, 1980; O'Farrell and McCue, 1981; USFWS, 1983; Kato, 1986).

The major urbanization developments within the SJKF range include cities, towns, aqueducts, irrigation canals, surface mines, road networks, non-petroleum industrial projects, power lines, and wind farms. While these actions are detrimental to suitable habitat, the SJKF may still utilize adjacent areas to expand prey bases or construct additional den sites. For example, individuals have been observed with dens in a number of urban habitats: along canals and in levees (Jones and Stokes, 1981; Hansen, 1988), abutting highways (ESA Planning and Environmental Services, 1986b, Hansen, 1988), near wind farms (Hall 1983, Orloff *et al.*, 1986), along power line corridors (Swick, 1973), and at sanitary land fills (R. Faubion, pers. comm.). The SJKF have also been documented utilizing habitats adjacent to towns and behaving differently from individuals living in more remote populations. These individuals scavenge for food in dumpsters, forage from smaller areas, demonstrate diurnal activity patterns, and are comparatively tame. It is thought that this variability in behavior and temperament reflects their degree of ecological plasticity (*e.g.*, Grinnell *et al.*, 1937, p.411; T. Murphy, pers. comm., B.L. Cypher pers. comm.).

Habitat loss due to petroleum field development affects the SJKF in the southern half the San Joaquin Valley. Habitat is lost to construction and grading of roads, well pads, tank settings, pipelines, and settling ponds. Habitat is degraded from noise, ground vibrations, venting of toxic and noxious gases, and the release of petroleum products and waste waters. SJKFs are also at higher risks to traffic-related mortalities on oil fields. However, it is thought that as long as mitigation efforts accompany drilling developments, oil sites may provide adequate habitat for the SJKF (O'Farrell *et al.*, 1980; Spiegel *et al*., in press). The effects of oil activities on SJKF population density, reproduction, dispersal, and mortality were examined at the Elk Hill Naval Petroleum Reserves, California. No differences were reported between populations living on developed or undeveloped areas of the reserve (Berry *et al*., 1987a). Likewise, a correlation between population size and habitat disturbance for SJKF populations in Kern County is unclear. Nonetheless, habitat loss remains the most significant factor affecting SJKF survival on oil fields (Spiegel *et al*., in press).

11.2.1.2 Invasive Species, Predation, and Competition

The SJKF is threatened by coyotes, non-native red foxes, domestic dogs(*Canis familiaris*), bobcats (*Felis rufus*), and large birds of prey (Hall, 1983; Berry *et al.,* 1987; O'Farrell, *et al*., 1987; White *et al*. 1994, Ralls and White, 1995; CDFG, 1987). Due to land conversions and human activities (White, 2000), the non-native red fox is currently expanding its distribution in central California (Orloff *et al*., 1986; Lewis *et al*., 1993) and may be a factor in SJKF declines in Santa Clara Valley (T. Rado, pers.comm.). Red foxes are also implicated in declines in the northwestern portion of their range (USFWS, 1998). Red foxes may potentially limit opportunities for the SJKF through competition or predation. Competition arises from the foxes utilizing similar den sites and preying on the same species. The effects of red foxes have not been well-studied, and neither their historical impacts nor continuing expansion on the SJKF are clear (USFWS, 1998).

Predation and competition are not listed explicitly as threatening the survival of the SJKF in the Recovery Plan (USFWS, 1998), but their effects are nonetheless mentioned as adversely affecting the species. For example, coyote injuries are listed as the primary factor in the majority of SJKF mortalities (White *et al.*, 2000). Coyotes will "aggressively dominate encounters with red foxes and will pursue and kill" red, gray (Sargent and Allen, 1989), and kit foxes. However, a coyote control program at the Naval Petroleum Reserves in California reported no increase in survivorship among the SJKF, nor was there a decrease in coyote-induced mortality (Cypher and Scrivner, 1992; Scrivner and Harris, 1986; Scrivner, 1987). Incidentally, the coyote may simultaneously benefit the SJKF by reducing the competition and predation by the non-native red fox. The gray fox is a native species which may compete with the SJKF, although to what degree is unknown (USFWS, 1998).

11.2.1.3 Disease and Climatic Extremes

Both disease and climatic extremes are implicated in SJFK mortality (USFWS, 1983). At Camp Roberts, rabies caused a 6.3 percent decline in radio-collared individuals (Standley *et al.* 1992), and the disease may be linked to the population decline at Camp Roberts in recent years (P.J. White, pers. comm.). A significant threat is posed by stochastic environmental events, particularly drought and flooding. Drought affects the reproductive success of the SJKF by affecting the availability of prey (White and Ralls, 1993; Spiegel *et al.* in press). There is a concern of local extinctions in certain isolated areas. Excessive rainfall also adversely affects prey populations. The above-average rainfall in 1994-1995 caused a "rapid and severe" decline of small mammal populations. Fewer pupping dens were found in the Elk Hills region in 1995, and the proportion of pup rears was relatively small (B.L. Cypher pers. comm., L.K. Spiegel pers. comm.).

11.2.2 Baseline Status

Though the SJKF has been listed for over 30 years, the status throughout most of its range is poorly known, in part due to limited access on private properties. Despite the lack of a comprehensive survey, research projects, incidental sightings, and local surveys indicate that kit foxes currently inhabit some areas of suitable habitat on the San Joaquin Valley floor. Suitable habitat is also found in the surrounding foothills of the coastal mountain ranges, the Sierra Nevada, Tehachapi Mountains (from southern Kern County north to Contra Costa, Alameda, and San Joaquin Counties), near La Grange, Stanislaus County on the east side of the Valley, and some of the larger scattered islands of natural land on the Valley floor in Kern, Tulare, Kings, Fresno, Madera, and Merced Counties. Kit foxes occur westward into the interior coastal ranges in Monterey, San Benito, and Santa Clara Counties (Pajaro River watershed), in the Salinas River watershed, Monterey and San Luis Obispo, and in the upper Cuyama River watershed in northern Ventura and Santa Barbara Counties and southeastern San Luis Obispo County. Kit foxes are also known to live within the city limits of the city of Bakersfield in Kern County (USFWS, 1998).

"The largest extant populations of kit foxes are in western Kern County on and around the Elk Hills and Buena Vista Valley, Kern County, and in the Carrizo Plain Natural Area, San Luis Obispo County. The kit fox populations of Elk Hills and the City of Bakersfield, Kern County, Carrizo Plain Natural Area, San Luis Obispo County, Ciervo-Panoche Natural Area, Fresno and San Benito Counties, Fort Hunter Liggett, Monterey County, and Camp Roberts, Monterey and San Luis Obispo Counties have been recently, or are currently, the focus of various research projects. Though monitoring has not been continuous in the central and northern portions of the range, populations were recorded in the late 1980s at San Luis Reservoir, Merced County, North Grasslands and Kesterson National Wildlife Refuge area on the Valley floor, Merced County, and in the Los Vaqueros watershed, Contra Costa County in the early 1990s. Smaller populations and isolated sightings of kit foxes are also known from other parts of the San Joaquin Valley floor, including Madera County and eastern Stanislaus County" (USFWS, 1998, p.124)

Population Dynamics

The SJKF is known for its instability in population size. It is not uncommon for the SJKF population to fluctuate fivefold or more from year to year. These large variations are correlated with the intrinsic fluctuations of desert ecosystems. The volatility in precipitation affects seed bank and vegetative biomass, leading to associated fluctuations in leporids (rabbits and hares) and rodents. Ultimately, these density independent variables lead to fluctuations in kit fox reproductive rates. Both extended episodes of drought or rainfall can contribute to low reproduction and population crashes (White *et al.* 2000).

The San Joaquin kit fox is found within three or four core populations, and approximately nine satellite populations that vary in size and degree of isolation (White *et al.* 2000).

These core populations are thought to “anchor the spine” of the metapopulation (USFWS, 1998). Three core populations are: (1) Carrizo Plain Natural Area in San Luis Obispo County; (2) Natural lands of western Kern County inhabited by kit foxes (e.g., Elk Hills, Buena Vista Hill, and the Buena Vista Valley, Lokern Natural Area and adjacent natural land); and (3) the Ciervo-Panoche Natural Area of western Fresno and eastern San Benito Counties. Each core population is distinct and has different environmental conditions, as manifested in their unsynchronized population dynamics (B.L. Cypher, pers. comm.). Population viability studies suggest that extinction probabilities increase dramatically if either the Carrizo Plain or western Kern County population is extirpated. The combination of these three populations is advantageous compared to other groupings, because they are all more or less interconnected by grazing lands (although steep and rocky in certain places) (USFWS, 1998). An important population, although not a core population, is located in the Salinas-Pajaro Region at Camp Roberts and Fort Hunter Liggett. This population has natural connections to the Carrizo Plain Natural Area and the San Joaquin Valley, although the extent of migration between these areas is unknown (K. Ralls pers. comm.).

Safe harbor and other land owner incentive programs have been initiated to reduce the degree of isolation between populations and promote conservation of SJKF populations on agricultural lands. These initiatives should ensure that farmers are not penalized, farming is not disrupted, and habitats are protected that maintain and enhance kit foxes. The American Farmland Trust has already proposed a project addressing the SJKF fragmentation. In addition to the goals listed above, the project also aims to establish small populations of breeding SJKFs on farm lands. These breeding grounds would serve as bridges between isolated patches and the larger populations “along the spine of the metapopulation” (USFWS, 1998). There is one safe harbor agreement listed on the San Joaquin kit fox (*Vulpes macrotis mutica*) species profile USFWS webpage: Paramount Farming SHA, Artificial Escape Den Project. The USFWS issued a permit for 3 years in 2003 for the take of the SJKF on approximately 1000 acres of land from Paramount Farms, Kern County. Twenty one Habitat Conservation Plans for the SJKF are also listed on the species profile USFWS.

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12: VALLEY ELDERBERRY LONGHORN BEETLE

12.1. CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the valley elderberry longhorn beetle (VELB) (*Desmocerus californicus dimorphus*) that are reasonably certain to occur in the action area. Future federal actions unrelated to the proposed action are not considered because they are subject to consultation pursuant to Section 7 of the Endangered Species Act (Act). Numerous non-federal actions that could affect the VELB are reasonably certain to occur within the action area. These will typically include maintenance of levees and canals, agriculture, park management, grazing activities, and urban development. Many of these activities are linked and create complex effects on listed species or their habitat in the action area. For example, maintenance of levees and canals facilitate continued farming activities and restrict water flow into downstream riparian habitats. Farming utilizes the fertile soil in riparian habitats and further contributes to its net habitat loss.

12.2. ENVIRONMENTAL BASELINE

The environmental baseline is defined as the effects of past and ongoing human induced and natural factors leading to the status of the species, its habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of the valley elderberry longhorn beetle's status at this time. Details of the valley elderberry longhorn beetle's habitat description and known locations are included in Attachment III. However, the baseline condition of each assessed VELB species and habitat varies across locations. Given the large number of occupied habitats and extent of the action area included of this assessment, the environmental baseline includes a general discussion of factors that may affect VELB within the action area. This information is presented in Section 12.2.1. Additional information on the current distribution and population dynamics of the VELB is presented as part of the baseline status in Section 12.2.2. Tables 12.1, 12.2, and 12.3, present restoration and preservation projects that have been completed, or are currently being completed, on behalf of the VELB and its habitat. The majority of information included in the environmental baseline section is a summary of information presented in the 5-Year Review: Summary and Evaluation of the Valley Elderberry Longhorn Beetle (USFWS, 2006).

12.2.1 Factors affecting species environment within the action area

The VELB is primarily affected by riparian habitat loss, due to factors such as maintenance of levees and canals, and park management. To a lesser degree, the VELB is also affected by predation and invasive species. When the species was first listed by the United States Fish and Wildlife Service (USFWS), 90% of the riparian habitats that once occupied the Central Valley 150 years ago had become obsolete (Barr, 1991; USFWS, 2007a). While the rate of habitat loss has declined within the past 25 years, the lo1ss has yet to fully subside. Urbanization projects continue within the species range, although for mitigation purposes, adjoining habitat conservation plans for each action are also underway. These plans are in conjunction with former and ongoing riparian restoration and conservation activities. Projects to protect existing land, and enhance or restore current riparian habitats, have been or are currently being completed by private organizations (*ie*. land trusts), and by Federal, State or local agencies. Sections 12.2.1.1 through 12.2.1.3 provide details on each of the major issues and actions affecting the VELB within its range of habitats (USFWS, 2006).

12.2.1.1 Habitat Loss and Degradation

Habitat loss is the number one factor contributing to VELB decline. In particular, the removal of riparian habitat, and as a consequence the removal of the elderberry host plant (*Sambucus* spp.), has been the primary factor since the species was first listed as "Threatened" in 1980. The overall loss continues to this day (USFWS, 1984 and 2007a). This reduction in habitat is linked to urban development, flood controls, levees and canals, agriculture, and park management. These factors have the potential to remove, degrade or fragment VELB habitat. The species is particularly sensitive to these adverse factors because of its high habitat specificity and its limited dispersal ability. Isolated subpopulations are vulnerable to extirpation from environmental, demographic, and genetic events (Schonewald-Cox et al. 1983).

The loss of habitat has historically been a major threat to the VELB. Between 1900 and 1990, there was a 96% loss of riparian habitat in the southern portion of the Central Valley (Kern to Fresno Counties), with only 16,000 acres remaining. There was an 84% loss with 21,000 acres remaining in the middle Valley (Merced to San Joaquin Counties), and an 80% loss with 96,000 acres remaining in the northern Valley (Sacramento and Solano Counties to Shasta County). Loss rates slowed between 1960 and 1990, but were still relatively "high" with 59% loss in the south, 65% loss in the middle, and 35% loss in the northern Central Valley (Geographic Information Center, 2003). Quantifying the actual loss of elderberry is difficult, however. It is uncertain how much of the riparian habitat contained elderberry, or even if it was occupied by the VELB (USFWS, 2006).

The VELB "is in long-term decline due to human activities that have resulted in widespread alteration and fragmentation of riparian habitats, and to a lesser extent, upland habitats, which support the beetle" (USFWS, 1996 and 2007 a). Historically, riparian habitats in the Central Valley were utilized heavily for their significant quantity of wood and for their fertile soil. In the last 25 years, the rate of riparian loss has slowed significantly due to the decreasing number of existing riparian habitats, restoration efforts, and protections provided under the Act for both the VELB and other species. Non-riparian, upland habitats where elderberry occurs have also declined: savanna and grassland adjacent to riparian areas, oak woodland, and mixed chaparral-woodland.

Since 1980, the USFWS has allowed an estimated incidental take of 10,000 to 20,000 acres of beetle habitat, for the purposes of urbanization, water management, transportation, and flood control (Talley *et al. 2006*). This area amounts to 12,000 to 15,000 elderberry shrubs or 40,000 to 50,000 elderberry stems one inch in diameter. Between 1983 and 2005, the USFWS completed 526 formal Section 7 consultations. One jeopardy opinion was issued in the early 1980s to the U.S. Army Corps of Engineers for the Sacramento River Bank Protection Program. Two other programmatic biological opinions were issued in 1997 to the U.S. Army Corps of Engineers and the Federal Highways Administration (USFWS, 2006). Non-Federal parties are issued section 10(a)(1)(B) permits for the take of listed species. To date, 18 permits for habitat conservation plans, totaling 970,000 acres for the VELB, have been issued (USFWS, 2006).

The current maintenance of levees and canals for the purposes of flood control and agriculture could result in a loss of habitat for the VELB (Talley et al. 2006). Historically, artificial levees, river channelization, dam construction, water diversion, and heavy groundwater pumping have all been attributed to the destruction of VELB habitat (USFWS, 1984). Flood control levees constructed in the lower Sacramento River have apparently reduced the number of elderberry shrubs and associated VELBs compared to the unobstructed upper Sacramento River (Talley *et al.* 2006). Future restoration potential for the lower Sacramento River is hindered by the availability of suitable sites (USFWS, 2006). Due to concerns of VELB habitat interfering with flood-fighting, or with risking the chance of costly mitigation afterwards, the Reclamation Board has limited the planting of elderberry in flood plains over the past decade (USFWS, 2006). By excluding elderberry from restoration projects, the potential benefits to the beetle are undermined. However, land owner incentive programs, discussed later, offer a means to mitigate the threat to the VELB caused by levee and canal maintenance (USFWS, 2006).

12.2.1.2 Predation, Invasive Species, and Other Issues

The potential predators of the VELB are the Argentine ant (*Linepithema humile*), insectivorous birds and lizards, and the European earwig (*Forficularia auricularia)*. However, little is known of the overall impacts these predators have on the beetle (USFWS, 2006). The Argentine ant is an aggressive competitor and appears to be the main predator of the VELB (Huxel, 2000). The ant displaces native populations of arthropods and is currently expanding its range throughout riparian habitats in California. Between 1998 and 2002, 30 sites were examined along Putah Creek and American River and the number of sites infested with the ant during that time period increased by a factor of 3 (Huxel, 2000; Holyoak and Talley, 2001). According to Huxel (2000), there was a negative correlation between the observed presence of the ants and VELB exit holes on elderberry along Putah Creek in 1997. The Argentine ant may affect the success of VELB mating or feeding, or prey on eggs and larvae (Way *et al.* 1992). Due to the moisture requirements of the ants, riparian and irrigation areas may provide suitable habitats for the ant to thrive in (Huxel, 2000).

Insectivorous birds, insectivorous lizards and the European earwig are potential predators of the VELB because they all forage on the stems of elderberry (Klasson *et al.* 2005).

The European earwig is known to be a scavenger and omnivore. It may prey on VELB larvae because the earwig is known to feed on the larvae of tethered mealworms (*Tenebrio monitor)*. The earwig is often found in urban areas, and like the Argentine ant, also in riparian and irrigation areas because of the availability of moisture (Klasson *et al.* 2005). According to Klasson (*et al.* 2005), mitigation sites generally have the highest densities of European earwigs because of the associated irrigation. However, this fact has yet to be proven statistically (Klasson *et al.* 2005).

The host plant elderberry is threatened by the colonization of invasive species (Talley et al., 2006). Invasive woody species include: black locust (*Robinia pseudoacacia*), giant reed (*Arundo donax*), red sesbania (*Sesbania punicea*), Himalaya blackberry (*Rubus armeniacus*), tree of heaven (*Ailanthus altissima*), Spanish broom (*Spartium junceum*), Russian olive (*Eleagnus angustifolia*), edible fig (*Ficus carica*), and Chinese tallowtree (*Sapium sebiferum*)" (Talley et al., 2006). Non-woody invasive species include ripgut brome (*Bromus diandrus*), foxtail barley (*Hordeum murinum*), *Lolium multiflourum*, and starthistle/knapweed (*Centaurea* spp.). Non-woody species may affect elderberry establishment or germination, and increase fire risk. One study on *Robinia pseudoacacia* demonstrated a positive correlation between the VELB and the black locust. However, this positive association was thought to be temporary as elderberry would soon be crowded out (Talley et al., 2006).

Other issues attributed to VELB habitat decline include grazing animals and park/forest management (Talley *et al*., 2006). Browsing deer and cattle are known to feed on elderberry shoots and bark, and according to the California Department of Fish and Game (CDFG) biologists, cattle "readily" forage on new elderberry growth (USFWS, 1984). Additionally, rodents may damage new saplings, and voles have reportedly girdled elderberry shrubs at two different conservation banks (Talley *et al.* 2006). In some State and local parks, the clearing of undergrowth and the institution of lawns has reduced suitable VELB habitat. Similar practices occur in the few natural woodlands remaining, further reducing suitable habitat (USFWS, 1980).

12.2.1.3 Habitat Protection and Restoration

The number of riparian restoration projects and protective actions in the Central Valley has increased in recent years (USFWS, 2006). The actions are categorized into one of the following three groups: the protection of existing riparian habitats, the enhancement or restoration of riparian habitats plus the planting of elderberry shrubs, or the enhancement or restoration of riparian habitats without the planting of elderberry. Table 8.1 provides a summary of projects in which riparian habitat was acquired or protected in the Central valley since 1980 (excerpted from Talley *et al.* 2006). The table shows that 22 agencies and organizations have protected approximately 50,000 acres of existing land. Table 8.2 provides a summary of projects in which riparian habitat was restored and elderberry were plated in the Central Valley (excerpted from Talley *et al.* 2006). Seven agencies are responsible for 19 projects which have or are restoring 5,193 acres of habitat and planting 130,345 elderberry saplings. Table 8.3 provides a summary of projects in which riparian habitat was restored, yet no elderberry were planted in the Central Valley (excerpted from Talley *et al.* 2006). This table reveals that 1,592 acres have been or are being restored.

Table 12.1: Summary of projects in which riparian habitat was acquired or protected in the Central Valley since 1980 (excerpted from Talley *et. al.* 2006).

Project/Program	Floodplain acres (approx.)	Comments
Sacramento Valley:		
Sacramento River Natural Wildlife Reserve (NWR)	11,000	May acquire up to 18,000 acres
The Nature Conservancy (TNC) Sacramento	~3000	Many projects turned over to the Sacramento River NWR
Big Chico Creek Ecological Preserve, California State University (CSU) Chico Research Foundation	4000	
Fenwood Ranch, Shasta Land Trust	2160	2.5 miles river frontage. Conservation easement
Gover Ranch/Bloody Island, Bureau of Land Management	800	Conservation Easement
Hamilton City levee setback	1500	
[Bobelaine Sanctuary, Audubon]	[400]	[Acquisition in 1975 pre-dates listing (1978) but was then considered outside VELB range]
Feather River Wildlife Area, California Depart. of Fish and Game (SDFG)	2500	Units flank Bobelaine Sanctuary [May pre-date listing]
American River Parkway	4600	Much park area pre-dates listing
Consumnes River:		
Consumnes River Preserve, TNC and partners	5500	Approx. 40,000 acres non-floodplain
Stone Lakes NWR	4000	May acquire up to 18,200 acres
San Joaquin Valley:		
San Joaquin River NWR	6600	May acquire up to 12,900 acres
Partners for Fish and Wildlife, Natural Resource Conservation Service (NRCS)		23+ miles river frontage. Conservation easements
San Joaquin River Parkway	~2000	http://www.sjrc.ca.gov/docs/Parkway_map_01-06.pdf
Bobcat Flat, Friends of the Tuolumne	300	
Big Bend, Tuolumne River, Natural Resource Conservation Service (easement)	250	Conservation easement
Grayson River Ranch, Tuolumne River, NRCS	137	Conservation easement
Mining Reach-7/11 Segment, Tuolumne River, Turlock Irrigation District	87	2.2 river miles. Don Pedro 1996 Federal Energy Regulatory Commission Settlement Agreement
Merced River Salmon Habitat Restoration Program		Mostly for channel restoration
Fine Gold Creek, CDFG	708	
Kaweah River watershed, Sequoia Riverlands Trust	2200+	In fee and conservation easements

Kern River Preserve, Audubon California	1000	Benefit to VELB not established
Total	~45,000	

Table 12.2 Summary of projects in which riparian habitat was restored and elderberry were planted in the Central Valley (excerpted from Talley *et al.* 2006).

Project/Name	**Owner/ Manager**	**Planted by**	**River**	**Acres**	**# EB[1] planted**	**Comments**
Llano Seco	USFWS	River Partners	Sacramento	271	1472	
Ord Bend	USFWS	River Partners	Sacramento	111	1616	
Turtle Bay	McConnell Arboretum, Turtle Bay Exploration Park	River Partners	Sacramento	100	1323	Has FWS Biological Opinion 1-1-03-F-189, appears pure restoration
Flynn	USFWS	TNC	Sacramento	247	5605	
Kopta	State Controller's Trust	TNC	Sacramento	105	2086	
Lohman		TNC	Sacramento	20	882	
Ohm	USFWS	TNC	Sacramento	206	7613	
O'Connor Lakes Ecological Reserve	CDFG	River Partners	Feather	471	900	300-400 more plantings planned
Packer Island	USFWS	TNC	Sacramento	175	7633	
Partners for Fish and Wildlife projects	private		Sacramento	700		Elderberry planted, # not recorded
Phelan Island	USFWS	TNC	Sacramento	117	2730	
Pine Creek	USFWS	TNC	Sacramento	270	6781	
Princeton Ferry	USFWS	TNC	Sacramento	44	2700	
Rio Vista	USFWS	TNC	Sacramento	799	36735	
River Unit	California Department of Water Resources (DWR)	TNC	Sacramento	27	486	
Ryan	USFWS	TNC	Sacramento	164	6164	
Sam Slough	DWR	TNC	Sacramento	72	7200	
Shaw	DWR	TNC	Sacramento	11	383	
Southam	USFWS	TNC	Sacramento	65	2574	
Sul Norte	USFWS	TNC	Sacramento	46	1271	
Mohler Tract II	USFWS	River Partners	Stanislaus	35	520	Anadromous Fish Restoration Program
McHenry Ave Recreation Area	US Army Corps of Engineers	River Partners	Stanislaus	32	512	
Merced NWR	USFWS	USFWS	San Joaquin	40	160	
San Luis NWR	USFWS	USFWS	San Joaquin	210	840	
San Joaquin River NWR	USFWS	River Partners	San Joaquin	800	32512	

Mining Reach-7/11 Segment	Turlock Irrigation District	HART Restor-ation Group	Tuolumne	87	160	2.2 river miles. Don Pedro 1996 Federal Energy Regulatory Commission Settlement Agreement
Totals:				**5,193**	**130,345**	

[1]EB = Elderberry

Table 12.3: Summary of projects in which riparian habitat was restored yet no elderberry were planted in the Central Valley (excerpted from Talley *et al*. 2006).

Project/Name	**Owner/ Manager**	**Planted by**	**River**	**Acres**	**Comments**
Battle Creek	CDFG	River Partners	Battle Creek	21	
Beehive Bend	CDFG	River Partners	Sacramento	59	
Big Bend	Tuolumne R., Preservation Trust	River Partners	Tuolumne	250	Planning in 2003
Butler Slough	CSU Chico Restoration Foundation	River Partners	Sacramento	54	
Cottonwood Creek	CDFG	River Partners	Cottonwood Creek	15	
Del Rio	CDFG?[1]	River Partners	Sacramento	259	Acquisition. Adj. to Llano Saco NWR unit. Future SHA?[1]
Drumheller Slough	USFWS	River Partners	Sacramento	135	
Gianella Landing/Beard	CDFG	River Partners	Sacramento	20	
Howard Slough, Butte Basin	CDFG	River Partners	Butte Creek	51	
Jacinto	CDFG	River Partners	Sacramento	37	
Moulton Weir	CDFG	River Partners	Sacramento	46	
Partners for Fish & Wildlife, NRCS projects	Private		San Joaquin		23+ river miles
Pine Creek	CDFG	River Partners	Sacramento	235	
Princeton	CDFG	River Partners	Sacramento	34	
River Ranch	Private	River Partners	Sacramento	3	
Sacramento R., Big Chico Ck., Mud Ck., [confluence]	Cal. Dept. of Pesticide Regulation (CDPR)?[1] [Bidwell-Sac RSP]	TNC	Sacramento	217	Acquisition and restoration planning only at this stage?[1]

Thomas	CDFG	River Partners	Sacramento	19	
Merced River Salmon Enhancement	CDWR/CDFG		Merced	unknown	Planning stage for vegetation?[1]
Grayson River Ranch	Natural Resource Conservation Service (easement)		Tuolumne	137	
Total:				**1592**	

[1]Question marks appear in the 5-Year Review (USFWS, 2006), and no explanation is provided regarding their meaning.

Each riparian restoration and enhancement project will not necessarily benefit the VELB. Certain constraints limit the VELB's ability to disperse to or inhabit certain areas, so actual advantages to the beetle vary with site. Elderberry plants need to be mature enough so that they provide a dependable habitat from year-to-year. The colonization of the beetle may be difficult to overcome because of a barrier or dispersal limitation. The site should also be ecologically suitable for elderberry species over the long term (USFWS, 2006). For example, although some restoration projects do not plan on directly introducing elderberry, the projects may still serve the VELB if restored areas can support long term recruitment and survival of the hostplant. Elderberries will likely colonize the area, but will do so at a slower rate than if it had been directly planted (River Partners, 2003, River Partners 2004b). Factors such as the distance from known beetle locations, suitability of the habitat for elderberry shrubs, and post-placement monitoring of the beetle determine the success of each restoration project (USFWS, 2006).

Safe Harbors Agreement and other land owner incentives
The Safe Harbor Agreement was developed to "encourage landowners to enhance, restore, or otherwise encourage listed species to use their property for foraging, breeding, resting, or other activities" (USFWS, 2006). The program accomplishes this goal by allowing private property owners to restore or maintain their property for the benefit of endangered species through issuance of Section 10 permits, without incurring additional regulatory restrictions (EDF, 2008; USFWS, 2006). The landowner opines to return the property to its "baseline condition", the mutually agreed upon state of property between the USFWS and landowner (USFWS, 2006). Three Safe Harbor agreements have so far been made for the VELB: 7,450 acres from the Burrows and Big Bluff Ranches in Tehama County, a programmatic agreement for up to 3,500 acres along the Mokelumne River, and 259 acres in Glenn County (USFWS, 2006). Other landowner incentive programs are available to private owners and organizations that want to support the restoration of VELB habitat without the ensuing negative consequences (USFWS,2006).

12.2.2 Baseline Status

The current distribution of the VELB ranges from the southern portion of Shasta County south to Fresno County in the San Joaquin Valley (Barr, 1991). Records primarily based on exit holes indicate that there are roughly 190 locations in the Central Valley where the beetle is known to occur (California Department of Fish and Game, 2006). This number is up from less than 10 locations in 1980, along the American River, Putah Creek, and Merced River in the Central Valley (USFWS, 1980). Historically, there have been records indicating the beetle's presence in Kern County, but current observations of living specimens have yet to be found (Talley *et al.* 2006). However, an increase in known locations can primarily be attributed to an increased effort to look for the beetle. Also, the number of locations is not representative of the known populations, as there are 24 records of VELB occupancy located within 2 miles of the American River (no exact locations specified) (CDFG, 2006).

A suite of occupancy surveys have been conducted to gain a better understanding of VELB population numbers, densities, and locations. A survey in 1982 identified one individual at Rossmoor Bar from among "1,247 elderberries along the American River from Rancho Cordova to Johnson Industrial Park in Sacramento" (USFWS, 1984). However, emergence holes were found on 27% of the elderberry trees. Similarly, no adults were found from a survey of 228 elderberries along Putah Creek, spanning Solano and Yolo Counties, yet 44% of the trees had exit holes (USFWS, 1984). The frequency of VELB presence along the American River was found to be 20-50%, while the frequency was less than 20% along the Sacramento River (LSA, 2004). "VELB occupancy ranged from 2.9% in a non-riparian scrub area to 7-11.2% of shrubs in riparian reaches of Putah Creek and the American River" (Talley et al., 2006). "Beetle density averaged about 2 new exit holes per 'site'" (Talley et al., 2006). The VELB is generally found "with 1-2 exit holes per occupied shrub or per site and occupancy rates of 2-10% of shrubs or 25% of sites" (Talley et al. in press; Talley et al., 2006). "Neither pruning nor topping affected the colonization or loss of VELB from shrubs, or the length of time that a shrub was either occupied or unoccupied by VELB" (Talley et al., 2006).

It is worth mentioning the factors that may affect the accuracy of population estimates. The two main types of survey error include overestimates due to misidentification of exit holes and underestimates due to difficulty of locating exit holes. Beetle sightings are rare and exit holes provide a proxy for observing beetle presence. These exit holes could be misidentified as being created by the VELB, when they were actually produced by such animals as horntails, wood wasps (Siricidae), or solitary bees. Locating the exit holes proves to be even more difficult because of dense shrubbery and the low number of holes per tree. Along the American River, exit holes per tree average from 1.6 in non-riparian habitats to 2.2-2.9 in riparian habitats (Talley *et al.* in press; Talley *et al.* 2006). Locating the beetle exit holes, as opposed to misidentifying the holes, is thought to have a greater influence on our understanding of the beetle's distribution and numbers (Talley *et al.* 2006).

Population Dynamics

Little information exists on the range-wide population trends of the VELB. One long-term data set was acquired by Collinge *et al.* (2001). The data set examined a majority of locations surveyed by Barr (1991). Both studies observed the VELB at roughly 20% of the sites and on approximately 25% of the elderberries examined. Although the proportions of occupancy between the two studies appear similar, the study by Corringe *et al.* actually found fewer occupied sites and groups of shrubs (Collinge *et al.* 2001). Elderberry density and the number of examined sites containing elderberry had decreased since the study conducted by Barr (1991) (Collinge *et al.* 2001). Another study, although not long-term, recorded occupancy rates at 64% for the Sacramento River between Sacramento and Red Bluff (Lang *et al.* 1989). However, 64% was an average between the sites. Occupancy rates ranged from 28% between Sacramento and Colusa to 94% between Chico and Red Bluff. This variation was attributed to greater flood control efforts along the southern reach of the Sacramento River, reducing the width of the riparian corridor (Lang *et al.* 1989). However, a straight comparison of occupancy rates is difficult because the methodologies utilized by Corringe *et al.* (2001) and Barr (1991) are different from those utilized by Lang *et al.*(1989).

While VELB habitat continues to be lost, today that loss is typically accompanied by compensatory measures (Talley *et al.* 2006). However, uncertainty remains about the extent to which natural environments are compensated for by restored habitats (Talley *et al.* 2006). Compared to the historic extent of habitat reduction, the total area of restored and enhanced habitats still remains quite small. Restoration and mitigation efforts are valuable, however, with the hope that proactive efforts will provide habitat for persistent and sustainable metapopulations. Mitigation efforts to replace elderberry shrubs more than compensate for the number of elderberry lost to development, but the quality, size, and persistence of these shrubs is questionable (Talley *et al.* 2006). According to a survey by Talley *et al.*, the VELB occupied 47% of now developed habitat, while the beetle now occupies 43% of the accompanying mitigation habitats. One possibility for the discrepancy in percentages is that a significant, unquantified number of elderberries may be planted in upland, non-riparian, areas which are not preferred habitats for the VELB. Incidentally, even with an overall net population loss, there is not a strong trend toward decline or recovery (Talley *et al.* 2006).

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13: TIDEWATER GOBY

13.1. CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local, private, or other non-federal entity activities on the tidewater goby (*Eucyclogobius newberryi*) that are reasonably certain to occur in the action area. Future federal actions not directly related to the proposed action are not considered because they require additional consultation under Section 7 of the Endangered Species Act (Act). Actions that may occur within the action area consist of coastal development, water diversions, and exotic species introductions. Water resource diversions include those for agriculture, channelization for flood control, groundwater overdrafting, and anthropogenic sandbar breaching. There is an inverse relationship between the expansion of development and water diversions and the resulting flow volume of freshwater into lagoons and marshes on which the tidewater goby depends. These changes in turn impact the complex balance of sediment loads, salinity, and temperature. Coupled with natural seasonal perturbations in the goby's habitat, annual droughts, and the introduction of exotic species, isolated metapopulations may be more vulnerable to reduced flow volumes, although the degree of impact on long-term survival of the tidewater goby is uncertain.

13.2. ENVIRONMENTAL BASELINE

The environmental baseline is defined as combined effects of past and present factors, both natural and anthropogenic, leading to the status of the species, its habitat, and ecosystem. The environmental baseline is a snapshot of the tidewater goby's status at this time. However, the baseline condition of the tidewater goby and habitat varies across locations. Given the large number of occupied areas and extent of the action area included in this assessment, the environmental baseline includes a general discussion of factors that may affect tidewater goby within the action area. This information is presented in Section 1.2.1. Additional information on the current distribution and population dynamics of the tidewater goby is presented as part of the baseline status in Section 1.2.2.

13.2.1. Factors affecting species environment within the action area

At the time of its listing in 1994, the 43 remaining tidewater goby populations were small and threatened by a variety of anthropogenic and natural factors (USFWS, 1994). The main threat to the tidewater goby discussed in the USFWS listing notice is the loss of saltmarsh habitat due to coastal development (USFWS, 1994, 2005 and 2007). The USFWS "Recovery Plan for the Tidewater Goby (*Eucyclogobius newberryi*)" and the "Tidewater Goby (*Eucyclogobius newberryi*) 5-Year Review: Summary and Evaluation" (5-Year Review) list loss of habitat, agricultural and sewage discharges, sedimentation from cattle and feral pigs, breaching of lagoons, drought in combination with human-induced water reductions, population isolation, alteration of upstream sediment flows, introductions of exotic fish predator and competitor species, habitat damage, and pollution from vehicular activity near lagoons, as factors that may threaten the goby (USFWS, 2005). As of 2007, when the 5-Year Review was completed, there was no comprehensive information showing that pollution and cattle grazing were having an effect on the tidewater goby, and therefore these were not considered further in the 5-year Review (USFWS, 2007) or in this document.

13.2.1.1. Habitat alteration, curtailment, and degradation

Habitat loss or modification is the major factor threatening the tidewater goby (USFWS, 1994 and 2005). These losses can be direct, such as those resulting from coastal residential and industrial development or from dredging of waterways for navigation purposes or they can be indirect losses resulting from effects such as changes in salinity (USFWS, 1994). Road projects break the connection between lagoons and the ocean, changing temperatures and salinity, and upstream diversions or alterations of flow reduce the size of marshes downstream as well as change the salinity profile (USFWS, 1994). Habitat is also lost or modified from stream channelization and groundwater overdrafts (USFWS, 2005).

Development and Habitat Loss

It is estimated that 75% to 90% of the estuarine wetlands of California have been lost, most likely to development (USFWS, 2007). Railroads and early coastal highways were the first to break the connection between marshes and lagoons with the ocean (USFWS, 2005). Railroad development often completely filled marshes, resulting in tidewater goby population losses in San Luis Obispo, Santa Barbara, and San Diego Counties before 1900 (USFWS, 2005). Lagoons and tributary streams were also channelized to protect structures such as bridges and farmland, further isolating lateral marshes from the main stream and increasing velocity of flows (USFWS, 2005). The scouring action of high velocity water washes away sand needed by gobies to breed (USFWS, 2005). Without access to marshes which provide cover for growth and refuge from scouring winter flows, gobies are more susceptible to flood events (USFWS, 2005).

The likelihood of major habitat loss has been reduced due to current laws and regulations, but limited amounts of habitat will continue to be altered, with limited impacts on the tidewater goby (USFWS, 2007). Ongoing activities include annual dredging in places like Goleta Slough in Santa Barbara County, habitat restoration projects in Malibu Lagoon in Los Angeles County, and at Mission Creek in Santa Barbara County, and bridge widening projects such as at Mission Creek in Santa Barbara County (USFWS, 2007). Small projects can have large effects on the goby, as in the case of repair work on San Mateo Creek Lagoon railroad trestle crossings in San Diego County in 1998, which involved dredging parts of the creek and lagoon and filling parts of the marsh (USFWS, 2007). After the project, surveys found no gobies, which were once abundant (USFWS, 2007).

Increased sedimentation from urban development upstream may make lagoons shallower, which may allow water temperatures to fluctuate between extremes (USFWS, 2007). Although tidewater gobies are adapted to a wide range of temperatures, they are usually found in water 44 to 77^{o}F (8 to 25^{o}C) when breeding, and temperature changes may allow predators and competitors to increase in abundance (USFWS, 2007). Shallower lagoons may also be more susceptible to freshwater inputs, which may reduce habitat, expose burrows, and flush gobies out to sea (USFWS, 2007). Limited flows may result in the desiccation of shallower lagoons (USFWS, 2007).

Hydrologic Changes

Hydrologic changes include channelization, diversion of water, groundwater pumping, and in some cases, restoration projects (USFWS, 2007). Channelization causes high flow events that can flush gobies out to sea and degrade downstream habitat by scouring stream channels, reducing or eliminating sediments needed for burrows, and altering salinity regimes (USFWS, 2007). An altered salinity regime downstream may adversely affect the size and distribution of tidewater goby populations (USFWS, 2005). Water diversions and groundwater pumping reduce freshwater inputs into lagoons and estuaries, and changing flow rates may also alter water availability needed during breeding season to cover burrows and eggs and provide foraging habitat (USFWS, 2005 and 2007). Water diversions reduce the size of marshes at the mouths of rivers and creeks, and exacerbate the negative effects of natural events like drought (USFWS, 2005). Three populations were lost in San Luis Obispo County due to a drought worsened by water diversions between 1986 and 1990 (USFWS, 2005). In the summer, Penasquitos Creek lagoon in San Diego County and Aliso Creek lagoon in Orange County are drained every month, preventing their habitation by gobies (USFWS, 2007). Reduced flows may also allow aggressive plants to colonize sand and mud substrates at the margins of lagoons, which in turn degrades these substrates needed by gobies for breeding (USFWS, 2005). Stream depths are also reduced from plant colonization and increased sedimentation, preventing gobies from swimming upstream from the lagoons (USFWS, 2005).

Man-made barriers, typically at the upstream end of a channelization, create reservoirs for flood protection and modify or eliminate the brackish zone required by the tidewater goby (USFWS, 2005). Dikes and levees are placed around land that subsides below sea level just inland of beaches as a result of oil or water extraction, blocking freshwater flows which then fall over the structure or seep through into the higher salinity water on the downstream side, made saltier due to reduced inflows (USFWS, 2005). Increased evaporation, the opening of the barrier sandbar to the ocean, and underground saltwater intrusion, may also raise the salinity of brackish lagoons and estuaries downstream (USFWS, 2005).

Groundwater overdrafting from increasing numbers of wells as coastal populations increase reduces the amount of freshwater that reaches the lagoons and thus eliminates or reduces the brackish zone (USFWS, 2005). Streams that receive municipal wastewater discharges have more water in them during the year, and much more water in the dry season than would naturally occur (USFWS, 2005). High nutrient loads enrich the lagoons and can decrease dissolved oxygen (USFWS, 2005). More water can cause an increase in the frequency of lagoon breaching and cause erratic fluctuations in the level of water, which decreases habitat and increases risk of predation (USFWS, 2005). Spawning burrows can be exposed to air, and if the lagoon suddenly drains in late spring or summer, marine water can dominate until the rains of winter (USFWS, 2005) making conditions unsuitable for goby reproduction.

Anthropogenic breaching of lagoons during the dry season adversely affects gobies in numerous ways (USFWS, 2005). Although lagoons may reform in about a week, they typically stabilize at a lower water level, usually by a meter (3.3 ft) or more, which strands gobies in shallow pools and exposes breeding burrows to predation and desiccation (USFWS, 2005). Salinity increases, because of lower or no inputs of freshwater in the dry season, can be tolerated but are not ideal for gobies (USFWS, 2005). Adjacent marshes that are shallower than lagoons and are used for cover from predators and as areas to grow faster and larger (compared with the open water) become unavailable because they are sensitive to small changes in lagoon water levels (USFWS, 2005). People may cause the breaching of lagoons to remedy actual or perceived stagnation and water odor, to prevent flooding of structures or agricultural fields within the lagoon flood plain, to deal with mosquitoes by reducing the adjacent vegetated marsh, and, to an unknown extent, to create a new "fan" of freshly deposited sediment in the ocean to improve the surfing conditions (USFWS, 2005). The significance of this threat to long-term survival can be severe as in the case of San Onofre Creek lagoon, which was breached and resulted in a 56% decline of tidewater gobies (USFWS, 2007). Breaching occurs regularly at Lake Earl in Del Norte County, Santa Clara River in Ventura County, and Malibu Lagoon in Los Angeles County (USFWS, 2007). However, goby populations continue to survive in these locations; therefore, breaching may not be as severe a threat as previously feared (USFWS, 2007). Sediment supplies from upstream flows determine the barrier sandbar and the content of sand in the lagoon, and interruption of the upstream flow may cause the sandy beaches to degrade and impede the formation of barrier sandbars (USFWS, 2005). This may contribute to the effects of anthropogenic breaching by allowing tides to alter the breeding substrate and salinity (USFWS, 2005).

Estuary restorations use jetties and dredging to create open tidal areas, which do not allow for the seasonally closed habitat that the tidewater goby depends upon in such places as Bolsa Chica Lagoon in Orange County and Batiquitos Lagoon in San Diego County (USFWS, 2007). A Highway 1 bridge replacement and restoration that removed levees at Pescadero Marsh/Butano Creek in San Mateo County caused the sandbar to form in late summer to early fall instead of in early to mid-summer (USFWS, 2007). The extensive goby habitat at North Marsh has been eliminated by this habitat conversion (USFWS, 2007). Even with this habitat loss, the population is still considered reasonably secure; information is unavailable as to the magnitude and frequency of occurrence of these efforts and how they affect goby abundance, productivity, and/or adult and juvenile survival (USFWS, 2007). Additionally, channelization and habitat removal continue, but the impact is less severe than thought prior to the listing of tidewater goby, and improvements in technology like floating weirs and biostabilization techniques have also reduced impacts (USFWS, 2007).

13.2.1.2. Disease and Predation

Parasites

The fluke, *Cryptocotyle lingua*, is a common marine parasite that was introduced from the eastern Atlantic (USFWS, 2005 and 2007). It causes an infection that can kill host fish in high numbers, especially juveniles, and also facilitates secondary infections in broken skin (USFWS, 2005). It could also increase the goby's vulnerability to predation from increased visibility of the black cysts, or alter predator avoidance behavior (USFWS, 2005). The fluke has been documented infesting tidewater gobies in Gannon Slough, Humboldt County, Pescadero Creek, San Mateo County, and possibly in Cocoran Lagoon, Santa Cruz County, though other places may potentially support it (USFWS, 2007). A parasitic microsporidian (*Kabatana newberryi*) identified in 2007 may be specific to tidewater gobies, and overlaps geographically with the goby in northern California (USFWS, 2007). This microsporidian attacks muscle tissue, turning it white (USFWS, 2007). Surveys in 2003-2004 found tidewater gobies in the northern part of their range in a reasonably large and secure population infected with what was initially identified as *Kabatana newberryi*; similar infections were seen in 2005 in gobies from Rodeo Lagoon, Marin County although it could not be identified because of the specimen preservation techniques (USFWS, 2007). The parasite has not been seen in the southern part of the goby range, but the dispersal mechanism is unknown and it is uncertain whether it is a significant threat that can contribute to goby decline (USFWS, 2007). At this time, the effect of parasites on tidewater goby populations is not well understood (USFWS, 2007).

Exotic and Native Predators

California has a high volume of shipping, and it is estimated that more than 10,000 marine species are transported each day in ballast water (USFWS, 2007). Exotic species are a threat to the tidewater goby, including predators like striped bass (*Morone chrysops*), which have been introduced several times in central and southern California bays and coastal lagoons since the early 1900's (USFWS, 2005). Exotic species may prey on goby adults, larvae, or eggs (USFWS, 2007). Tidewater gobies can be particularly affected because they are distributed across small isolated populations that fluctuate in size and may be extirpated, or populations may be made more vulnerable to natural perturbations (USFWS, 2007). Native predators like salmonids also have a greater effect on the tidewater goby when their populations and habitat are reduced (USFWS, 2007).

It has been estimated that approximately 30 species of introduced marine, brackish, and freshwater fish are important carnivores in the San Francisco Bay and Delta (USFWS, 2007). Species known to prey upon gobies such as the striped bass, chameleon goby (*Tridentiger trigonocephalus*), yellowfin goby (*Acanthogobius flavimanus*), and shimofuri goby (*Tridentiger bifasciatus*) have a wide range of salinity tolerances (USFWS, 2007). Largemouth bass (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*) are among the species identified that pose the greatest threat to tidewater gobies (USFWS, 2007). In the late 1980's or early 1990's, largemouth bass eliminated the goby population in Old Lagoon in San Luis Obispo County (USFWS, 2005). The introduction of the rainwater killifish (*Lucina parva*), chameleon goby, and yellowfin goby in the 1960's in San Francisco Bay coincided with the extirpation of tidewater gobies there (USFWS, 2005). While widespread in the San Francisco Bay and newly established in Upper Newport Bay, rainwater killifish have not established elsewhere (USFWS, 2005). Yellowfin gobies have spread to larger muddy tidal estuaries and were not found in smaller nontidal brackish goby habitat, but in 1992 and 1993 they were collected in the Santa Clara River and Santa Margarita lagoons, coinciding with the extirpation of tidewater gobies in the Santa Margarita River (USFWS, 2005). Chameleon gobies have been locally abundant in the San Francisco harbor since the 1960's and in the Los Angeles harbor since the 1970's (USFWS, 2005). However, these fish prefer hard substrates (USFWS, 2005).

Shimofuri gobies have invaded the San Francisco Bay Delta through the California Aqueduct into the Pyramid Reservoir and Piru Creek (USFWS, 2005). This goby species is freshwater adapted and invades from bilge or imported water, causing concern that water piped from the California Aqueduct to central coastal California is a potential threat to tidewater gobies there (USFWS, 2005). Shimofuri gobies have been shown in laboratory experiments to aggressively intimidate, outcompete, and prey upon tidewater gobies (USFWS, 2005). Shimofuri gobies also prefer hard substrates and may not interact with tidewater gobies in lagoons, but increases in hard substrates would aid their establishment, in addition to the concern that breaching of lagoons and other factors can lower the water level and the shimofuri gobies could move out of the rocky areas and establish themselves in the tidewater goby habitat (USFWS, 2005).

Freshwater fish have also been introduced that may affect the tidewater goby, like the centrarchid sunfish and bass that are the main seasonal predators in the upper brackish area of estuaries (USFWS, 2005). These introduced centrarchids are in nearly every tributary to tidewater goby habitat, and in 1993 their movement downstream to Santa Margarita River Lagoon severely reduced or eliminated tidewater gobies there (USFWS, 2005). In San Diego County, green sunfish (*Lepomis cyanellus*) eliminated the tidewater gobies in the San Mateo Creek Lagoon (USFWS, 2005). A small number of tidewater gobies were present following the elimination of the green sunfish, which may have been washed out or eliminated by salinity changes (USFWS, 2005). Introduced African clawed frogs (*Xenopus laevis*) also prey on tidewater gobies, but aren't as prevalent in tidewater goby habitat (USFWS, 2005). However, bullfrogs (*Rana catesbeiana*) have also been introduced and it is suspected that they have a significant negative impact on tidewater gobies, and are thought to be responsible for the extirpation of the population at Old Creek, San Luis Obispo County (USFWS, 2007). Also, because the habitat of the goby is nearly freshwater for part of the year they may be vulnerable to largemouth bass, green sunfish, and African clawed frogs even though these species, among others, have limited salinity tolerance (USFWS, 2007).

The full extent of the impact of predation on the abundance of the tidewater goby has not been determined (USFWS, 2007). Eighty-four goby locations (64%) are affected by the risk of predation, with the Central Coast recovery unit at the greatest risk (USFWS, 2007). In the future, the ranges of African clawed frogs and yellowfin gobies may expand, and new non-native species like Chinese mitten crabs (*Eriocheir sinensis*) may become problematic, perhaps enabled by water redistribution plans (USFWS, 2007).

13.2.1.3. Other Factors affecting the species' continued existence

Metapopulation structure is important to tidewater goby biology and conservation (USFWS, 2007). This structure has been altered by the loss of "stepping-stone" populations and the loss or reduction of connectivity due to distance and lack of intermediate habitats (USFWS, 2007). Populations can be isolated from habitat destruction, diverting or drying of waterways, and urbanization (USFWS, 2007). Lost stepping stone populations include Waddell Creek in Santa Cruz County that was between San Mateo County in the north and southern locations, the Schwans and Woods lagoons between the Baldwin/Wilder population to the north and the Corcoran/Moran population to the south, and San Vicente and Liddell Creeks between Scot and Laguna Creeks (USFWS, 2007). Wide gaps exist between Gaviota Creek and Winchester/Bell Canyon in Santa Barbara County, and between there and Arroyo Burro and Mission Creek-Laguna Channel (USFWS, 2007). These areas are in turn far from Ventura River and the Santa Clara River (USFWS, 2007). Other locations like Lagunitas Creek and Rodeo Lagoon are so isolated that if the population were to be extirpated they would not likely be recolonized (USFWS, 2007). Central and northern California populations naturally separated by 20 miles or more such as those in the Ten-Mile River-Virgin Creek-Pudding Creek group, Mendocino County, are unlikely to be recolonized by dispersing gobies if they were to be lost, or to contribute gobies to the north or south (USFWS, 2007). Long-shore currents are from north to south, and thus as weak swimmers, gobies may be limited to recolonizing locations only to the south of occupied locations (USFWS, 2007).

Genetic drift and inbreeding may occur without genetic exchange in a metapopulation, and this reduces the ability of the population to survive in the long term given environmental changes (USFWS, 2007). Deleterious alleles may be expressed and decrease fitness, destroy local adaptation, and break up co-adapted gene complexes (USFWS, 2007).

Drought is the most significant natural factor, affecting the tidewater goby, with the resulting coastal and riparian habitat alteration (USFWS, 2007). However, the tidewater goby may be more resilient to drought than previously thought, based on an increase of locations occupied by gobies since the last drought in 2006-2007 (USFWS, 2007). Anthropogenic water reductions, in combination with natural drought, degrade and stress coastal and riparian ecosystems (USFWS, 2007). This can decrease tidewater goby populations to very low levels or extirpation, as stated in the final listing rule that noted population declines because of the reduced availability of lagoon habitats and goby disappearance when lagoons dried up (USFWS, 2007). Flooding also threatens the goby, and though they do occur naturally and may be necessary for recolonization, floods are worsened in duration and intensity by the affects of channelization upstream (USFWS, 2007).

Animal waste and agricultural and oil field runoff contaminate the tributaries of coastal lagoons, and smaller lagoons receive septic tank effluent especially during winter flooding (USFWS, 2005). Anoxia is also a problem in Estero San Antonio, Arroyo del Oso, Pismo Creek, Santa Ynez River, and other locations in the summer and fall from oxidation of excessive nutrients, which also stimulate the growth of oxygen-consuming microalgae as well as toxic algae blooms (USFWS, 2005). Tidewater gobies may have been extirpated from the Salinas River from the discharge of poorly treated sewage (USFWS, 2005).

The introduced beaver (*Castor canadensis*), giant reed (*Arundo donax*), salt cedar (*Tamarix pentandra*), and smooth cordgrass (*Spartina alterniflora*) in tributaries and lagoons could potentially affect tidewater goby habitat (USFWS, 2005). Beaver ponds trap nutrients and thus affect the nutrient level and sediment composition in lagoons (USFWS, 2005). Salt cedar stabilizes banks and increases channelization of tributary streams (USFWS, 2005). Giant reed windrows in Santa Margarita River Lagoon existed as of 1993 and could provide substrate for shimofuri gobies should they access the area thereby preventing the tidewater goby from re-establishing itself (USFWS, 2005). Smooth cordgrass could degrade goby habitat because it alters channel forming characteristics of flowing tidal and non-tidal areas (USFWS, 2005).

Vehicles disturb wetland vegetation and pollute waterways with petroleum, and highway or railways spills of toxic substances are a threat when they occur in the vicinity of goby habitat (USFWS, 2005). Recreational vehicles may also dump waste and grey water tanks when close to lagoons (USFWS, 2005).

13.2.2. Baseline Status

At the time of its listing in 1994, the tidewater goby had disappeared from nearly 50% of its historic range in the coastal lagoons of California, from 74% of the lagoons south of central California's Morro Bay down to only three locations south of Ventura County (USFWS, 2007). As of 1984 the goby occurred or had been known to occur in 87 locations extending from the extreme northern and southern ends of its historical geographic range (USFWS, 2006 and 2008). A 1993 assessment of the species' distribution using records from the area between the Monterey Peninsula in Monterey County and the United States-Mexico border found four additional locations occupied by tidewater gobies (USFWS, 2006 and 2008). When the goby was listed in 1994, only 48 of the 87 locations were occupied by gobies (USFWS, 2007). Since then, other locations have been identified and tidewater gobies have been documented at 135 locations within its historical geographic range (USFWS, 2006). However, of these 135 localities, 29 (21 percent) are no longer occupied by tidewater gobies, leaving 106 locations, probably only a small subset which actually function as metapopulations (USFWS, 2007). The 5-Year review recommended downlisting the goby to threatened, because the number of locations occupied by tidewater gobies had more than doubled since the time of its listing; this increase cannot be explained by increased survey efforts alone (USFWS, 2007). Habitat destruction and alteration had been reduced, and the goby was no longer in imminent danger of becoming extinct (USFWS, 2007).

Population Dynamics

No long-term monitoring program is available for the tidewater goby, population dynamics are not well documented, and estimating population size is complex because populations are controlled by environmental conditions (USFWS, 2005 and 2007). Seasonal variation in distribution and abundance, short lifespan, and annual conditions such as drought result in a paucity of estimates of population sizes (USFWS, 2007). It has been estimated though that the number of tidewater gobies in a population at Aliso Creek Lagoon ranged from 1,000 to 1,500 individuals in the late winter to early spring and 10,000 to 15,000 individuals in the late summer to early fall (USFWS, 2005).

A metapopulation viability analysis based on monitoring over 10 years has not been initiated, and is required before the species can be downlisted to threatened (USFWS, 2007). The only individual management plan developed to address threats to a metapopulation is the Integrated Natural Resource Management Plan (INRMP) for Marine Corps Base Camp Pendleton, though management plans are under development for the populations at Mission Creek in Santa Barbara County, the Santa Clara River estuary in Ventura County, and Malibu Lagoon in Los Angeles County (USFWS, 2007).

Pronounced differences in the genetic structure evidenced by genetic markers demonstrate that tidewater gobies in some locations are genetically distinct (USFWS, 2006 and 2008). The results of a 1985 study suggest a low level of gene movement between populations in the northern, central and southern parts of the range, but the sites that were sampled were far apart geographically and so results may not show gene flow on local levels (USFWS, 2000 and 2002). Mitochondrial analysis indicates that the tidewater goby in Orange and San Diego counties differentiated from tidewater goby populations to the north about two million years ago and may be a distinct species, a finding which is also supported by morphological studies (USFWS, 2007).

Six major phylogeographic groups were identified in a study of mitochondrial DNA and cytochrome b sequences collected at 31 locations throughout the range and include: (1) the North Coast (NC) Unit comprised of Tillas Slough (Smith River) in Del Norte County to Lagoon Creek in Mendocino County; (2) the Greater Bay (GB) Unit comprised of Salmon Creek in Sonoma County to Bennett's Slough in Monterey County; (3) the Central Coast (CC) Unit comprised of Arroyo del Oso to Morro Bay in San Luis Obispo County; (4) the Conception (CO) Unit comprised of San Luis Obispo Creek in San Luis Obispo County to Rincon Creek in Santa Barbara County; (5) the Los Angeles-Ventura (LV) Unit comprised of Ventura River in Ventura County to Topanga Creek in Los Angeles County; and (6) the South Coast (SC) Unit comprised of San Pedro Harbor in Los Angeles County to Los Penasquitos Lagoon in San Diego County (USFWS, 2006 and 2008). These recovery units are based upon molecular or morphological data or on geomorphology where the former data are lacking (USFWS, 2007).

The abundance of each of the recovery units was assessed for the 5-Year Review with presence/absence surveys over time, and each classified as extirpated, intermittent, or regular (USFWS, 2007). Regular populations are source populations for other locations (USFWS, 2007). The status of each of these units is summarized in **Table 1** below:

Table 1: Status of Tidewater Goby Recovery Units (USFWS, 2007).

Recovery Unit	Locations occupied at time of listing (1994)	Total localities following additional survey	Abundance Classification			
			Presumed extirpated	Intermittent	Regular	Unknown
North Coast	10	22	3	8	4	7
Greater Bay	9	34	11	15	7	1
Central Coast	9	21	5	10	5	1
Conception	15	36	2	17	17	-
LA/Ventura	2	8	2	4	6	-
South Coast	3	14	6	7	1	-

13.3. REFERENCES

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