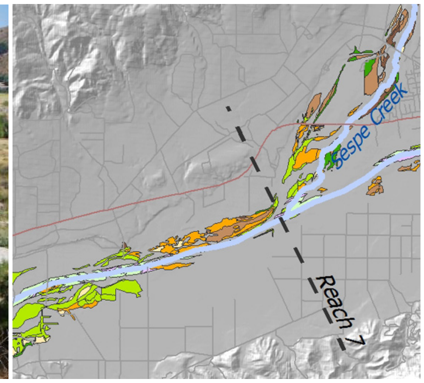
Santa Clara River Parkway
Floodplain Restoration Feasibility Study





Focal Species Analysis and Habitat Characterization for the Lower Santa Clara River and Major Tributaries
Ventura County, California

FINAL REPORT
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Cover photographs:
Left - Least Bell's vireo (photograph by U.S. Fish and Wildlife Service)
Center – Example of habitat conditions in the lower Santa Clara River corridor (photograph by Stillwater Sciences)
Right – Excerpt from habitat distribution map produced as part of this report.

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1 INTRODUCTION

This report discusses the Santa Clara River in the context of focal species, in which the habitat needs of selected species that use the river help to focus analysis and synthesis of existing information. For each focal species, we identify the different life history stages that occur in the Santa Clara River, the habitats used by each of those life history stages, the ecological processes that create and maintain those habitats, and the management actions that influence those ecological processes and habitat conditions.

This report describes the process and criteria used to select these focal species and the key habitat types that they use or represent in the Santa Clara River corridor.

The 116-mile long Santa Clara River flows in a westerly direction from headwaters on the northern slopes of the San Gabriel Mountains in Los Angeles County through the Santa Clara River Valley and the Oxnard Plain in Ventura County, and finally empties into the Pacific Ocean near the City of Ventura. Many large coastal southern California rivers (*i.e.*, the Los Angeles, Santa Ana, and San Gabriel rivers) have been confined to concrete channels in their lower reaches to provide flood protection for surrounding urban areas, dramatically reducing (or eliminating) riparian vegetation and the fluvial geomorphic processes that maintains a functioning river corridor ecological system. The Santa Clara River riparian corridor, however, has retained a significant amount of high quality aquatic and riparian habitat supporting threatened and endangered species, including the arroyo toad (*Bufo microscaphus californicus*), southwestern willow flycatcher (*Empidonax traillii extimus*), least Bell's vireo (*Vireo bellii pusillus*), and slender-horned spineflower (*Dodecahema leptoceras*).

The present-day Santa Clara River is a dynamic semi-arid ecological system driven primarily by periodic short duration, high intensity flood events (Stillwater Sciences 2007a). The channel is functionally on the boundary between meandering and braided river forms in terms of the relationship between gradient, discharge, and bed material grain size. The result (where natural processes prevail) is an unusual compound channel morphology that is essentially braided at lower flows but more akin to a low sinuosity meandering channel during large flood discharges. The channel morphology is affected primarily by large flood flows rather than by the moderate discharges that are frequently used to characterize channel form response in temperate climate river channels. These factors result in a shifting mosaic of riparian vegetation throughout the corridor.

Although the Santa Clara River riparian corridor is relatively intact, flood protection infrastructure, diversions, roads, agriculture, and urbanization have constrained or disrupted natural geomorphic and hydrologic processes causing riparian and aquatic habitat degradation. A number of studies and planning efforts have begun on the river to address these issues. Understanding the physical drivers for riparian vegetation distribution, composition, and health is a crucial part of river management and restoration planning, particularly because riparian vegetation serves as an indicator for important environmental variables, including habitat quality and quantity, and because this understanding allows restoration planners to better predict vegetation response to restoration actions. A variety of physical factors are known to influence the recruitment, growth, persistence, distribution, and composition of riparian vegetation, including relative depth to groundwater, frequency of disturbance, substrate composition, and salinity. The relative roles these factors play in determining the pattern and composition of riparian vegetation within the lower Santa Clara River system, however, is only partially understood (Stillwater Sciences 2007b).

1.1 Objectives

1.1.1 Estimate spatial extent of focal species' habitat

The primary objective of this report is to estimate the current spatial extent of selected "focal" species' habitat within the Study Area. The spatial extent is an estimate of current potential habitat based upon recent field studies, aerial photographic interpretation of riparian vegetation and channel planform evolution, reviews of scientific literature, and interviews with local experts. We will use these results within the Santa Clara River Parkway Floodplain Restoration Feasibility Study to develop recommended restoration actions and to assess the potential effects of these actions across the set of focal species' habitat to aid in restoration planning within the Study Area. The overall approach is based on the premise that maintaining and restoring physical and ecological processes will provide properly functioning habitat for well-distributed populations of native species. An analysis of the life history and habitat requirements of certain species of animals and plants was used to identify the relative importance of various habitat features along the lower Santa Clara and to evaluate the degree to which restoration strategies may benefit these individual species, many of which have declined and may require habitat restoration to persist in the area.

1.1.2 Focal species selection and habitat description

We selected focal species from a list of candidate species that currently occur or historically occurred along the lower Santa Clara River. Focal species were selected based on their status under state and federal Endangered Species Acts, the occurrence of suitable habitat within the vicinity of the project area, and the ecological niche they represent. They cover a range of aquatic, riparian, and upland habitat requirements and represent various taxonomic groups and guilds within the river corridor ecosystem. A few of the selected species no longer occur in the project area, but were included because they might re-colonize or be re-introduced to if habitat is restored. We compiled information on focal species' life history and habitat requirements to detail a set of habitat features that could be spatially identified from existing information. This information was then used to estimate and map the extent of focal species' habitat under current conditions.

1.2 Study Area

The analysis area (Figure 1-1) encompasses the extent of riparian vegetation within the 500-year floodplain along the lower mainstem Santa Clara River in Ventura County, a reach of approximately 38 mi (61 km).

2 METHODS

2.1 Focal Species Selection

Stillwater Sciences has developed a set of criteria and a vetting process for selecting focal species, as illustrated in Figure 2-1. Though this process involves the application of criteria to a pool of candidate species, it is intended to assist the process of selecting focal species by facilitating comparisons among species, rather than serving as a rigid procedure that emits a list of species based on quantitative scores.

2.1.1 Step 1: Determine the historical or current distribution within the Study Area

The first step of the vetting process involves determining if a candidate species currently exists, or existed historically, within the study reach. This step effectively eliminates consideration of introducing a non-native species to the study area because of uncertainty about a new species' ability to exist in current conditions, because of uncertainty about how a new species can affect current species assemblages through ecological interactions, and because of ethical concerns associated with deliberately introducing a non-native species.

This step does not eliminate non-native species that currently occupy the study reach from consideration as a focal species. It is usually infeasible to eradicate a non-native species once it has become widely established in an environment, so a target species assemblage must often include a mix of both native and non-native species. There can be value in selecting a non-native species as a focal species, because a better understanding of its habitat requirements can facilitate identification of changes to the ecosystem that have conferred a competitive advantage to the species relative to native species, which can, in turn, assist the definition of management actions designed in part to control the distribution or abundance of the non-native species. This first step of the vetting process also allows for the potential re-introduction of an extirpated species.

2.1.2 Step 2: Determine whether the species is federally- or state-listed as endangered or threatened

Step two of the vetting process acknowledges that the recovery of listed species constitutes a high social and regulatory priority, and it recognizes that listed species are often at the center of resource management conflicts. As a result, listed species usually serve as focal species. However, the number of listed species that occur in the Santa Clara River corridor generally precludes the selection of every listed species as a focal species, because the sheer number of listed species would undermine the use of focal species as a means of focusing and organizing the synthesis and analysis of information, and the development of restoration strategies.

2.1.3 Step 3: Apply additional criteria for non-listed species

The third step of the selection process provides much of the information used to compare candidate focal species by applying a series of criteria to non-listed species. It is often important to include non-listed species in the group of focal species in order to capture potential ecosystem changes that are reducing their populations, which could necessitate future protection that would exacerbate resource conflicts.

- **Other special-status designation**. The first criterion asks whether an unlisted species has some other special-status designation (*e.g.*, species of concern) that indicates concern with the health of the population. For example, California Department of Fish and Game (CDFG) has designated the Arroyo Toad as a species of concern because of population trends, indicating that further reductions could necessitate listing and protection.

- **High economic or public interest value**. The second criterion recognizes the economic or social importance of certain species, such as species that are sportfish that are the focus of recreational angling (*e.g.*, steelhead).

- **Narrow habitat requirements**. The third criterion tests whether a species has narrow habitat requirements such that loss of that habitat type would pose a significant threat to the health of the species. For example, Arroyo toads are habitat specialists, primarily located on third- to sixth-order floodplains with highly dynamic fluvial processes, which are necessary for the removal of vegetation and provide suitable, open, riparian habitats (Sandburg 2004). Current populations are restricted to shallow (<0.5 m [1.5 ft]) gravelly pools, adjacent to sandy terraces (Court *et al.* 2000).

- **Weak disperser**. The fourth criterion identifies species that have difficulty dispersing to new areas, preventing establishment of new sub-populations that can mitigate the potential loss of an existing population from a catastrophic event. For example, tidewater goby are found in lagoons, estuaries, and stream mouths separated by intolerable marine environments, and are absent from steep coastline areas and streams without lagoons or estuaries (USFWS 2005a). The fish's current distribution is entirely within its observed historical range, but local extirpation has occurred in 17% (23/134) of once populated sites, while another 40–50% (55–70/134) of historical sites maintain such small populations that long-term persistence is uncertain (USFWS 2005a). As a consequence, a natural or anthropogenic event that eliminates habitat in one of these original localities could reduce the species' range.

- **Strong interactor**. The fifth criterion indicates that particular species can significantly influence natural communities through ecological interactions with other species. For example, a species may serve as an important prey source for a number of other species, such that a decline in its population can reduce the food base for other species and depress the abundance of an entire community. Similarly, other species can affect a community by monopolizing available habitat and resources or by preying on a wide variety of species, which is the concern underlying the introduction of the African clawed frog (*Xenopus laevis*) in the Santa Clara watershed. The invasive giant reed (*Arundo donax*) is another example of a non-native species that is a strong interactor in the Santa Clara River floodplain.

- **Loss of habitat**. The sixth criterion addresses a key factor contributing to reductions in abundance or distribution of a species: habitat loss and degradation from system-wide anthropogenic changes. For example, several native riparian bird species have experienced losses of breeding habitat due to agricultural and urban development. This criterion suggests that changes in current resource management (*e.g.*, levee placement, large woody debris (LWD) abundance, increases in available floodplain area) can improve ecosystem condition despite habitat loss and degradation.

- **Local and/or regional population declines**. The final criterion applied in step three of the vetting process acknowledges that population abundance and distribution are two key metrics for assessing a species' health. Local and regional population declines are an indicator of system-wide change, and give further motivation to identify factors affecting local and regional populations. Continued population declines may require future federal or state protection.

2.1.4 Step 4: Assess availability of information

If a species satisfies one step or more of the three criteria, then it passes to step four, which assesses available information about that species. At a minimum, the general habitat requirements and life history stages of a species must be known for it to qualify as a focal species. Ideally, quantitative data on a species' habitat preferences will exist, and although it is preferable for these data to be specific to the Santa Clara River basin, knowledge from a similar system is also valuable. For example, there is little information about the abundance and distribution of steelhead in the Santa Clara River basin, but data from other river systems about general habitat preferences may be useful and applicable to the Santa Clara River basin. A species with detailed habitat information provides greater utility as a focal species. For example, several studies have identified the general range of preferred flow velocities, flow depths, and water temperatures of steelhead (Shapavolov and Taft 1954, Bjornn and Reiser 1991).

2.1.5 Step 5: Rank species

The information produced for each candidate species in steps two, three and four provides the foundation to rank species in step five of the vetting process, used to inform focal species selection in step six. Selection of focal species also emphasizes using species that represent different assemblages or guilds and species utilizing a broad range of habitat types within the study reach, so synthesis and analysis are relevant to a broad range of local species.

2.1.6 Step 6: Select focal species

Selecting too many focal species can undermine the purpose of a focal species approach, which is to focus and organize the discussion and analysis in a manner that is still relevant to a broad array of species. We determined that a total of six to twelve focal species would allow us to engage and organize much of the information available for the Santa Clara River and cover a broad range of habitat types that occur in the river corridor. Nine native and two non-native species were selected for this study (see Section 3).

2.2 Focal Species Habitat Characterization and Distribution

Focal species habitats were described according to key vegetative, geomorphic, and physical characteristics identified from detailed reviews of each species' current and historical distribution, and life history requirements. The current potential distribution of focal species habitat was estimated within a geographic information system (GIS) using these habitat characteristics, recently collected vegetation and geomorphic data (Stillwater Sciences 2007a,c), and available spatial data. Potential habitat represents areas where, according to known habitat preferences and current vegetative cover, focal species are most likely to be found or establish themselves. Stillwater Sciences and URS Corporation (2007) present detailed descriptions of vegetation alliances occurring within the Project Area that were used to identify potential habitat. These alliances were further grouped into distinct habitat types that integrate vegetative characteristics and biological and physical factors constraining their distribution (Table 2-1).

Table 2-1. Habitat types and associated alliances.

HABITAT TYPE	ALLIANCE	ACRES	HECTARES
Agriculture	Agriculture	8,141.0	3,294.6
	Agriculture (TOTAL)	**8,141.0**	**3,294.6**
Arundo donax	*Arundo donax*	890.2	360.3
	***Arundo donax* (TOTAL)**	**890.2**	**360.3**
Coastal sage scrub	*Artemisia californica*	115.5	46.8
	Artemisia californica - Eriogonum fasciculatum	23.7	9.6
	Encelia californica	14.1	5.7
	Eriogonum fasciculatum	23.8	9.6
	Lotus scoparius	5.7	2.3
	Malosma laurina	1.0	0.4
	Mixed scrub	30.7	12.4
	Salvia mellifera	5.3	2.1
	Sambucus mexicana	4.6	1.9
	Coastal sage scrub (TOTAL)	**224.5**	**90.9**
Cottonwood-willow forest	Mixed willow forest	332.9	134.7
	Populus balsamifera	285.0	115.3
	Populus fremontii	205.0	83.0
	Salix laevigata	349.2	141.3
	Salix lasiolepis	249.9	101.1
	Salix lucida	67.3	27.2
	Cottonwood-willow forest (TOTAL)	**1,489.2**	**602.7**
Desert riparian scrub	*Artemisia tridentata*	92.2	37.3
	Atriplex lentiformis	5.3	2.2
	Lepidospartum squamatum	235.3	95.2
	Yucca whipplei	5.5	2.2
	Desert riparian scrub (TOTAL)	**338.4**	**136.9**
Developed/Disturbed	Developed	6,484.4	2,624.2
	Disturbed	59.6	24.1
	Developed/Disturbed (TOTAL)	**6,544.1**	**2,648.3**
Freshwater wetland	Floodplain wetland superalliance	480.5	194.5
	Leymus triticoides	8.5	3.4
	Phragmites australis	0.2	0.1
	Scirpus spp.	35.5	14.4
	Freshwater wetland (TOTAL)	**524.8**	**212.4**
Herbaceous	*Ambrosia psilostachya*	1.1	0.4
	Lessingia filaginifolia	6.6	2.7
	Leymus condensatus	12.3	5.0
	Riverwash herbaceous	1,355.6	548.6
	Sambucus mexicana	3.8	1.5
	Herbaceous (TOTAL)	**1,379.4**	**558.2**

HABITAT TYPE	ALLIANCE	ACRES	HECTARES
Mixed non-native trees	Eucalyptus	66.7	27.0
	Mixed exotic trees	4.0	1.6
	Myoporum laetum	2.1	0.9
	Myoporum laetum - Arundo donax	0.4	0.1
	Nicotiana glauca	8.7	3.5
	Nicotiana glauca - Artemisia californica	1.4	0.6
	Olea europaea	12.7	5.1
	Ricinus communis	2.7	1.1
	Schinus molle	65.0	26.3
	Tamarix spp.	11.4	4.6
	Mixed non-native trees (TOTAL)	**175.0**	**70.8**
Mixed riparian forest	*Juglans californica*	1.2	0.5
	Mixed riparian forest	110.6	44.8
	Platanus racemosa	1.4	0.6
	Quercus agrifolia	72.3	29.3
	Mixed riparian forest (TOTAL)	**185.5**	**75.1**
Mixed riparian scrub	*Baccharis pilularis*	136.1	55.1
	Mixed riparian scrub	90.0	36.4
	Pluchea sericea	9.9	4.0
	Riverwash scrub	294.9	119.4
	Salix exigua	133.9	54.2
	Salix exigua - Arundo donax	83.3	33.7
	Salix exigua - Baccharis salicifolia	40.7	16.5
	Mixed riparian scrub (TOTAL)	**788.8**	**319.2**
Mixed willow scrub	*Baccharis salicifolia*	187.4	75.8
	Mixed willow scrub	101.9	41.3
	Mixed willow scrub (TOTAL)	**289.3**	**117.1**
Non-native vegetation	Non-native grasses and forbs	556.3	225.1
	Non-native vegetation (TOTAL)	**556.3**	**225.1**
Restoration site	Restoration site	3.8	1.6
	Restoration site (TOTAL)	**3.8**	**1.6**
Riverwash	Riverwash	2,095.8	848.2
	Riverwash (TOTAL)	**2,095.8**	**848.2**
Sand dune/Beach	*Abronia spp. - Ambrosia chamissonis*	169.8	68.7
	Beach	102.4	41.4
	Carpobrotus spp. - Mesembryanthemum crystallinum	17.1	6.9
	Sand dune/Beach (TOTAL)	**289.2**	**117.0**

HABITAT TYPE	ALLIANCE	ACRES	HECTARES
Tidal marsh	*Distichlis spicata*	10.8	4.4
	Jaumea carnosa	1.7	0.7
	Potentilla anserina	1.3	0.5
	Salicornia virginica	5.6	2.3
	Tidal marsh (TOTAL)	**19.4**	**7.8**
Water	Water	856.5	346.6
	Water (TOTAL)	**856.5**	**346.6**
GRAND TOTAL		**24,791.3**	**10,032.9**

3 RESULTS

The vetting process was adapted by selecting a pool of candidate focal species that we hypothesized were highly responsive to changes in the Santa Clara River's flow regime. We also identified species that are at the center of resource management conflicts or the object of significant study in the basin. We selected these species by conducting an initial search within the California Natural Diversity Database (CNDDB) and the California Native Plant Society (CNPS) inventory of rare and endangered plants (Appendix A). The number of species was narrowed after reviewing recent information about the Project Area (*e.g.*, AMEC 2005, Court *et al.* 2000, Penrod *et al.* 2006), and from professional judgment.

There is particular concern in the basin with the invasion of giant reed (*Arundo donax*) and saltcedar (*Tamarix ramosissima*) into riparian areas. The physical processes in the Santa Clara River basin are conducive to dispersal of these two species. Mono-specific stands of these species provide little habitat for native species, and limit native wildlife habitat by outcompeting native plant species. Therefore, giant reed and saltcedar were chosen as focal species at the outset, because control of these two species is likely to be a major restoration strategy on any parcel purchased in the Parkway.

The following sections describe the vetting process for the eleven focal species to explain their inclusion in the final group, and their potential habitat characterization and distribution. Their key habitat components described in Appendix B. The amount of detail varies for each species based upon availability of information and unique life history components. For example, southern steelhead and tidewater goby are not directly associated with any vegetation species or community. The descriptions provide the basis for the estimated habitat area and distribution presented below.

3.1 Arroyo Toad (*Bufo californicus*)

3.1.1 Vetting process

Arroyo toad was historically found in the upper and lower Santa Clara River basin and currently persists in large numbers along Sespe Creek from Hot Springs Canyon upstream to the mouth of Tule Creek (Sweet 1992, as cited in USFWS 1999) (Step 1). The toad is a federal endangered species (Step 2) and a California species of special concern (Step 3) (USFWS 1994). They are habitat specialists (Step 3) that are primarily found on third- to sixth-order floodplains shaped by dynamic fluvial processes, which scour vegetation and provide open riparian habitats (Sandburg 2004). Their breeding habitat consists of open sites, such as pools and old flood channels that provide still water and have little emergent vegetation (Sweet 1992, as cited in USFWS 1999). Juveniles and adults prefer to forage and burrow on terraces enclosed by dense riparian forest with little herbaceous cover. Arroyo toads require habitat near water, but many populations have been reduced or extirpated by extensive habitat loss (Step 3) from anthropogenic activities (*e.g.*, flood control, road building, agriculture, and recreation) (USFS 1999). There is a significant volume of information available for the arroyo toad in the Santa Clara River basin and the surrounding region (Step 4), which produced a high priority ranking (Step 5) leading to its selection as a focal species (Step 6).

3.1.2 Habitat characterization and distribution

The arroyo toad prefers vegetation-free open floodplains for breeding, and terraces with mature cottonwood (*Populus spp.*), coastal live oak (*Quercus agriflolia.*), sycamore (*Platanus spp.*), and willow (*Salix spp.*), and sparse understory with little or no vegetative cover for juvenile rearing and juvenile and adult foraging (Sweet 1992, as cited in USFWS 1999). Larval stages require shallow pools (water depth < 12 in

[30 cm]) with minimal current, usually found along the channel margin and in overflow channels (Holland 1997, as cited in USFWS 1999). The toad is generally found from 1,000-4,600 ft (300-1400 m) above sea level and along low gradient channels flanked by uneven-aged geomorphic surfaces (*i.e.*, floodplains and terraces) that support a mix of riparian seral stages (Sandburg 2004).

We estimated 893 ac (360 ha) of potential arroyo toad habitat within the project area (Figure 3-1). Among the habitat types listed in Table 2-1, those most likely to support all of the toad's life history stages are freshwater wetland (larval), riverwash (breeding), and cottonwood-willow and mixed riparian forest (juvenile rearing, juvenile and adult foraging) (Table 3-1). Although Table 3-1 indicates that Piru Creek does not have freshwater wetland habitat to support larval stages, it was included assuming that there are shallow pools with minimal current along the main channel margins. We also constrained habitat to areas > 650 ft (200 m) above sea level, which is a lower elevation than identified in the species summary. This was intended to capture current potential habitat occurring at the lower elevational range of the toad's distribution. Most habitat was predicted to occur within the Piru reach, with small patches also present in Reaches 9 and 10.

Table 3-1. Arroyo toad potential habitat area and distribution.

REACH	HABITAT TYPE									
	Cottonwood-willow forest		Freshwater wetland		Mixed riparian forest		Riverwash		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
10	5.4	2.0	22.0	8.9	10.8	3.6	456.0	184.4	**494.3**	**198.8**
11	108.1	43.8	24.8	10.0	33.9	13.7	28.3	11.5	**195.1**	**79.0**
Piru	148.6	60.1	0.0	0.0	37.3	15.1	17.8	7.2	**203.6**	**82.4**
TOTAL	**262.2**	**105.9**	**46.8**	**18.9**	**82.0**	**32.4**	**502.1**	**203.0**	**893.0**	**360.2**

3.2 Western Pond Turtle (*Clemmys marmarota*)

3.2.1 Vetting process

Western pond turtles have been observed throughout the lower Santa Clara River basin (CDFG 2005) (Step 1). Several known western pond turtle populations occur in the upper Santa Clara River watershed near Santa Clarita and in the vicinity of Piru Creek. Though the turtle is not currently listed as an endangered or threatened species (Step 2), it has been designated as a state species of concern (Step 3). Western pond turtle populations have experienced declines due to extensive conversion of wetland and riparian habitat for urban and agricultural use (Step 3) (Jennings and Hayes 1994, Germano and Bury 2001). Local population trends in the Santa Clara River watershed are currently unknown, but it is likely that most turtles in the Santa Clara mainstem are "fossil" populations consisting of old individuals, and immigrants from side drainages like Sespe Creek (Step 3) (S. Sweet, UC Santa Barbara, *pers. comm.*, March 28, 2006). It is unlikely that reproduction occurs in the mainstem Santa Clara due to a lack of suitable habitat and a high density of predators (raccoons). Still, populations appear to be stable in lower Sespe Creek and in parts of the Piru Creek system (S. Sweet, UC Santa Barbara, *pers. comm.*, March 28, 2006).

Though western pond turtles are known to occur in the Santa Clara River, there is relatively little information about their distribution within the corridor. Nevertheless, research conducted on other rivers provides a general understanding of their life history stages and habitat requirements that can guide inquiry in the Santa Clara River (Step 4). These factors led to a medium priority ranking (Step 5).

But, the general habitat requirements and preferences of western pond turtle provide a linkage with a range of off-channel habitat types (*e.g.*, oxbow lakes, sloughs, side channels) that are not well-covered by other candidate focal species. Further, the distribution and abundance of these off-channel habitats are strongly linked to management actions being evaluated by this study, such as levee and riprap alignment. In addition, the western pond turtles are unique among the pool of candidate species because they use both aquatic and terrestrial habitats. For these reasons, the western pond turtle was chosen as a focal species for the lower Santa Clara River (Step 6).

3.2.2 Habitat characterization and distribution

The western pond turtle prefers nesting in grasslands and meadows away from trees and shrubs, and overwintering in upland areas in early- and late- stage riparian scrub and forest (Holland 1994, Reese 1996, Reese and Welsh 1997, Reese and Welsh 1998, Buskirk 2002). Juveniles and adults are generally found within lentic habitats (oxbows, side channels) near terrestrial areas used for basking and nesting (Holland 1994, Jennings and Hayes 1994, Ashton *et al.* 1997).

We estimated that 1015 ac (411 ha) of potential western pond turtle habitat occurred within the lower Santa Clara River (Table 3-2). The potential habitat includes freshwater wetland and herbaceous vegetation for nesting, and mixed riparian and willow scrub and cottonwood-willow forest for overwintering habitat. Holland and Bury (in press, as cited in Spinks *et al.* 2003) observed turtles building nests an average of 150 ft (45 m) away, but within a range of 5–1,300 ft (2–400 m) from the wetted channel. As such, we limited habitat to within 330 ft (100 m) of the wetted channel to encompass potential nesting area. If a reach did not have habitat types to support all life history stages, it was not included as potential habitat area, which excluded Reaches 0 and 1. Within these limits, current potential habitat was distributed throughout Reaches 2 to 11, with most concentrated in Reaches 6, 7, and 8 (Figure 3-2).

Table 3-2. Western pond turtle potential habitat area and distribution.

REACH	HABITAT TYPE											
	Cottonwood-willow forest		Freshwater wetland		Herbaceous		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
02	34.0	13.8	22.4	9.1	45.3	18.3	10.7	4.3	8.3	3.4	120.6	48.8
03	5.0	2.0	9.9	4.0	3.9	1.6	15.4	6.2	0.0	0.0	34.1	13.8
04	12.2	4.9	10.9	4.4	19.9	8.0	8.3	3.3	0.0	0.0	51.3	20.7
05	4.2	1.7	12.8	5.2	12.8	5.2	6.6	2.7	0.0	0.0	36.3	14.7
06	6.8	2.7	26.5	10.7	118.3	47.9	26.6	10.8	0.0	0.0	178.1	72.1
07	3.2	1.3	30.7	12.4	100.6	40.7	1.1	0.4	0.5	0.2	136.1	55.1
08	3.7	1.5	71.3	28.8	164.2	66.4	0.0	0.0	0.0	0.0	239.1	96.8
09	1.2	0.5	0.0	0.0	3.2	1.3	0.0	0.0	0.0	0.0	4.4	1.8
10	3.4	1.4	0.6	0.3	36.5	14.8	7.8	3.2	0.0	0.0	48.3	19.6
11	55.5	22.4	24.8	10.0	32.8	13.3	52.9	21.4	0.4	0.1	166.3	67.3
TOTAL	**129.1**	**52.2**	**209.8**	**84.9**	**537.4**	**217.5**	**129.3**	**52.3**	**9.1**	**3.7**	**1,015**	**410.7**

3.3 Least Bell's Vireo (*Vireo bellii pusillus*)

3.3.1 Vetting process

Labinger and Greaves (2001a) reported that least Bell's vireo was the most abundant and widely distributed endangered bird species within the lower Santa Clara River area (Step 1). Between 1994 and 1999, they found 81 nesting pairs in the lower Santa Clara River and then again found many pairs at the same locations in 2000 (Labinger and Greaves 2001a, 2001b). The birds are state and federal endangered species (Step 2). Habitat fragmentation from development within riparian areas, and the establishment and spread of non-native plant species are primary factors in population decline (Step 3). Habitat fragmentation can result in small populations that are spread out among remaining suitable patches. These smaller, more isolated, populations are more vulnerable to habitat destruction, disease, low production years, and parasitism (USFWS 1998a, Labinger and Greaves 2001a). The Least Bell's vireo was once abundant, but underwent sharp declines in abundance and range during the first half of the 20th century (Step 3) (USFWS 1998a, Labinger and Greaves 2001a, Kus 2002). The ecological interaction of primary management concern for least Bell's vireo populations is brood parasitism by brown-headed cowbirds (*Molothrus ater*). USFWS (1998) describe least Bell's vireo as a host species that readily accepts cowbird eggs (Step 3). The least Bell's vireo received a high priority ranking (Step 5) because of its current distribution in the lower Santa Clara River, it is state and federally endangered, met multiple criteria under Step 3, and recent studies, including several recent conservation and recovery plans (USFWS 1998a, Kus 2002), provide adequate information to characterize the bird's habitat needs (Step 4). The least Bell's vireo was chosen as a focal species (Step 6).

3.3.2 Habitat characterization and distribution

Least Bell's vireo prefers dense vegetative cover within 3–6 ft (1–2 m) of the ground for nesting, and a dense, stratified canopy for foraging (Goldwasser 1981, USFWS 1998a, Labinger and Greaves 2001a) (Appendix B). Labinger and Greaves (2001a) observed least Bell's vireo within early successional cottonwood-willow forest, willow woodland, and mulefat scrub along the lower Santa Clara River. The birds are known to prefer cottonwood forest, willow and mulefat scrub, and sycamore woodland habitats (Kus 2000).

We estimated 2,524 ac (1,022 ha) of potential least Bell's vireo habitat occurring throughout the lower Santa Clara River and lower Piru, Santa Paula, and Sespe creeks under current conditions. We limited potential habitat to the full extent of cottonwood-willow forests as foraging habitat , and mixed riparian and willow scrub as nesting habitat (Table 3-3) (Goldwasser 1981, USFWS 1998a, Kus 2000, Labinger and Greaves 2001a). Unlike less motile species (*i.e.*, western pond turtle), we included reaches that did not have habitat types to support all life history stages, assuming that the bird will satisfy life history needs with habitat among selected reaches. The largest concentration of potential habitat was found within Reach 6, with additional concentrations in Reach 1 and lower Piru Creek (Figure 3-3).

Table 3-3. Least Bell's vireo potential habitat area and distribution.

REACH	HABITAT TYPE							
	Cottonwood-willow forest		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
00	86.1	34.9	31.0	12.5	0.0	0.0	117.1	47.4
01	286.2	115.8	128.8	52.1	16.0	6.5	431.1	174.5
02	142.6	57.7	92.7	37.5	44.5	18.0	279.8	113.2
03	89.2	36.1	44.6	18.0	28.7	11.6	162.5	65.8
04	18.5	7.5	23.3	9.4	8.4	3.4	50.2	20.3
05	40.3	16.3	149.6	60.5	17.9	7.2	207.8	84.1
06	372.4	150.7	118.2	47.8	54.1	21.9	544.7	220.4
07	40.5	16.4	74.8	30.3	5.9	2.4	121.2	49.1
08	109.1	44.1	0.0	0.0	0.0	0.0	109.1	44.1
09	10.2	4.1	20.7	8.4	12.2	4.9	43.1	17.4
10	4.8	1.9	7.3	2.9	0.8	0.3	12.8	5.2
11	107.0	43.3	75.1	30.4	2.7	1.1	184.8	74.8
Piru	148.5	60.1	0.0	0.0	87.2	35.3	235.7	95.4
Santa Paula	4.0	1.6	0.0	0.0	1.9	0.8	5.9	2.4
Sespe	4.6	1.9	7.8	3.2	6.2	2.5	18.7	7.6
TOTAL	**1,464.0**	**592.5**	**773.9**	**313.2**	**286.5**	**116.0**	**2,524.4**	**1,021.6**

3.4 Southwestern Willow Flycatcher (*Empidonax trailli extimus*)

3.4.1 Vetting process

Between 1990 and 2002, southwestern willow flycatcher was recorded in locations along the Santa Clara River (Step 1) (CDFG 2005). The species is federally and state endangered (Step 2). Water diversion and groundwater pumping, changes in flood and fire frequency, grazing, and establishment of invasive non-native plants have caused extensive loss of breeding habitat and reduced populations (Step 3) (USFWS 2002). Historical accounts suggest that willow flycatchers were once abundant in the inland valleys and coastal regions of central and northern California (Bombay *et al.* 2000). In the last five to six decades, however, southwestern willow flycatchers have been eliminated from most of the lower elevation habitat in California (Step 3) (Unitt 1987, Marshall 2000, Sogge *et al.* 2003). Their populations are potentially impacted by interaction with brown-headed cowbirds, which lay their eggs in the nests of other host species, who then incubate the cowbirds eggs and raise their young (Step 3). Because cowbird eggs hatch after relatively short incubation and hatchlings develop quickly, they often out-compete the host's own young for parental care. The southwestern willow flycatcher received a high ranking (Step 5) and was chosen as a focal species (Step 6) because it met criteria under steps 1, 2, and 3, and there is adequate information to understand general habitat requirements within the Study Area (Step 4).

3.4.2 Habitat characterization and distribution

Southwestern willow flycatcher are generally found in riparian areas, preferring trees and shrubs with dense canopy for nesting and breeding, typically cottonwood (*Populus* spp.), willow (*Salix* spp.), mulefat

(*Baccharis salcifolia*), and saltcedar (*Tamarix ramosissima*) (USFWS 2002) (Appendix B). Critical habitat for southwestern willow flycatcher includes riparian areas within the 100-year flood plain or flood prone areas, where dense vegetation is present or expected to become established through succession (USFWS 2005b). Water is typically present within southwestern willow flycatcher territories, particularly at the beginning of the breeding season Sogge *et al.* (1997). Territory sizes have been reported to range from 1.5–5.0 ac (0.06–1.5 ha), with generally larger ranges for polygamous males (Williams and Craig 1998).

Suitable habitat typically consists of the following habitat features (USFWS 2005b):

- Nesting habitat with trees and shrubs that include, but are not limited to, willow (*Salix* spp.) species and boxelder (*Acer negundo*).
- Nesting habitat with a dense (*i.e.*, 50 to 100 percent) tree and/or shrub canopy.
- Dense riparian vegetation with thickets of trees and shrubs
- Dense patches of riparian forest interspersed with small areas of open water or marsh, creating a mosaic; patch size may be as small as 0.25 ac (0.1 ha) or as large as 175 ac (70 ha).

We estimated that 2,125 ac (860 ha) of potential southwestern willow flycatcher habitat occurs throughout Reaches 2 to 11, and along lower Piru, Santa Paula and Sespe Creeks (Table 3-4, Figure 3-4). Among habitat types shown in Table 2-1 those likely to support nesting and foraging for the bird are cottonwood-willow forest, mixed willow and riparian scrub, and mixed riparian forest (USFWS 2005a). The birds are found at elevations ranging from 100–6000 ft (30–1,860 m) above sea level (Grinell and Miller 1944, as cited in USFWS 2002). We included the full extent of the above habitat types, but constrained habitat to a minimum elevation of 100 ft (30 m), eliminating potential habitat in Reaches 0 and 1. Most potential habitat is concentrated in Reach 6, but large amounts also occurred within Reaches 5, 8, 11, and lower Piru Creek.

Table 3-4. Southwestern willow flycatcher potential habitat area and distribution.

REACH	HABITAT TYPE									
	Cottonwood-willow forest		Mixed riparian forest		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
02	101.4	41.0	4.3	1.8	75.0	30.4	44.8	18.1	225.6	91.3
03	89.2	36.1	0.0	0.0	44.6	18.0	29.0	11.7	162.8	65.9
04	18.5	7.5	0.0	0.0	24.6	10.0	8.4	3.4	51.5	20.8
05	40.3	16.3	14.9	6.0	147.3	59.6	17.9	7.2	220.4	89.2
06	374.4	151.5	6.1	2.5	129.7	52.5	54.8	22.2	565.0	228.7
07	41.2	16.7	19.1	7.7	75.1	30.4	6.1	2.5	141.5	57.3
08	109.1	44.1	0.0	0.0	0.0	0.0	0.0	0.0	109.1	44.1
09	10.2	4.1	2.7	1.1	21.1	8.6	12.2	4.9	46.2	18.7
10	4.9	2.0	8.8	3.6	7.6	3.1	0.8	0.3	22.0	8.9
11	108.7	44.0	35.9	14.5	77.3	31.3	3.2	1.3	225.1	91.1
Piru	149.5	60.5	37.3	15.1	0.1	0.0	87.2	35.3	274.1	110.9
Santa Paula	4.2	1.7	22.3	9.0	0.0	0.0	1.9	0.8	28.5	11.5
Sespe	5.9	2.4	32.9	13.3	7.8	3.2	7.0	2.8	53.6	21.7
TOTAL	**1,057**	**427.9**	**184.3**	**74.6**	**610.3**	**247.0**	**273.3**	**110.6**	**2,125**	**860.1**

3.5 Western Yellow-billed Cuckoo (*Coccyzus americanus occidentalis*)

3.5.1 Vetting process

Documented sightings of cuckoos in the Santa Clara River watershed are sparse, although, suitable habitat does exist throughout the watershed (Laymon and Halterman 1989). One cuckoo sighting was recorded near Santa Paula on the Santa Clara River in 1971 (Step 1) (CDFG 2005) (Appendix B). Labinger and Greaves (2001a) sighted two in 1997 and 1998 in the upper Santa Clara River basin, although they noted that these were one-time sightings and most likely migrants. In 2003, a cuckoo was sighted on the Santa Clara River west of Fillmore (CDFG 2005). The western yellow-billed cuckoo is a state endangered species (Step 2) and is a federal endangered species candidate. The bird has narrow habitat requirements, with field studies and habitat suitability modeling concluding that vegetation type (*i.e.*, cottonwood-willow forest), patch size, distance to water, and ratio of high to medium and low tree canopy height are critical factors determining the suitability of habitat for yellow-billed cuckoo breeding pairs (Step 3) (Laymon and Halterman 1989, Greco 1999). Adequate patch size and loss of habitat are the primary threats to western yellow-billed cuckoo populations (Step 3). Loss of habitat is attributed to riparian and floodplain land conversion for agricultural and urban development, and to water management (*e.g.*, dams, channelization, ground water pumping and diversion) that alters the hydrologic regime and precludes the renewal and establishment of riparian trees and shrubs. In California, yellow-billed cuckoos have shown both historic and recent population declines (Step 3). In 1977, there were an estimated 123 to 163 pairs in the state (Laymon 1998). This estimate fell to 30 to 33 pairs ten years later, a 73 to 82 percent decline (Laymon 1998).

Recent research and monitoring have contributed to a reasonable understanding of the habitat requirements and current distribution of the species (Step 4), giving the western yellow-billed cuckoo a high priority ranking (Step 5). The bird was selected as a focal species (Step 6) because it was historically found in the study area, is a state endangered species, met criteria under Steps 3, 4, and 5, and because of its unique habitat needs.

3.5.2 Habitat characterization and distribution

Western yellow-billed cuckoos typically inhabit densely foliated, stands of deciduous trees and shrubs, particularly willows, with a dense understory formed by blackberry, nettles, and/or wild grapes, adjacent to slow-moving watercourses, backwaters, or seeps (CDFG 1983) (Appendix B). River bottoms and other mesic habitats, including valley-foothill and desert riparian habitats, are necessary for breeding. Dense low-level or understory foliage with high humidity is preferred (Gaines 1974, 1977).

Field studies and habitat suitability modeling have concluded that vegetation type (*e.g.*, willow scrub and cottonwood-willow forest), patch size, patch width, and distance to water are important factors determining the suitability of habitat for yellow-billed cuckoo breeding (Laymon and Halterman 1989, Greco 1999). Patch size is an important variable determining presence of cuckoos in California (Halterman 1991, as cited in Laymon 1998), with a trend toward increasing occupancy with increased patch size. Few cuckoos have been found in forested habitat of less than 25 ac (10 ha) (Anderson *et al.* 1994). Willow-cottonwood habitat patches greater than 1,970 ft (600 m) in width were found to be optimal, and typically anything less than 328 ft (100 m) is unsuitable (Laymon and Halterman 1989). Halterman (1991, as cited in Greco 1999) and Laymon *et al.* (1997, as cited in Greco 1999) also observed nesting more frequently in areas where the distance to water was less than 328 ft (100 m).

Greco (1999, adapted and modified from Laymon and Halterman 1989; Laymon *et al.* 1997) described optimum to unsuitable habitat:

- **Optimum:** cottonwood-willow forest, >198 ac (80 ha) area, >1970 ft (600 m) width, <330 ft (100 m) to water
- **Suitable:** cottonwood-willow forest, >100–197 ac (40–80 ha) area, >660–1970 ft (200–600 m) width, <330 ft (100 m) to water
- **Marginal:** cottonwood-willow forest, > 42–99 ac (17–40 ha) area, >330–660 ft (100-200 m) width, <330 ft (100 m) to water
- **Unsuitable:** cottonwood-willow forest, <42 ac (<17 ha) area, <330 ft (<100 m) width, >330 ft (100 m) to water

We estimated current potential habitat using a minimum patch size of 37 ac (15 ha) (Laymon and Halterman 1989), slightly smaller than given in Greco (1999) (42 ac [17 ha], but likely the minimum patch size, assuming micro-habitat requirements (width, distance to water) are fulfilled. The current potential habitat for the cuckoo covered an area of 139 ac (56 ha) and was limited to cottonwood-willow forest, mixed willow forest, and mixed riparian and willow scrub habitats (Figure 3-5, Table 3-5). Based on Greco (1999), we also limited potential habitat to within 330 ft (100 m) of the wetted channel.

Table 3-5. Western yellow-billed cuckoo potential habitat area and distribution.*

REACH	HABITAT TYPE					
	Cottonwood-willow forest		Mixed riparian scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha
5	0.0	0.0	48.0	19.4	**48.0**	**19.4**
8	51.6	20.9	0.0	0.0	**51.6**	**20.9**
Piru	39.4	15.9	0.0	0.0	**39.4**	**15.9**
TOTAL	**91.0**	**36.8**	**48.0**	**19.4**	**139.0**	**56.3**

* Constrained by a minimum patch size of 37 ac (15 ha)

The Riparian Habitat Joint Venture (RHJV 2004) recommends restoring habitat in 25 locations across the state to support 625 pairs (25 pairs per location), including 4 locations (supporting 100 pairs) in Southern California. The RHJV bases this restoration target on simulation modeling, which suggests that “a minimum of at least 25 pairs in a subpopulation with interchange with other subpopulations should be reasonably safe from extinction by stochastic events.” Meeting a 25 pair target within the lower Santa Clara River would require increasing suitable habitat from its current estimated level of 139 ac (56 ha) to 2,500 ac (1,010 ha), while total suitable habitat throughout the state would need to increase from 4,240 hectares to 21,040 hectares to support the state target of 625 pairs (RHJV 2004).

If the potential habitat patch size (37 ac [15 ha]) is ignored under the assumption that patches will increase in area as riparian vegetation expands through removal of non-native species (*i.e.*, giant reed) and restoration of fluvial geomorphic processes (*e.g.*, scour and deposition) associated with river meandering, then current potential habitat increases to 390 ac (158 ha) (Table 3-6). This value is still lower than the 2,500 ac (1,010 ha) target, but provides insight into future potential habitat. While still limited to within 300 ft (100 m) of the channel, potential habitat unconstrained by a minimum patch size is distributed throughout all reaches and lower Piru, Santa Paula, and Sespe creeks with most found along Piru Creek and within Reach 11 (Figure 3-6).

Table 3-6. Western yellow-billed cuckoo potential habitat area and distribution.*

REACH	HABITAT TYPE									
	Cottonwood-willow forest		Mixed riparian forest		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
00	0.9	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.4
01	32.9	13.3	0.0	0.0	0.4	0.1	0.0	0.0	33.3	13.5
02	25.8	10.4	0.0	0.0	0.0	0.0	0.1	0.0	25.9	10.5
03	15.6	6.3	0.0	0.0	7.3	2.9	0.1	0.0	22.9	9.3
04	13.9	5.6	0.0	0.0	0.0	0.0	8.3	3.4	22.3	9.0
05	4.2	1.7	0.0	0.0	13.5	5.5	0.0	0.0	17.6	7.1
06	7.8	3.2	0.0	0.0	3.2	1.3	0.0	0.0	11.0	4.5
07	7.7	3.1	0.0	0.0	13.2	5.3	0.0	0.0	20.9	8.4
08	3.7	1.5	0.0	0.0	0.0	0.0	0.0	0.0	3.7	1.5
09	1.2	0.5	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.5
10	2.4	1.0	0.6	0.3	0.0	0.0	0.0	0.0	3.1	1.2
11	56.3	22.8	9.5	3.8	0.4	0.2	2.0	0.8	68.1	27.6
Piru	119.9	48.5	0.0	0.0	0.1	0.0	26.2	10.6	146.2	59.2
Santa Paula	1.8	0.7	3.1	1.3	0.0	0.0	1.7	0.7	6.6	2.7
Sespe	2.5	1.0	0.0	0.0	1.2	0.5	1.9	0.8	5.6	2.3
TOTAL	**296.6**	**120.0**	**13.3**	**5.4**	**39.3**	**15.9**	**40.3**	**16.3**	**389.4**	**157.6**

* Not constrained by a minimum patch size of 37 ac (15 ha).

3.6 Nevin's Barberry (*Berberis nevinii*)

3.6.1 Vetting process

There is one extant occurrence of Nevin's Barberry in the Santa Clara River basin recorded in the CNDDB (Step 1) (CDFG 2005). Nevin's Barberry is a federal and state endangered species (Step 2) and is a California Native Plant Society list 1B.1 species (seriously endangered in California) (Step 3). Population decline is likely related to low fecundity and habitat loss (Boyd1987, Mistretta 1989). Populations that occur in alluvial washes are threatened by urban and agricultural development, competition by non-native plant species, off-road vehicle activity, road maintenance, and vegetation clearing and channelization for flood control (Mistretta 1989, USFWS 1998b, CNPS 2006, NatureServe 2006) (Step 3). While population sizes vary considerably among extant groups, the majority of occurrences are comprised of only one to a few individuals, with little to no reproduction observed (Step 3) (Boyd 1987, CDFG 2006). While there is no recovery plan for this species, there are several multi-species conservation plans (MSCPs) that address Nevin's barberry habitat (USFWS 2006). The plant received a high ranking because it currently exists in the lower Santa Clara River (Step 1), is endangered (Step 2), met multiple criteria under Step 3, is the focus of recent conservation plans (Step 5), and was chosen as a focal species (Step 6).

3.6.2 Habitat characterization and distribution

Nevin's barberry generally grows within sandy, gravelly soil, on north facing slopes or low gradient washes (Boyd 1987, Hickman 1993, CDFG 2006). On north facing slopes, it is associated with coastal scrub and chaparral habitat, while in low gradient washes it is found in alluvial and riparian scrub (Boyd 1987, CDFG 2006). In general, the plant occurs from 800-5200 ft (240-1580 m) above sea level, with local distribution potentially related to the presence of groundwater (NatureServe 2006, CDFG 2006).

Among the habitat types listed in Table 2-1, we associated Nevin's barberry habitat requirements with the full extent of coastal sage and desert riparian scrub (coastal scrub and chaparral) and mixed riparian and willow scrub (alluvial and riparian scrub) above an elevation of 330 ft (100m). The elevation constraint is lower than described in the species summary and is intended to capture current potential habitat occurring at the edge of the plant's distribution. As such, habitat in Reaches 0-5 was excluded from current potential habitat. We estimated that 979 ac (396 ha) of Nevin's barberry potential habitat is currently present, primarily concentrated along Piru and Sespe creeks and within Reaches 6, 7, and 9 (Table 3-7, Figure 3-7).

Table 3-7. Nevin's barberry potential habitat area and distribution.

REACH	HABITAT TYPE									
	Coastal sage scrub		Desert riparian scrub		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
06	3.2	1.3	45.5	18.4	129.7	52.5	54.8	22.2	233.2	94.4
07	1.7	0.7	21.9	8.8	75.1	30.4	6.1	2.5	104.7	42.4
08	2.9	1.2	4.9	2.0	0.0	0.0	0.0	0.0	7.8	3.2
09	0.0	0.0	69.5	28.1	21.1	8.6	12.2	4.9	102.8	41.6
10	2.1	0.9	66.9	27.1	7.6	3.1	0.8	0.3	77.3	31.3
11	0.8	0.3	0.0	0.0	77.3	31.3	3.2	1.3	81.3	32.9
Piru	86.2	34.9	42.4	17.2	0.1	0.0	87.2	35.3	215.9	87.4
Santa Paula	4.6	1.8	0.0	0.0	0.0	0.0	1.9	0.8	6.5	2.6
Sespe	46.3	18.7	88.2	35.7	7.8	3.2	7.0	2.8	149.3	60.4
TOTAL	**147.7**	**59.8**	**339.2**	**137.3**	**318.7**	**129.0**	**173.2**	**70.1**	**978.8**	**396.1**

3.7 Slender-horned Spineflower (*Dodecahema leptoceras*)

3.7.1 Vetting process

We found three CNDDB occurrences of the slender-horned spineflower within the Santa Clara River basin (Step 1) (CDFG 2005). The species is federally and state endangered (Step 2), and is a California Native Plant Society list 1B.1 species (seriously endangered in California) (Step 3). The flower is found on stabilized alluvial fans, floodplains, and terraces from 660–2,500 ft (200–760 m) in elevation (Step 3) (CNPS 2006). These geomorphic surfaces are greater than 100 years in age (Wood and Wells 1996) and are inundated every 50 to 100 years (Prigge *et al.* 1993, as cited in Dudek and Associates 2000). In general, urbanization and stream channelization are the main causes of population decline (Step 3) (CNPS 2006). Thirty-four historical southern California populations have been observed, but 11 are now presumed extirpated (CDFG 2006). Most of the known occurrences support only a small number of subpopulations.

Preservation of older, stable alluvial surfaces in the historical range of *D. leptoceras* should be the primary focus for the protection of the species (Wood and Wells 1996). There is a recent recovery plan (USFWS 1996) and a recent habitat conservation plan (USFWS 2006) that detail life history requirements and conservation objectives, providing sufficient information (Step 4). The slender-horned spineflower received a high priority ranking (Step 5) and was chosen as a focal species (Step 6) because it is federally and state endangered, satisfied multiple criteria under Step 3, and had recent information about its life history.

3.7.2 Habitat characterization and distribution

Slender-horned spineflower is found on stabilized alluvial fans, floodplains, stream terraces, washes, and associated benches from 660–2,500 ft (200–760 m) in elevation (CNPS 2006). These geomorphic surfaces are usually alluvial deposits greater than 100 years in age (Wood and Wells 1996) that receive overbank deposits every 50 to 100 years (Prigge *et al.* 1993, as cited in Dudek and Associates 2000). They are found in slightly acidic silt soil with low salinity, organic matter, and nutrient content. Preferred microhabitats include silt filled, shallow depressions on relatively flat surfaces (Allen 1996, Wood and Wells 1996). The spineflower occurs in chaparral, cismontane woodland, and coastal alluvial fan scrub habitat, and is generally found in open areas with other spineflower species (Allen 1996).

Among the habitat types listed in Table 2-1, we associated slender-horned spineflower's habitat requirements with the full extent of coastal sage scrub (chapparal), mixed riparian, and willow scrub (alluvial fan scrub habitat) above an elevation of 330 ft (100m). As with Nevin's barberry, the elevation constraint is lower than described in the species summary (660–2,500 ft [200–760 m]), but is intended to capture current potential habitat occurring at the edge of the plant's distribution. This constraint excluded Reaches 0–5. We estimated that 640 ac (259 ha) of current potential habitat is currently distributed along Reaches 6–11 and lower Piru, Santa Paula, and Sespe Creeks (Figure 3-8). Most potential habitat occurs along Piru Creek, and within Reach 6 (Table 3-8).

Table 3-8. Slender-horned spineflower potential habitat area and distribution.

REACH	HABITAT TYPE							
	Coastal sage scrub		Mixed riparian scrub		Mixed willow scrub		TOTAL	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
06	3.2	1.3	129.7	52.5	54.8	22.2	187.7	76.0
07	1.7	0.7	75.1	30.4	6.1	2.5	82.9	33.5
08	2.9	1.2	0.0	0.0	0.0	0.0	2.9	1.2
09	0.0	0.0	21.1	8.6	12.2	4.9	33.3	13.5
10	2.1	0.9	7.6	3.1	0.8	0.3	10.4	4.2
11	0.8	0.3	77.3	31.3	3.2	1.3	81.3	32.9
Piru	86.2	34.9	0.1	0.0	87.2	35.3	173.5	70.2
Santa Paula	4.6	1.8	0.0	0.0	1.9	0.8	6.5	2.6
Sespe	46.3	18.7	7.8	3.2	7.0	2.8	61.1	24.7
TOTAL	**147.7**	**59.8**	**318.7**	**129.0**	**173.2**	**70.1**	**639.6**	**258.8**

3.8 Southern Steelhead (*Oncorhynchus mykiss*)

3.8.1 Vetting process

Steelhead historically spawned and reared in tributaries of the lower Santa Clara River basin, downstream of the Santa Clara River and Piru Creek confluence (Step 1) (Kelley 2004, Harrison *et al.* 2006). Steelhead is also a federally listed threatened species (Step 2). The fish generates high public interest because it is prized by recreational anglers, and appeals to the broader public as a charismatic megafauna associated with wild places and California history (Step 3). Steelhead have specific habitat requirements for each life history stage (egg, fry, juvenile, smolt, and adult) (Step 3). The current distribution of anadromous steelhead in the Santa Clara River basin is influenced by several complete and partial migration barriers (Step 3). The Vern Freeman Diversion, approximately 16 km (10 mi) upstream from the mouth of the mainstem, is likely a partial migration barrier. Since 1991, only 14 adult steelhead are known to have successfully passed through the diversion's fish ladder. Upstream of the Vern Freeman Diversion, passage within Santa Paula Creek is limited by a fish ladder damaged during the 2005 floods[1], and the emplacement of the Santa Felicia Dam (1955) eliminated access to Piru Creek, leaving Sespe Creek as the only unregulated and potentially accessible spawning tributary available to upstream migrants (Titus *et al.*, in preparation 2005). Although there is minimal life history information for southern California steelhead, several unique traits have been identified, including increased temperature tolerance, duration and timing of life stages, and environmental flexibility, and we can use information derived from other sub-populations to understand the general habitat requirements of steelhead in the Santa Clara River (Step 4) (Stoecker and Kelley 2005, Titus *et al.*, in press).

Steelhead received a high priority ranking (Step 5), and were chosen as a focal species (Step 6), because they are a listed species, satisfied multiple criteria in the third step of the vetting process, and we know enough about their general life history stages and habitat requirements to understand how changes in the system may affect them.

3.8.2 Habitat characterization and distribution

We assumed potential steelhead habitat to be the entire length of the lower Santa Clara River. The fish use the river as a migration corridor from the ocean to spawning and rearing habitats upstream in Santa Paula, Sespe, and Piru creeks (Stoecker and Kelley 2005). Fish passage barriers along the mainstem (*e.g.*, Vern Freeman diversion) and within tributaries prevent steelhead from traveling upstream and reaching potential habitat. Rainbow trout, the non-anadromous form of steelhead, occur in headwater reaches and may provide outmigrating smolts, but upstream access is limited (Stoecker and Kelley 2005). Currently, the mainstem supports low quality steelhead habitat, but historically may have provided important over-summering habitat for adult fish (Stoeker and Kelley 2005). Stoecker and Kelly (2005) estimated the amount (in miles) of potential habitat within the Santa Clara River basin based on field surveys (Table 3-9), while Boughton et al (2006) estimated habitat quality within a GIS, based upon geomorphic, hydrologic, and climatic features, concluding that high quality habitat likely exists in the western headwaters of Sespe and Piru creeks, and in a "small but significant" patch on Santa Paula Creek. We present these estimates to show the potential habitat that could become accessible to steelhead through

[1] Restoration and enhancement of fish passage at the Vern Freeman Diversion and within Santa Paula Creek is currently being considered. The Vern Freeman Diversion Dam is undergoing formal Section 7 consultations under the federal Endangered Species Act, lead by the National Marine Fisheries Service. Barriers within the Santa Paula Creek sub-basin are being addressed through studies funded by the California Department of Fish and Game (see www.santaclarariverparkway.org/wkb/projects/santapaulacreek) and the California Department of Transportation.

barrier modification and to demonstrate the importance of the lower Santa Clara River as a migration corridor.

Table 3-9. Steelhead potential habitat length and distribution.

REACH	LENGTH	
	mi	km
Mainstem Santa Clara	32.6	52.5
Piru Creek	128.0	205.9
Santa Paula Creek	18.4	29.6
Sespe Creek	123.0	197.9
TOTAL	**302.0**	**485.9**

3.9 Tidewater Goby (*Eucyclogobius newberryi*)

3.9.1 Vetting process

Tidewater goby has been observed in the Santa Clara River estuary as far as three miles upstream (Step 1) (AMEC 2005). The Central California Coast Evolutionary Significant Unit (ESU) of the tidewater goby is a federal threatened species (Step 2). The tidewater goby is an estuarine species that disperse infrequently through marine habitat, but has no dependency on marine habitat for its life cycle (Step 3) (Swift *et al.* 1989, Lafferty *et al.* 1999). Floods and estuary breaching events can disperse tidewater gobies to nearby suitable habitat, but survival is likely low and dispersal is limited. They are an important part of estuarine food webs, as they provide prey for larger fish and piscivorous birds (Step 3) (Swenson and McCray 1996). However, tidewater goby are highly susceptible to predation by introduced species, especially piscivorous fish and amphibians (Lafferty *et al.* 1999, Lafferty and Page 1997). Current distribution is within the originally observed range of the species, but 20% of historical populations have been extirpated and 50% are likely too small or too degraded to persist long-term (Step 3) (USFWS 2005). The main threats to tidewater goby populations are changes in water quality, degradation and loss of habitat due to urbanization, and predation from invasive species such as the African clawed frog. It is estimated that tidewater goby has disappeared from 74 percent of the coastal lagoons south of Morro Bay (Step 3). In 1999 populations of tidewater goby north of Orange County were proposed to be removed from the federal endangered species list, and the United States Fish and Wildlife Service (USFWS) completed a recovery plan for the fish in 2005 (USFWS 2005a), providing a good source for understanding habitat needs of the species (Step 4). The tidewater goby received a high ranking (Step 5) because there is extensive information about life history requirements, it is a listed species, and met multiple criteria under Step 3, and is recommended as a focal species (Step 6).

3.9.2 Habitat characterization and distribution

We estimated 280 ac (113 ha) of tidewater goby habitat, encompassing the tidal estuary of the lower Santa Clara River (Figure 3-9). The fish require shallow water (< 3 ft [1 m]) at the upper end of tidal lagoons and estuaries, sandy substrate for breeding, and velocity refuge during floods to prevent dispersal to the ocean or areas that are too saline (Moyle 2002, USFWS 2005). Historically, the tidal estuary was approximately 300 ac (122 ha), but is now closer to 30 ac (12 ha) (Stoeker and Kelley 2005). We included the entire tidal estuary and areas just upstream in Reaches 1 and 2, because tidewater goby often migrate a short distance (0.6 mi [1 km]) upstream. Conservation of tidewater goby habitat likely requires conserving the hydro- and morphodynamics of the entire estuary system.

3.10 Giant Reed (*Arundo donax*)

3.10.1 Vetting process

On the Santa Clara River, giant reed grows in large stands or monocultures along floodplains and terraces, and has also invaded most native riparian vegetation types (Step 1) (Stillwater Sciences and URS Corporation 2007). It thrives in open riparian areas with abundant water and nutrients as well as any area susceptible to burning (Coffman2007). The species is non-native, not endangered or threatened (Step 2), but is a B Rated California Department of Food and Agriculture Noxious Weed, and received a High rating from the California Invasive Plant Council (Step 3). The Ventura County Resource Conservation District (VCRCD) initiated the Upper Santa Clara Arundo River Watershed Removal Plan (SCARP) with the goal of implementing a long-term (20-year) removal plan for giant reed and saltcedar. (Step 3) (VCRCD 2006). Due to its clonal growth strategy, efficient use of resources, and high growth rate, giant reed is one of the most successful riparian weedy invaders in California (Rieger and Kreager 1989, Coffman 2007). In California, giant reed is known to increase the risk of flooding, create fire hazards, out-compete indigenous plant species for scarce water resources, and reduce the value of riparian habitat for wildlife (Step 3) (Bell 1994, Bell 1997, DiTomaso 1998). The least Bell's vireo and other riparian birds require structural diversity provided by riparian scrub and mature forest communities for breeding (Zembal 1990, Bell 1994, Bell 1997). When natural riparian vegetation types are replaced by thick stands of giant reed, bird species abundance and other native wildlife have been found to decline (Step 3) (Bell 1994, Bell 1997, Herrera and Dudley 2003, Kisner 2004, Labinger and Greaves 2001). Labinger and Greaves (2001a, 2001b) observed over the course of their study (1994–2000) that while dense thickets of giant reed supported very low bird diversity, "a low to moderate mixture of giant reed with native willow woodland supported high bird diversity in some areas…(and) giant reed was also used for nesting."

Although giant reed is a non-native, invasive species, it received a high rating (Step 5) because it met multiple criteria under Step 3 and is the focus of management documents that detail long-term eradication strategies (Step 4). Giant reed was chosen as a focal species (Step 6).

3.10.2 Habitat characterization and distribution

Giant reed was found throughout the lower Santa Clara River and appeared to be the most invasive of all non-native plant species (Stillwater Sciences and URS 2007). Its presence was recorded as a visual estimate of the percent cover for every mapped polygon, regardless of the assigned vegetation type. This method was used to provide an estimate of actual area occupied by giant reed and to estimate the degree to which it has established throughout the lower Santa Clara River. Based on this method, we found that giant reed has invaded all study reaches (Table 3-10 and Figure 3-10). Most vegetation strata (polygon areas) had less than 25 % cover of giant reed, but since the spread can be rapid and have adverse impacts to riparian ecology, these areas should still be considered for eradication or restoration efforts.

Table 3-10. The percent cover and area of giant reed within mapped vegetation polygons along the lower Santa Clara River.

REACH	GIANT REED							
	1–25%		26–50%		51–75%		76–100%	
	Ac	Ha	Ac	Ha	Ac	Ha	Ac	Ha
00	65.0	26.3	1.8	0.7	8.3	3.4	14.1	5.7
01	430.8	174.4	91.8	37.1	32.8	13.3	10.2	4.1
02	488.9	197.9	14.6	5.9	64.1	25.9	17.1	6.9
03	252.4	102.2	5.1	2.0	37.9	15.3	16.7	6.7
04	192.5	77.9	10.4	4.2	136.6	55.3	0.0	0.0
05	189.1	76.5	48.8	19.8	14.3	5.8	3.1	1.3
06	789.8	319.6	56.9	23.0	55.1	22.3	61.8	25.0
07	376.4	152.3	145.6	58.9	22.6	9.1	6.8	2.7
08	372.2	150.6	20.7	8.4	5.7	2.3	78.4	31.7
09	188.7	76.3	1.8	0.7	4.5	1.8	0.4	0.2
10	497.9	201.5	47.9	19.4	5.1	2.1	2.0	0.8
11	191.8	77.6	44.0	17.8	17.3	7.0	5.4	2.2
Piru	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Santa Paula	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sespe	98.1	39.7	0.0	0.0	0.0	0.0	0.0	0.0
TOTAL	**4,133.6**	**1,672.8**	**489.3**	**198.0**	**404.2**	**163.6**	**215.9**	**87.4**

3.11 Saltcedar (*Tamarix ramosissima*)

3.11.1 Vetting process

Saltcedar is found sporadically along terraces and floodplains of the Santa Clara River (Step 1), as well as other streams and rivers in Ventura County. The species is non-native, not endangered or threatened (Step 2), but is a B Rated California Department of Food and Agriculture Noxious Weed, and received a High rating from the California Invasive Plant Council (Step 3). Invasion of *Tamarix* in river systems of the southwestern U.S. has created both environmental and economic impacts (Step 3). The extensive lateral root system of saltcedar makes it an extremely strong competitor with native riparian phreatophytes, such as willows and cottonwoods. Once plants have become well-established, they can use prodigious amounts of water and tolerate long periods of both inundation and drought (Duncan 1997). Before large floods in winter 2005, saltcedar was sparse along floodplains with a few large accumulations along adjacent terraces, particularly in upstream reaches (Coffman personal observation). After the floods, seedlings established across floodplains, likely from sustained moist soil conditions (Stillwater Sciences and URS Corporation 2007). Future monitoring will be required to determine whether the seedling cohort becomes widely established (*i.e.*, survives to maturity). In general, invading saltcedar lowers wildlife habitat value in riparian ecosystems heavily by decreasing available food sources and altering structural characteristics (Step 3) (Shafroth *et al.* 2005). Monotypic stands provide limited cover for large mammals, and fewer nesting sites for birds and herpetofauna in more southern latitudes due to lack of shading in mid- to late-summer (Hunter *et al.* 1988, Lovich and DeGouvenain 1998, Shafroth *et al.* 2005). Both the endangered southwestern willow flycatcher and the candidate for federal endangered species list western yellow-billed cuckoo prefer native forests in some cases, but incorporate some habitat with saltcedar into their breeding territory (Shafroth *et al.* 2005).

Although the plant is non-native and invasive, saltcedar received a high rating (Step 5) because it met multiple criteria under Step 3 and is the focus of management documents that detail long-term eradication strategies (Step 4). Saltcedar was chosen as a focal species (Step 6).

3.11.2 Habitat characterization and distribution

Saltcedar was also found throughout the lower Santa Clara system in 2005 and 2006, primarily as seedlings and scattered mature individuals (Figure 3-11). Its presence was recorded as either present or absent within a mapped polygon, but in few polygons was it the dominant vegetation or habitat type, although a few stands dominated saltcedar were observed in Reach 11 (Stillwater Sciences and URS 2007). The field surveys did not include a visual estimate of percent cover because saltcedar distribution was sparse compared to other native and non-native (*i.e.*, *Arundo donax*) habitat types and a percent cover scale would show a narrow range of values, likely within 0-5%. The data were meant to estimate potential saltcedar habitat and areas of potential saltcedar invasion. We observed 1486 ac (601 ha) of invaded habitat, with most found in Reaches 10 and 6 (Table 3-11), which also had high rates of giant reed invasion (Table 3-10).

Table 3-11. Saltcedar potential habitat area and distribution.

REACH	HABITAT TYPE	
	Saltcedar	
	ac	ha
00	0.0	0.0
01	69.3	28.0
02	33.6	13.6
03	99.8	40.4
04	18.6	7.5
05	27.8	11.2
06	281.1	113.7
07	125.4	50.8
08	22.4	9.1
09	54.6	22.1
10	392.2	158.7
11	194.0	78.5
Piru	134.0	54.2
Sespe	32.6	13.2
TOTAL	**1486**	**601.2**

4 DISCUSSION

Within the lower Santa Clara River, restoration actions should initially be targeted within reaches and habitat types that provide the greatest potential benefit to focal species. Using the above analysis, we found that Reaches 10 and 11 supported the greatest number of focal species, while Reach 6 and Piru Creek supported the greatest habitat area (Table 4-1, Figure 4-1). Additionally, we found that mixed riparian and mixed willow scrub habitat types supported the greatest number of focal species, followed by cottonwood-willow forest, which was the most abundant habitat type (Table 4-2). To further target potential restoration actions, we examined habitat types within reaches, finding that Reaches 6 and 11, and Piru contained the greatest areas of mixed riparian and mixed willow scrub, and cottonwood-willow forest. These results suggest that restoration actions should be targeted to Reaches 6, 11, and Piru Creek.

These reaches are similar in location, with the exception of Piru Creek, to those suggested by Court *et al.* (2000). They prioritized areas for species conservation using a site selection model (SITES) that considered habitat usage by endangered species (at the parcel-scale) and the cost of land parcel acquisition under $8 and $16 million budget scenarios. Their recommended parcels were clustered in between Santa Paula and Sespe Creeks, and upstream of Piru Creek in roughly the same locations as Reaches 6 and 11. They considered eight focal species, the cost of land parcels, and limited their analysis to the 500-year floodplain of the lower Santa Clara River, which may explain differences in habitat area and the exclusion of lower Piru Creek from their recommended sites.

Restoration opportunities may also be afforded by giant reed and saltcedar eradication to allow for re-establishment of native habitat types. Reaches 6 and 10 had the greatest amount of area invaded by giant reed and saltcedar, but large areas were also observed in Reaches 1, 2, 7, and 8 (Table 3-10 and Table 3-11). Non-native species removal should be targeted to Reach 10, which supports seven focal species, and Reaches 7 and 8, which support fewer focal species, but greater potential habitat area (Table 4-1). Restoration within these reaches will also encourage the formation of contiguous habitat corridors along Reaches 6, 7, and 8, and Reaches 10 and 11.

The active channel of the lower Santa Clara River is an important corridor for aquatic and terrestrial species dispersal and movement. Wildlife corridors and landscape linkages provide a conduit for movement between habitat patches, allow gene flow between populations, and allow recolonization of disconnected and potentially extinct habitat patches (Forman and Godron 1986, SCRPSC 1996a). Important linkages were described within the South Coast Missing Linkages project, which identified those essential to maintaining the ecological integrity of the South Coast Ecoregion (Penrod *et al.* 2006). The Santa Clara River Enhancement and Management Plan (SCREMP) also discussed potential linkages along the river and to adjacent uplands, identifying connections by study segment and reach, and assigning a conservation priority (high, medium, low) to each linkage (SCRPSC 1996a,b). The lower Santa Clara River connects east-west along its active channel, links with the Sierra Madre to the north along Santa Paula, Sespe, and Piru Creeks, and links with the Santa Susana Mountains to the south (SCRPSC 1996a, Penrod *et al.* 2006). While Reaches 5, 9, and lower Santa Paula and Sespe Creeks potentially support fewer focal species and less habitat area than Reaches 6, 7, 8, 10, 11, and Piru Creek (Table 4-1), restoration actions targeted to these reaches, in addition to Reaches 6, 7, 8, 10, 11, and Piru Creek, would provide a local benefit by creating an east-west habitat corridor along the lower Santa Clara River and a regional benefit by creating a set of north-south corridors from the Sierra Madre to the Santa

Susana Mountains. Restoration along these reaches would also potentially conserve all high and medium priority connections along the lower Santa Clara River, as identified in SCRPSC (1996a,b) (Figure 4-2).

Table 4-1. Focal species habitat area and usage by reach.

REACH	Total focal species habitat area		# of focal species potentially supported by habitat types in reach
	Ac	Ha	
00	118.1	47.79	4[a,b,c]
01	464.4	187.93	4[a,b,c]
02	651.9	263.83	5[b,c]
03	382.4	154.74	5[b,c]
04	175.3	70.92	5[b,c]
05	482.2	195.14	5[b,c]
06	1,718.9	695.64	7[b,c]
07	607.3	245.79	7[b,c]
08	471.6	190.87	7[b,c]
09	231.0	93.48	7[b,c]
10	661.3	267.61	8[b,c]
11	1,008.4	408.10	8[b,c]
Piru	1,249.6	505.72	7[b,c]
Santa Paula	54.0	21.84	6[b,c]
Sespe	287.6	116.38	6[b,c]
TOTAL	**8,563.9**	**3465.75**	**9[c]**

[a] Includes tidewater goby
[b] Includes steelhead
[c] Excludes giant reed and saltcedar

Table 4-2. Habitat area and focal species usage.

HABITAT TYPE*	Total focal species habitat area		# of focal species potentially supported by habitat type
	Ac	Ha	
Coastal sage scrub	294.7	119.28	2
Cottonwood-willow forest	3,300.3	1335.62	5
Desert riparian scrub	338.4	136.94	1
Freshwater wetland	256.6	103.83	2
Herbaceous	537.4	217.49	1
Mixed riparian forest	279.6	113.16	3
Mixed riparian scrub	2,238.1	905.73	6
Mixed willow scrub	955.7	386.78	6
Riverwash	502.1	203.19	1
TOTAL	**8,702.9**	**3522.02**	**7**

*Excludes tidewater goby, steelhead, giant reed, and saltcedar.

5 CONCLUSION

Based on the above results, the highest priority restoration sites within the lower Santa Clara River floodplain corridor are Reaches 6 and 11, and lower Piru Creek. These reaches provide potential habitat for the greatest number of focal species (7, 8, and 7, respectively) and the greatest potential habitat areas (Table 4-1). These reaches also contain the greatest areas of mixed riparian and mixed willow scrub, and cottonwood-willow forest, which support the greatest number of focal species and are critical for restoration (Table 4-2). Additional restoration opportunities may arise from non-native species eradication along Reaches 7, 8, and 10, which have been highly invaded by giant reed and are at risk for more widespread establishment of saltcedar (Table 3-10 and Table 3-11). Restoration that also includes Reaches 5, 9, and lower Santa Paula and Sespe Creeks would allow formation of habitat corridors along the Santa Clara River and north-south to the Sierra Madre and Santa Susana Mountains (Figure 4-2), which were identified in SCRPSC (1996a) and Penrod *et al.* (2006) as essential to maintaining local and regional diversity.

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Figures

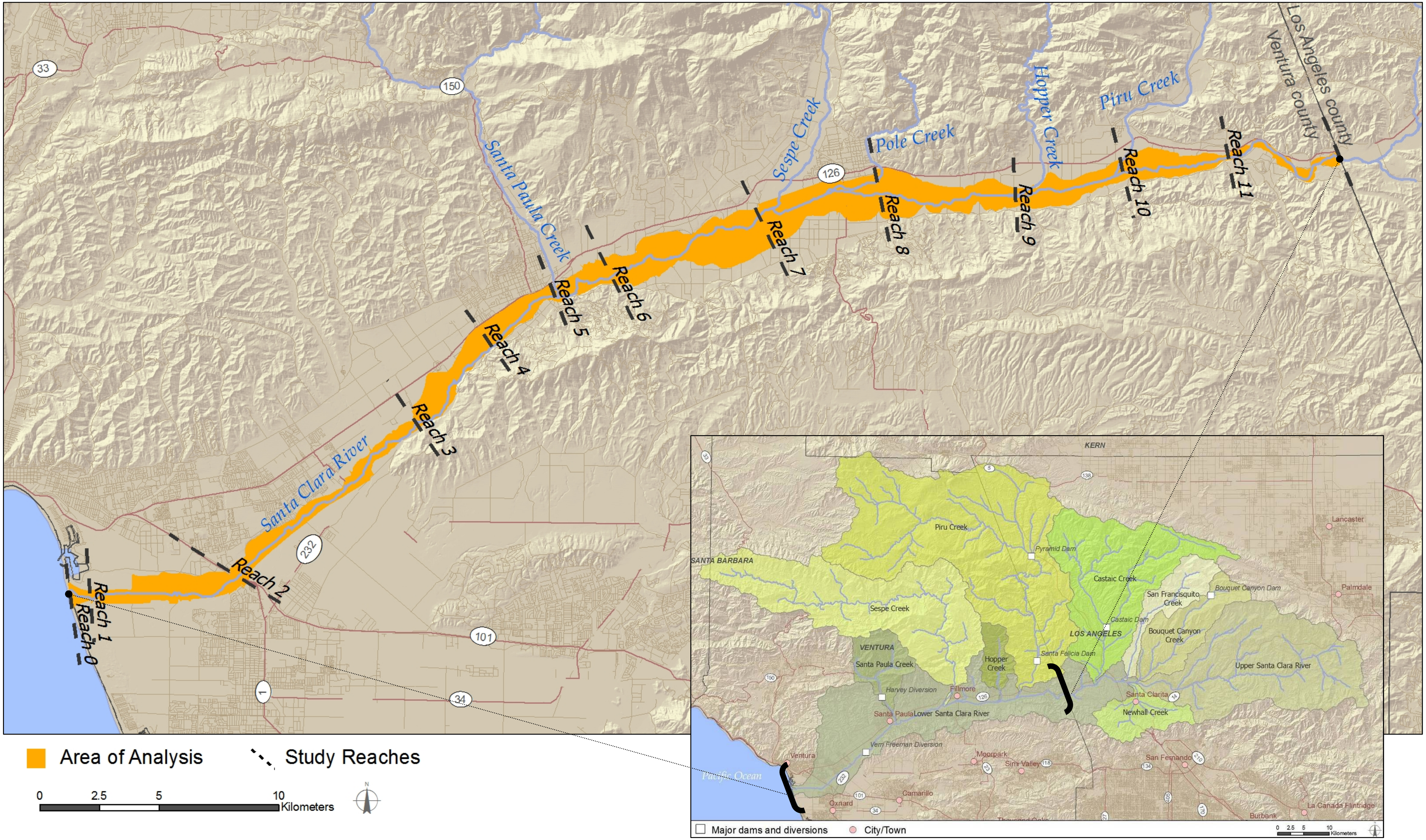


Figure 1-1. The Santa Clara River watershed (inset) and focal species habitat assessment Study Area and Reach delineation.

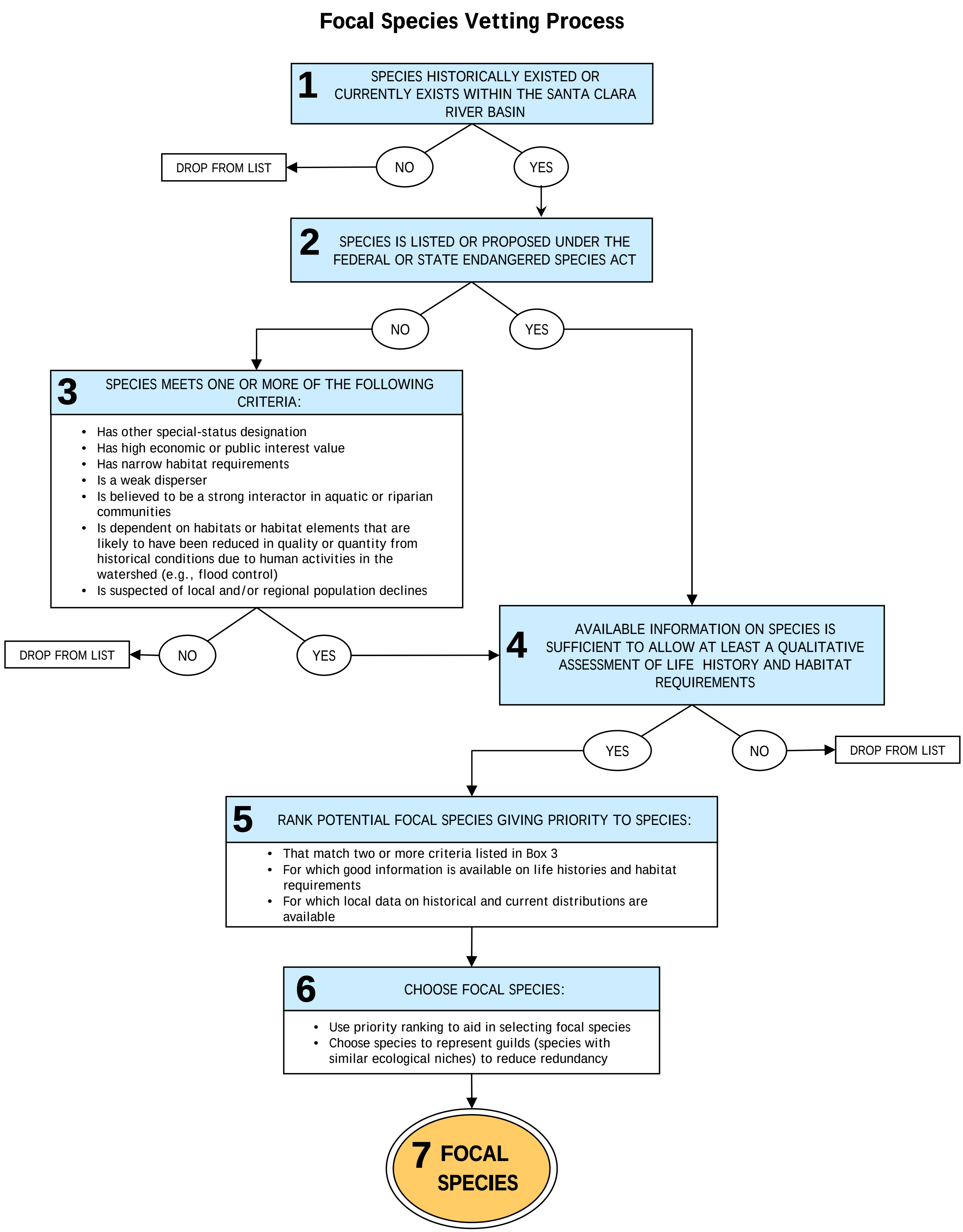


Figure 2-1. Focal Species Vetting Process.

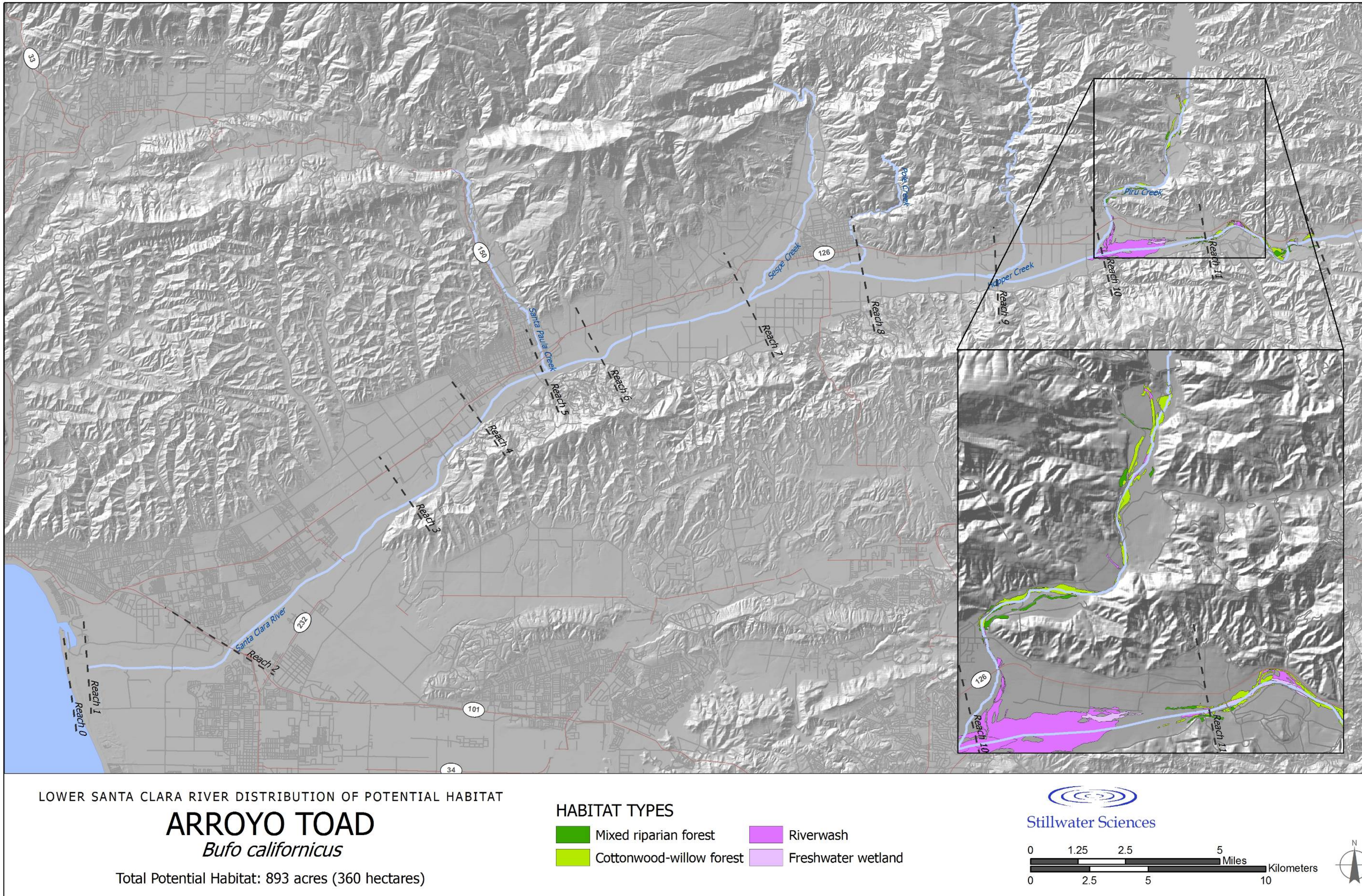


Figure 3-1. Distribution of potential habitat for the arroyo toad under current conditions in the lower Santa Clara River study area.

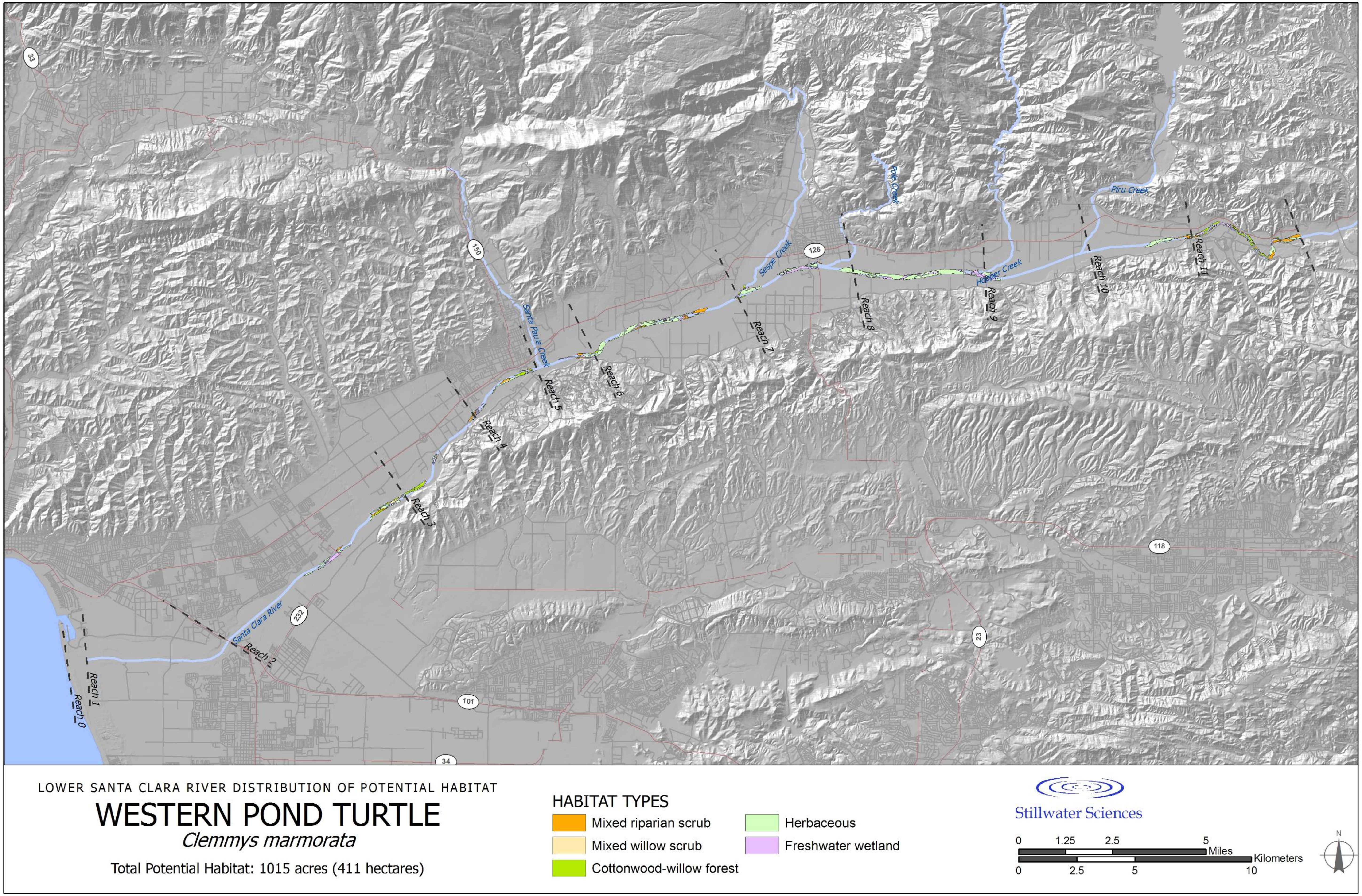


Figure 3-2. Distribution of potential habitat for the western pond turtle under current conditions in the lower Santa Clara River study area.

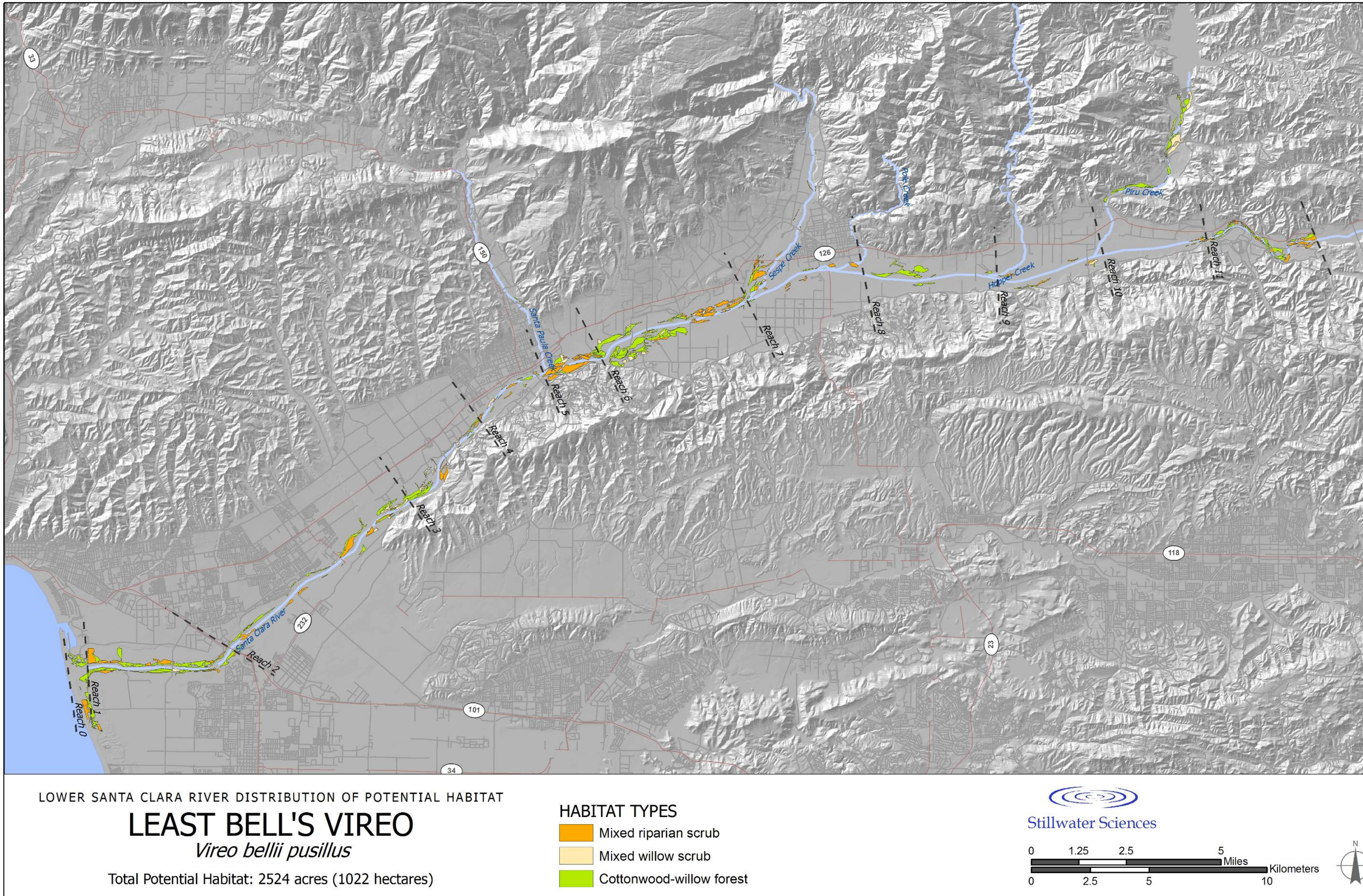


Figure 3-3. Distribution of potential habitat for the least Bell's vireo under current conditions in the lower Santa Clara River study area.

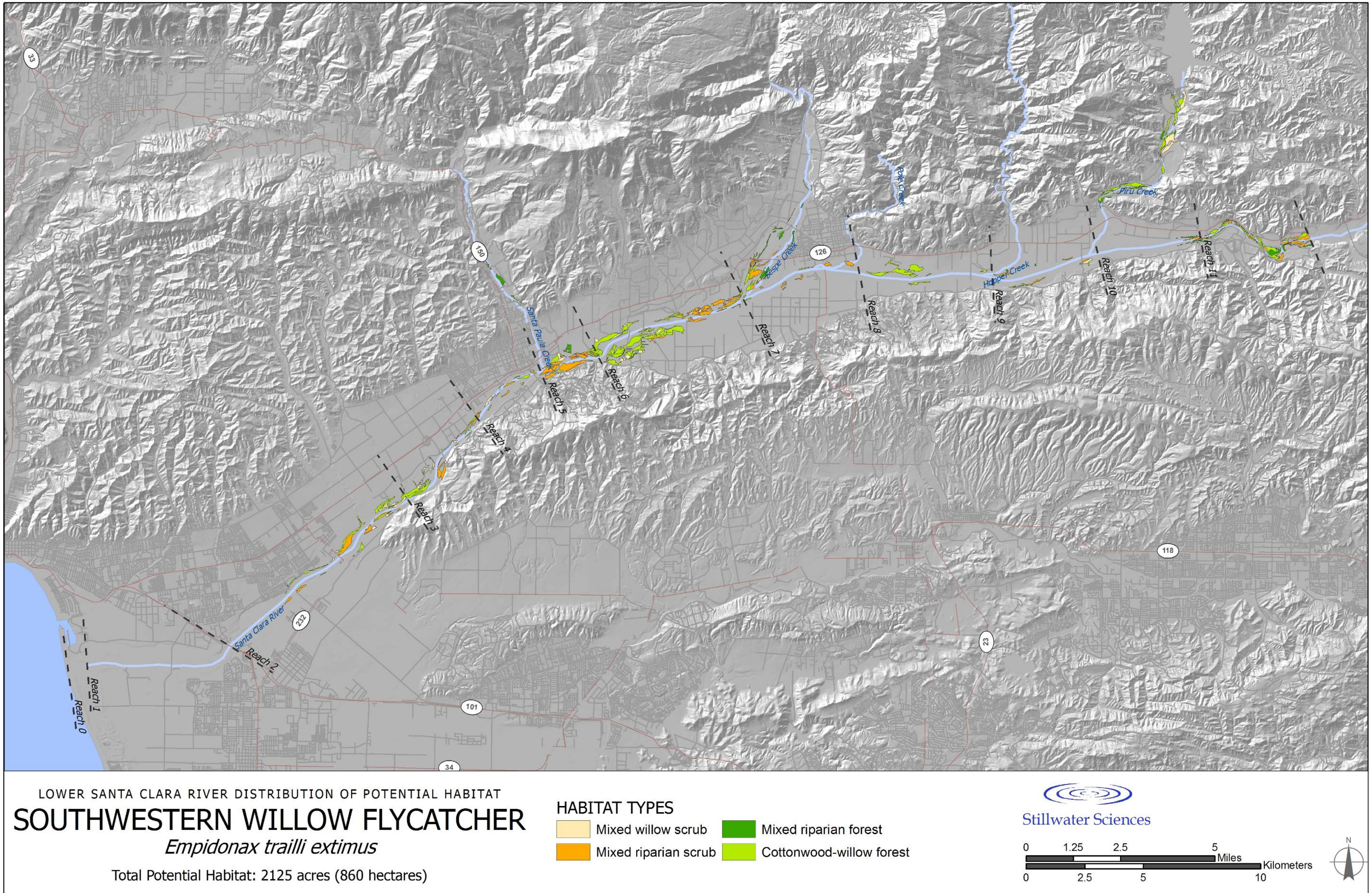


Figure 3-4. Distribution of potential habitat for the southwestern willow flycatcher under current conditions in the lower Santa Clara River study area.

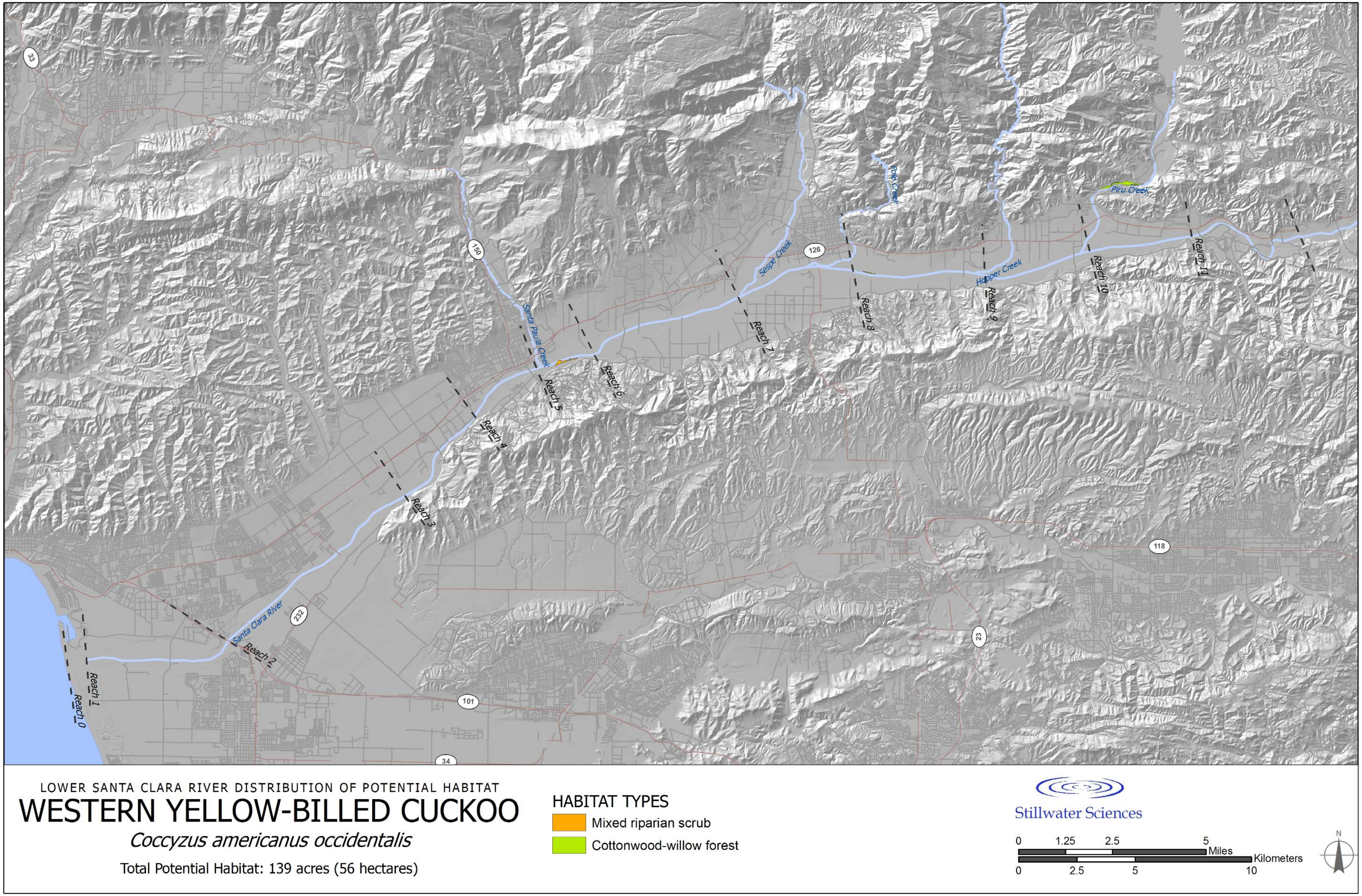


Figure 3-5. Distribution of potential habitat for the western yellow-billed cuckoo under current conditions in the lower Santa Clara River study area constrained by 37 acre (15 ha) patch size.

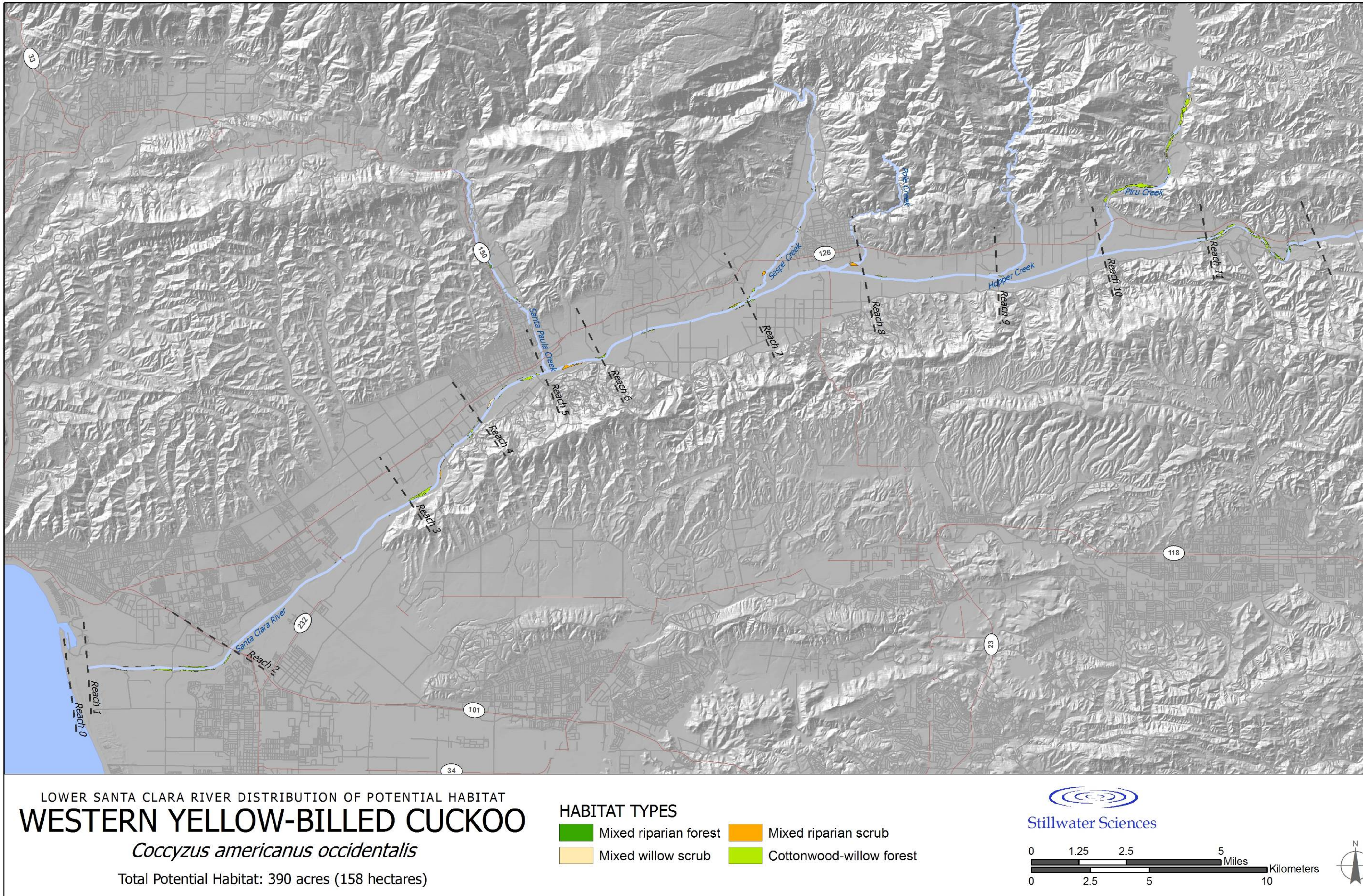


Figure 3-6. Distribution of potential habitat for the western yellow-billed cuckoo under current conditions in the lower Santa Clara River study area unconstrained by patch size.

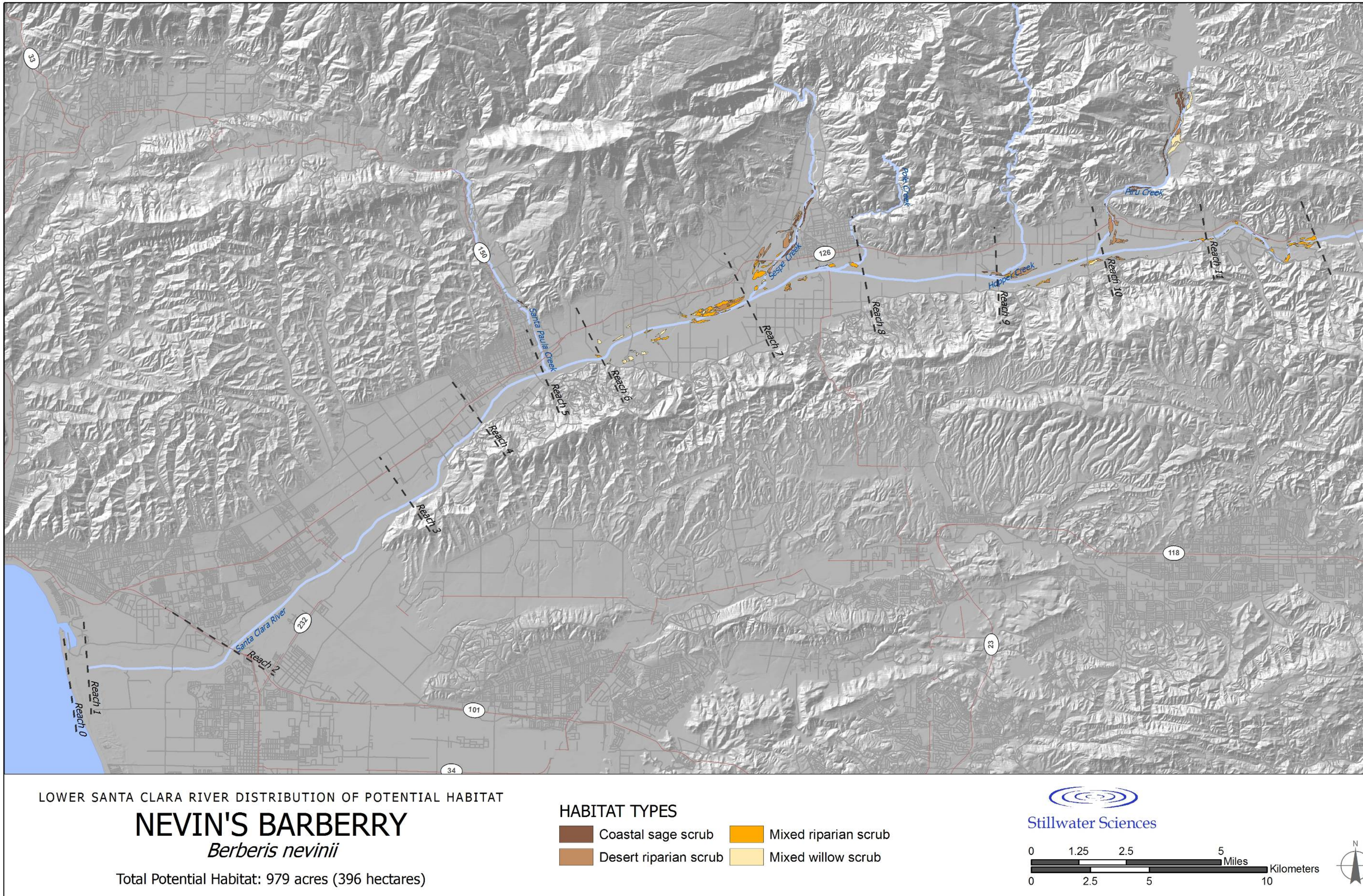
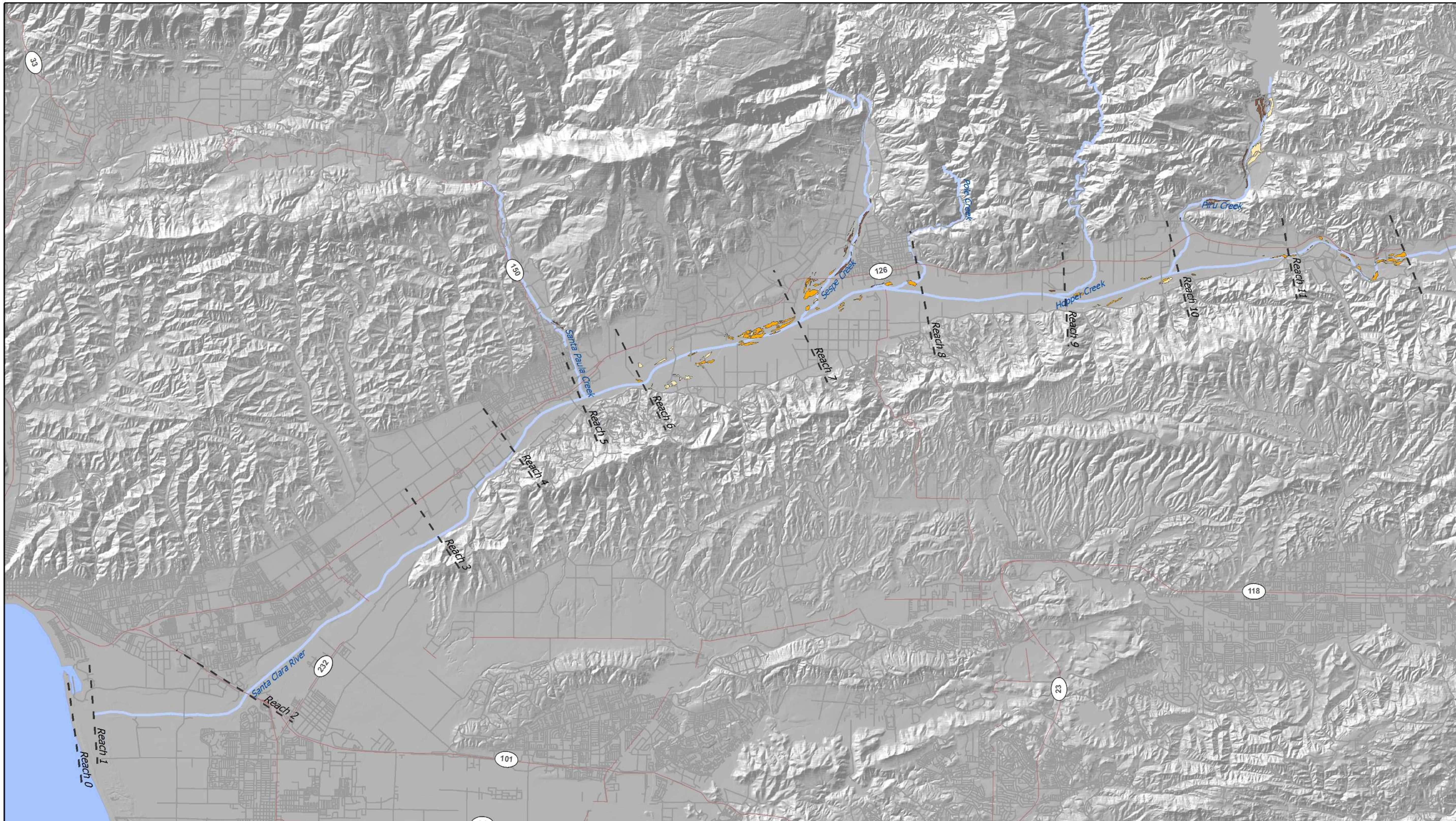


Figure 3-7. Distribution of potential habitat for the Nevin's barberry under current conditions in the lower Santa Clara River study area.



LOWER SANTA CLARA RIVER DISTRIBUTION OF POTENTIAL HABITAT

SLENDER-HORNED SPINEFLOWER

Dodecahema leptoceras

Total Potential Habitat: 640 acres (259 hectares)

HABITAT TYPES

- Coastal sage scrub
- Mixed riparian scrub
- Mixed willow scrub

Stillwater Sciences

0 1.25 2.5 5 Miles

0 2.5 5 10 Kilometers

N

Figure 3-8. Distribution of potential habitat for the slender-horned spineflower under current conditions in the lower Santa Clara River study area.

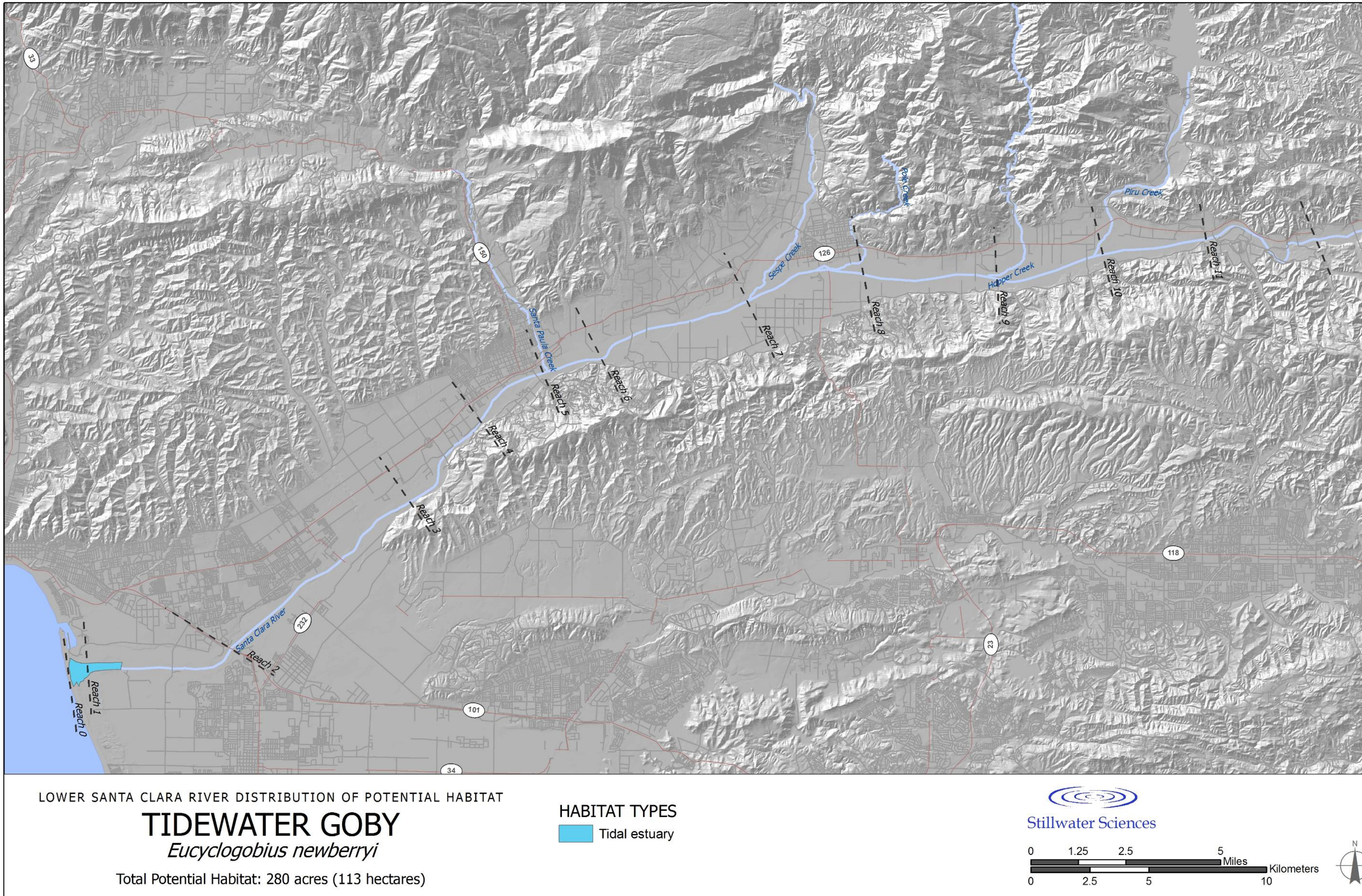


Figure 3-9. Distribution of potential habitat for the tidewater goby under current conditions in the lower Santa Clara River study area.

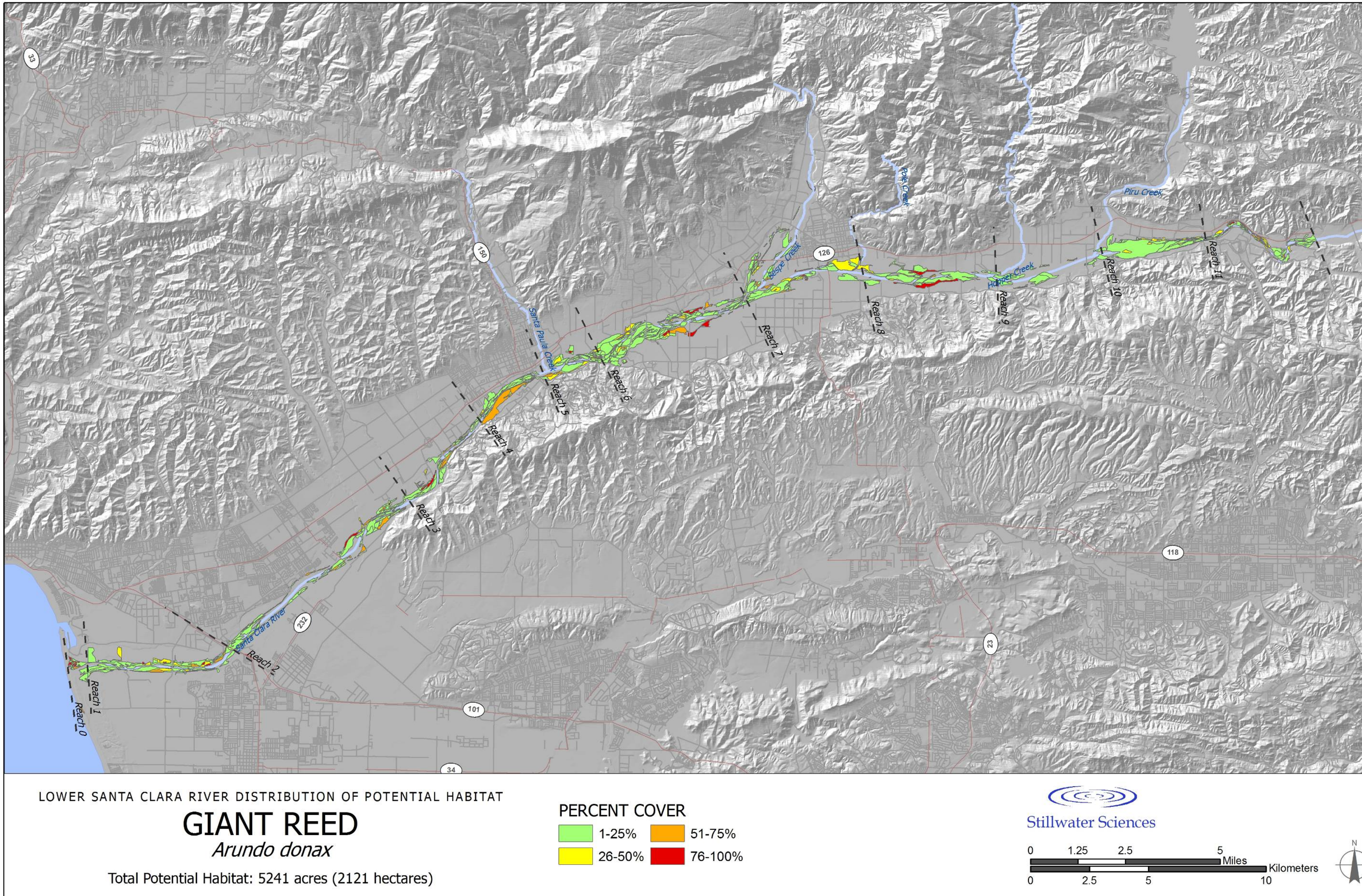


Figure 3-10. Current distribution and abundance (percent cover) of giant reed in the lower Santa Clara River study area, based on recent vegetation mapping and field surveys.

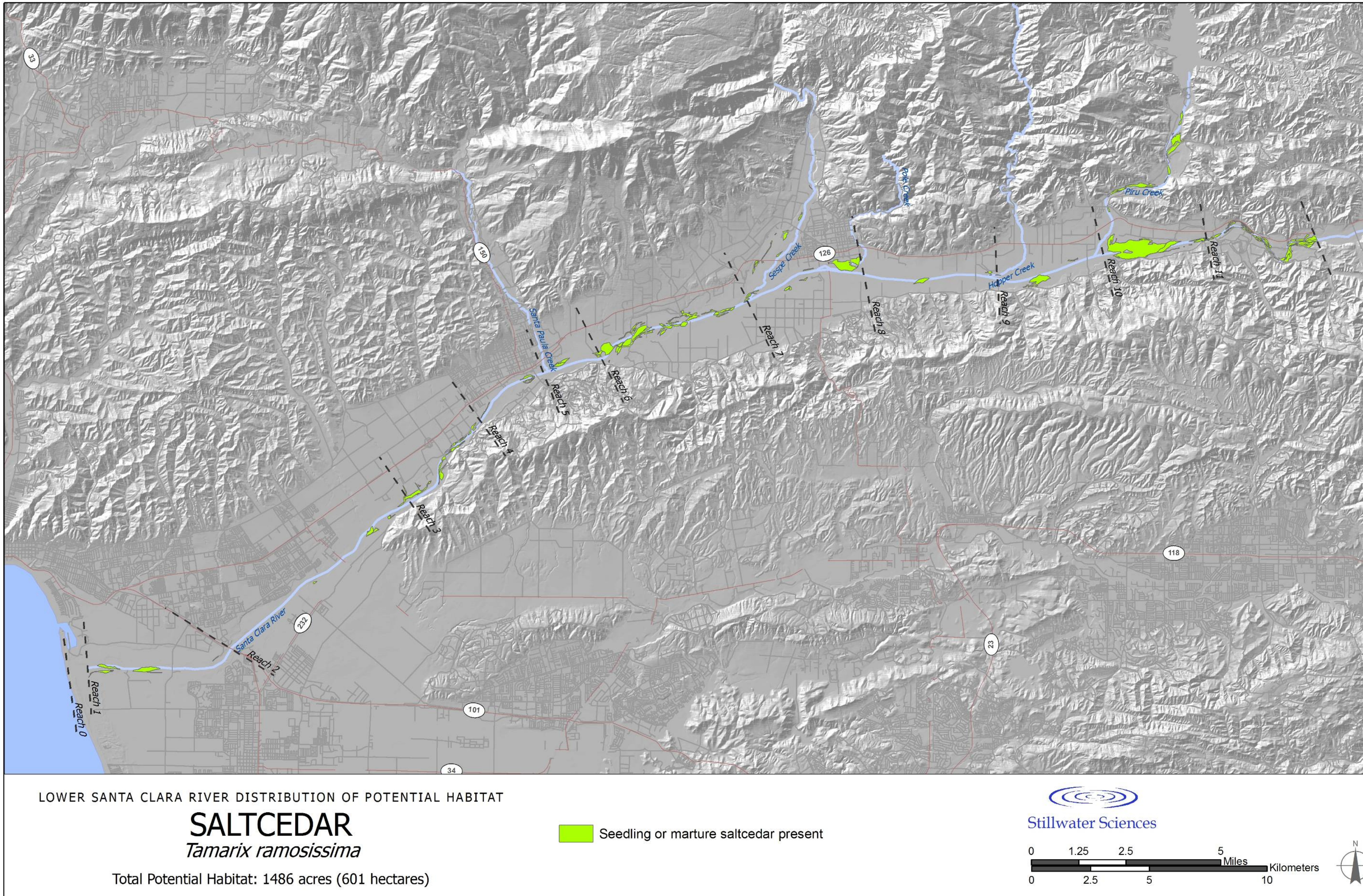


Figure 3-11. Current distribution of saltcedar in the lower Santa Clara River study area, based on observations of seedlings and mature individuals during recent (2005 and 2006) field surveys.

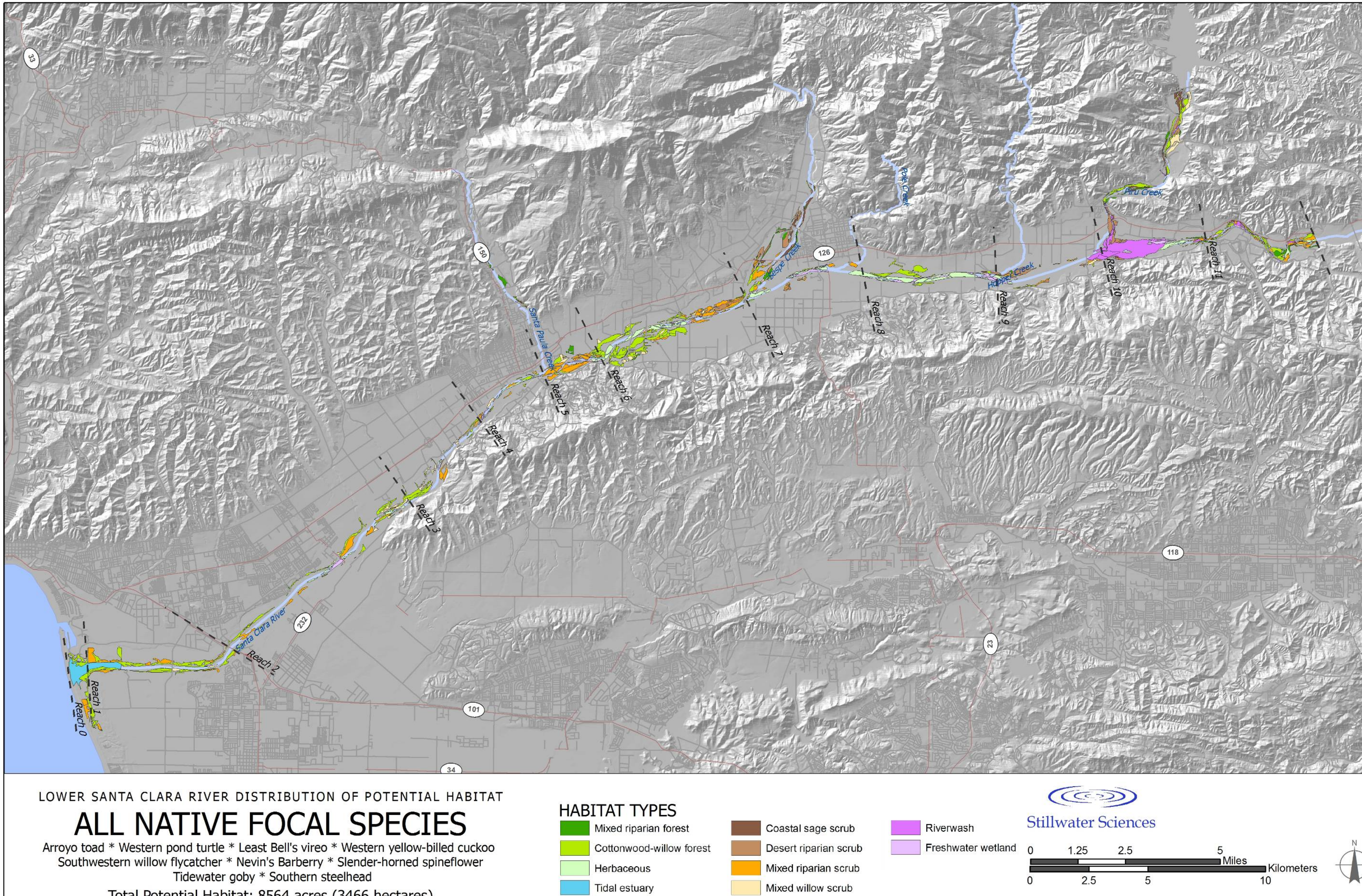


Figure 4-1. Distribution of potential habitat for the nine native focal species under current conditions in the lower Santa Clara River study area. Potential southern steelhead habitat is indicated in the map as the entire Santa Clara River mainstem and major tributaries (see Section 3.8 for more detail).

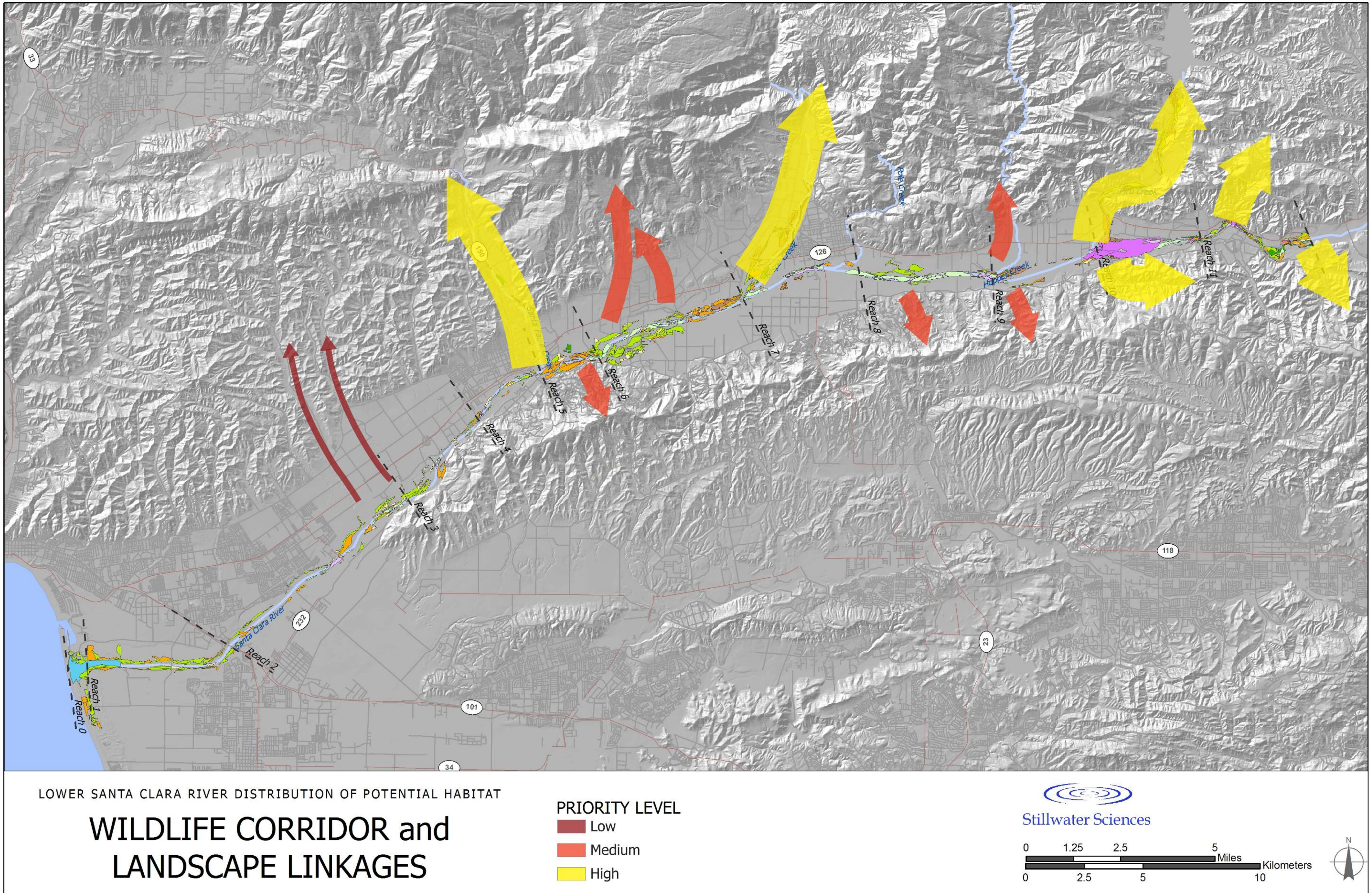


Figure 4-2. Location and conservation priority of important landscape linkages along the lower Santa Clara River study area (adapted from SCRPSC 1996a, b).

Appendices

APPENDIX A

Threatened, Endangered, and Special Status Species within the Lower Santa Clara River

Table A-1. Threatened, endangered, and special status animal species potentially occurring in the lower Santa Clara River.

Scientific Name	**Common Name**	**Status** Federal	**Status** State	**Status** Other	**Habitat Requirements (General + Micro)**
INVERTEBRATES					
Streptocephalus woottoni	Riverside fairy shrimp	FE	--	--	Endemic to Riverside, Orange, and San Diego counties in areas of tectonic swales/earth slump basins in grassland and coastal sage scrub; inhabit seasonally static pools filled by winter/spring rains. Hatch in warm water later in the season.
FISH					
Gila orcutti	Arroyo chub	--	--	CSC	Los Angeles basin south coastal streams; slow water stream sections with mud or Sand bottoms. Feed heavily on aquatic vegetation and associated invertebrates.
Catostomus Santaanae	Santa Ana sucker	FT	--	CSC	Endemic to Los Angeles basin south coastal streams; habitat generalists, but prefer Sand-rubble-boulder bottoms, cool, clear water, and algae.
Oncorhynchus mykiss irideus	Southern steelhead - southern California ESU	FE	--	CSC	Fed listing refers to pops from Santa Maria River south to southern extent of range (San Mateo Creek in San Diego County); southern steelhead likely have greater physiological tolerances to warmer water and more variable conditions.
Eucyclogobius newberryi	Tidewater goby	FE	--	CSC	Brackish water habitats along the California coast from Agua Hedionda lagoon, San Diego County to the mouth of the Smith River; found in shallow lagoons and lower stream reaches, they need fairly still but not stagnant water and high oxygen levels.
Gasterosteus aculeatus williamsoni	Unarmored threespine stickleback	FE	SE	FP	Weedy pools, backwaters, and among emergent vegetation at the stream edge in small southern California streams; cool (<75 F [24 C]), clear water with abundant vegetation.

Scientific Name	Common Name	Status: Federal	Status: State	Status: Other	Habitat Requirements (General + Micro)
AMPHIBIANS					
Bufo californicus	Arroyo toad	FE	--	CSC	Semi-arid regions near washes or intermittent streams, including valley-foothill and desert riparian, desert wash, etc; rivers with sandy banks, willows, cottonwoods, and sycamores; loose, gravelly areas of streams in drier parts of range.
Rana muscosa	Mountain yellow-legged frog	FE	--	CSC	Federal listing refers to populations in the San Gabriel, San Jacinto and San Bernardino mountains only; always encountered within a few feet of water. Tadpoles may require up to 2 yrs to complete their aquatic development.
Spea (=Scaphiopus) hammondii	Western spadefoot	--	--	CSC	Occurs primarily in grassland habitats, but can be found in valley-foothill hardwood woodlands; vernal pools are essential for breeding and egg-laying.
REPTILES					
Phrynosoma coronatum (blainvillii)	Coast (San Diego) horned lizard	--	--	CSC	Inhabits coastal sage scrub and chaparral in arid and semi-arid climate conditions; prefers friable, rocky, or shallow sandy soils.
Anniella pulchra pulchra	Silvery legless lizard	--	--	CSC	Sandy or loose loamy soils under sparse vegetation; soil moisture is essential. They prefer soils with a high moisture content.
Clemmys (=Emys; =Actinemys) marmorata pallida	Southwestern pond turtle	--	--	CSC	Inhabits permanent or nearly permanent bodies of water in many habitat types; below 6000 ft elev; require basking sites such as partially submerged logs, vegetation mats, or open mud banks. Need suitable nesting sites.
Thamnophis hammondii	Two-striped garter snake	--	--	CSC	Coastal California from vicinity of Salinas to northwest Baja California. From sea to about 7,000 ft elevation; highly aquatic, found in or near permanent fresh water. Often along streams with rocky beds and riparian growth.

Scientific Name	Common Name	Status: Federal	Status: State	Status: Other	Habitat Requirements (General + Micro)
BIRDS					
Riparia riparia	Bank swallow	--	ST	--	(Nesting) colonial nester; nests primarily in riparian and other lowland habitats west of the desert; requires vertical banks/cliffs with fine-textured/Sandy soils near streams, rivers, lakes, ocean to dig nesting hole.
Passerculus Sandwichensis beldingi	Belding's savannah sparrow	--	SE	--	Inhabits coastal salt marshes, from Santa Barbara south through San Diego County; nests in *salicornia* on and about margins of tidal flats.
Athene cunicularia	Burrowing owl	--	--	CSC	(Burrow sites) open, dry annual or perennial grasslands, deserts and scrublands characterized by low-growing vegetation; subterranean nester, dependent upon burrowing mammals, most notably, the California ground squirrel.
Gymnogyps Californianus	California condor	FE	SE	--	Require vast expanses of open savannah, grasslands, and foothill chaparral in mountain ranges of moderate altitude; deep canyons containing clefts in the rocky walls provide nesting sites. Forages up to 100 miles from roost/nest.
Eremophila alpestris actia	California horned lark	--	--	CSC	Coastal regions, chiefly from Sonoma co. To San Diego co. Also main part of San Joaquin valley and east to foothills; short-grass prairie, "bald" hills, mountain meadows, open coastal plains, fallow grain fields, alkali flats.
Sterna antillarum browni	California least tern	FE	SE	FP	(Nesting colony) nests along the coast from San Francisco bay south to northern Baja California; colonial breeder on bare or sparsely vegetated, flat substrates: Sand beaches, alkali flats, land fills, or paved areas.
Polioptila californica californica	Coastal California gnatcatcher	FT	--	CSC	Obligate, permanent resident of coastal sage scrub below 2500 ft in southern California; low, coastal sage scrub in arid washes, on mesas and slopes. Not all areas classified as coastal sage scrub are occupied.

Scientific Name	Common Name	Status			Habitat Requirements (General + Micro)
		Federal	State	Other	
Accipiter cooperii	Cooper's hawk	--	--	CSC	(Nesting) woodland, chiefly of open, interrupted or marginal type; nest sites mainly in riparian growths of deciduous trees, as in canyon bottoms on river flood-plains; also, live oaks.
Vireo bellii pusillus	Least Bell's vireo	FE	SE	--	(Nesting) summer resident of southern calif in low riparian in vicinity of water or in dry river bottoms; below 2000 ft; nests placed along margins of bushes or on twigs projecting into pathways, usually willow, baccharis, mesquite.
Falco mexicanus	Prairie falcon	--	--	CSC	(Nesting) inhabits dry, open terrain, either level or hilly; breeding sites located on cliffs. Forages far afield, even to marshlands and ocean shores.
Aimophila ruficeps canescens	Southern California rufous-crowned sparrow	--	--	CSC	Resident in southern California coastal sage scrub and sparse mixed chaparral; frequents relatively steep, often rocky hillsides with grass and forb patches.
Empidonax traillii extimus	Southwestern willow flycatcher	FE	SE	--	(Nesting) riparian woodlands in southern California. State listing includes all subspecies;
Agelaius tricolor	Tricolored blackbird	--	--	CSC	(Nesting colony) highly colonial species, most numberous in central valley and vicinity. Largely endemic to California; requires open water, protected nesting substrate, and foraging area with insect prey within a few km of the colony.
Charadrius alexandrinus nivosus	Western snowy plover	FT	--	CSC	(Nesting) federal listing applies only to the pacific coastal population; Sandy beaches, salt pond levees and shores of large alkali lakes. Needs Sandy, gravelly or friable soils for nesting.
Coccyzus americanus occidentalis	Western yellow-billed cuckoo	FC	SE	--	(Nesting) riparian forest nester, along the broad, lower flood-bottoms of larger river systems; nests in riparian jungles of willow, often mixed with cottonwoods, w/ lower story of blackberry, nettles, or wild grape.

		Status			
Scientific Name	**Common Name**	**Federal**	**State**	**Other**	**Habitat Requirements (General + Micro)**
Elanus leucurus	White-tailed kite	--	--	FP	(Nesting) rolling foothills/valley margins w/scattered oaks and river bottomlands or marshes next to deciduous woodland; open grasslands, meadows, or marshes for foraging close to isolated, dense-topped trees for nesting and perching.
Dendroica petechia brewsteri	Yellow warbler	--	--	CSC	(Nesting) riparian plant associations. Prefers willows, cottonwoods, aspens, sycamores, and alders for nesting and foraging; also nests in montane shrubbery in open conifer forests.
Icteria virens	Yellow-breasted chat	--	--	CSC	(Nesting) summer resident; inhabits riparian thickets of willow and other brushy tangles near watercourses; nests in low, dense riparian, consisting of willow, blackberry, wild grape; forage and nest w/in 10 ft of ground.
MAMMALS					
Taxidea taxus	American badger	--	--	CSC	Most abundant in drier open stages of most shrub, forest, and herbaceous habitats, with friable soils; need sufficient food, friable soils and open, uncultivated ground. Prey on burrowing rodents. Digs burrows.
Chaetodipus californicus femoralis	Dulzura pocket mouse	--	--	CSC	Variety of habitats including coastal scrub, chaparral and grassland in San Diego Co; attracted to grass-chaparral edges.
Choeronycteris mexicana	Mexican long-tongued bat	--	--	CSC	Occasionally found in San Diego co. Which is on the periphery of their range; feeds on nectar and pollen of night-blooming succulents. Roosts in relatively well-lit caves, and in and around buildings.
Neotoma lepida intermedia	San Diego desert woodrat	--	--	CSC	Coastal scrub of southern California from San Diego county to San Luis Obispo County; moderate to dense canopies preferred. They are particularly abundant in rock outcrops and rocky cliffs and slopes.

Scientific Name	**Common Name**	**Status**			**Habitat Requirements (General + Micro)**
		Federal	**State**	**Other**	
Onychomys torridus ramona	Southern grasshopper mouse	--	--	CSC	Desert areas, especially scrub habitats with friable soils for digging. Prefers low to moderate shrub cover; feeds almost exclusively on arthropods, especially scorpions and orthopteran insects.
Euderma maculatum	Spotted bat	--	--	CSC	Occupies a wide variety of habitats from arid deserts and grasslands through mixed conifer forests; feeds over water and along washes. Needs rock crevices in cliffs or caves for roosting.
Eumops perotis californicus	Western mastiff bat	--	--	CSC	Many open, semi-arid to arid habitats, including conifer and deciduous woodlands, coastal scrub, grasslands, chaparral etc; roosts in crevices in cliff faces, high buildings, trees and tunnels.

FE = Federally listed as endangered
FT = Federally listed as threatened
FC = Candidate for federal listing
SE = State listed as endangered (California)
ST = State listed as threatened (California)
CSC = listed by CDFG as a California species of special concern
FP = listed by CDFG as fully protected

Table A-2. Threatened, endangered, and special status plant species potentially occurring in the lower Santa Clara River.

		Status			
Scientific Name	**Common Name**	**Federal**	**State**	**CNPS**	**Habitat Requirements (General + Micro)**
PLANTS					
Oxytheca parishii var. abramsii	Abrams's oxytheca	--	--	1B	Chaparral; shale to sandy places.
Aphanisma blitoides	Aphanisma	--	--	1B	Coastal bluff scrub, coastal dunes, coastal scrub; on bluffs and slopes near the ocean in Sandy or clay soils. In steep decline on the islands and the mainland.
Dudleya blochmaniae ssp. blochmaniae	Blochman's dudleya	--	--	1B	Coastal scrub, coastal bluff scrub, valley and foothill grassland; open, rocky slopes; often in shallow clays over serpentine or in rocky areas w/little soil.
Orcuttia californica	California Orcutt grass	FE	SE	1B	Vernal pools.
Eriogonum crocatum	Conejo buckwheat	--	SR	1B	Chaparral, coastal scrub, valley and foothill grassland; conejo volcanic outcrops; rocky sites.
Lasthenia glabrata ssp. coulteri	Coulter's goldfields	--	--	1B	Coastal salt marshes, playas, valley and foothill grassland, vernal pools; usually found on alkaline soils in playas, sinks, and grasslands.
Delphinium parryi ssp. blochmaniae	Dune larkspur	--	--	1B	Chaparral, coastal dunes (maritime); on rocky areas and dunes.
Aster greatae	Greata's aster	--	--	1B	Chaparral, cismontane woodland; mesic canyons.
Calochortus weedii var. vestus	Late-flowered mariposa lily	--	--	1B	Chaparral, cismontane woodland; dry, open coastal woodland, chaparral; on serpentine.
Helianthus nuttallii ssp. parishii	Los Angeles sunflower	--	--	1A	Marshes and swamps (coastal salt and freshwater). Historical from southern California.
Pentachaeta lyonii	Lyon's pentachaeta	FE	SE	1B	Chaparral, valley and foothill grassland; edges of clearings in chap., usually at the ecotone btwn grassland and chaparral or edges of firebreaks.
Stylocline masonii	Mason's neststraw	--	--	1B	Chenopod scrub, pinyon-juniper woodland; sandy washes.
Castilleja gleasonii	Mt. Gleason Indian paintbrush	--	SR	1B	Lower montane coniferous forest; on open flats or slopes in granitic soil. Restricted to the san gabriel mountains. .

Scientific Name	Common Name	Status			Habitat Requirements (General + Micro)
		Federal	State	CNPS	
Berberis nevinii	Nevin's barberry	FE	SE	1B	Chaparral, cismontane woodland, coastal scrub, riparian scrub; on steep, n-facing slopes or in low grade sandy washes.
Fritillaria ojaiensis	Ojai fritillary	--	--	1B	Broadleaved upland forest (mesic), chaparral, lower montane coniferous forest; rocky sites; one reported as "moist shale talus."
Chaenactis glabriuscula var. orcuttiana	Orcutt's pincushion	--	--	1B	Coastal bluff scrub, coastal dunes; sandy sites.
Calochortus plummerae	Plummer's mariposa lily	--	--	1B	Coastal scrub, chaparral, valley and foothill grassland, cismontane woodland, lower montane coniferous forest; occurs on rocky and sandy sites, usually of granitic or alluvial material. Can be very common after fire.
Senecio aphanactis	Rayless ragwort	--	--	2	Cismontane woodland, coastal scrub; drying alkaline flats..
California macrophyll	Round-leaved filaree	--	--	2	Cismontane woodland, valley and foothill grassland; clay soils.
Cordylanthus maritimus ssp. maritimus	Salt marsh bird's-beak	FE	SE	1B	Coastal salt marsh, coastal dunes; limited to the higher zones of the salt marsh habitat.
Chorizanthe parryi var. fernandina	San Fernando Valley spineflower	FC	SE	1B	Coastal scrub; sandy soils.
Deinandra minthornii	Santa Susana tarplant	--	SR	1B	Chaparral, coastal scrub; on sandstone outcrops and crevices, in shrubland.
Opuntia basilaris var. brachyclada	Short-joint beavertail	--	--	1B	Chaparral, joshua tree woodland, mohavean desert scrub, pinyon juniper woodland, riparian woodland; sandy soil or coarse, granitic loam.
Calochortus clavatus var. gracilis	Slender mariposa lily	--	--	1B	Chaparral, coastal scrub; shaded foothill canyons; often on grassy slopes within other habitat.
Dodecahema leptoceras	Slender-horned spineflower	FE	SE	1B	Chaparral, coastal scrub (alluvial fan sage scrub); flood deposited terraces and washes; assoc include *encelia, dalea, lepidospartum*, etc.

		Status			
Scientific Name	**Common Name**	**Federal**	**State**	**CNPS**	**Habitat Requirements (General + Micro)**
Navarretia fossalis	Spreading navarretia	FT	--	1B	Vernal pools, chenopod scrub, marshes and swamps, playas; San Diego hardpan and San Diego claypan vernal pools; in swales and vernal pools, often surrounded by other habitat types.
Delphinium umbraculorum	Umbrella larkspur	--	--	1B	Cismontane woodland; mesic sites.
Astragalus pycnostachyus var. lanosissimus	Ventura Marsh milk-vetch	FE	SE	1B	Coastal salt marsh; within reach of high tide or protected by barrier beaches, more rarely near seeps on sandy bluffs.
Dudleya verityi	Verity's dudleya	FT	--	1B	Chaparral, cismontane woodland, coastal scrub; on volcanic rock outcrops in the Santa Monica mountains.

FE = Federally listed as endangered
FT = Federally listed as threatened
FC = Candidate for federal listing
SE = State listed as endangered (California)
ST = State listed as threatened (California)
SR = State listed as rare (California)
1A = Presumed extinct in California
1B = Rare, threatened, or endangered in California and elsewhere
2 = Rare, threatened, or endangered in California, but more common elsewhere.

APPENDIX B

Focal Species Summaries

Table of Contents

ARROYO TOAD
*Bufo californicus**

Legal Status

Federal	Endangered
State	None
Other	CDFG Species of Special Concern

Taxonomy

Although the arroyo toad has been treated as a subspecies of *B. microscaphus* (Arizona toad), recent genetic analysis has shown that *B. m. californicus* may be morphologically differentiated enough from Arizona populations of *B. m. microscaphus* that species recognition is justified (Price and Sullivan 1988, Frost and Hillis 1990, Stebbins 1985). Recent genetic work also indicates that the arroyo toad is morphologically differentiated from the other two subspecies *B. m. microscaphus* and *B. m. mexicanus* (Jennings and Hayes 1994). A comparison of allozyme (enzyme variants) frequencies between all three subspecies found discrete differences and indicated mutually exclusive evolutionary lineages, which supports the hypothesis that each taxon should be recognized as a unique, separate species (Gergus 1998 as cited in USFWS 1999).

Geographic Distribution

Historical arroyo toad distribution extended west of the desert in coastal mountains and valleys from Monterey County and the upper Salinas River system in the vicinity of Santa Margarita (San Luis Obispo County), throughout southern California in the Los Angeles basin and the coastal drainages of Orange and Riverside counties to the San Diego River system (Miller and Miller 1936 as cited in USFWS 1999), to as far south as the Rio Santo Domingo system in San Quentin–San Simon in Baja California (Myers 1930, Tevis 1944, and Sanders 1950, all as cited in USFWS 1999; Stebbins 2003).

Although the arroyo toad occurs mainly along coastal drainages, it has also been recorded at several locations on the desert slopes of the Transverse and Peninsular mountain ranges south of the Santa Clara River, Los Angeles County (Patten and Myers 1992 as cited in USFWS 1999, Jennings and Hayes 1994, USFWS 1999). Although the elevation range has historically extended from sea level to 4600 ft(1, 400 m)(and extending to a maximum of 8,000 feet (2,440 m) in Baja California del Norte) (Lannoo 2005). Currently most arroyo toad populations in the northern and central parts of the range are restricted to elevations of 300 to 1,400 m (1,000 to 4,600 ft), perhaps due to widespread habitat loss at lower elevations (USFWS 1999, Stebbins 2003). The inability to withstand cooler temperature regimes, especially during the larval stage (Sweet 1992 as cited in USFWS 1999), may limit the species in the upper elevations (USFWS 1999).

The current distribution of arroyo toads is limited to several populations that occur on privately owned lands, primarily in holdings within the Cleveland National Forest (USFWS 1994); extant populations of

* Formerly *Bufo microscaphus californicus*

more than a dozen adults have been recorded at only six of the 22 known locations south of Ventura (USFWS 1994). There are several populations on the Mojave River, Little Rock Creek, Whitewater River, San Felipe Creek, Vallecito Creek, and Pinto Canyon (USFWS 1994, Patton and Myers 1992 as cited in USFWS 1999, Stebbins 1985).

Local Distribution

The arroyo toad was historically found in the upper and lower Santa Clara River watershed. The Santa Clara River basin crosses Ventura and Los Angeles counties, with Los Angeles County encompassing most of the upper watershed and Ventura County containing most of the lower. Many historical records of arroyo toad populations in the watershed are outdated or no longer exist, but it is almost certain that toads occupied much of the mainstem Santa Clara from the Los Angeles County line to a few miles from the ocean, as they do in drainages on MCB (Marine Corps Base) Camp Pendleton today (S. Sweet, UC Santa Barbara, pers. comm., 28 March 2006; Lannoo 2005). A large flood in 1928 and extensive agricultural modification of the lower floodplain beginning as early as 1880 likely extirpated a significant amount of the arroyo toad habitat (S. Sweet, pers. comm., 28 March 2006). For example, arroyo toads were found in the Santa Clara River basin on May 22, 1912, at Santa Paula, Ventura County (USFWS 1999). This site (now located along Highway 150) apparently was part of a formerly extensive oak *(Quercus* spp.) woodland on the floodplain near Santa Paula Creek (USFWS 1999). The current creek floodplain (75 to 120 m [250 to 400 ft] in elevation) has been urbanized extensively for approximately for 4.8 kilometers (3 miles) along the river, and arroyo toads have been extirpated from the area (USFWS 1999).

Arroyo toads currently persist in large numbers along Sespe Creek in the Los Padres National Forest, Ventura County, from about Hot Springs Canyon upstream to the mouth of Tule Creek (Sweet 1992, USFWS 1999). The maximum elevation is approximately 1,040 m (3,400 ft) and there are 24 km (15 mi) of suitable arroyo toad stream habitat in Sespe Creek (USFWS 1999). The upper half of the portion of Sespe Creek inhabited by arroyo toads has generally contained large areas of excellent habitat and numerous high quality breeding pools, while the lower portion supports few stream terraces with suitable substrates, and fewer pools appropriate for use as arroyo toad breeding sites (Sweet 1992). Sweet (1992, 1993) found through repeated surveys of Sespe Creek during the 1980s and 1990s that the arroyo toad population fluctuated between approximately 130 and 250 adults. The Lions Creek fire in 1991 reduced vegetative cover and led to severe erosion in approximately half of the upland habitat in the upper half of the creek basin, reducing the extent and quality of the upland and breeding habitat (USFWS 1999).

Arroyo toads have been historically found along Piru Creek (Ventura and Los Angeles counties) between the confluence of the Santa Clara River (elevation 205 m [660 ft]) and Bear Gulch (elevation 945 m [3,100 ft]) (USFWS 1999). With the construction of Lake Piru in the 1950s and Pyramid Lake in the 1970s, arroyo toads were eliminated from much of their historic range in the drainage and now are restricted to short segments above each of the two reservoirs (Sweet 1992 as cited in USFWS 1999). Upper Piru Creek supports small populations of arroyo toads distributed in a range of good to marginal habitats, while lower Piru Creek generally has larger numbers of arroyo toads distributed over areas of good to excellent habitat that generally are undisturbed by human activities (Sweet 1992, USFWS 1999). The lower segment is from Blue Point Campground upstream to lower Piru Gorge (elevation 340 to 410 m [1,100 to 1,350 ft]), a distance of 5.6 km (3.5 mi), and the upper segment is from the headwaters of Pyramid Lake upstream to Bear Gulch (elevation 760 to 945 m [2,500 to 3,100 feet]), a distance of 7.2 km (4.5 mi) (USFWS 1999).

Potential habitat for arroyo toads probably exists in the upper Santa Clara River basin, Los Angeles County, and in some of the other canyons that drain from the north (USFWS 1999). Drainages that are potential candidates for arroyo toad habitat include parts of the San Francisquito Canyon drainages and Bouquet Canyon drainages (S. Sweet, pers. comm., 1997 as cited in USFWS 1999). Additionally, along Castaic Creek, Los Angeles County, on California Department of Water Resources land and the Angeles National Forest, arroyo toads were recently found below the dam at Castaic Lake, throughout a 3.2-kilometer (2-mile) segment of the creek, as well as above the reservoir in the dredge spoils (Campbell *et al.* 1996, F. Hovore, Planning Consultants Research, pers. comm., 1997; both as cited in USFWS 1999). Arroyo toads were likely more widespread in the Castaic Creek drainage before the reservoir was constructed in the 1970s (USFWS 1999).

California Natural Diversity Database (CNDDB) searches identified five 7.5-minute U.S. Geological Survey (USGS) quadrangles within the Santa Clara River watershed that contained records of arroyo toad sightings: Devil's Heart Peak, Agua Dulce, Newhall, Cobblestone Mountain, and Whitaker Peak.

Population Trends

The arroyo toad has been extirpated from 75 percent of its former range (USFWS 1994), and populations have declined in abundance (often to extirpation) at most sites where historical records exist (Lannoo 2005). In general, population densities of arroyo toads are relatively low (12 per hectare) in foothill and montane areas but are often found in comparatively higher densities (10 per 100 m) along coastal streams (Lannoo 2005).

Life History

The arroyo toad has evolved in a system that is inherently dynamic, with marked seasonal and annual fluctuations in climatic factors, particularly rainfall (USFWS 1999). Adult and juvenile toads may aestivate or hibernate during summer and winter months, emerging to feed and hydrate (Sandburg 2004). Burrow locations are generally in dry or lightly damp, fine sand, particularly in the canopy edge of willow (*Salix* spp.) or cottonwood (*Populus* spp.) (Sandburg 2004). Arroyo toads may travel as far as 1.2 km (0.70 mi) from the edge of the riparian corridor for burrowing and night foraging (Holland *et al*. 2001).

Breeding generally occurs from late January or February to early July, although timing is dependent on the water year and local weather conditions (USFWS 1999). In the northern portion of their range, adult arroyo toads begin breeding in late March (Sweet 1992), and as early as January in the coastal areas of southern California (USFWS 1999). A study of arroyo toads in the northern part of the range found breeding activity was associated with rainfall and air temperatures above 7 °C (45 °F) (Dudek & Associates 2000, Sweet 1992). Males usually begin calling when water temperatures reach 14 °C (57 °F) and may breed with several females during the course of the season (Sweet 1992 as cited in USFWS 1999). Breeding may continue into early July depending on when individual females are no longer reproductively active, and when the males stop calling (Sweet 1992 as cited in USFWS 1999).

Males generally call from edge habitat of shallow pools (less than 5 cm [2 inches] deep), although they may call from sandbars out of the water (Holland 1997 as cited in USFWS 1999). During the mating season, calling males are particularly susceptible to predation (Sweet 1992). Males advertise or call until females locate the calling males, engage in amplexus, and then lay their eggs (Sweet 1992).

Female arroyo toads release one clutch of 2,000 to 10,000 eggs approximately 3.5 centimeters (1.4 inches) deep at a single site, and are probably unable to produce a second clutch in the same year (Sweet 1992 as cited in Dudek & Associates 2000). Embryos usually hatch 4 to 6 days after fertilization at water temperatures of 12 to 16 ºC (54 to 59 ºF) (Griffin 1999, USFWS 1999). Once eggs hatch into larvae, individuals may take 8 to 14 days to become free-swimming, depending on the water temperature (Sweet 1992, USFWS 1999). Larvae are excellent swimmers and distribute themselves along the shallow bottom of the breeding pool once they are mobile. Arroyo toad larvae generally forage individually, unlike other Californian toads (*e.g.*, *Bufo boreas, B. canorus)*, which generally aggregate (Griffin 1999). Mature larvae swim in short bursts and often remain motionless for 1 to 4 minutes between movements (Sweet 1992, USFWS 1999). Larvae feed on interstitial algae, bacteria, and diatoms by inserting their heads into the substrate and ingesting loose organic material (Dudek & Associates 2000). Arroyo toad larvae do not forage on macroscopic vegetation (Sweet 1992, Jennings and Hayes 1994, USFWS 1999).

Metamorphosis may occur at any time between April and the beginning of September, depending on the time of breeding, weather, and water quality. Metamorphosis occurs after a period of larval growth of at least 65 days (Griffin 1999). The period of time required for metamorphosis to occur can exceed 85 days (Griffin 1999). Peak metamorphosis occurs from the end of June to mid-July in the northern part of the toad's range (Sweet 1992, 1993), and from late-April to mid-May in southern California (USFWS 2000). Under suitable conditions, juvenile arroyo toads remain along the margins of the breeding pools as long as 6 months (Sweet 1992, Atkinson *et al.* 2003).

After approximately 8–9 weeks, juvenile arroyo toads reach a snout-to-vent length (SVL) of 28–30 millimeters (1.1–1.2 in) (USFWS 1999). Once they have reached approximately 30 mm (1.2 in) SVL, juvenile arroyo toads begin to shift their behavior, dispersing away from streamside habitat into nearby willow scrub and sand terrace margins (Sweet 1992 as cited in Dudek & Assoiciates 2000). The timing of dispersal may be delayed until October or November and is affected by local drying conditions and the presence of suitable microhabitat for burrowing (USFWS 1999). Post-metamorphic toads have been observed burrowing at SVL as small as 12 mm (0.5 in); juveniles and adults take refuge underground within the riparian zone and disperse farther away following the dampening of stream terraces by fall and winter rains (Sweet 1992, Griffin 1999). Toads buried in the substrate most likely experience less daily fluctuation in temperature than if they were at the surface, and remain hidden from potential predators (Griffin 1999). In addition, arroyo toads probably lose less water from their bodies when they are below the surface of the soil because substrates below the surface generally contain more moisture than substrates directly on the surface (Griffin 1999). Juveniles favor areas that remain damp and contain less than 10 percent cover, as these sites possess the thermal and refuge characteristics required for juvenile survival and rapid growth (Sweet 1992 as cited in USFWS 1999).

Adult male arroyo toads may reach sexual maturity one year after metamorphosis, while females require at least two years to reach the minimum size for sexual maturity (Griffin 1999). Nocturnal activity is normal for arroyo toad adults and larger juveniles, but they occasionally may be observed during the day (USFWS 1999). Adult arroyo toad movement may cover as much as 0.8 km (0.5 mi) and over 1.0 km (0.6 mi) in a few cases for juveniles and males moving along stream corridors (Sweet 1993 as cited in USFWS 1999). Arroyo toads, like many toads, are essentially terrestrial and adults may move several hundred meters, up to 1.1 km (0.7 mi), from the watercourse where breeding takes place (Griffin 1999). In the central portion of the range (Orange, Riverside, and San Diego counties), arroyo toads may be active all year (USFWS 1999). Activity is generally associated with rainfall and moderate temperatures above 7 ºC (45 ºF) (USFWS 1999). All age classes of post-metamorphic toads may be active on rainy nights and on some nights of very high relative humidity (USFWS 1999).

There are little or no data available on the longevity of arroyo toads, although age-size distributions indicate that many individuals live only about 5 years (Sweet 1992, 1993, both as cited in USFWS 1999).

Habitat Requirements

Arroyo toads are habitat specialists usually found in riparian environments in the middle reaches of third-order streams (Sweet 1989). Adult arroyo toads are primarily located on third- to sixth-order floodplains with highly dynamic fluvial processes, which are necessary for the removal of vegetation and provide suitable, open, riparian habitats (Sandburg 2004). Preferred aquatic habitat includes low stream gradients in coastal sage scrub, oak, and chaparral with persistent water from March to mid-June (Sweet 1992 as cited in Dudek & Associates 2000). Current populations seem to be restricted to the headwaters of large streams that have shallow gravely pools less than 0.46 m (1.5 ft), and adjacent sandy terraces (Dudek & Associates 2000).

Breeding Habitat

Arroyo toads use open sites such as overflow pools, old flood channels, and pools with shallow margins on streams for breeding habitat (Sweet 1992 as cited in USFWS 1999). These habitats rarely have closed canopies over the lower banks of the stream channel because of regular flood events. The importance of these open areas is due to the unicellular algae that generally grow in shallow, sunny pools, which seem to be a principal food source for arroyo toad larvae (Griffin 1999). Heavily shaded pools are generally unsuitable for larval and juvenile arroyo toads because of the lower water and soil temperatures and reduced algal mat development (Sweet 1992 as cited in USFWS 1999). Episodic flooding is critical to keep the low stream terraces relatively free of vegetation, and sand dominant soils friable enough for juvenile and adult toads to create burrows to avoid predation and desiccation (Griffin 1999, Jennings and Hayes 1994, USFWS 1999).

Suitable breeding pools must be shallow (less than 30 cm [12 in] deep), with clear water (Sweet 1992, 1993, both as cited in USFWS 1999), minimal current (less than 5 cm/second [~2 in/sec]), and with a sand or pea gravel substrate overlain with sand or flocculent silt (Sandburg 2004, USFWS 1999, Dudek & Associates 2000). The eggs are laid on open substrates of sand, gravel, cobble, or mud. Eggs are generally located in areas free of vegetation in the shallow margins of the pool (Sweet 1992 as cited in USFWS 1999). Adjacent banks must provide open, sandy or gravely terraces with very little herbaceous cover for adult and juvenile foraging areas, within a moderate riparian canopy of cottonwood, willow, or oak (Dudek & Associates 2000).

Adult and Juvenile Habitat

Areas that are used by juveniles for feeding and shelter consist primarily of sand or fine gravel bars with varying amounts of large gravel or cobble and adjacent stable sandy terraces and oak flats. Arroyo toads prefer sand or fine gravels and tend to avoid cobble because they need places to bury themselves to avoid predation and desiccation (Griffin 1999). Areas that are damp and have some (less than 10 percent) vegetation cover such as American brooklime *(Veronica americana)* are favored by juvenile toads. These habitats usually possess refugia as well as thermal characteristics necessary for juvenile survival and rapid growth (Sweet 1992 as cited in USFWS 1999).

Adults use terraces in the 100-year flood zone, which may extend up to 100 m (328 ft) from the stream (Campbell *et al.* 1996 as cited in USFWS 1999). However, more recent data suggest that they may move between 1 and 2 km (0.6–1.2 mi) into adjacent upland habitats to aestivate during dry periods (Griffin 1999, Dudek & Associates 2000). River drainages with straighter courses generally have broader marginal zones and fewer terraces but may have associated oak flats that provide suitable adult habitat (Campbell *et al.* 1996 as cited in USFWS 1999).

Vegetation on adjacent sandy terraces may be sparsely to heavily vegetated with brush and trees including mulefat *(Baccharis salicifolia*), California sycamore *(Platanus racemosa)*, cottonwoods *(Poputus* spp.), coast live oak *(Quercus agrifolia)*, and willow *(Salix* spp.) (USFWS 1999). Preferred arroyo toad habitat in the under story of stream terraces may consist of scattered short grasses, herbs, and leaf litter, with patches of bare or disturbed soil, or have no vegetation at all (USFWS 1999). Juveniles and adult arroyo toads have been documented around the drip lines of oak trees because these areas lack vegetation, yet have appropriate levels of prey (Sweet 1992 as cited in USFWS 1999). They have also been documented in strawberry, cucumber, and tomato fields, directly under transparent plastic sheeting or next to irrigation hoses on elevated crop ridges (Griffin 1999). Soils in agricultural fields are generally loose and friable, and usually have higher silt contents than those found in the river channel, but still appear to provide suitable foraging and burrowing habitat for adult toads.

Ecological Interactions

Although some research has shown that arroyo toads do not use small mammal burrows in areas where soils are compacted, Griffin (1999) movement study observed four arroyo toads using rodent burrows and burrowing in the piles of loose substrate next to rodent burrows (USFWS 1999).

Juvenile arroyo toads feed almost exclusively on ants (USFWS 1999). By the time juveniles reach 17 to 23 mm (0.7 to 0.9 inches) SVL, they feed on beetles along with the ants (Sweet 1992, USFWS 1999). The diet of adult toads includes a wide variety of insects and arthropods including ants, beetles, spiders, larvae, caterpillars, and others (USFWS 1999).

All life stages of the arroyo toad are susceptible to predation (Sweet 1992 as cited in USFWS 1999). Although not ingested, disturbance and fragmentation of egg strands by mallards *(Anas platyrhynchos)* can reduce hatching rates. Predators of larvae include giant water bugs *(Abedus indentatus),* two-striped and common garter snakes *(Thamnophis hammondii* and *T. sirtalis),* green sunfish, largemouth bass, fathead minnows, and bullfrogs. Griffin (1999) documented a flock of crows foraging at a streamside area where post-metamorphic toads had been observed in high abundance. Predators of juveniles and adults include killdeer *(Charadrius vociferus),* two-striped garter snakes, bullfrogs, green-backed herons *(Butorides striatus),* and great blue herons *(Ardea herodias).* Other potential predators of arroyo toad larvae, juvenile, or adults include black bullheads, prickly sculpins, African clawed frogs, western pond turtles *(Emys (*formerly *Clemmys) marmorata),* raccoons (*Procyon lotor*), opossums (*Didelphis virginiana*), American crows *(Corvus brachyrhynchos),* and common ravens *(C. corax)* (USFWS 1999).

Sensitivity to Anthropogenic Watershed Disturbances

The arroyo toad has evolved in an area where watersheds have dynamic flows, with marked seasonal and annual fluctuations in climatic factors, particularly in rainfall (USFWS 1999). Severe flooding or droughts are two common natural climatic variations with which arroyo toad populations have evolved. Other stochastic events such as fires and earthquakes, coupled with the species' specialized habitat requirements, are likely to lead to natural annual fluctuations in arroyo toad populations (USFWS 1999). Human alterations of habitat also can have unpredictable effects on arroyo toad populations (USFWS 1999).

Locally, toads in Piru Creek are affected by recreational activities such as placer mining and off-highway vehicle use (USFWS 1999). The lower section has been affected by the introduction of Louisiana red-swamp crayfish (*Procambarus clarkii*), bullfrogs, exotic fishes (especially green sunfish, black bullhead, prickly sculpin [*Cottus aspen*], and largemouth bass [*Micropterus salmoides*]), recreational activities in and around campgrounds, flow regulation from Pyramid Lake, and grazing of the riparian zone by livestock (Sweet 1992, USFWS 1999). There are a number of similar issues in Sespe Creek and Castaic Creek that may affect the local arroyo toad population. These impacts include recreational activities such as off-highway vehicles, fishing, camping, stochastic events such as fires and floods, and the spread of introduced aquatic predators such as green sunfish (*Lepomis cyanellus*), black bullheads *(Ameiurus melas)*, and bullfrogs (*Rana catesbeiana*) (USFWS 1999). Arroyo toads in the lower reaches of Castaic Creek are also threatened by water flow regulation and potential urban development of the surrounding hillsides (Campbell *et al.* 1996 as cited in USFWS 1999).

Arroyo toads require habitat near water, and due to construction and development activities (such as flood control structures, dams, roads, agriculture, urbanization, and recreational facilities), many arroyo toad populations have been reduced in size or extirpated due to extensive habitat loss from 1920 to 1980 (USFS 1999). Habitat loss coupled with habitat alteration due to the manipulation of water levels in many central and southern California streams and rivers, as well as predation from introduced aquatic species such as the bullfrog, have extirpated arroyo toads from about 75 percent of the previously occupied habitat in California (Jennings and Hayes 1994). The arroyo toad was listed as an endangered species because of these threats and due to the current limited natural occurrence of arroyo toads. The remaining populations are small and highly susceptible to extinction from naturally occurring events (such as extended droughts or fires) (USFWS 1999).

Key Uncertainties

As with many amphibian species, little is known about the overwintering habitats and threats to adult toads during the non-breeding season (USFWS 1999). Factors influencing survival during this time may include desiccation, starvation, predation by native and non-native species, road mortality, and activities that may disturb non-breeding habitats (Sweet 1992).

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WESTERN POND TURTLE
Clemmys marmorata[*]

Legal Status

Federal	None
State	None
Other	Species of Special Concern

Taxonomy

Clemmys marmorata (also known as *Emys marmorata* or *Actinemys marmorata*) has undergone numerous name changes since the species was first identified in 1852 (Bettelheim 2005, Buskirk 2002, Parham and Feldman 2002, Shaffer *et al.* 1997). Phylogenetic research has variously suggested that western pond turtle may belong to any of three genera: *Emys*, *Actinemys*, and *Clemmys*. Recent molecular phylogenetic work points to the *Emys* genus, based on the evolution of shell kinesis (shell formation and movement) in the western pond turtle (Buskirk 2002, Parham and Feldman 2002). Given that the *Clemmys* is not monophyletic, it is expected that some revision of the genus will probably occur after more extensive research is completed (Spinks *et al.* 2003).

Geographic Distribution

In 1945, two subspecies of western pond turtle were distinguished in California by M. Seeliger: the southwestern pond turtle (*C. m. pallida*) and the northwestern pond turtle (*C. m. marmorata*) (Buskirk 2002). This taxonomy is at odds with results from recent studies of molecular genetics, which suggest that western pond turtle populations fall into four distinct groupings or clades: (1) a northern clade spanning the widest range, from San Luis Obispo County, California, to Washington, and including the northern Central Valley populations (Spinks 2005, Spinks *et al.* 2003, Buskirk 2002, and Shaffer *et al.* 1997); (2) a San Joaquin Valley clade in the southern Central Valley; (3) a geographically restricted clade in a short coastal stretch in Santa Barbara and Ventura counties, California; and (4) a southern clade that ranges from the Tehachapi Mountains to Baja California, including areas to the west of the Transverse Ranges (Spinks 2005). Based on the results of the molecular genetics studies, the genetic variations in the southern California populations of western pond turtle have been described as cryptic, and are now the subject of ongoing research and consideration in conservation planning (Spinks 2005, Germano 2005).

Local Distribution

Based on the California Natural Diversity Database (CDFG 2006), western pond turtles have been observed throughout the lower Santa Clara River watershed. Many records from the CNDDB search were listed as “sensitive” and did not include specific locations. Several known western pond turtle populations occur in the upper Santa Clara River watershed near Santa Clarita and in the vicinity of Piru Creek.

[*] Also known as *Emys marmorata*, or *Actinemys marmorata*

Population Trends

Western pond turtle populations have experienced declines due to extensive conversion of wetland and riparian habitat for urban and agricultural use (Jennings and Hayes 1994, Germano and Bury 2001). The most extensive extirpation of western pond turtle populations may be in southern California and the San Joaquin Valley (USFWS 1992, Germano and Bury 2001). Local population trends in the Santa Clara River watershed are currently unknown, but it is likely that most turtles currently observed in the Santa Clara mainstem are mainly "fossil" populations consisting of old individuals (and immigrants from side drainages like Sespe Creek) (S. Sweet, UC Santa Barbara, pers. comm., 28 March 2006). It is unlikely that there is much reproduction occurring in the mainstem Santa Clara anymore due to a lack of suitable habitat and high densities of raccoons (predators of turtles); however, western pond turtle populations appear to be stable in lower Sespe Creek and in parts of the Piru Creek system (S. Sweet, UC Santa Barbara, pers. comm., 28 March 2006).

Life History

Although primarily an aquatic reptile, the western pond turtle needs terrestrial habitat for basking, overwintering, nesting, and traveling between ephemeral sources of water (Reese 1996).

Breeding activity peaks in June and July, but may occur throughout the year (Holland 1994, Reese 1996). The turtles are philopatric (*i.e.*, they tend to return to a specific location year after year in order to breed or feed), which implies that continuity of nesting habitat from year to year may be an important consideration for conservation. A tendency for clustering of turtle nests has been noted but is poorly understood (Holland 1994). Western pond turtles have low fecundity, laying up to 14 eggs per clutch (Holland 1994, Reese 1996, Stebbins 2003).

The incubation period for eggs averages 80 days, but in some cases may exceed 100 days in California (Bettelheim 2005). Incubating eggs are extremely sensitive to increased soil moisture, which can cause high mortality (Bettelheim 2005, Shaffer 2005, Ashton *et al.* 1997). In wet conditions, eggs can literally explode from internal pressure caused by water absorption (Ashton et al 1997).

In colder climates, hatchlings often overwinter in their nests, emerging in the following spring (Bettelheim 2005). In warmer climates, such as southern and central California, many hatchlings tend to emerge from the nest in the early fall (Bettelheim 2005). Hatchlings spend much of their time in shallow water, within dense vegetation of submergent or short emergent macrophytes (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Hatchling and juvenile survivorship is considered to be low because they are often more prone to predation compared with adults (Holland 1994).

Western pond turtles in California reach sexual maturity in 7 to 11 years. Survivorship for adults is thought to be high (Jennings *et al.* 1992). The turtle has a potentially long lifespan; one recaptured individual is known to have survived at least 42 years in Trinity County (Jennings and Hayes 1994), although 25 years is generally considered to be the approximate upper limit on age for most adults in natural settings (Bury 2005).

Western pond turtles have a widely variable home range, and although they may disperse overland due to environmental stressors such as droughts or floods, most movement is associated with normal movement within a terrestrial home range (Holland 1994, Reese 1996, Bettleheim 2005). In southern California, linear aquatic home ranges can vary between 32 and 4,200 meters (2.6 miles) and total aquatic home ranges can vary between 294 and 7,284 square meters (Goodman and Stewart 2000, Bettleheim 2005).

Habitat Requirements

The western pond turtle inhabits a wide range of fresh or brackish water habitats including ponds, lakes, ditches, perennially filled pools of intermittent streams, and backwater and low-flow areas of perennial streams and rivers (Jennings and Hayes 1994). A key requirement is proximity to potential nesting sites.

Nesting Habitat

Although some general nesting habitat parameters have been quantified, data are sparse. Females tend to build nests between 10 and 12 centimeters (4–5 inches) deep, in dry clayey, loamy, or silty soils (Bettelheim 2005, Ashton *et al.*1997, Reese 1996, Holland 1994), on gentle (<15 percent), south- or west-facing slopes (Holland 1994), at distances ranging from 1.5 to 402 m (5–1,319 feet) (average = 45 m [148 ft]) away from water (Holland and Bury in press, as cited in Spinks *et al.* 2003; Nussbaum *et al.* 1983; Holland 1994; Reese 1996). Nests are generally located in grassy meadows, away from trees and shrubs (Holland 1994); with canopy cover commonly less than about 10 percent (Reese 1996). No data are available on the relative elevations of western pond turtle nests and water levels in adjacent water bodies; these are important habitat parameters that require further research (B. Bury, USGS Forest and Rangeland Ecosystem Science Center, Corvallis, Oregon, pers. comm., 22 March 2005; J. D. Germano, Professor, Department of Biology, CSU Bakersfield, pers. comm., 16 February 2005).

Aquatic Habitat

This turtle species is not an especially strong swimmer. Hence, suitable aquatic habitats generally have standing (lentic) or slow-moving (lotic) water, which typically occur in off-channel areas, such as oxbows and sloughs. Oxbows that are better connected to the hydraulics of the active river are also generally more strongly affected by fluctuations in mainstem flow. However, this kind of variability is probably not an important regulator of oxbow habitat quality for western pond turtle, because juveniles and adults can readily abandon areas that become unsuitable, and travel large distances over land in search of better aquatic habitats (J. D. Germano, pers. comm., 16 February 2005). Overwintering in terrestrial habitats may be an adaptation that helps western pond turtle escape high winter stream flows (Ashton *et al.* 1997).

Western pond turtles, being weak swimmers, can be easily displaced downstream by fast-moving water (Ashton *et al.* 1997). The mainstem channels of large rivers are not generally expected to provide optimal habitat for western pond turtles (Reese and Welsh 1998b; Germano and Bury 2001; B. Bury, pers. comm., 22 March 2005). Observations from the Trinity River (northern California) and other large rivers in the Pacific Northwest appear to support the hypothesis that there is an inverse relationship between river size (measured by stream order) and density of western pond turtles in mainstem habitats (Reese and Welsh 1998b).

Western pond turtles that do occur in mainstem riverine areas are generally concentrated in side channels and backwaters, and generally migrate to off-channel habitats, such as oxbows, during high flows (Holland 1994, Ashton *et al.* 1997). These western pond turtles may also overwinter, generally for 1 to 2 months, but sometimes for up to 6.5 months, in upland areas under leaf litter (Reese 1997, Buskirk 2002, Bettelheim 2005). On the Trinity River, in un-dammed riverine habitat, western pond turtles appear to prefer deep, water, moderate amounts of riparian vegetation, warm water and/or ample basking sites, and large woody debris and rocks (Reese 1996, Reese and Welsh 1997, 1998a, 1998b), which provide underwater cover from predators such as otters and minks.

Canopy cover in both riverine and off-channel habitats is thought to provide western pond turtles with protection from avian predators. However, juveniles and adults have been documented to occur in habitat conditions ranging from unvegetated gravel bars to immature riparian vegetation (*i.e.*, early seral stage willow scrub), to those of mature, late-seral stage riparian vegetation (Reese 1996, Reese and Welsh 1998b).

Habitat for Hatchlings and Juveniles

Whereas adults and older juveniles are considered aquatic habitat generalists, hatchlings and young juveniles require specialized habitat for survival through their first few years. For example, in addition to requiring low-flow and backwater areas of rivers, hatchlings need to spend much of their time feeding in shallow water amongst dense submergent and short emergent vegetation (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Young western pond turtles growth rates are thought to be closely tied to the abundance of food, particularly the concentration of zooplankton in the water column (Jennings and Hayes 1994, Holland 1994). Juveniles prefer habitats similar to adults, but generally with lower water flow (Bettleheim 2005). Often these low-flow habitats are scarce, and may be especially sensitive to anthropogenic and natural disturbances (Jennings *et al.* 1992).

Basking Habitat

Western pond turtles are poikilothermic ("cold-blooded") and generally must spend a portion of each day basking (Jennings and Hayes 1994, Zeiner *et al.* 1988), either on land or in aquatic thermal refugia. Terrestrial basking sites may include rocks, logs, banks, emergent vegetation, root masses, open banks, and tree limbs (Reese 1996; Reese and Welsh 1998,b; Zeiner *et al.* 1988). Emergent woody debris, overhanging vegetation, and rock outcrops associated with deep (>0.5 m [2 ft]), still water provide optimal basking habitat for older western pond turtles life stages (Bury 1972). Terrestrial basking promotes synthesis of vitamin D, controls parasites (Reese 1996), and permits the turtles to thermoregulate, and thus assists digestive processes.

Basking in water can permit western pond turtles to attain body temperatures of up to 30–34ºC (86–93ºF) for several hours per day, even in relatively cold streams (Bury 2005). A comparative study of aquatic basking in the Trinity River system showed that turtles in colder waters seem to spend more time seeking aquatic thermal refugia and basking than turtles in warmer waters (Bettaso *et al.* 2005).

Observations from the Russian River (Sonoma County, California) indicate that typical basking sites have water depths of 1 to 2 m [3 to 7 ft] (average = 1.45 m [5 ft]) and include some overhead riparian canopy cover, with more than 64 percent of observed sites having canopy cover of 20 percent or greater (Cook and Martini-Lamb 2005). Basking western pond turtles on the Russian River seem to prefer live, downed trees or new snags over older, worn or decomposed snags (Cook and Martini-Lamb 2005). Juveniles and adults of both sexes (Bettelheim 2005) have been known to compete aggressively for basking sites (Nussbaum *et al.* 1983) by biting, pushing, and making open-mouthed threats (Bettelheim 2005).

Warm summer air temperatures in the Santa Clara River watershed may make thermoregulation activities of western pond turtles less important than they are elsewhere, in colder environments (Germano and Bury 2001). Moreover, western pond turtles in the Santa Clara River may be able to reach suitable body temperatures by basking in beds of aquatic vegetation (*e.g.*, algae or vascular aquatic macrophytes). Dense beds of aquatic macrophytes can create thermal stratification, with warmer water in vegetated areas near the surface (Collins et al 1985). Hence, by remaining within warm stratification layers western pond turtles can meet thermoregulation requirements while remaining in cover that reduces risk of predation. Germano and Bury (2001) observed a significant number of turtles in Dry Creek (Fresno County) using algal mats for thermoregulation.

Temperature Requirements

Temperatures preferences and requirements of western pond turtles are not well understood. Adults do not seem to allow body temperatures to exceed 34 ºC (93 ºF) (Lovich 1999), and also seem to avoid water temperatures greater than 39–40 ºC (102–104 ºF) (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Data from the Trinity River indicate that juveniles tend to prefer water temperatures of 12–33 °C (54–91 ºF), whereas adults prefer water temperatures of 10–17 °C (50–63 ºF) (Ashton *et al.* 1997). Water temperature appears to have a strong effect on activity levels of western pond turtles, with notably higher activity in water temperatures that consistently exceed 15 ºC (59 ºF) (Jennings and Hayes 1994). Downstream of dams with hypolimnetic low-level summer flow releases, temperatures are generally much cooler than they would be under natural conditions, and may result in unnaturally slow western pond turtles growth rates, thus affecting body size and age at maturity (Reese 1997, Reese and Welsh 1998a). A case in point comes from studies on the Trinity River, where water temperatures in the regulated mainstem are more than 10ºC colder than they are on the unregulated South Fork Trinity River (Ashton 2005), and where turtles in colder reaches appear to be stunted, reaching sexual maturity at an unnaturally early age (Ashton 2005). Data from a separate series of studies on the Trinity River suggest that, in regions with cold winters, western pond turtles generally take refuge from the main river from October or November until April or even later (Reese 1996, Reese and Welsh 1998a), with a majority seeking terrestrial overwintering sites and a smaller fraction choosing lentic aquatic sites (Reese and Welsh 1998a, Reese 1996, Holland 1994).

Ecological Interactions

The introduction of non-native species can be detrimental to native species assemblages. Of particular concern are non-native red-ear slider turtles (*Trachemys scripta elegans*), which have been introduced throughout California, largely as a result of escape or release from pet owners (Bettelheim 2005). Red-ear sliders may compete directly with native western pond turtles for basking habitat, food, and nesting habitat (Spinks 2003, Reese 1996, and Holland 1994). Studies completed by Spinks *et al.* (2003) observed significant reduction in western pond turtle use of optimal habitat when red-ear sliders were present. Moreover, red-ear sliders are a vector for an unidentified upper respiratory disease, which can be fatal for western pond turtles (Holland 1994). Both turtle species favor lentic waters and have similar diets, and share the need for aerial basking (warming body temperature out of the water) as a component of metabolism (Campbell 2004). Other species may have more indirect effects on western pond turtle habitat and food resources. Introduced centrarchid fish may compete with hatchlings for zooplankton and other invertebrate prey items.

In general, it is expected that western pond turtle populations may be distributed according to pressure from predators, as well as physical habitat conditions. A number of species may prey on one or more life stages. Raccoons are an important predator and can prey on western pond turtles during all life stages, and have been observed in higher densities in areas where western pond turtle populations were found (Reese 1996, Holland 1994, Germano 2005, J. D. Germano, pers. comm., 16 February 2005). In the San

Simeon area of coastal California, in which fewer western pond turtles were observed when raccoon numbers were high. Other possible non-native predators include largemouth bass and bullfrogs, which would be expected to target hatchlings in particular, and are found in increasing numbers throughout California (Bettelheim 2005). Bullfrogs have been observed feeding on both hatchlings and juveniles (Holland 1994, Moyle 1973).

Sensitivity to Anthropogenic Watershed Disturbances

Reduction in nesting habitat quality may increase the risk of nest failure for a number of reasons. Semi-suitable nesting habitat may exist in agricultural areas adjacent to river corridors. Western pond turtles may utilize this habitat to nest, which may increase the chance of egg mortality from inundation (via irrigation) or predation. Abnormally high raccoon populations (which have been linked with habitat fragmentation, supplemental feeding from garbage, and increased edge habitat) may severely limit western pond turtle recruitment if raccoons encounter western pond turtle nesting areas (Holland 1994, D. C. Holland, PhD, Wildlife Diversity Program, Oregon Department of Fish and Wildlife, Portland, Oregon, pers. comm., 15 February 2005).

Key Uncertainties

- The relative elevations of western pond turtle nests and water levels in adjacent water bodies
- Competition for nesting habitat, food, and basking habitat between slider turtles (*Trachemys scripta*) and western pond turtles
- Predation by raccoons and bullfrogs, and the influences of anthropogenic disturbances on predator populations
- Western pond turtle dispersal and terrestrial habitat use in the Santa Clara River
- Connectivity of western pond turtle aquatic habitat and nesting/hatchling habitat in the Santa Clara River
- Microhabitat preferences at nesting locations of western pond turtles (*e.g.,* vegetation effects on soil temperatures, soil water potential, etc.)

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LEAST BELL'S VIREO*
Vireo bellii pusillus

Legal Status

Federal	Endangered
State	Endangered
Other	None

Geographic Distribution

The historical distribution of least Bell's vireo ranged from central northern California through the Sacramento and San Joaquin valleys and Sierra Nevada foothills, and from the southern Coast Ranges (including the Santa Clara River watershed) to Baja California, Mexico (Wilbur 1979, 1980; Kus 2002; USWFS 1998). Historical populations were also documented in Owens Valley, Death Valley, and scattered locations in the Mojave Desert (USFWS 1998, Kus 2002).

Today, the breeding range of least Bell's vireo is limited primarily from Santa Barbara County south to San Diego County (where the majority of remaining populations occur) (Franzreb 1989 as cited in Labinger and Greaves 2001a, Kus 2002). Breeding pairs have also been sighted near Gilroy (Santa Clara County) (Roberson *et al.* 1997, as cited in Kus 2002) and along the Santa Clara River (Ventura County) (Labinger and Greaves 2001a), Mojave River (San Bernadino County) (Kus and Beck 1998, as cited in Kus 2002), and San Joaquin River (San Joaquin County) (River Partners 2005).

Critical habitat for the species has been designated in Santa Barbara, Ventura, Los Angeles, San Bernardino, Riverside, and San Diego counties (USFWS 1992). Critical habitat patches occur on the Santa Ynez, Santa Clara, Santa Margarita, San Luis Rey, Sweetwater, San Diego, and Tijuana rivers (USFWS 1992).

Local Distribution

A number of historical records document the presence of least Bell's vireo near and in the Santa Clara River watershed. One museum record confirms the presence of a nesting pair of least Bell's vireo in Foster Memorial Park on the Ventura River in 1911 (CDFG 2005). Two least Bell's vireo observations were reported in 1980 and 1988: one 5.6 km (3.5 mi) east of Piru and one at Newhall Ranch (CDFG 2005).

In 1990–1991, three separate records of least Bell's vireo territorial males and nesting pairs were reported near Saticoy and southwest of Santa Paula (CDFG 2005). Multiple active least Bell's vireo nests have been observed on the Santa Clara River at the Fillmore Fish Hatchery since 1991 (Labinger and Greaves 2001a, CDFG 2005). Greaves and Labinger (1997) report capturing and banding 266 least Bell's vireo individuals between 1991 and 1996 in the lower half of the Santa Clara River between I-5 downstream to the Highway 118 bridge near Saticoy.

* The least Bell's vireo, *Vireo bellii pusillus*, is one of four subspecies of Bell's vireo recognized in North America (Brown 1993).

More recently, Labinger and Greaves (2001a) reported that least Bell's vireo was the most abundant and widely distributed endangered bird species within the lower Santa Clara River area. Between 1994 and 1999, they found 81 nesting pairs in the lower Santa Clara River, including nine pairs between the McBean Parkway bridge (Santa Clarita, Los Angeles County) and Las Brisas (Ventura County), 25 pairs from the Fillmore Fish Hatchery downstream to the Highway 23 bridge, three pairs from just downstream of the confluence with Sespe Creek to 2 km (1.2 mi) east of Santa Paula, 38 pairs along a 5 km (3.1 mi) segment upstream of the Highway 118 bridge (including the Freeman Diversion), and six pairs between the Highway 118 bridge and the Victoria Avenue bridge (downstream of Highway 101, Ventura) (Labinger and Greaves 2001a). Breeding pairs were found at many of these same locations in 2000 (Labinger and Greaves 2001b). In 2003, there was a record of at least one nesting pair on the Santa Clara River south of the Highway 101 bridge in 2003 (CDFG 2005).

The USFWS has designated critical habitat for the species along the Santa Clara River between Ventura and Los Angeles counties, including "all land within 3,500 feet perpendicularly and generally southward or westward" of State Highway 126 between approximately Piru and Interstate 5 near Castaic Junction (USFWS 1994).

Population Trends

Least Bell's vireo is reported to have been common to abundant in its historical range before undergoing a sharp decline in abundance and range during the first half of the 20th century (USFWS 1998, Labinger and Greaves 2001a, Kus 2002). USFWS (1998) report that:

> By 1986, the population [of least Bell's vireo] had declined to an estimated 300 pairs, with the majority occurring in San Diego County. Restoration efforts and brown-headed cowbird control have allowed populations to increase in recent years. In 1998, the population size was estimated at 2,000.

Based on population monitoring and compiled data covering 1991 to 2000, Labinger and Greaves (2001a and 2001b) characterize the Santa Clara River population as relatively stable to slightly increasing and with increasing distribution along the river corridor. Between 1994 and 1999, Labinger and Greaves (2001a) documented 124 least Bell's vireo territories and a doubling of the vireo population from 25 to 57 pairs in their original cumulative study area of 29 km (18 mi). In 1999, after expanding their study area to include an additional 14 km (9 mi), Labinger and Greaves (2001a) documented a total of 163 territories and 80 pairs. In 2000, they the documented 81 pairs in their expanded study area (Labinger and Greaves 2001b). Count data from 18 points in the lower 50 km (31 m) of the river reveal that the mean relative abundance of least Bell's vireo steadily increased from 0.28 in 1994 to 1.25 in 1998, before decreasing to 0.75 in 1999 (Labinger and Greaves 2001a). In an upper reach of the river that had been affected by an oil spill, the mean relative abundance of least Bell's vireo decreased from 0.1 in 1994 to 0.03 in 1995 and then steadily returned to 0.1 in 1998 and 1999 (Labinger and Greaves 2001a).

Life History and Timing

The draft recovery plan for least Bell's vireo (USFWS 1998) describes the species as:

> …a sub-tropical migrant, traveling some 2,000 miles annually between breeding and wintering grounds. Preliminary results of studies of color-banded birds indicate that least Bell's vireos have a life span ranging to seven years.

Least Bell's vireos generally arrive in California from mid- to late-March for a breeding season that typically ends in late September (USFWS 1986, Kus 2002). During this period they are known to breed almost exclusively within riparian habitats (USFWS 1998). Least Bell's vireos have been documented to return to the same breeding site year after year (Greaves 1989).

According to the U.S. Fish and Wildlife Service (USFWS) (1998):

> Males establish and defend territories through counter-singing, chase and sometimes physical combat with neighboring males. Territory size ranges from .20 ha to 3.03 ha (0.5 to 7.5 acres). … Newman (1992) investigated the relationship between territory size, vegetation characteristics, and reproductive success for populations of vireos at the San Diego and Sweetwater Rivers, but found no significant factors which could account for the variability in territory size observed at his sites.

Nests are built by both breeding pair members within a few days of pair formation, and generally take between four and five days to complete (Kus 2002). Typically three to four eggs are laid beginning a day or two after nest completion (Kus 2002). Both male and female share in egg incubation, although females incubate more than males during the day and appear to be the exclusive incubator at night (USFWS 1998 Kus 2002). Incubation lasts about 14 days, and nestlings fledge 10–12 days after hatching (USFWS 1998, Kus 2002).

The fledgling stage is described in USFWS (1998):

> Least Bell's vireo may attempt as many as five nests in a breeding season (B. Kus, pers. comm.), although most fledge young from only one or two nests. … Adults continue to care for the young for at least two weeks after fledging when territorial boundaries may be relaxed as family groups range over larger areas. Fledglings generally remain in the territory or its vicinity for most of the season, although the behavior of older fledglings produced early in the year has not been well studied.

Habitat Requirements and Associated Vegetation

Territory Habitat

Least Bell's vireos primarily occupy riparian habitats along open water or dry parts of intermittent streams, generally below 460 m (1,500 ft) in elevation (USFWS 1986; Small 1994 as cited in Dudek and Associates 2005, Kus 2002). They are generally associated with the following vegetation types: southern willow scrub; cottonwood forest; mule fat scrub; sycamore alluvial woodland; coast live oak riparian forest; arroyo willow riparian forest; wild blackberry; and mesquite in desert localities (Kus 2002).

Most vireo territories contain both dense vegetative cover within 1–2 meters of the ground, the preferred habitat for nesting, and a dense, stratified overstory canopy, the preferred habitat for foraging (Goldwasser 1981, USFWS 1998, Labinger and Greaves 2001a). In the Santa Clara River watershed, Labinger and Greaves (2001) documented least Bell's vireo territories in early successional cottonwood/willow forest, southwestern willow woodland, and mulefat scrub. While vegetative structure was found to be more important in territory selection than the presence of particular plant species, willow trees and shrubs were found to be the most common plant species in the vicinity of vireo territories and the preferred species for nest placement (Labinger and Greaves 2001a).

Least Bell's vireos have been observed to maintain territories that include upland habitats adjacent to riparian areas, such as coastal sage scrub (USFWS 1998). Upland habitats have also been documented for foraging and for nesting when early spring floods inundate riparian areas (Kus and Miner 1989, USFWS 1998). It has also been hypothesized that berry-producing upland vegetation, such as laurel sumac (*Malosma laurina*) and elderberry (*Sambucus mexicana*), may supplement the vireo diet in marginal habitats (Kus and Miner 1989).

Nesting Habitat

Least Bell's vireo primarily nests in small remnant segments of vegetation typically dominated by willows (*Salix* spp.) and mulefat (*Baccharis salicifolia*) but may also use a variety of shrubs, trees, and vines (Olsen and Gray 1989). Nests are typically built within one meter (3.3 ft) of the ground in the fork of understory vegetation (Franzreb 1989 as cited in Kus 2002). Cover surrounding nests is moderately open midstory with an overstory of willow, cottonwood (*Populus* sp.), sycamore (*Platanus* sp.), or oak (*Quercus* sp.). Crown cover is usually more than 50 percent and contains occasional small openings. On the Santa Clara River, Labinger and Greaves (2001a) found that the dominant plant species used for nest support (in 57 percent of observed nests) were willows (*Salix lasiolepis*, *S. exigua*, *S. laevigata*, and *S. lasiandra*), followed by mulefat (28 percent of nests). The remaining 11 percent of nests observed were scattered throughout a variety of tree, shrub, and forb species, including poison oak (*Toxicodendron diversilobum*), white alder (*Alnus rhombifolia*), mugwort (*Artemisia douglasiana*), and Fremont cottonwood (*Populus fremontii*). Vireo's were also found nesting in two invasive, non-native plants: four percent of nests observed were found in giant reed (*Arundo donax*) stands and two of the 426 nests observed were found in tamarisk (*Tamarix* sp.) plants (Labinger and Greaves 2001a).

Foraging Habitat

Kuss (2002) indicates that the vireo typically forages in riparian and adjoining upland habitat. Grinnell and Miller (1944) indicated that foraging occurs at all levels of the canopy, but appears to be concentrated in the lower to mid-strata, particularly when pairs have active nests. Salata (1983) found that 69 percent of 131 foraging observations were within 4 meters (12 feet) of the ground. Miner (1989) found a similar peak in foraging activity in vegetation between 3–6 meters (9–18 feet) in height. Moreover, she determined that the distribution of vireo foraging time across all heights was not simply a function of the availability of vegetation at those heights, but rather represented an actual preference for the 3–6 meter zone (Miner 1989).

Ecological Interactions

Least Bell's vireos are insectivores, preying on a wide variety of insect types including bugs, beetles, grasshoppers, moths, and particularly caterpillars (Chapin 1925; Bent 1950). It is likely that vireos do not require water for drinking (Kus 2002). They obtain prey primarily by foliage gleaning (picking prey from leaf or bark substrates), and hovering (removing prey from vegetation surfaces while fluttering in the air). Both Salata (1983) and Miner (1989) observed vireos occasionally capturing prey by hawking (pursuit and capture of flying prey), and Miner (1989) noted a behavior she called "clinging," which she described as hovering, but with the feet in contact with the vegetation.

The invasion of exotic plant species into riparian habitats increases habitat fragmentation and can decrease suitable vireo nesting habitat in some cases. Invasive non-native plants found in current least Bell's vireo habitat include castor bean (*Ricinus communis*), cocklebur (*Xanthium strumarium*), tamarisk, and giant reed (USFWS 1998). Giant reed is of prime concern due to its ability to disperse throughout an entire drainage and its rapid growth rate, which allows it to outcompete and restrict the growth of other native riparian plants (Kus 2002). When natural riparian vegetation types, such as the structurally diverse native riparian scrub and mature forest communities required by the vireo, are replaced by thick stands of giant reed, bird species abundance and other native wildlife have been found to decline (Bell 1994, Bell 1997, Herrera and Dudley 2003, Kisner 2004). Labinger and Greaves (2001a) observed over the course of their study (1994–1999) that while dense thickets of giant reed supported very low bird diversity:

> … a low to moderate mixture of giant reed with native willow woodland supported high bird diversity in some areas [such as near the Freeman Diversion]…. In such areas giant reed was also used for nesting, as noted by at least 17 nests of least Bell's vireo, one nest of southwestern willow flycatcher, and several other species such as Anna's hummingbird, bushtit, and common yellowthroat.

The ecological interaction of most concern to least Bell's vireo populations is brood parasitism by brown-headed cowbirds (*Molothrus ater*). Dudek and Associates (2000) provide a description of brood parasitism:

> Cowbirds lay their eggs in the nests of other songbirds. The cowbird often removes a number of the host's eggs and replaces them with an equal number of cowbird eggs. Cowbird eggs require a relatively short incubation period, thus the young cowbird hatches earlier than the host's eggs. The effects of parasitism include reducing nest success rate and egg-to-fledgling rate and delaying successful fledgling. A common response to parasitism is abandonment of the nest. The success rate of re-nesting is often reduced and there may be inadequate time to prepare to migration. In California, parasitism rates range from 50 percent to 80 percent, considered to be a high parasitism rate.

USFWS (1998) describe least Bell's vireo as a common host species that readily accepts cowbird eggs. In the Santa Clara River watershed, Labinger and Greaves (2001a) documented cowbird parasitism of vireo nests and suggest that such parasitism may be limiting productivity of host species. Labinger and Greaves' (2001a) study "…did not find a significant correlation between cowbird abundance and vireo productivity. Parasitism rates of subpopulations of least Bell's vireos were never more than 20 percent and typically less than 10 percent." Removal of cowbird eggs and chicks from vireo nests during nest monitoring has been cited to enhance vireo productivity by up to 44 percent in some studies (USFWS 1998).

Sensitivity to Anthropogenic Watershed Disturbances

Least Bell's vireo is sensitive to the direct loss and degradation of habitat and increased rates of cowbird parasitism that result from or are exacerbated by urbanization and other development within and near riparian areas. Urbanization and agriculture, including runoff from agricultural fields and roadways, livestock grazing, water diversion projects, traffic noise, feral pets, and recreational use of habitat can result in the direct loss of vireo habitat, and degrade and fragment habitat to the extent that it is no longer usable or increases the vulnerability of the population. Least Bell's vireo often nest near recreational open spaces or trails. Nest failure and abandonment can be caused by human disturbance such as trampling of nests or nest sites or clearing of vegetation (USFWS 1998).

Habitat fragmentation is thought to be one of the primary factors responsible for vireo population decline, has been attributed to development within riparian areas and the establishment and spread of non-native plant species. Habitat fragmentation results in smaller populations spread out among remaining suitable patches. These smaller, more isolated populations then become more vulnerable to habitat destruction (through flooding or development, for example), disease, low production years, and parasitism (USFWS 1998, Labinger and Greaves 2001a).

The abundance of brown-headed cowbirds is believed to increase in areas with development near riparian areas (USFWS 1998). Brood parasitism by brown-headed cowbirds is the other primary factor, in addition to habitat fragmentation, responsible for the decline of least Bell's vireo (Kus 2002, Labinger and Greaves 2001a).

Key Uncertainties

The following key uncertainties and information gaps regarding least Bell's vireo were identified by Kus (2002):

- Whether any reproductive parameters are density-dependent.
- Whether dispersal is density-dependent.
- The effect of different cowbird control regimes on vireo parasitism rates and reproductive success.
- The use of restored habitat by vireos.
- The status of wintering habitat and identification of current or potential threats.
- Identification of predators and establishing means of control.
- Identification of additional and potential least Bell's vireo breeding habitat within its historical range.

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SOUTHWESTERN WILLOW FLYCATCHER
Empidonax traillii extimus

Legal Status

Federal	Endangered
State	Endangered
Other	None

Geographic Distribution

The southwestern willow flycatcher is the southern most occurring subspecies of willow flycatcher, which historically nested in riparian deciduous shrub vegetation throughout California primarily between 30 m (100 ft) and 1,830 m (6,000 ft) in elevation (Grinnel and Miller 1944 as cited in USFWS 2002). Southwestern willow flycatchers occurred in southern California, with a historical distribution ranging from Inyo County, through Kern, Santa Barbara, Los Angeles, and San Bernardino counties, to Riverside County (Unitt 1987, as cited USFWS 2002). Within these counties, populations were documented along the South Fork Kern River, the San Fernando Valley, Colorado River, Santa Ana River, Mohave River, and in swampy thickets around Los Angeles (Belding 1890 as cited in Williams and Craig 1998). Southwestern willow flycatchers are also believed to have occurred in Arizona, Colorado, Utah, Nevada, New Mexico, southwestern Texas, northern Sonora, Mexico, and Baja California Norte (Marshall 2000, USFWS 2002).

The final recovery plan for southwestern willow flycatcher (USFWS 2002) summarizes the California river systems where the species is known to persist: Colorado, Owens, Kern, Mojave, Santa Ana, Pilgrim Creek, Santa Margarita, San Luis Rey, San Diego, San Mateo C reek, San Timoteo Creek, Santa Clara, Santa Ynez, Sweetwater, San Dieguito, and Temecula Creek.

In addition to southern California, breeding southwestern willow flycatchers have also been recently recorded in extreme southern Nevada, southern Utah and Colorado, Arizona and New Mexico (McKernan 1997, Marshall 2000, USFWS 2002, Sogge *et al.* 2003)

Critical habitat for the southwestern willow flycatcher has been designated along streams segments in southern California, southern Nevada, southwestern Utah, Arizona, and New Mexico (USFWS 2005b). In California critical habitat has been designated specifically along segments of the Santa Ynez, Santa Ana, Santa Margarita, San Luis Rey, Owens, South Fork Kern, and Mojave rivers, as well as Holcomb, Deep, and San Felipe creeks (USFWS 2005b). Critical habitat was not designated within the Santa Clara River because the watershed did not meet the population size or habitat connectivity criteria established by the U.S. Fish and Wildlife Service (2005b).

Local Distribution

Between 1990 and 2002, southwestern willow flycatcher was recorded in scattered locations along the Santa Clara River during the breeding season (CDFG 2005). Several pairs nested in the vicinity of Balcom Canyon near Santa Paula in 1992, and a juvenile was sighted in Balcom Canyon in 1993 (CDFG 2005).

Labinger and Greaves (2001a) documented four territorial southwestern willow flycatchers during their 1994-1999 avian study on the Santa Clara River, and reported that:

> Southwestern willow flychatchers were found throughout the study period in several locations. However, the majority of these sightings were in late May and early June when willow flycatchers of several races are migrating in concentrated numbers. No breeding was observed during this study (the first successful breeding did occur in 2000 at the Fillmore Fish Hatchery, unpublished data), but several individuals held territories for extended periods of time.

During monitoring in 2000, Labinger and Greaves (2001b) documented three territorial southwestern willow flycatchers and reported:

> One individual was observed singing on several visits to each of the territories in the reference sites. …Four willow flycatchers were present at or in the vicinity of the Fillmore Fish Hatchery on more than one occasion at each location. In one area, on the state hatchery property, a pair of flycatcher bred.

Population Trends

Historical accounts suggest that willow flycatchers were once abundant in the inland valleys and coastal regions of central and northern California (Bombay *et al.* 2000). In the last five to six decades, however, southwestern willow flycatchers have been eliminated from most of the lower elevation habitat in California (Unitt 1987, Marshall 2000, Sogge *et al.* 2003). U.S. Fish and Wildlife Service (1997) describes the population decline of southwestern willow flycatcher:

> In a review of historical and contemporary records of *Empidonax traillii extimus* throughout its range, Unitt (1987) noted that the species has "declined precipitously" and "the population is clearly much smaller now than 50 years ago." He believed the total was "well under" 1,000 pairs, more likely 500.

General population trends are illustrated by declines from 1989 to 1992 at the South Fork Kern River, where the largest remaining population in California is located (Whitfield *et al.* 1997). Brown-headed cowbird (*Molothrus ater*), a brood parasite, control programs are credited with stabilizing or increasing southwestern willow flycatcher populations at this location, as well as at Camp Pendleton and the upper San Luis Rey River (Griffith Wildlife Biology 1995, Whitfield *et al.* 1997).

Marshal (2000) estimated the total population of southwestern willow flycatchers to be 549 territories with 935 individuals range-wide and 201 individuals in California. More recently, survey data from 1993 to 2001 indicated 986 territories range wide and 256 territories in California (Sogge *et al.* 2003). U.S. Geological Survey reports 1,256 total territories range wide and 200 territories in California (Durst *et al.* 2005). Sogge *et al.* (2003) warn that the reported "increase in territories should not be interpreted as a southwestern willow flycatcher population increase. Rather, it is mostly a function of increased survey effort over time."

In the Santa Clara River watershed, southwestern willow flycatcher populations can be described as generally increasing to stable, based on reports of four and three territorial individuals between 1994 and

2000 (Labinger and Greaves 2001a and 2001b), 12 territories between 1993 and 2001 (Sogge *et al.* 2003), and 10 territories in 2004 (UWCD 2004, Durst *et al.* 2005).

Life History and Timing

Southwestern willow flycatchers typically arrive at breeding grounds between early May and early June. (Sogge *et al.* 1997, USFWS 2002). As a neotropical migrant, the subspecies spends only three to four months at their breeding grounds. The remainder of the year is spent on migration and in wintering areas south of the United States (USFWS 2002). Although most southwestern willow flycatchers return to former breeding areas, they have also been observed to move among sites within and between years, particularly as a result of natural or human-induced habitat loss (Netter *et al.* 1998, Kenwood and Paxton 2001, M. W hitfield unpubl. data, all as cited in USFWS 2002).

Males are usually monogamous, but annual polygamy rates of approximately 10–15 percent have been recorded at the Kern River Preserve in California (Williams and Craig 1998). Breeding males sing to advertise their territory to prospective mates and other nearby males. Males sing from a series of song perches throughout their territory, usually from tall perches but sometimes from within dense vegetation (Finch and Stoleson 2000).

Nest building usually begins within a week of pair formation. USFWS (2002) reports that "females build the nest over a period of four to seven days, with little or no assistance from the male." Egg laying generally begins in early to mid-June On the South Fork Kern River, and full clutches ranged from two to four (Whitfield and Enos 1996, Whitfield *et al.* 1997, both as cited in Williams and Craig 1998). The southwestern willow flycatcher recovery plan (USFWS 2002) describes incubation and fledgling stages:

> Incubation begins after the last egg is laid, and lasts 12 to 13 days. Most incubation is by the female, although male incubation is also known (Gorski 1969, H. Yard, B. Brown, and Arizona Game and Fish Department unpubl. data). Most eggs in a nest hatch within 48 hours of each other (McCabe 1991). …The female provides most of the initial care of the young. As demand for food increases with nestling growth, the male also brings food to the nest. Generally, only the female broods the young. …Nestlings fledge 12 to 15 days after hatching. …Fledglings stay close to the nest and each other for 3 to 5 days, and may repeatedly return to and leave the nest during this period (Spencer *et al.* 1996). Fledglings typically stay in the general nest area a minimum of 14 to 15 days after fledging, possibly much longer

Chicks can be present in nests from mid-June through early August. Young typically fledge from nests from late June through mid-August (Finch and Stoleson 2000). At the South Fork Kern River, of the 58 nesting pairs studied in 1996 and 1997, 34 pairs successfully fledged young for a 59 percent success rate (Williams and Craig 1998). Adults depart from breeding territories as early as mid-August, but may stay until mid-September if they fledged young late in the season (Finch and Stoleson 2000).

Willow flycatchers forage by either aerially gleaning (capturing an insect from a substrate while hovering) from trees, shrubs, and herbaceous vegetation or hawking larger insects by waiting on exposed forage perches and capturing insects in flight (Williams and Craig 1998). Hawking appears to be more common than aerial gleaning in mountain meadows and the opposite appears to be the case in lowland riparian areas (Sanders and Flett 1989 as cited in Williams and Craig 1998).

Habitat Requirements and Associated Vegetation

Critical habitat for southwestern willow flycatcher includes riparian areas within the 100-year flood plain or flood prone areas, where dense vegetation is present or expected to become established in the future through successional processes (USFWS 2005b). Sogge *et al.* (1997) report that water is almost always present at southwestern willow flycatcher territories, particularly at the beginning of the breeding season. Territory sizes have been reported to range from 0.06 to 1.5 ha (1.5 to 5.0 acres), with generally larger ranges for polygamous males (Williams and Craig 1998).

Based on a summary of available literature, USFWS (2005a) described suitable habitat for the species as typically consisting of the following six biological and physical habitat features:

1. Nesting habitat with trees and shrubs that include, but are not limited to, willow (*Salix*) species and boxelder (*Acer negundo*).
2. Nesting habitat that contains a dense (*i.e.*, 50 to 100 percent) tree and/or shrub canopy.
3. Dense riparian vegetation with thickets of trees and shrubs ranging in height from 2 to 30m (6 to 98 feet), with lower-stature thickets from 2 to 4 m (6 to 13 feet) tall found at higher elevation riparian forests, and tall-stature thickets found at middle- and lower-elevation riparian forests.
4. Areas of dense riparian foliage from ground level to approximately 4 m (13 ft), or dense foliage only at the shrub level, or as a low, dense tree canopy.
5. Dense patches of riparian forests that are interspersed with small areas of open water or marsh or shorter/sparser vegetation, which creates a mosaic that is not uniformly dense; patch size may be as small as 0.1 hectare (ha) (0.25 acre) or as large as 70 ha (175 acres).
6. A variety of insect prey populations, including but not limited to, wasps and bees (Hymenoptera), flies (Diptera), beetles (Coleoptera), butterflies/moths and caterpillars (Lepidoptera), and spittlebugs (Homoptera).

USFWS (2002) summarize the breeding habitat of southwestern willow flycatcher:

> The flycatcher breeds in different types of dense riparian habitats, across a large elevational and geographic area. Although other willow flycatcher subspecies in cooler, less arid regions may breed more commonly in shrubby habitats away from water (McCabe 1991), the southwestern willow flycatcher usually breeds in patchy to dense riparian habitats along streams or other wetlands, near or adjacent to surface water or underlain by saturated soil. Common tree and shrub species comprising nesting habitat include willows (*Salix* spp.), seepwillow (aka mulefat; *Baccharis* spp.), boxelder (*Acer negundo*), stinging nettle (*Urtica* spp.), blackberry (*Rubus* spp.), cottonwood (*Populus* spp.), arrowweed (*Tessaria sericea*), tamarisk (aka saltcedar; *Tamarix ramosissima*), and Russian olive (*Eleagnus angustifolia*) (Grinnell and Miller 1944, Phillips *et al.* 1964, Hubbard 1987, Whitfield 1990, Brown and Trosset 1989, Brown 1991, Sogge *et al.* 1993, Muiznieks *et al.* 1994, Maynard 1995, Cooper 1996, Skaggs 1996, Cooper 1997, McKernan and Braden 1998, Stoleson and Finch 1999, Paradzick *et al.* 1999). Habitat characteristics such as plant species composition, size and shape of habitat patch, canopy structure, vegetation height, and vegetation density vary across the subspecies' range. However, general unifying characteristics of flycatcher habitat can be identified. Regardless of the plant species composition or height, occupied sites usually consist of dense vegetation in the patch interior, or an aggregate of dense patches interspersed with openings. In most cases this dense vegetation occurs within the first 3 - 4 m (10-13 ft) above ground. These dense patches are often interspersed with small openings, open water, or shorter/sparser vegetation, creating a mosaic that is not uniformly dense. In almost

> all cases, slow-moving or still surface water and/or saturated soil are present at or near breeding sites during wet or non-drought years.

Nesting site habitat preferences are further described by USFWS (2002):

> Thickets of trees and shrubs used for nesting range in height from 2 to 30 m (6 to 98 ft). Nest sites typically have dense foliage from the ground level up to approximately 4 m (13 ft) above ground, although dense foliage may exist only at the shrub level, or as a low dense canopy. Nest sites typically have a dense canopy, but nests may be placed in a tree at the edge of a habitat patch, with sparse canopy overhead. The diversity of nest site plant species may be low (*e.g.*, monocultures of willow or tamarisk) or comparatively high. Nest site vegetation may be even- or uneven-aged, but is usually dense (Brown 1988, Whitfield 1990, Muiznieks *et al.* 1994, McCarthey *et al.* 1998, Sogge *et al.* 1997, Stoleson and Finch 1999). Historically, the southwestern willow flycatcher nested in native vegetation such as willows, buttonbush, boxelder, and *Baccharis*, sometimes with a scattered overstory of cottonwood (Grinnell and Miller 1944, Phillips 1948, Unitt 1987). Following modern changes in riparian plant communities, the flycatcher still nests in native vegetation where available, but also nests in thickets dominated by the non-native tamarisk and Russian olive and in habitats where native and non-native trees and shrubs are present in essentially even mixtures (Hubbard 1987, Brown 1988, Sogge *et al.* 1993, Muiznieks *et al.* 1994, Maynard 1995, Sferra *et al.* 1997, Sogge *et al.* 1997, Paradzick *et al.* 1999).

Ecological Interactions

Food analysis from southwestern willow flycathers has revealed a diet consisting of a variety of insects including, but not limited to, wasps and bees (Hymenoptera), flies (Diptera), beetles (Coleoptera), butterflies/moths and caterpillars (Lepidoptera), spittlebugs (Homoptera), and dragonflies (Odonata) (Bent 1960, USFWS 2002). There is no documentation that water is required for drinking (Williams and Craig 1998).

USFWS (2002) summarizes documented predators of southwestern willow flycatcher eggs and nestlings:

- common kingsnake (*Lampropeltis getulus*);
- gopher snake (*Pituophis m elanoleucus affinis*);
- Cooper's hawk (*Accipiter cooperii*);
- red-tailed hawk (*Buteo jamaicensis*);
- great horned owl (*Bubo virginianus*);
- western screech owl (*Otus kennicottiii*);
- yellow-breasted chat (*Icteria virens*); and
- Argentine ants (*Linepithema humili*).

USFWS (2002) further reports that:

> Other potential predators of flycatcher nests include other snakes, lizards, chipmunks, weasels, racoons, ringtailed cats, foxes, and domestic cats (McCabe 1991, Sogge 1995, Langridge and Sogge 1997, Paxton *et al.* 1997, Sferra *et al.* 1997, McCarthey *et al.* 1998, Paradzick *et al.* 2000). Predatory birds such as jays, crows, ravens, hawks (especially accipiters), roadrunners, and owls may hunt in flycatcher habitat.

While these predators likely influence populations, documented predation rates for southwestern willow flycatcher are within the typical range for open-cup nesting passerine birds (Williams and Craig 1998). However, it should be noted that normal predation rates may disproportionately impact populations when the species is endangered.

The ecological interaction that has the greatest potential to impact southwestern willow flycatcher populations is brood parasitism by brown-headed cowbird (*Molothrus ater*). Cowbirds lay their eggs in the nests of other host species, which then incubate the cowbirds eggs and raise their young. Because cowbird eggs hatch after relatively short incubation and hatchlings develop quickly, they often out-compete the hosts' own young for parental care. Cowbirds may also prey upon nests by directly removing eggs and nestlings of host species from nests. Cowbirds, therefore, have the potential to negatively affect reproductive success of flycatchers, although available evidence indicates that cowbirds are not frequent predators of flycatcher nests (Whitfield *et al.* 1999, Whitfield 2000). In the Santa Clara River watershed, where brown-headed cowbirds are abundant, Labinger and Greaves (2001a) found no evidence of southwestern willow flycatcher nests being parasitized by cowbirds, although nests of other host species (least Bell's vireo, Hutton's vireo, yellow warbler, yellow-breasted chat, and song sparrow) were regularly found to be parasitized.

Sensitivity to Anthropogenic Watershed Disturbances

The southwestern willow flycatcher has experienced extensive loss and modification of breeding habitat. Habitat losses and modifications have been caused mainly by: reduction or elimination of surface and subsurface water due to diversion and groundwater pumping; changes in flood and fire regimes due to dams and stream channelization; clearing and controlling vegetation; livestock grazing; changes in water and soil chemistry due to disruption of natural hydrologic cycles; and establishment of invasive non-native plants (USFWS 2002).

Timber harvest, ground-disturbing activities, groundwater extraction, and water impoundments can alter hydrology, geomorphology, and associated biological processes that directly and indirectly affect flycatcher habitat. Regulated stream flow from dams, levees, and channelization is thought to be one of the most important factors explaining the decline of cottonwood and willow woodlands that flycatchers use as breeding habitat (Finch and Stoleson 2000). Eradication of tamarisk, which southwestern willow flycatcher frequently nests in, can be detrimental when it is implemented in or near occupied habitat or where there is not a plan to restore suitable native riparian vegetation following the eradication effort (USFWS 2002).

Brown-headed cowbirds are associated with pack stations, groups of livestock, and suburban and agricultural (Verner and Ritter 1983, Stafford and Valentine 1985, both as cited in Craig Williams 1998, USFWS 2002). When these land uses occur near flycatcher habitat, the potential for brood parasitism increases.

Key Uncertainties

One of the primary uncertainties related to southwestern willow flycatcher recovery is whether small clusters of breeding sites, which is how most populations are characterized, are collectively as productive as one larger site. Studies of population structure (whether the species functions as a series of metapopulations), dispersal patterns (whether flycatchers disperse from unsuitable to suitable habitat patches as they become available), and habitat vulnerability (whether small patches function as population sinks) are needed to resolve this issue (Williams and Craig 1998, USFWS 2002, Sogge *et al.* 2003).

A better understanding of how habitat type influences flycatcher productivity and survival is also needed, particularly in light of how frequently flycatchers have been observed to nest in invasive, non-native vegetation (Williams and Craig 1998, USFWS 2002, Sogge *et al.* 2003). This information becomes more critical when large-scale tamarisk eradication efforts are planned, which may negatively impact flycatcher populations, particularly when subsequent native revegetation efforts are not sufficient or successful.

Much remains to be learned about how water management affects riparian vegetation, and how water management can be altered to benefit riparian southwestern willow flycatcher habitat (Finch and Stoleson 2000).

Some southwestern willow flycatcher populations suffer heavy cowbird parasitism, while others do not. Work is needed to determine which factors of landscape, habitat, avian community structure, or land use affect cowbird abundance and parasitism rates. Cowbird control through trapping has been proven to be effective in reducing rates of parasitism in some willow flycatcher populations (*e.g.*, Whitfield and Strong 1995, Whitfield *et al.* 1997) but not in others (*e.g.*, Whitfield *et al.* 1999, Whitfield 2000).

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WESTERN YELLOW-BILLED CUCKOO
Coccyzus americanus occidentalis

Legal Status

Federal	Candidate
State	Endangered
Other	None

Geographic Distribution

The western yellow-billed cuckoo ranges across most of the U.S. and northern Mexico, and winters in South America. The western subspecies of the yellow-billed cuckoo historically nested from British Columbia south to Mexico and was known to breed in all regions of California except the central and northern Sierra Nevada, the Great Basin, and the Colorado Desert. It was thought to range from Redding (Shasta County), south to Bakersfield and Weldon (Kern County), and from Sebastopol (Sonoma County) south to the Mexican border (Grinnell and Miller 1944 as cited in Laymon 1998). Small populations were also found further north in Shasta, Siskiyou, and Modoc counties, as well as east of the Sierra Nevada in Owens Valley (Laymon 1998).

Today, the western yellow-billed cuckoo has been extirpated as a nesting species from most of the state and its current distribution is limited to scattered locations in California. Breeding populations of greater than five pairs that persist every year in California are currently limited to the Sacramento River from Red Bluff to Colusa and the South Fork Kern River from Isabella Reservoir to Canebrake Ecological Reserve. Other sites where small populations of cuckoos (<5 pairs) breed or possibly breed (but not necessarily every year) are: the Feather River from Oroville to Verona (Butte, Yuba and Sutter counties); the Prado Flood Control Basin (San Bernardino and Riverside counties); the Amargosa River near Tecopa (Inyo County); the Owens Valley near Lone Pine and Big Pine (Inyo County); the Santa Clara River near Santa Clarita (Los Angeles County); the Mojave River near Victorville (San Bernardino County); and the Colorado River from Needles (San Bernardino County) to Yuma (Imperial County) (Laymon and Halterman 1987 as cited in Laymon 1998).

Local Distribution

Populations of western yellow-billed cuckoos occur in a number of southern California locations. The Santa Ana River, Prado Park, Lake Elsinore, and Temecula Creek have all been documented to support key western yellow-billed cuckoo populations (Laymon and Halterman 1989, Laymon 1998). Documented sightings of cuckoos in the Santa Clara River watershed are sparse, although, suitable habitat does exist throughout the watershed (Laymon and Halterman 1989). One cuckoo sighting was recorded near Santa Paula on the Santa Clara River in 1971 (CDFG 2005). Labinger and Greaves (2001a) sighted two western yellow-billed cuckoos in 1997 and 1998 in the upper portion of the Santa Clara River watershed (ne ar Magic Mountain and above McBean Freeway in Valencia), although they noted that these were one-time sightings and most likely migrants. In 2003, a cuckoo was sighted on the Santa Clara River west of Fillmore (CDFG 2005).

Population Trends

Historical accounts describe western yellow-billed cuckoos as a common species throughout much of lowland California, and particularly in the Central Valley (Belding 1890 as cited in Laymon 1998). In the United States, from 1966 to 1996, yellow-billed cuckoos experienced an annual decline in population of 1.6 percent (Sauer *et al.* 1997). During the same time period, yellow-billed cuckoos in the western breeding region had an estimated annual decline of 4.7 percent, although the number of cuckoo sighting was so low (n = 17) the estimate is considered unreliable (Sauer at al. 1997 as cited in Laymon 1998). In California, yellow-billed cuckoos have shown both historic and recent population declines. In 1977, there were an estimated 123 to 163 pairs in the state (Laymon 1998). This estimate fell to 30 to 33 pairs ten years later, a 73 to 82 percent decline (Laymon 1998).

During surveys in 1986, Laymon and Halterman (1989) observed no western yellow-billed cuckoos on the Santa Clara River, but estimated a population of two, based on the presence of available habitat. Labinger and Greaves (2001a) heard one individual cuckoo in 1997 and observed two birds in 1998 in upper portions of the Santa Clara River watershed. They found no cuckoos during subsequent searches and presumed that the observed individuals were migrants rather than breeding pairs (Labinger and Greaves 2001a and 2001b).

Life History

The yellow-billed cuckoo is a neotropical migrant that winters in South and Central America (Deschauensee 1970 as cited in Suckling *et al.* 1998). The majority of yellow-billed cuckoos arrive at breeding grounds in western North America in June, though the time period can range from late-April to early-July (Phillips *et al.* 1964, Ryser 1985, both as cited in Suckling *et al.* 1998; Gaines and Laymon 1984 as cited in Laymon 1998). While breeding pairs maintain territories of 4 to 40 hectares (10 to 100 acres), they do not actively defend these territories after nest sites have been chosen (Laymon 1980, Halterman 1991; both as cited in Suckling *et al.* 1998, Laymon and Halterman 1985, Laymon 1998).

Most females appear to breed within their first year, while breeding in males may be delayed because of a high male:female ratio (Laymon 1998). Most pairs are monogamous, with both sexes sharing in incubation of eggs and feeding of young during mid-June to late July. Two to three eggs are laid on average, although up to five have been reported (Laymon 1998). Incubation lasts from 11 to 12 days, and the nestling period lasts from 5 to 8 days (Laymon 1998). Although they cannot fly yet, most newly hatched cuckoos are adept climbers and typically fledge six to eight days after hatching (Suckling *et al.* 1998). Fledglings continue to partially depend on their parents for two to four weeks before beginning the migration south (Suckling *et al.* 1998, Laymon 1998, Anderson *et al.* 1994).

Cuckoos typically begin their fall migration in early August and most have left California with their young by mid-September (Laymon 1998).

Habitat Requirements and Associated Vegetation

Cuckoos inhabit densely foliated, deciduous trees and shrubs, particularly willows (*Salix* spp.), with a dense understory formed by blackberry (*Rubus* spp.), stinging nettle (*Urtica dioica*), and/or California wild grape (*Vitis californica*) adjacent to slow-moving watercourses, backwaters, or seeps (CDFG 1983). River bottoms and other mesic habitats, including valley-foothill and desert riparian habitats, are necessary for breeding. Dense low-level or understory foliage with high humidity is preferred (Gaines 1974, 1977). This taxon may avoid Valley oak (*Quercus lobata*) riparian habitats where scrub jays are abundant (Laymon 1998).

Field studies and habitat suitability modeling have concluded that vegetation type (*i.e.*, cottonwood-willow), patch size, distance to water, and ratio of high to medium and low tree canopy height are critical factors determining the suitability of habitat for yellow-billed cuckoo breeding pairs (see Table 1) (Laymon and Halterman 1989, Greco 1999).

Table 1. Habitat suitability for western yellow-billed cuckoos.

Habitat Suitability	Habitat Type	Area (ha)	Width (m)	Distance to water (m)	Ratio of Tree Height Classes (H:L+M)	Ratio of Floodplain Age (YNG:OLD) (<45yrs: >45yrs)
Optimum	Willow-Cottonwood	> 80	> 600	< 100	0.8–1.249	2.1–4.0
Suitable	Willow-Cottonwood	41–80	200–600	*	0.25–0.799 1.25–2.0	1.1–2.0 4.1–7.0
Marginal	Willow-Cottonwood	17–40	100–199	*	<0.249	0.6–1.0 >7.1
Unsuitable	Willow-Cottonwood	< 17	< 100	> 100	>2.1	<0.5

Source: Greco 1999 (adapted and modified from Laymon and Halterman 1989; Laymon *et al.* 1997).
* Excluded (presense/absence basis)

Patch size was the most important variable determining presence of cuckoos on the Sacramento River from 1987 to 1990 (Halterman 1991 as cited in Laymon 1998), with a trend toward increasing occupancy with increased patch size. Patch sizes greater than 80 ha (~200 ac) were always occupied by cuckoos; patches 41–80 ha (~101–200 ac) had 58.8 percent occupancy; and patches 20–40 ha (~50–100 ac) had 9.5 percent occupancy (Laymon and Halterman 1989). Few cuckoos have been found in forested habitat of less than 10.1 ha (25 ac) (Anderson *et al.* 1994). Willow-cottonwood habitat patches greater than 600 m (1,969 ft) in width were found to be optimal, and typically anything less than 100 m (328 ft) was unsuitable (Laymon and Halterman 1989). Halterman (1991, as cited in Greco 1999) and Laymon *et al.* (1997 as cited in Greco 1999) also observed more frequent nesting in areas less than 100 m (328 ft) from water and with vegetation below 20 m (66 ft) in height.

Young, rapidly growing stands of riparian vegetation provide preferred nest sites, higher productivity of invertebrate prey, and lower prevalence of predators compared with older stands (Laymon 1998, Halterman 1991 as cited in Laymon 1998). Greco (1999) defined suitable vegetation stand age to be less than 45–60 years, and stressed the importance of meandering riparian systems with intact erosional and depositional processes that create new areas for riparian vegetation to establish.

Cuckoos typically build their nests on horizontal branches of willow trees, where they are hidden from view from the ground or surrounding trees by foliage (Laymon *et al.* 1997, Hanna 1937, both as cited in Laymon 1998, CDFG 2000). Nest sites are typically located near surface water, in areas of high local humidity and cooler temperatures (Launer *et al.* 1990 as cited in Laymon 1998, Gaines and Laymon 1984 as cited in Suckling *et al.* 1998). Dense vegetation less than 20 m (66 ft) in height is especially important for nesting (Laymon *et al.* 1997 as cited in Greco 1999). At the South Fork Kern River nests were built at an average height of 4.8 m (15.7 ft) (Laymon *et al.* 1997, as cited in Laymon 1998). Similar nest heights have been reported in other areas (Laymon 1998).

In addition to willows, cuckoos have also been reported to nest in cottonwood trees (*Populus* spp.), and occasionally box elder trees (*Acer negundo*), mesquite shrubs (*Prosopis* spp.), and walnut and almond orchards (Hamilton and Hamilton 1965, Laymon 1998, CDFG 2000). In a survey of nesting sites along the South Fork Kern River, the dominant canopy species was Gooding's black willow (*Salix gooddingii*), with red willow (*Salix laevigata*) and Fremont cottonwood (*Populus fremontii*) as co-dominants, and mulefat (*Baccharis salicifolia*) as the dominant shrub species (Laymon *et al.* 1997, as cited in Laymon 1998). Forb species in the vicinity of nests included stinging nettle (*Urtica dioica* ssp. *holosericea*), mugwort (*Artemisia douglasiana*), and goldenrod (*Solidago occidentalis*) (Laymon *et al.* 1997, as cited in Laymon 1998).

Preferred foraging habitat is in cottonwoods with greater overall foliage density than where nesting occurs (Anderson and Laymon 1989).

Ecological Interactions

Yellow-billed cuckoos are foliage-gleaning insectivores, typically hopping from location to location slowly, watching for the motion of their prey on the green leaf background (Laymon 1998). The cuckoos feed primarily on caterpillars (especially the large sphinx moth larva), grasshoppers, cicadas, and other large insects, and occasionally on small vertebrates and fruits (Bent 1940, Preble 1957, Anderson *et al.* 1994). Food availability varies from year to year and can have a significant impact on cuckoo densities and reproductive success (Forbush 1927, Nolan and Thompson 1975; all as cited in Suckling *et al.* 1998, Laymon *et al.* 1997 as cited in Laymon 1998).

Laymon (1998) has observed red-shouldered hawks and northern harriers preying on nestlings and suggests that Cooper's hawks are the only potential predator of adult cuckoos. An aversion to nesting where western scrub-jays and loggerhead shrikes occur suggests that these species may prey upon cuckoo nestlings (Laymon 1998).

Sensitivity to Anthropogenic Watershed Disturbances

Loss of habitat and adequate patch sizes is the primary threat to western yellow-billed cuckoo populations. In California, it is estimated that only 2,769 ha (6,842 ac) of suitable cuckoo habitat exists (outside of the Colorado River watershed). Loss of habitat is attributed to riparian and floodplain land conversion for agricultural and urban development, and to water management (*e.g.*, dams, channelization, ground water pumping and diversion) that alters the hydrologic regime of rivers and precludes the renewal and establishment of preferred cuckoo habitat. Grazing has also impacted habitat for this species, as have invasions of salt cedar (*Tamarix* spp.) and giant reed (*Arundo donax*). Non-native trees such as English walnut (*Juglans regia*) and domestic fig (*Ficus carica*) provide limited nesting and foraging habitat (Laymon 1998).

There has been mixed evidence of the effects of pesticides on western yellow-billed cuckoos (Laymon 1998). Laymon and Halterman (1987) reported that pesticides caused eggshell thinning and potentially reproductive failure in cuckoo nests. Laymon personally observed a reduction in insectivorous birds, including cuckoos, after insecticides were aerially broadcast along the Stanislaus River, as well a odd nestling behavior following spraying of orchards (Laymon 1998).

Key Uncertainties

While preservation of existing cuckoo habitat is critical to maintaining populations, the limited area of existing habitat cannot insure the survival of the species. Creation and/or restoration of suitable habitat is required if populations are to increase. One key uncertainty in this regard is the ability of riparian and floodplain restoration and revegetation projects to provide the area and quality of habitat required by the species in a quick enough timeframe. On the Colorado River, cuckoos were observed foraging at a restored site two years after revegetation, and three pairs nested at the site four years after revegetation (Anderson and Laymon 1989). On the Kern River, where vegetative growth is slower than the Colorado River, cuckoos were neither observed foraging nor nesting three years after revegetation (Anderson and Laymon 1989).

Laymon (1998) outlines cuckoo research needs:

> Yellow-billed cuckoos are at such low densities in California that monitoring them with traditional methods such as point counts or spot mapping is not possible. Surveys using playback of the contact call are the only acceptable way to monitor the species. Several sites in California (Kern River, Sacramento River and Colorado River) should be monitored on a yearly basis and a statewide survey every ten years is needed to monitor trends for this species. Research on the movement patterns between subpopulations is needed to determine the potential for genetic interchange. Research on the effects of pesticide residues, especially on migration cuckoos is needed. More research on the effects of riparian habitat restoration is needed to determine the optimum mix of willows and cottonwoods to plant on these sites.

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NEVIN'S BARBERRY
Berberis nevinii

Legal Status

Federal	Endangered
State	Endangered
CNPS	1B.1 (rare, threatened, or endangered in CA and elsewhere; seriously endangered in CA)

Recovery Plan: A recovery plan has not been drafted for this species.

HCPs: The following Multi-species Conservation Plans (MSCPs) and Habitat Conservation Plans include consideration of this species (USFWS 2006):

- MSCP, City of Chula Vista Subarea Plan
- MSCP, City of La Mesa Subarea Plan
- MSCP, City of Poway Subarea Plan
- MSCP, City of San Diego Subarea Plan
- MSCP, County of San Diego Subarea Plan
- San Diego Gas & Electric
- Western Riverside MSHCP

Morphology

Berberis nevinii is a rounded, evergreen shrub in the barberry family (Berberidaceae) that can reach 4 m (13 ft) in height (Hickman 1993, CPC 2006). It has blue-green, spiny pinnately-compound leaves, bright yellow flowers, and produces yellow-red berries year round (CDFG 2005, CPC 2006).

Geographic Distribution

B. nevinii is endemic to cismontane southwestern California (Munz 1974, Hickman 1993). The Center for Plant Conservation (2006) reports: "Historically, this species was distributed from San Francisquito Canyon in the Liebre Mountains to San Fernando Valley and the Arroyo Seco near Pasadena (Los Angeles County), to San Antonio Wash along the southern base of the San Gabriel Mountains (San Bernardino County), to Scott Canyon and San Timoteo Canyon near Redlands, and to Dripping Springs/Vail Lake area (Riverside County)."

Currently, localized populations of *B. nevinii* are restricted to the interior foothills of the San Gabriel Mountains (Los Angeles and San Bernardino counties) and the foothills of the Agua Tibia Mountains (southwestern Riverside County) (Munz 1974, Hickman 1993). In 1987, the California Natural Diversity Database (CNDDB) reported a total of 9 extant populations of *B. nevinii*:

- in Los Angeles County at San Francisquito Canyon, the confluence of San Francisquito Creek and the Santa Clara River, and Arroyo Seco;
- in San Bernadino County at Rim Forest, Scott Canyon, and San Timoteo Canyon;
- in Riverside County at Dripping Springs and Vail Lake; and
- in San Diego County at Palomar (CDFG 1985 as cited in Boyd 1987).

A study of the habitat parameters associated with these populations by Boyd (1987) reports that the populations at the confluence of San Francisquito Creek and the Santa Clara River and at Rim Forest appear to be extirpated, and that the Arroyo Seco and Palomar populations appear to have been planted.

Local Distribution

There is one extant occurrence of *B. nevinii* within the Santa Clara River watershed (occurrence no. 12 in the Newhall USGS 7.5 minute quadrangle), and two nearby occurrences (occurrences no. 11 and 19 in the Warm Springs Mountain quadrangle), recorded in the CNDDB (CDFG 2005). These populations represent the northern- and western-most occurrences reported for the species. Descriptions of these three population occurrences, from the CNDDB (CDFG 2005), are included in Table 1.

Table 1. Descriptions of the three populations of *Berberis nevinii* occurring within or nearby the Santa Clara River watershed (CDFG 2005).

Occ. No.*	Status	General Location	Population Size and Date Last Observed	Ecological Information	Threats to Population
12	Natural/native; possibly extirpated	San Francisquito Canyon, near confluence with Santa Clara River	Species seen in 1965, but not in 1987 field visit.	None provided.	Area now has a nursery under power lines, crops in the floodplain, and is a popular ORV area. Erosion also threatens population.
11	Reintroduced to native habitat/range; presumed extant	San Francisquito Canyon, on both sides of highway, below Powerhouse #2, north of Saugus	75 seedlings seen in 1986, 130+ plants in 1987, 200 plants observed in 1988. *Berberis* planted here in 1929 by Payne may have naturalized at this site.	On rocky, gravelly cliffs and wash bottom in chaparral with cost live oak, black sage. Mostly in northwest facing slopes.	Dumping, invasion by tamarisk, road widening, and gold extraction activities.
19	Reintroduced to native habitat/range; presumed extant	Approx. 0.5 mi north of San Francisquito Powerhouse in San Francisquito Canyon	1 mature plant observed. Payne planted *Berberis nevinii* in this vicinity.	On alluvial terrace, associated with *Eriodictyon* sp., *Prunus ilicifolia* and Yucca.	Proposed highway construction.

*Occurrence number is the unique population identifier used in the CNDDB.

Population Trends

B. nevinii was listed as federally endangered in its entire range on October 13, 1998 (USFWS 2006). Since 1965, there have been a total of 29 occurrences of *B. nevinii* reported in southern California, 6 of which have been or are presumed extirpated, and 14 of which are believed to have been introduced (*i.e.*, transplants back into native habitat/range or outside of native habitat/range).

The total number of individuals is estimated to be fewer than 1,000, although only half of those are believed to be naturally occurring individuals (CDFG 2006). While population sizes vary considerably among extant groups, the majority of occurrences are comprised of only one to few individuals, with little to no reproduction observed (Boyd 1987, CDFG 2006). In 1987, the San Francisquito Canyon population was the largest population known, with greater than 253 individuals (Boyd 1987). In 1998, however, the complex of approximately 16 populations at Vail Lake, with 200 total individuals, was believed to be the largest population (USFWS 1998).

Life History

There is little information available on the life history, population demographics, breeding system, and pollination biology of *B. nevinii*. What information is available comes largely from the experience of horticulturalists, as the species is a popular cultivar. The plant blooms in March through April (CPC 2006). Despite prolific berry production, fertile seed development appears to occur very sporadically (Benson 1943 as cited in Mistretta 1989). When fertile seeds are available, germination rates are generally quite high (Mistretta 1989). *Berberis nevinii* has been observed to sprout from a basal burl following wildfire, although vegetative propagation has not been successful in cultivation (Mistretta 1989). Mistretta (1989) concludes that *B. nevinii* recruitment is likely dependent on sporadic seed production.

Habitat Requirements

B. nevinii generally grows in sandy/gravelly soils on steep, north facing slopes or in low gradient sandy washes, between 240–1,575 m (787–5,167 ft) in elevation (Boyd 1987, Hickman 1993, CDFG 2006, CNPS 2006). In a study of the habitat parameters of *B. nevinii* (which excluded the populations believed to have been planted; see above), Boyd (1987) found that populations on slopes occur in coarse, sandy, non-marine derived soils, and populations on washes occur in recently derived sandy/gravelly alluvium. NatureServe (2006) reports that the presence of groundwater flow may be a habitat requirement of *B. nevinii*, but this does not appear to have been evaluated by other sources. In cultivation, *B. nevinii* demonstrates a wide tolerance to varying substrate and water conditions (Lenz and Dourley 1981 as cited in Mistretta 1989).

Where *B. nevinii* occurs on steep, north-facing slopes, it is generally found in coastal scrub and chaparral habitat. Where the species occurs in low gradient washes, it is found in alluvial and riparian scrub habitats (Boyd 1987, CDFG 2005, CDFG 2006, CNPS 2006, NatureServe 2006). Boyd (1987) reports that *Eriogonum fasciculatum*, *Artemisia californica*, *Rhus ovata*, *Salvia mellifera*, *Sambucus mexicana*, *Encelia farinosa*, and *Rhamnus crocea* are species associated with *B. nevinii*. Further, Boyd (1987) notes that several desert species, not characteristic of cismontane chaparral habitat, are frequently associated with *B. nevinii*, including *Chrysothamnus nauseosus*, *Artemesia tridentata*, *Chilopsis linearis*, *Yucca schidigera*, and *Atriplex canecens*.

Ecological Relationships

Because of its radial, yellow flowers, *B. nevinii* is believed to be pollinated by a variety of insects (Mistretta 1989). Birds are believed to be the primary seed dispersal mechanism for *B. nevinii*; a variety of bird species have been observed eating the berries (Wolf 1940 as cited in Mistretta 1989).

B. nevinii has been observed to stump-sprout following wildfires, a trait of many chaparral species that depend on particular fire frequencies and intensities for regeneration (Mistretta 1989). Therefore, fire regime (frequency and intensity) may be an important ecological variable in determining *B. nevinii* population persistence, although the specific effects of fire on this species have not been studied (USFWS 1998).

Sensitivity to Anthropogenic Watershed Disturbances

Boyd (1987) and Mistretta (1989) both consider that the decline in *B. nevinii* may be related to low fecundity and habitat loss. *Berberis nevinii* populations that occur in alluvial washes are threatened by a variety of activities associated with habitat loss, including urban and agricultural development, competition by non-native plant species, off-road vehicle activity, road maintenance, vegetation clearing and channelization for flood control, and fire fighting activities (Mistretta 1989, USFWS 1998, CNPS 2006, NatureServe 2006). As discussed earlier, *B. nevinii* may be sensitive to alterations to the natural fire regime, but these effects are unknown. Similarly, factors effecting observed low fecundity are unknown.

Key Uncertainties

A key uncertainty in the recovery of *B. nevinii* is the lack of reproduction and recruitment observed at most sites. While seed viability generally appears to be high, and germination under cultivated conditions has been highly successful, seed production appears to be sporadic and plant establishment in the wild is limited (CDFG 2006b). This is further confounded by the lack of information on *B. nevinii* life history, population demographics, breeding system, and pollination biology.

The CPC (2006) recommends that additional population surveys and annual monitoring be conducted to identify additional occurrences of *B. nevinii* and evaluate reproduction and seedling recruitment within populations (CPC 2006). Further, genetic studies are warranted to improve understanding of population dynamics, particularly between native and introduced populations (CPC 2006).

The lack of life history information for *B. nevinii*, in contrast with the success of cultivation efforts to germinate and grow the species, suggests that in the short-term, active revegetation of *B. nevinii* plants collected from viable, local populations and grown in a nursery may be the most effective means of increasing populations.

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SLENDER-HORNED SPINEFLOWER
Dodecahema leptoceras

Legal Status

Federal	Endangered
State	Endangered
CNPS	1B.1 (rare, threatened, or endangered in CA and elsewhere; seriously endangered in CA)

Recovery Plan: USFWS (1996) has published a draft recovery plan for this species.

*HCP*s: The Lake Mathews environmental assessment and Western Riverside MSHCP environmental impact statement include consideration of this species (USFWS 2006).

Morphology

Dodecahema leptoceras is a small, spreading annual in the buckwheat family (Polygonaceae). It has a basal rosette of leaves from which rise dense, flowering stalks. The plant reaches 3–15 cm (1.18–5.91 in) across, with the size depending on annual available moisture (Ferguson *et al.* 1996). It is distinguished from other spineflowers by the presence of six terminal awns and six hooked basal awns on each involucre (group of bracts subtending a flower), which encloses three white to pink flowers, 1.2 to 2 mm (0.47 to 0.79 in) in length (Hickman 1993, Ferguson *et al.* 1996).

Geographic Distribution

D. leptoceras occurs in the foothills of the Transverse and Peninsular Ranges of southern California (Young *et al.* 2000). Reveal and Hardham (1989 as cited in Boyd and Banks 1995) report that, historically, its range extended from the northwestern San Gabriel Mountains (Soledad Canyon), east across the southern San Gabriel and San Bernardino Mountains, and south along the western San Jacinto mountains to the north base of the Palomar Range at Agua Tibia Mountain. In Riverside County, it historically occurred along the eastern base of the Santa Ana Mountains in Temescal Canyon.

Currently, this species is known to occur in eight watersheds:

- in Los Angeles County at Santa Clara River, Big Tujunga Wash;
- in Riverside County at Temescal Canyon, San Jacinto River, Bautista Creek, Arroyo Seco, Kolb Creek, and at Vail Lake; and
- in San Bernadino County at Santa Ana River and Lytle Creek (Dudek and Associates 2000, CDFG 2006).

Local Distribution

There are three recorded occurrences of *D. leptoceras* within the Santa Clara River watershed reported in the CNDDB, representing the northern- and western-most occurrences reported for the species (CDFG 2005). Populations are clustered in five USGS 7.5 minute quadrangles: Mint Canyon, Newhall, Val Verde, Santa Susana, and Agua Dulce (CDFG 2005). Descriptions of these three population occurrences, taken directly from the CNDDB (CDFG 2005) are included in Table 1.

Table 1. Descriptions of the three populations of *Dodecahema leptoceras* occurring within the Santa Clara River watershed (CDFG 2005).

Occ. No.*	Status	General Location	Population Size and Date Last Observed	Ecological Information	Threats to Population
5	Possibly extirpated	Mint Canyon, Los Angeles County	Based on 1937 collection. 1979: none observed in south end of canyon; north end may still have habitat.	None provided.	Alteration of natural hydrology
6	Possibly extirpated	Newhall	Based on 1893 collection.	None provided.	Developme nt
27	Presumed extant	Bee Canyon Wash tributary of the Santa Clara River	1991: ~500 plants in ~30 x 60 m (100 x 200 ft) area 1993: >1,000 observed	Occur in large barren openings in old growth alluvial scrub/woodland on stabilized alluvial bench. Soil composed of silt with some gravel. Associated with Juniperus californica, Yucca whipplei, and Ephedra spp. Other associates include Lepidospartum squamatum, Chorizanthe coriacea, Stylocline gnaphalioides, Schismus barbatus.	Developme nt

* Occurrence number is the unique population identifier used in the CNDDB.

Population Trends

The status of *D. leptoceras* in 1999 was listed as "stable to declining" by the US Fish and Wildlife Service (USFWS 2006). Of the 34 reported population occurrences in southern California, 23 are existing and 11 have been or are presumed extirpated (CDFG 2006). Most of the known occurrences support only a small number of subpopulations. The Santa Ana River (in Riverside County) supports as many as 22 subpopulations, although eight of those have not been seen in recent years. At one site on the Santa Ana River, there was a marked decline in the number of plants from 1994–96 to 1998–99 (Ferguson and Ellstrand 1999). The Vail Lake area in Riverside County may support 28 subpopulations (CDFG 2005). Two of the three reported occurrences of the species within the Santa Clara watershed are possibly extirpated populations from historical collections (1893 and 1937), while the third population is quite large, with over 1,000 plants last reported in 1993 (CDFG 2005).

It has been demonstrated that *D. leptoceras* has a higher level of genetic diversity, mostly within populations, than is typical for annual and endemic plant species (Ferguson and Ellstrand 1999). This species is protandrous (anthers develop earlier than the stigma), suggesting that *D. leptoceras* is an obligate outcrosser (USFWS 1996). Additionally, genetic evidence suggests that the breeding system for this species is highly outcrossed, although it has been determined that *D. leptoceras* is self-compatible

(Reveal and Hardman 1989 as cited in Boyd and Banks 1995, Ferguson *et al*. 1996). Ferguson and Ellstrand (1999) found that, despite large differences in population size between locations and fluctuations within populations between years, there is no evidence that any particular population maintains significantly less genetic diversity or has experienced increases in homozygosity (the presence of identical alleles at one or more loci in homologous chromosomes) relative to other surveyed populations of *D. leptoceras*. Reveal and Hardman (1989 as cited in Dudek and Associates 2000) concluded that population sizes at the sites studied were large enough (hundreds to thousands of individuals) to prevent a genetic bottleneck.

Life History and Timing

D. leptoceras flowers from April through June (CNPS 2006). The flowers produce small (1.7 to 2 mm long [0.07 to 0.08 in]), brown or black achenes (Munz 1974, Hickman 1993). Seeds are believed to remain viable in the soil for a number of years, with abundant germination known to occur following successive years of little or no seed production (Ferguson and Ellstrand 1999). The seed bank appears to be critical to replenishing populations of *D. leptoceras* both demographically and genetically (Ferguson and Ellstrand 1999). *D. leptoceras* germinates from late February to early March in response to winter rains. Ferguson *et al.* (1996) found that available soil moisture had a strong influence on survivorship to reproduction and subsequent seed rain.

Habitat Requirements

D. leptoceras is found on stabilized alluvial fans, floodplains, stream terraces, washes, and associated benches from 656 to 2,493 ft (200 to 760 m) in elevation (CNPS 2006). These geomorphic surfaces are usually alluvial deposits greater than 100 years in age (Wood and Wells 1996) that receive overbank deposits every 50 to 100 years (Prigge *et al.* 1993 as cited in Dudek and Associates 2000).

According to studies by Allen (1996), *D. leptoceras* are found in slightly acidic silt soil with low salinity, organic matter, and nutrient content. Preferred microhabitats include silt filled, shallow depressions on relatively flat surfaces (Allen 1996, Wood and Wells 1996). The location of the largest population reported in the Santa Clara watershed, in the Bee Canyon Wash tributary, is consistent with these reports, since it occurs on a stabilized alluvial bench in silty soil with some gravels (CDFG 2005). It appears that where *D. leptoceras* occurs, sediment deposition is generally driven by local lower-impact processes, such as overland flow during rain events and/or windgaps (which deposit fine-grained wind-borne sediment), rather than extreme flood events (Wood and Wells 1996). Wood and Wells (1996) recommend research into the role of overland flow as a seed dispersal agent for this species and CDFG is planning to examine microtopographic features, soil flora, and factors that may limit seed dispersal in an upcoming study (CDFG 2005).

The moisture requirements of *D. leptoceras* are not well understood and reports vary in their interpretation of moisture related factors that favor *D. leptoceras* . Wood and Wells (1996) indicate that the old alluvial deposits upon which *D. leptoceras* is typically found have low permeability and enhanced run-off, while Boyd and Banks (1995) suggest that the cryptogamic crust (associations of mosses, algae, lichens, and some xerophytic liverworts) that is often found at *D. leptoceras* sites may help retain soil moisture. However, cryptgamic crusts are not found consistently with *D. leptoceras.* Young *et al.* (2000) suggest that *D. leptoceras* may depend on upwelling zones both for water and nutrients, and that the difference between occupied and unoccupied habitat may be related to groundwater rather than surface features.

D. leptoceras occurs in chaparral, cismontane woodland, and coastal alluvial fan scrub habitat, and is generally found in open areas with other spineflower species. In the Santa Clara watershed, one occurrence of the species is found in large barren openings in old growth Riversidean/Mojavean alluvial scrub (CDFG 2005). In the Vail Lake area in Riverside County, *D. leptoceras* occurs in gravel soils in association with open chamise chaparral, but this population appears to be anomalous (Boyd and Banks 1995). CDFG (2005) reports that *D. leptoceras* is often associated with *Encelia* spp., *Dalea* spp. and other *Lepidospartum* spp. However, studies by Allen (1996) indicate that the plant co-occurs with a variety of other alluvial fan plant species, to the extent that sites with *D. leptoceras* can occur in substantially different community types (*e.g.*, sites with juniper, cottonwood, or no trees, and sites with 75 percent cover of cryptogamic crust, or virtually no crusts). Allen (1996) concluded that there was no particular indicator species that could be used to detect *D. leptoceras* habitat. *D. leptoceras* can co-occur with exotic grass species, but is generally found in areas with low densities of exotic grasses and other introduced weedy species (Allen 1996, CPC 2006). Where percent cover of exotic grasses is very high, *D. leptoceras* populations have been shown to decrease (Allen 1996).

Cryptogamic crusts are frequently present in areas occupied by *D. leptoceras*, and may help suppress recruitment of non-native plant species (Boyd and Banks 1995). Allen (1996) found that, while cryptogamic crusts are frequently present, they are not consistently present in association with *D. leptoceras*. The range of cryptogamic crust cover at *D. leptoceras* sites, from dominate to nearly absent, may be related to the age of the site's geomorphic surfaces. This suggests that *D. leptoceras* is associated with specific soils that may vary in age, but meet the species' other edaphic requirements (Allen 1996).

Ecological Relationships

Studies of eight locations of *D. leptoceras* revealed that the species can form mycorrhizal associations, although not obligatory and not likely mutualistic, with arbuscular-mycorrhizal fungi (Young *et al.* 2000). The authors infer that absence of arbuscular-mycorrhizal fungi was not a limiting factor in suitable but unoccupied habitat (Young *et al.* 2000).

D. leptoceras flowers are likely pollinated by insects, but floral visitation is extremely difficult to observe. Ferguson *et al.* (1996) observed mostly ants and flying insects visiting *D. leptoceras* flowers, and a small wasp (*Plenoculus davisii*) carrying pollen has been collected (CDFG 2005).

Dispersal of *D. leptoceras* seeds has not been extensively studied, but it has been suggested that, given the shape of the involucre (*i.e.*, six ascending awns and six descending awns), the seed is suited for animal dispersal. Animals responsible for seed dispersal could include coyotes, rabbits, rodents and deer. Additionally, dispersal may also occur via flood water or wind (Prigge *et al.* 1993 as cited in Dudek and Associates 2000, USFWS 1996), although these mechanisms are in need of further study (Wood and Wells 1996).

Sensitivity to Anthropogenic Watershed Disturbances

Historical occurrences of *D. leptoceras* have generally been lost to urbanization and stream channelization activities (CNPS 2006). Currently, the species is threatened by development, sand and gravel mining, flood control, proposed reservoir construction, and vehicles (CNPS 2006). Preservation of older, stable alluvial surfaces in the historical range of *D. leptoceras* should be the primary focus for the protection of the species (Wood and Wells 1996).

Key Uncertainties

The prospects for restoring *D. leptoceras* through revegetation are uncertain. Previous efforts at growing *D. leptoceras* from germinated seeds have failed and additional research is needed to establish successful germination protocols and growing conditions for the species (Ferguson *et al.* 1996, CPC 2006). Further, the high genetic diversity of the seed bank coupled with high mortality of adult plants in the field and a limited seed rain suggest that revegetation to establish new populations will require a large investment of seeds over many generations (Ferguson 1999, Ferguson and Ellstrand 1999).

The lack of information on pollination and seed dispersal mechanisms could also confound attempts to restore *D. leptoceras* populations. The ability of revegetation efforts to restore self-sustaining populations of *D. leptoceras* may be limited if appropriate pollinators and/or seed dispersal conditions do not occur at selected restoration sites.

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TIDEWATER GOBY
Eucyclogobius newberryi

Legal Status

Federal	Endangered
State	None
Other	Species of Special Concern

Native Origin and Geographic Distribution

This species is endemic to California and is generally located along the California coast, mainly in small coastal lagoons and near stream mouths in the uppermost brackish portion of larger bays (SCR Project Steering Committee 1996). Historically this species ranged from the mouth of the Smith River, Del Norte County near the Oregon border to Agua Hedionda Lagoon in northern San Diego County (USFWS 2005). Tidewater goby localities include discrete lagoons, estuaries, or stream mouths separated by mostly marine conditions, and are generally absent from areas where the coastline is steep and streams do not form lagoons or estuaries (USFWS 2005).

Local Distribution

Tidewater gobies have been observed in the Santa Clara River as far as three miles from the estuary/lagoon, between Ventura and Oxnard (SCR Project Steering Committee 1996), and also in the Santa Ynez River (USFWS 2005).

Population Trends

Estimating tidewater goby population trends is complicated because populations are controlled by environmental conditions (USFWS 2005). For example, when lagoons are breached due to flood events during the rainy seasons, populations of tidewater gobies generally decrease and then recover during the following summer (USFWS 2005). Current distribution still remains entirely within the original known range of the species, however, 23 (17 percent) of the 134 documented localities are considered extirpated and 55 to 70 (41 to 52 percent) of the localities are so small in size or have been degraded over time that long-term persistence is uncertain (USFWS 2005).

Life History and Timing

Although tidewater goby are short-lived (generally 1 year), they have relatively high fecundity (females produce 300–500 eggs/batch and spawn multiple times per year), with males defending eggs in burrows. Reproduction and spawning typically occurs during the spring and summer (April to June) in slack shallow waters of seasonally disconnected or tidally muted lagoons, estuaries, and sloughs. Males dig burrows vertically into sand, 100-200 mm [4 to 8 inches] and guard eggs (SCR Project Steering Committee 1996). Juveniles and adults can be found year-round, although they are most abundant in summer/fall.

Reproduction occurs at all times of the year, but generally male tidewater gobies begin digging breeding burrows in relatively unconsolidated, clean, coarse sand (averaging 0.5 mm [0.02 inch] in diameter), in April or May after lagoons close to the ocean (USFWS 2005). Individual burrows are at least 70 to 100 mm (3 to 4 inches) from each other (USFWS 2005). Female tidewater gobies aggressively spar with each other for access to males with burrows for laying their eggs (USFWS 2005). Female tidewater gobies can lay 300 to 500 eggs per clutch, depending on the size of the individual female tidewater goby, and can lay up 6 to 12 clutches per year (Swift *et al.* 1989, Swenson 1999). Male tidewater gobies remain in the burrow to guard the eggs that are attached to sand grains in the burrow ceiling and walls (USFWS 2005). Embryos require 9 to 11 days to hatch, during which the male tidewater goby cares for the embryos, rarely emerging from the burrow to feed (USFWS 2005). Tidewater gobies spawn regularly in water with salinities 8 to 15 parts per thousand (ppt) and temperatures 17 to 22 Celsius (°C) (62 to 71 degrees Fahrenheit) (USFWS 2005). Tidewater goby standard length at hatching is approximately 4 to 5 millimeters (0.17 to 0.25 inch), and are planktonic (unable to swim freely) for 1 to 3 days before they become benthic (USFWS 2005). The average size of tidewater gobies tends to be significantly larger in marshes (43 to 45 millimeters [1.7 to 1.8 inches] standard length) when compared to tidewater gobies from lagoons or creek habitats (USFWS 2005, Swenson 1999). This may be because the more stable physical conditions of the marsh foster improved growth or a more consistent or abundant supply of prey (USFWS 2005, Swift et al? 1997).

Habitat Requirements

The lagoons in which tidewater gobies are found range in size from a few square meters of surface area to about 800 hectares (2,000 acres). Most lagoons are much smaller, ranging from about 0.5 to 5 hectares (1.25 to 12.5 acres) (USFWS 2005). Tidewater gobies can use habitat in water that is comprised of 75 percent sea water (*i.e.*, salinity of 28 parts per thousand), but generally are found in areas where water salinity is 12 parts per thousand or less (USFWS 2005). Tidewater gobies are usually collected in areas with water less than 1 meter (3.3 feet) deep (USFWS 2005). Tidewater gobies often migrate upstream into tributaries, as far as 1.0 kilometer (0.5 mile) from the estuary (SCR Project Steering Committee 1996). However, in the Santa Ynez River, Santa Barbara County, tidewater gobies are often collected 5 to 8 kilometers (3 to 5 miles) upstream of tidal lagoons areas, sometimes in sections of stream impounded by beavers (*Castor canadensis*) (SCR Project Steering Committee 1996, USFWS 2005).

Tidewater gobies are an estuarine/lagoon adapted species that may infrequently disperse via marine habitat but with no dependency on marine habitat for its life cycle (Swift *et al.* 1989, Lafferty *et al.* 1999a). They can tolerate large temperature and salinity ranges. Reproduction takes place in water between 9 to 25 °C (48 to 77 degrees Fahrenheit) and at salinities of 2 to 27 parts per thousand (USFWS 2005). Tidewater gobies generally inhabit areas with water temperatures 4–21.5°C (39.2–70.7°F) (USFWS 2005). Preferred salinities (ppt) for reproduction/spawning were identified as ≤ 15 within a range of 2–27 ppt (Swenson 1999, USFWS 2005).

Tidewater gobies require stable lagoon or off-channel habitats, particularly during their relatively short larval stage (Lafferty *et al.* 1999a, Chamberlain 2006). Flood and breaching events can result in dispersal of tidewater gobies between estuarine/lagoon habitats, although survival is likely low and dispersal is limited. The distance between extirpated habitats and larger wetland source populations affects dispersal success and re-colonization potential (Lafferty *et al.* 1999a and 1999b). Gobies can persist in habitats that flood as long as a velocity refuge is present (Moyle 2002, Lafferty *et al.* 1999b).

The life stages that are likely most sensitive to changes in habitat conditions associated with flooding and breaching are eggs in burrows and pelagic larvae (Chamberlain 2006). Juveniles and adults can tolerate flooding/breaching in late fall/winter. Preferred substrates are sand, mud, gravel, and silt, particularly associated with submerged vegetation that is likely used for cover (USFWS 2005).

Ecological Interactions

Tidewater gobies feed mainly on small animals, usually mysid shrimp, gamarid amphipods, and aquatic insects, particularly chironomid midge larvae (Swift *et al.* 1989; Swenson 1995; Moyle 2002). Swenson (1996) found that juvenile tidewater gobies are generally day feeders, although adults mainly feed at night (USFWS 2005). Tidewater gobies use three different foraging styles to capture benthic prey: plucking prey from the substrate surface, sifting sediment in their mouth, and mid-water capture (USFWS 2005). The variety of foraging methods allows tidewater gobies to utilize a wide variety of prey items in various habitats.

Native predators of tidewater gobies include small steelhead (*Oncorhynchus mykiss*), prickly sculpin (*Cottus asper*), and staghorn sculpin (*Leptocottus armatus*) (Swift *et al.* 1989, USFWS 2005). Predation by the tule perch (*Hysterocarpus traski*), and historically by the Sacramento perch (*Archoplites interruptus*), has probably prevented tidewater gobies from inhabiting the San Francisco Bay delta, an otherwise ideal habitat for tidewater gobies (Swift *et al.* 1989). Garter snakes (*Thamnophis* spp.) also probably prey on tidewater gobies. Rathbun (1991) suggested that robust populations of tidewater gobies, as well as threespine stickleback and prickly sculpins, would provide food for the twostriped garter snake (*Thamnophis hammondii*) in Santa Rosa Creek Lagoon.

Gobies are an important part of estuarine food webs, as they provide prey for larger fish and piscivorous birds (Swenson and McCray 1996). However, tidewater goby are highly susceptible to predation by introduced species, especially piscivorous fish and amphibians (Lafferty *et al.* 1999a, Lafferty and Page 1997). Sunfishes (*Lepomis* spp.) and basses (*Micropterus* spp.), have been introduced in or near coastal lagoons and estuaries and could prey heavily on tidewater gobies (USFWS 2005), as well as African clawed frogs in some freshwater habitats (Lafferty *et al.* 1999a, and Swift *et al.* 1997, Lafferty and Page 1997). In addition, the shimofuri goby, which has become established in the San Francisco Bay region (Moyle 2002), competes with and preys upon the smaller tidewater goby (Swenson and Matern 1995). Introduced yellowfin goby and shimofuri goby may also compete with or prey on tidewater goby (Swenson and McCray 1996, Swenson 1999; both as cited in Moyle 2002). At least four species of Asian estuarine and freshwater gobies and the rainwater killifish (*Lucania parva*), have been introduced to California and may compete or displace tidewater goby when they occur in the same areas.

Many piscivorous birds, including egrets (*Egretta* spp.), herons (*Ardea herodias*, *Butorides striatus*, *Nycticorax nycticorax*), cormorants (*Phalacrocorax* spp.), terns (*Sterna* spp.), mergansers (*Mergus* spp.), grebes (*Podiceps* ssp., *Podilymbus* spp., *Aechmophorus* spp.), and loons (*Gavia* spp.), frequent the coastal lagoon habitats, mainly in fall and winter, and may feed on tidewater gobies (Rathbun 1991). Tidewater goby appear to prefer shallow depths (< 3 ft [1 m]) near emergent vegetation, possibly to avoid predation by wading birds and piscivorous fish (Moyle 2002). Reported shallow minimum depths of occurrence may be associated with depth thresholds for wading bird predators such as herons; in general, avian predation efficiency decreases with depths > 20 cm (8 in) (Gawlik 2002). However, reported depth preferences may be biased because sampling equipment commonly used to survey tidewater gobies, such as beach seines, are limited in their utility to sample deeper habitats.

Sensitivity to Anthropogenic Watershed Disturbances

The main threats to tidewater goby include changes in water quality, degradation and loss of habitat due to urbanization, and predation from invasive species such as the African clawed frog. It is estimated that tidewater goby has disappeared from 74 percent of the coastal lagoons south of Morro Bay.

Gobies are sensitive to impacts such as lack of freshwater due to diversions, pollution, siltation, and invasion of non-native species, such as the western mosquitofish (*Gambusia affinis*), which is a competitor, and the African clawed frog (*Xenopus laevis*), which is a predator (USFWS 2005, Lafferty et. al 1999a).

According to the U.S. Fish and Wildlife Service, which has prepared a final recovery plan for the tidewater goby, the key threats to the goby that are relevant to the Santa Clara River watershed include agricultural discharges, sewage treatment effluent, water diversions, and exotic species (USFWS 2005).

Key Uncertainties

- How do tidewater gobies re-colonize areas in Santa Clara, particularly after high flows due to storm events?
- Is it possible to improve connectivity of lagoon habitat?
- Are non-native fish (sunfish, bass) having a greater impact than non-native amphibians (African clawed frog, bullfrog) on tidewater gobies? If so, why?

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SOUTHERN STEELHEAD
Oncorhynchus mykiss

Legal Status

Federal Endangered
State None
Other Species of Special Concern

Taxonomy and nomenclature

Oncorhynchus mykiss is one of several related *Oncorhynchus* species that exhibit considerable life history plasticity, namely the ability to complete their life cycle entirely in freshwater or migrate to the ocean as juvenile "smolts" and return to spawn in freshwater as adults after 1-3 years at sea (Boughton *et al* 2006). The freshwater resident form is commonly termed "rainbow trout"; the sea-going or anadromous form is typically referred to as "steelhead". Adding to the complexity of *O. mykiss* life history is the apparent ability of rainbow trout to produce steelhead offspring (an anecdotally common occurrence in populations within the Santa Clara River watershed), and for steelhead to produce resident rainbow trout offspring. Further discussion of steelhead life history can be found below, and in Boughton *et al* (2006). This summary generally pertains to the anadromous form of *O. mykiss* within the southern California distinct population segment (NMFS 2006).

Geographic Distribution

Steelhead occur throughout the North Pacific Ocean and historically spawned in freshwater streams along the west coast of North America from Alaska to northern Baja California. Historically, *O. mykiss* occurred at least as far south as Rio del Presidio in Mexico, although spawning populations of steelhead did not likely occur that far south (NMFS 1997). At present, the southernmost stream used by steelhead for spawning is generally considered to be Malibu Creek, California (NMFS 1997); however, in years of substantial rainfall, spawning steelhead may be found as far south as the Santa Margarita River, in northern San Diego County (NMFS 1997).

Local Distribution

Historically (before circa 1946), steelhead likely spawned and reared in the major tributaries within the lower portion of the Santa Clara River system, west of the Piru Creek confluence (Kelley 2004, Harrison *et al.* 2006). These major tributaries included primarily Sespe and Piru creeks; Santa Paula and Hopper creeks likely provided significant steelhead habitat as well. A number of other tributaries in the upper (eastern) Santa Clara River system may have been used during wet years (Titus *et al.*, *in preparation*), though published information supporting their use by steelhead is generally lacking.

The present-day distribution of anadromous *O. mykiss* in the Santa Clara River watershed is modified by a number of complete and partial migration barriers that restrict upstream passage of adult steelhead, both in the lower mainstem river and most major tributaries. The Vern Freeman Diversion, approximately 16 km (10 mi) upstream from the mouth on the mainstem river, likely represents a partial barrier to upstream migration by returning adult steelhead; between 1994 and 1996 a total of four adult steelhead have navigated the fish ladder at the diversion (Entrix 2000). Passage within Santa Paula Creek, located approximately 8 km (5 mi) upstream of the Vern Freeman Diversion is limited by fish passage facilities damaged in the record 2005 floods[1]. Upstream migratory access to Piru Creek, approximately 43 km (27 mi) upstream of Vern Freeman Diversion, was eliminated by the completion of Santa Felicia Dam in 1955. Sespe Creek, located approximately 28 km (17 mi) upstream of the Vern Freeman Diverson, is the only major steelhead spawning tributary to the Santa Clara River watershed that remains unregulated and accessible to upstream migrants (Titus *et al.*, *in preparation*).

Population Trends

The National Marine Fisheries Service (NMFS) has concluded that populations of naturally reproducing steelhead have been experiencing a long-term decline in abundance throughout their range (NMFS 1996a). Populations in the southern portion of the range have experienced the most severe declines (NMFS 1996a); NMFS estimates that the current southern steelhead population represents less than 1 percent of its historical population size (as cited in Stoecker 2002).

Prior to 1940, the Santa Clara River watershed is thought to have supported an average annual run of approximately 7,000-9,000 steelhead (Titus *et al.*, *in preparation*). Steelhead runs in the Santa Clara River may have been one of the largest in southern California (Stoecker and Kelley 2005, Titus *et al.*, *in preparation*). Recent counts at fish passage facilities associated with the Vern Freeman Diversion Dam indicate approximately 14 adult steelhead have returned to spawn in the Santa Clara River watershed since 1990 (Stoecker and Kelley 2005, Titus et al., *in preparation*).

Southern steelhead in the Santa Clara River watershed have declined steeply since the 1950's, mainly because of an increase in surface water diversion in the lower Santa Clara River (Titus *et al.*, *in preparation*). Other causes include diversions along the Santa Clara River, such as the diversion near Saticoy and the Vern Freeman Diversion Dam (Titus *et al.*, *in preparation*). Early CDFG records indicate that 5,000 juvenile steelhead were stocked in 1938, and 21,600 were planted in the lagoon at the mouth of the Santa Clara River in 1944 (Titus *et al.*, *in preparation*). Most of the fish planted in the lagoon were rescued from the Santa Ynez River (Titus *et al.*, *in preparation*). A two year study completed in 1985 yielded less than 30 steelhead adults in the lower Santa Clara River, and no emigrating smolts (Titus *et al.*, *in preparation*). The study concluded that the lower Santa Clara River served primarily as a migration corridor for adult and juvenile steelhead, while the estuary may still provide potential rearing habitat (Puckett and Villa 1985 as cited in Titus *et al.*, *in preparation*).

Life History and Timing

Steelhead is the term used to distinguish anadromous populations of *O. mykiss* from resident populations. Much life history variability exists among steelhead populations; however, populations may be broadly categorized into two reproductive groups, most commonly referred to as either winter-run or summer-run. South of San Francisco Bay, all steelhead are all winter-run.

[1] Restoration and enhancement of fish passage at the Vern Freeman Diversion and within Santa Paula Creek is currently being considered. The Vern Freeman Diversion Dam is undergoing formal Section 7 consultations under the federal Endangered Species Act, lead by the National Marine Fisheries Service. Barriers within the Santa Paula Creek sub-basin are being addressed through studies funded by the California Department of Fish and Game (see www.santaclarariverparkway.org/wkb/projects/santapaulacreek) and the California Department of Transportation.

In the Santa Clara River watershed, the *O. mykiss* population appears to consist primarily of resident fish, possibly due to partial or complete migration barriers (both natural and anthropogenic) that preclude anadromous adults from reaching spawning tributaries. However, small numbers of anadromous juvenile steelhead (smolts) outmigrate from the Santa Clara River each year, presumably produced by the existing resident adult population. The relationship between anadromous and resident life history forms of this species is the subject of ongoing research. Current evidence suggests that either life history form can produce offspring that exhibit the alternate form (*i.e.*, resident rainbow trout can produce anadromous progeny and vice versa) (Shapovalov and Taft 1954, Burgner *et al.* 1992, Hallock 1989). The fact that little to no genetic differentiation has been found between resident and anadromous life history forms inhabiting the same basin supports this hypothesis (Busby *et al.* 1993, Nielsen 1994, but see Zimmerman and Reeves 2000).The life history patterns of southern California steelhead depend more strongly on rainfall and flow than steelhead populations found farther north (NMFS 1997, Titus *et al.* in press). In southern California, average rainfall is substantially lower and more variable than in regions to the north, resulting in increased duration of sand berms across the mouths of streams and rivers and, in some cases, complete dewatering of the lower reaches of these streams from late spring through fall (NMFS 1997, Entrix 2002). Steelhead in southern California appear to withstand higher temperatures than populations to the north (NMFS 1997). Although there is minimal life history information for southern California steelhead, several unique traits have been identified, including increased temperature tolerance, duration and timing of life stages, and environmental flexibility (Stoecker and Kelley 2005, Titus *et al.*, in press).

Adult Upstream Migration and Spawning

Adult steelhead return to spawn in their natal stream, usually in their fourth or fifth year of life (Shapovalov and Taft 1954, Behnke 1992). Access to natal streams is often impaired or blocked because of low flow conditions (Stoecker 2002). Southern steelhead time their upstream migration to follow sizable rainfall events in the fall (Stoecker 2002). A unique adaptation of southern steelhead is the ability to delay the upstream migration until adequate flows exist, or ascend another accessible and suitable stream nearby (Stoecker 2002). This is an important adaptation in the often stochastic and arid regions of southern California.

During spawning, female steelhead create depressions in streambed gravels by vigorously pumping their body and tail horizontally near the streambed. The optimal water depth for steelhead spawning is approximately 14 in (36 cm) (Stoecker 2002). These depressions, or redds, are approximately 4–12 inches (10–30 cm) deep, 15-in (38-cm) in diameter, and oval in shape (Needham and Taft 1934, Shapovalov and Taft 1954). Males do not assist with redd construction, but may fight with other males to defend spawning females (Shapovalov and Taft 1954).

Although most steelhead die after spawning, adults are capable of returning to the ocean and migrating back upstream to spawn in subsequent years, unlike most other Pacific salmon. Runs may include from 10% to 30% repeat spawners, the majority of which are females (Ward and Slaney 1988, Meehan and Bjornn 1991, Behnke 1992). Repeat spawning is more common in smaller coastal streams than in large drainages requiring a lengthy migration (Meehan and Bjornn 1991). Steelhead may migrate downstream to the ocean immediately following spawning or may spend several weeks holding in pools before outmigrating (Shapovalov and Taft 1954).

Egg Incubation, Alevin Development, and Fry Emergence

Hatching of eggs follows a 20- to 100-day incubation period, the length of which depends on water temperature (Shapovalov and Taft 1954, Barnhart 1991).

Juvenile Freshwater Rearing

Juvenile steelhead (parr) rear in freshwater before outmigrating to the ocean as smolts. Juvenile southern steelhead have extremely variable residence time due to the highly unpredictable and often stochastic environmental conditions that exist in watersheds in southern California (Stoecker and Kelley 2005). Some juvenile steelhead may never migrate, they remain in freshwater as coastal rainbow trout for their entire life cycle (Stoecker and Kelley 2005).

Steelhead may overwinter in mainstem reaches, particularly if coarse substrates in which to seek cover from high flows are available (Reedy 1995), or they may return to tributaries for the winter (Everest 1973 as cited in Dambacher 1991).

Smolt Outmigration

At the end of the freshwater rearing period, juvenile steelhead migrate downstream to the ocean as smolts. A length of 5.46 in (14 cm) is typically cited as the minimum size for smolting (Wagner *et al.* 1963, Peven *et al.* 1994). Evidence suggests that photoperiod is the most important environmental variable stimulating the physiological transformation from parr to smolt (Wagner 1974). During smoltification, the spots and parr marks characteristic of juvenile coloration are replaced by a silver and blue-green iridescent body color (Barnhart 1991) and physiological transformations occur that allow them to survive in salt water. Southern steelhead smolts may spend a considerable amount of time in lagoons and estuaries in order to acclimate to saltwater before outmigrating (Stoecker 2002). These lagoons and estuaries also provide a holding area where smolts can feed while waiting for adequate flow conditions to open the streams and lagoons to the ocean (sandbars build up and seal off many confluences in low flow conditions) (Stoecker 2002).

Estuarine Rearing

Estuarine rearing may be more important to steelhead populations in the southern half of the species' range due to greater variability in ocean conditions and paucity of high quality near-shore habitats in this portion of their range (NMFS 1996a). Estuaries may also be more important to populations spawning in smaller coastal tributaries due to the more limited availability of rearing habitat in the headwaters of smaller stream systems (McEwan and Jackson 1996). Most marine mortality of steelhead occurs soon after they enter the ocean and predation is believed to be the primary cause of this mortality (Pearcy 1992 as cited in McEwan and Jackson 1996). Because predation mortality and fish size are likely to be inversely related (Pearcy 1992 as cited in McEwan and Jackson 1996), the growth that takes place in estuaries may be very important for increasing the odds of marine survival (Pearcy 1992 [as cited in McEwan and Jackson 1996], Simenstad *et al.* 1982 [as cited in NMFS 1996a], Shapovalov and Taft 1954).

Ocean Phase

The majority of steelhead spend one to three years in the ocean, with smaller smolts tending to remain in salt water for a longer period than larger smolts (Chapman 1958, Behnke 1992). Steelhead staying in the ocean for two years typically weigh 7 to 10 lbs (3.15 to 4.50 kg) upon return to fresh water (Roelofs 1985). Unlike other salmonids, steelhead do not appear to form schools in the ocean. Steelhead in the southern part of the species' range appear to migrate close to the continental shelf, while more northern populations of steelhead may migrate throughout the northern Pacific Ocean (Barnhart 1991).

Habitat Requirements

Adult Upstream Migration and Spawning

During their upstream migration, adult steelhead require deep pools for resting and holding to minimize their energetic outputs (Puckett 1975, Roelofs 1983 as cited in Moyle *et al.* 1989, Stoecker and Kelly 2005). Southern steelhead require spawning areas to be at least 14 inches (36 cm) deep (SCR Project Steering Committee 1996). Steelhead need water with a minimum depth of 7 in (18 cm) and maximum velocity of 8 ft/s (240 cm/s) for successful upstream migration (Thompson 1972 as cited in Everest *et al.* 1985). Relatively cool water temperatures (between 50 and 59°F [10° and 15°C]) are preferred by adults, although they may survive temperatures as high as 80.6°F (27°C) for short periods (Moyle *et al.* 1989). The average

Pool tailouts or heads of riffles with well-oxygenated gravels are often selected as redd locations (Shapovalov and Taft 1954). The average area encompassed by a redd is 47–65.56 ft^2 (4.4–5.9 m^2) (Orcutt *et al.* 1968, Hunter 1973 as cited in Bjornn and Reiser 1991). Gravels ranging in size from 0.25 to 5.07 in (0.64 to 13 cm) in diameter are suitable for redd construction (Barnhart 1991).

Egg Incubation, Alevin Development, and Fry Emergence

Incubating eggs require dissolved oxygen concentrations, with optimal concentrations at or near saturation. Low dissolved oxygen increases the length of the incubation period and cause emergent fry to be smaller and weaker. Dissolved oxygen levels remaining below 2 ppm result in egg mortality (Barnhart 1991).

Juvenile Freshwater Rearing

Age 0+. After emergence from spawning gravels in spring or early summer, steelhead fry move to shallow-water, low-velocity habitats such as stream margins and low-gradient riffles and will forage in open areas lacking instream cover (Hartman 1965, Everest *et al.* 1986, Fontaine 1988). As fry increase in size in late summer and fall, they increasingly use areas with cover and show a preference for higher-velocity, deeper mid-channel waters near the thalweg (Hartman 1965, Everest and Chapman 1972, Fontaine 1988).

Age 1+ and older juveniles. Older age classes of juvenile steelhead (age 1+ and older) occupy a wide range of hydraulic conditions. They prefer deeper water during the summer and have been observed to use deep pools near the thalweg with ample cover as well as higher-velocity rapid and cascade habitats (Bisson *et al.* 1982, Bisson *et al.* 1988). Age 1+ fish typically feed in pools, especially scour and plunge pools, resting and finding escape cover in the interstices of boulders and boulder-log clusters (Fontaine 1988, Bisson *et al.* 1988). During summer, steelhead parr appear to prefer habitats with rocky substrates, overhead cover, and low light intensities (Hartman 1965, Facchin and Slaney 1977, Ward and Slaney 1979, Fausch 1993). Age 1+ steelhead appear to avoid secondary channel and dammed pools, glides, and low-gradient riffles with mean depths less than 7.8 in (20 cm) (Fontaine 1988, Bisson *et al.* 1988, Dambacher 1991).

As steelhead grow larger, they tend to prefer microhabitats with deeper water and higher velocity as locations for focal points, attempting to find areas with an optimal balance of food supply versus energy expenditure, such as velocity refuge positions associated with boulders or other large roughness elements close to swift current with high macroinvertebrate drift rates (Everest and Chapman 1972, Bisson *et al.* 1988, Fausch 1993). Reedy (1995) indicates that 1+ steelhead especially prefer high-velocity pool heads, where food resources are abundant, and pool tails, which provide optimal feeding conditions in summer due to lower energy expenditure requirements than the more turbulent pool heads. Fast, deep water, in addition to optimizing feeding versus energy expenditure, provides greater protection from avian and terrestrial predators (Everest and Chapman 1972).

Winter Habitat

Steelhead overwinter in pools, especially low-velocity deep pools with large rocky substrate or woody debris for cover, including backwater and dammed pools (Hartman 1965, Swales *et al.* 1986, Raleigh *et al.* 1984, Fontaine 1988). Juveniles are known to use the interstices between substrate particles as overwintering cover. Bustard and Narver (1975) typically found age 0+ steelhead using 3.9–9.7 in (10–25 cm) diameter cobble substrates in shallow, low-velocity areas near the stream margin. Everest *et al.* (1986) observed age 1+ steelhead using logs, rootwads, and interstices between assemblages of large boulders (39.0 in [>100 cm] diameter) surrounded by small boulder to cobble size (19.7–39.0 in [50–100 cm] diameter) materials as winter cover. Age 1+ fish typically stay within the area of the streambed that remains inundated at summer low flows, while age 0+ fish frequently overwinter beyond the summer low flow perimeter along the stream margins (Everest *et al.* 1986).

Ocean Phase

Little is known about steelhead use of ocean habitat. Some steelhead migrate extensively while others have short oceanic migrations (Stoecker and Kelly 2005). Steelhead appear to prefer ocean temperatures of 48.2°–52.7°F (9°–11.5°C) and typically swim in the upper 30–40 ft (9–12 m) of the ocean's surface (Barnhart 1991).

Ecological Interactions

Emergent *O. mykiss* fry initially feed on zooplankton and other microorganisms (Barnhart 1991). Juveniles feed on a wide range of items, primarily those associated with the stream bottom such as aquatic insects, amphipods, aquatic worms, fish eggs, and occasionally smaller fish (Wydoski and Whitney 1979). Juveniles may also feed on spiders, mollusks, and fish, including smaller steelhead (Roelofs 1985). Age 0+ steelhead prefer benthic invertebrates (Johnson and Ringler 1980); larger steelhead, having larger mouths, can consume a broader range of foods (Fausch 1991). In the ocean, steelhead feed on juvenile greenling, squids, amphipods, and other organisms (Barnhart 1991).

Major predators of adult steelhead include humans, marine mammals, and large pelagic fish. Eggs may be eaten by macroinvertebrates, crayfish, and other fish. Juvenile steelhead may be preyed upon by garter snakes, piscivorous fish such as older salmonids (including steelhead), freshwater sculpins, introduced piscivorous fish (*e.g.*, black bullhead, green sunfish, smallmouth bass, striped bass), mammals (*e.g.*, river otter, mink), and piscivorous birds (*e.g.*, mergansers, kingfishers, herons, ospreys, loons) (Stoecker and Kelly 2005). Juvenile steelhead have been observed feeding on emergent fry (Shapovalov and Taft 1954).

Sensitivity to Anthropogenic Watershed Disturbances

An anadromous life history and changes in habitat requirements at different life stages make steelhead vulnerable to a wide range of watershed disturbances, including dams, timber harvest, road construction, recreational use, and other human-related disturbances. The relative importance of anthropogenic or natural disturbances and ocean conditions for controlling steelhead populations is uncertain.

Physical Barriers to Migration and Movement

Dams without fish passage facilities block migration to historically available spawning and/or rearing areas, inundate spawning and rearing habitat beneath reservoirs, and alter hydrologic regimes, sediment and LWD budgets, water temperatures, nutrient cycling, and food supplies (Collins 1976). Where fish passage facilities are provided at dams, delays to upstream or downstream migration may occur, and stress, injury, or mortality may result from passage through juvenile bypass facilities. Stoecker and Kelly (2005) identified and assessed barriers to southern steelhead habitat throughout the Santa Clara River watershed. Severe barriers to steelhead passage were identified on tributaries to the Santa Clara River, including Santa Paula, Sespe, Hopper, and Piru Creeks (Stoecker and Kelly 2005). Additionally, the most significant barrier existing on the Santa Clara mainstem is the Vern Freeman Diversion Dam, which needs considerable improvement to allow unimpeded upstream and downstream migration over a wide range of flows, independent of water diversion operations, maintenance, debris blockage, or fish ladder damage (Stoecker and Kelly 2005).

Changes to Hydrologic Regimes

Changes to natural flow regimes may impact steelhead populations through changes to stimuli used for timing of upstream and downstream migrations, dewatering of redds, displacement of fry or juveniles, scouring of spawning gravels, and changes to the quality and quantity of habitat for different life stages. Rapid decreases in flow associated with hydroelectric project operations may cause stranding, especially of recently emerged fry because of their preference for stream margin areas of mainstem channels and because they are relatively weak swimmers (Hunter 1992). Vulnerability to stranding declines once juvenile steelhead reach lengths of 1.8 inches (45 mm) (R.W. Beck and Associates 1987). As juveniles grow, they are more likely to occupy deeper areas further from channel margins, reducing their susceptibility to stranding. Flow diversions may delay or stop adult migration if minimum water depths are not maintained (Everest *et al.* 1985).

Changes to Sediment Dynamics

Sedimentation of streams resulting from increased erosion may reduce spawning success of steelhead and the carrying capacity of juvenile rearing areas. Sedimentation due to land use activities has been recognized as a primary cause of habitat degradation for steelhead populations on the west coast (NMFS 1996a). Increased input of fine sediment resulting from natural or anthropogenic disturbance may be the principle cause of egg and alevin mortality in some areas (Shapovalov and Taft 1954). Filling of interstitial spaces with fine sediments reduces intragravel flow through redds, reducing dissolved oxygen concentrations and the rate of removal of metabolic wastes (Everest *et al.* 1985). Alevins that develop in oxygen-deficient gravels are smaller at emergence, placing them at a competitive disadvantage (Doudoroff and Warren 1965 as cited in Everest *et al.* 1985). Interstitial habitat used as cover by juvenile steelhead is also reduced if embedded in fine sediments. Bjornn *et al.* (1977) observed reduced juvenile steelhead abundance in Idaho streams characterized by a high degree of substrate embeddedness.

Changes to Stream Temperatures and Water Quality

Factors that result in increased stream temperatures, such as removal of riparian vegetation and changes to natural flow regimes may reduce steelhead populations both directly through increased mortality and indirectly through such factors as changes to growth rates or timing of emergence and downstream migration.

Warm water temperatures may favor competitors of juvenile steelhead, such as redside shiners (Reeves *et al.* 1987). Increases in water temperatures may also make juvenile anadromous salmonids more susceptible to mortality from diseases such as *Flexibacter columnaris* (Holt *et al.* 1975).

Estuary Impacts

Estuary conditions may have an important influence on anadromous fish survival, since anadromous fish must pass through these areas during upstream and downstream migration and since estuarine rearing prior to ocean entry is a life history strategy used by many juvenile anadromous fish to increase marine survival (Giger 1972, Healey 1991, McMahon and Holtby 1992). Degradation of estuary habitats due to diking and filling, increased temperatures, introduction of piscivorous fish, sedimentation due to upstream impacts, and other human activities may have contributed to anadromous fish declines in California

Key Uncertainties

- How do non-native species (*e.g.*, African clawed frogs, bullfrogs, smallmouth bass) impact steelhead in the Santa Clara River watershed?
- Would habitat restoration be beneficial for southern steelhead if barriers still exist downstream of the habitat restoration?
- Is it possible to improve habitat connectivity for southern steelhead?
- Is food availability a limiting factor for fry and juvenile steelhead success?
- Smolt utilization and survival in the estuary
- Steelhead ocean ecology
- How much straying occurs from natal streams?

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GIANT REED
Arundo donax

Non-Native Invasive Weed Rating

Federal No rating
State CDFA Noxious Weed "B" Rating
Cal-IPC High

Native Origin and Geographic Distribution

Arundo donax is indigenous to freshwaters of eastern Asia, specifically northern India and southern Nepal (Polunin and Huxley 1987). Approximately 2,000 years ago, *A. donax* was introduced around the Mediterranean Basin for use in erosion control, reeds for musical instruments and for construction of roofs, ceilings, fences and baskets (Perdue 1958). It has been introduced to most tropical and warm, temperate regions worldwide, including North and South America, Southern Africa and Australia, and thrives below ~1,150 ft (350 m) in elevation (Bell 1997, Brossard *et al.* 2000). In Southern California, *A. donax* was prevalent along the Los Angeles River as early as the 1820s and often planted for erosion control along streams and windbreaks (Robbins *et al.* 1951). Currently *A. donax* is rapidly invading rivers and streams in Mediterranean-type climates, including coastal watersheds and the central valley of California.

Local Distribution

Arundo donax can be found along most rivers and streams in Ventura County. On the Santa Clara River, it grows in large stands or monocultures along floodplains and terraces, and has invaded most riparian vegetation types (Stillwater Sciences and URS 2007). It thrives in open riparian areas with abundant water and nutrients as well as any area susceptible to burning (Coffman 2007).

Population Trends

Arundo donax is rapidly invading riparian ecosystems along rivers in Mediterranean-climate regions worldwide. Following an era of human alterations to river systems in Southern California, it was widely dispersed throughout riparian ecosystems in the floods of 1969, established in terrace and floodplain locations, and is now thriving in riparian ecosystems throughout this region (Coffman 2007). Factors such as quantity of water, nutrients, light, and fire that are abundant in riparian ecosystems of Mediterranean-climate regions increase the competitive ability of *A. donax* (Coffman 2007).

Life History and Timing

Due to its clonal growth strategy, efficient use of resources, and high growth rate, *A. donax* is one of the most successful riparian weedy invaders in California (Rieger and Kreager 1989). As a perennial bamboo-like member of the grass family (Poaceae), *A. donax* spreads vegatatively via a well-developed rhizome; its seeds are known to be sterile outside its native range (Decruyenaere and Holt 2001, Khudamrongsawat *et al.* 2004). *Arundo donax* plants are uprooted and dispersed downstream during large, winter flood events characteristic of Mediterranean-type climates (Bell 1994). Portions of the rhizome or culm break off, float downstream, land on a bare, moist substrate as flood waters recede and begin growing. Fragments of the rhizome or culm as small as 2 cm^2 (0.79 in^2) have been shown to sprout under most soil types, depths and soil moisture conditions (Else 1996, Boose and Holt 1999, Wijte *et al.* 2005). Growing at an extremely high rate (up to 6.25 cm [2.46 in] per day under ideal conditions), *A. donax* quickly establishes on unvegetated or sparsely vegetated soil and grows to a height of greater than ~20 ft (6 m) after only a few months (Rieger and Kreager 1989, Coffman 2007). It then expands outward in area, quickly displacing indigenous shrubs, herbs and grasses, and eventually even trees.

When above ground biomass of *A. donax* dies back in late summer and fall, riparian areas dominated by this plant become susceptible to fire (Scott 1994). Riparian terraces invaded by *A. donax* adjacent to shrubland communities are most vulnerable (Coffman *et al.* 2004). Indigenous riparian trees, shrubs and other vegetation not as well-adapted to fire are burned along with *A. donax* and resprout much slower (Coffman 2007). *Arundo donax* grows back immediately to completely replace the open burned areas originally dominated by indigenous riparian vegetation (Coffman 2007). In this manner, *A. donax* forms extensive stands or monocultures in riparian ecosystems, along floodplains and terraces of southern California's river and stream systems.

Habitat Requirements

Arundo donax grows primarily in floodplains and terraces of low-gradient river and stream systems in Southern California (DiTomaso 1998, Coffman 2007). *Arundo donax* successfully invades areas consisting of any soil type and once established can grow well in many soil moisture regimes (Singh *et al.* 1997, Boose and Holt 1999, Coffman 2007). It is most successful at colonizing open floodplains, containing elevated amounts of water and nutrients (Coffman 2007). However, it may be found on beaches, around homes, and next to hot springs where planted.

Ecological Interactions

In California, *A. donax* is known to increase the risk of flooding, create fire hazards, out-compete indigenous species for scarce water resources, and reduce the value of riparian habitat for wildlife (Bell 1994, Bell 1997, DiTomaso 1998). Wildfires ignited by man at unnatural and dangerous times of the year burn rapidly through riparian corridors infested with *A. donax* and may help spread fires across watersheds (Coffman 2007). The federally endangered least Bell's vireo (*Vireo bellii pusillus*) and other riparian birds require structural diversity provided by riparian scrub and mature forest communities for breeding (Zembal 1990, Bell 1994, Bell 1997). When natural riparian vegetation types are replaced by thick stands of *A. donax*, bird species abundance and other native wildlife have been found to decline (Bell 1994, Bell 1997, Herrera and Dudley 2003, Kisner 2004, Labinger and Greaves 2001). Labinger and Greaves (2001) observed over the course of their study (1994–2000) that while dense thickets of *A. donax* supported very low bird diversity:

> … a low to moderate mixture of giant reed [*Arundo donax*] with native willow woodland supported high bird diversity in some areas [such as near the Freeman Diversion]…. In such areas giant reed was also used for nesting, as noted by at least 17 nests of least Bell's vireo, one nest of southwestern willow flycatcher, and several other species such as Anna's hummingbird, bushtit, and common yellowthroat.

Greenhouse and field experiments suggest that *A. donax* does best in open areas, with high soil moisture and nutrients (Coffman 2007). However, *Salix laevigata* (red willow) is a strong competitor in these areas as well as in shaded areas with low soil moisture. *Baccharis salicifolia* (mule fat) outcompetes *A. donax* in shaded areas with high soil moisture and open, dry areas (Coffman 2007). Addition of nitrogen fertilizer under high moisture conditions significantly increased both *A. donax* and *S. laevigata* biomass, but had little effect on *B. salicofolia* or *Populus balsamifera* (black cottonwood) biomass (Coffman 2007). Due to its higher post-fire growth rate and immediate growth response compared to natives, fire appears to contribute to the *A. donax* invasion process especially in riparian terraces (Scott 1994, Coffman 2007).

Anthropogenic Watershed Disturbances that Promote Invasion

Human alterations associated with urbanization of watersheds in California in addition to the natural flood regime have created ideal conditions for *A. donax* invasion. Ever expanding residential and agricultural development in coastal Southern California has led to increased water availability and nutrient loading of riparian ecosystems. Consequently, open areas along floodplains formed by floods and clearing of terraces for development create an ideal location for weedy species like *A. donax* to establish. Mature riparian forests continue to be removed to make room for agriculture, golf courses, and residential and commercial development. Fire is more frequent in riparian corridors due to anthropogenic ignition during the dry summer and fall months when *A. donax* infested areas provide a large amount of dry fuel (Scott 1994, Coffman 2007).

Control Efforts

Management strategies for the control and removal of *A. donax* should be based on location and size of the infestation. The first priority management strategy recommended is removal of *A. donax* from riparian terrace habitats where infested areas are easily accessible and require less maintenance than along floodplains (Coffman *et al.* 2004). *Arundo donax* should be removed first from riparian terraces located adjacent to fire-prone shrubland plant communities where it poses a fire hazard (Coffman *et al.* 2004). Removal of *A. donax* infestations on riparian terraces with high soil moisture and nutrient availability will be most difficult and will likely require active revegetation with native plants. Watershed removal plans need to be developed to eradicate *A. donax* from floodplains. Unless *A. donax* is removed from floodplains on a watershed-scale working from the headwaters downstream, it is likely to recolonize removal areas after flood events. Both riparian terrace and floodplain areas may require revegetation with native plants to insure continued success of *A. donax* eradication, prevention of other weed infestations, and restoration of functional riparian ecosystems.

Both mechanical and hand clearing techniques may be used to remove *A. donax*. Mechanical clearing methods include mulching or total excavation of all above-ground and below-ground biomass. Hand clearing methods include either painting of *A. donax* stumps with herbicide (glyphosate) after cutting or foliar applications of herbicide (Sonoma Ecology Center and California State University 1999). Research on biocontrol agents for *A. donax* is underway on the Santa Clara River (T. Dudley, pers. comm.).

Several *A. donax* removal programs have been implemented or planned along the Santa Clara River recently. The first large *A. donax* removal project in riparian ecosystems of the Santa Clara River was conducted on a riparian terrace at the Valley View Ranch in Santa Paula through the Partners for Fish and Wildlife program. The landowner entered into an agreement with the U.S. Fish and Wildlife Service to remove all *A. donax* within the approximately 20-acre property between 2000 and 2002 (S. Hedrick, pers. comm.; D. Pritchett, pers. comm.). Several removal techniques were implemented including: total removal of all biomass using heavy equipment, mulching with hammer-flail equipment that left biomass in place as mulch, and foliar spraying by hand in less accessible areas. Routine foliar herbicide spraying maintenance has been on-going from 2000 to present on this property and most of the *A. donax* has been successfully removed.

The Nature Conservancy (TNC) removed *A. donax* from their 98-acre property in the lower Santa Clara River just upstream of the Santa Paula 12th Street bridge in September–October 2002 (E. Kelley, pers. comm.). Large stands of *A. donax* (above- and below-ground biomass) were removed mechanically using a bobcat, and *A. donax* interspersed within native riparian vegetation was treated with foliar spraying of herbicide. Resprouts were treated in January 2003 using the foliar spraying technique. In fall 2005, TNC completed a 5-acre *A. donax* removal project on their property upstream of Highway 101, hand removing *A. donax* post-flood debris and burning the debris piles (S. Matsumoto, pers. comm.). Monitoring and maintenance plans are underway. Visual inspections were conducted during winter 2005–2006, shoots that emerged in early spring 2006 were sprayed with herbicide, and any resprouts that remain in fall 2006 will be sprayed. Smaller removal projects have been implemented in conjunction with restoration efforts and infrastructure projects such as the Hedrick Ranch Nature Area restoration project (1998–2006; S. Hedrick, pers. comm.) and the Santa Paula airport emergency repairs in 2005 (C. Burns, pers. comm.).

The Ventura County Resource Conservation District (VCRCD) has completed the Upper Santa Clara River Watershed Arundo/Tamarisk Removal Plan (SCARP) and is initiating a similar plan for the lower watershed with the goal of implementing a long-term (20-years) removal plan for *A. donax* and *Tamarix* spp. (VCRCD 2006; N. Cabanting, pers. comm.). This plan includes a programmatic California Environmental Quality Act (CEQA) and National Environmental Policy Act (NEPA) document and related documentation for the implementation, maintenance, and monitoring of *A. donax* and *Tamarix* spp. removal projects within riparian ecosystems (500-year floodplain) of the upper Santa Clara River watershed. The goal of this comprehensive document is to streamline the permitting process for any agency or organization to perform *A. donax* and *Tamarix* spp. removal projects of any size within upper Santa Clara Watershed. From September 2005 to March 2006, the California Conservation Corps and the Los Angeles Agricultural Commissioner (for VCRCD) removed all *A. donax* from approximately 75 acres (15 acres of solid *A. donax*) of their 297-acre demonstration project at the confluence of the South Fork of the Santa Clara River and San Francisquito Creek in the City of Santa Clarita. A combination of cut/paint (1–2 acres), foliar spray (2–3 acres), and cut and spray regrowth (12–15 acres) methods were used in this removal project. The VCRCD plans to monitor and maintain this removal area for the next 4–5 years and expand the area of removal each year. According to visual inspection in May 2006, the cut/paint method performed the best with little regrowth (N. Cabanting, pers. comm.). Both the other two methods worked, but adjacent rhizomes had some new regrowth.

Key Uncertainties

Despite the recent research, pilot removal projects, and monitoring programs focused on *A. donax*, many key uncertainties still exist regarding interactions between *A. donax* and native riparian species, the effects of *A. donax* on ecosystem services and functions, and effective control and monitoring of infestations. Outstanding questions regarding *A. donax* ecology and management on the Santa Clara River are categorized and summarized below.

Interactions with Native Riparian Plant Species:

- How much water is used (transpired) by *A. donax* compared to native riparian species under various environmental conditions and seasons?
- What is the optimal nutrient uptake rate (and range) and efficiency of N, P and K for *A. donax* compared to native riparian plant species?
- How does root structure, function and interactions with the soil vary between *A. donax* and native riparian plants species?
- How do *A. donax* and native riparian plant species differ in the source (location) of water they used?

Effects on Ecosystem Services and Functions:

- How does leaf litter and denitrification contribute to nutrient cycling in *A. donax* stands versus native riparian vegetation?
- What is the relationship between biodiversity and riparian ecosystem function in streams like the Santa Clara River?

Control and Monitoring:

- What is the effectiveness of various biocontrol agents on *A. donax* in the Santa Clara River?
- What is the most effective remote sensing approach for mapping and methods for monitoring *A. donax* removal and spread?
- What is the long-term effectiveness of herbicide application on suppressing resprouting?

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SALTCEDAR
Tamarix ramosissima

Non-Native Invasive Weed

Federal No rating
State CDFA Noxious Weed "B" Rating
Cal-IPC High

Native Origin and Geographic Distribution

Over 50 species of the *Tamarix* genus (Tamaricaceae family) are known from the arid and semiarid regions of Eurasia (Baum 1978). Two major centers of speciation of *Tamarix* include central Asia and the eastern Mediterranean basin. Beginning in 1823, numerous species of *Tamarix* were introduced to temperate and subtropical regions worldwide as ornamental plants (Harris 1966, Baum 1967, Crins 1989). In the southwestern U.S., *Tamarix* species were planted primarily for erosion control and windbreaks, and many species were reported as naturalized by 1877 (Horton *et al.* 1960, Robinson 1965, Harris 1966). Throughout most of the arid and semi-arid southwestern U.S., several *Tamarix* species have become the dominant element of riparian vegetation and other low-lying wetland areas characterized by high salt content and irregular water availability (Harris 1966). The *Tamarix* species most common along the Santa Clara River, *T. ramosissima,* is indigenous to central Asia, from eastern Turkey to western China. It was introduced to the eastern U.S. in the early 1800s as an ornamental, but escaped from cultivation by the late 1800s and was planted by government agencies and private landowners in the southwest for stream bank erosion control (Duncan 1997). After the late 1920s, *T. ramosissima* spread rapidly along river floodplains and lakes in the western U.S., dominating vegetation in these areas below 2,625 ft (800 m) by the 1970s (Hickman 1993, Duncan 1997).

Local Distribution

Small *T. ramosissima* populations and individuals may be found sporadically along terraces and floodplains of the Santa Clara River, as well as other streams and rivers in Ventura County. Before the large floods in winter 2005, *T. ramosissima* was only sparsely dispersed along floodplains and only a few large populations of *T. ramosissima* could be found on higher riparian terraces (Coffman, pers. obs.). Shortly after the floods of 2005, *T. ramosissima* seedlings were observed throughout the floodplain of the lower Santa Clara River, presumably due to sustained high soil moisture conditions that followed the flooding (Stillwater Sciences and URS Corporation 2007). Future monitoring will be required to determine how many of these seedlings survive to maturity.

Population Trends

Tamarix ramosissima dominates riparian vegetation throughout rivers in the Southwestern U.S. and California. Zavaleta (2000) estimated that *T. ramosissima* encompasses an area of 1 to 1.6 million acres in North America. The longitudinal nature of rivers promotes its invasion by effectively dispersing its seeds along moist floodplains during the years that floodwaters are released in regulated river systems, and within natural river systems. *Tamarix ramosissima* produces a large number of very small wind-dispersed seeds through a long flowering season (April–September) (Hickman 1993). In addition, *Tamarix* species thrive in rivers with infrequent flooding (every 2–3 years) (D'Antonio *et al.* 1999). *Tamarix ramosissima* is a successful invader in riparian ecosystems of the western U.S. due to its ability to tolerate arid environments with saline soils. In addition, *T. ramosissima* readily colonizes recently scoured floodplain surfaces and quickly establishes in areas with continued high soil moisture levels. Altered hydrologic and geomorphic processes associated with damming of western rivers and water diversions that resulted in soil salinization led to expansion of *T. ramosissima* along rivers in the last century (Everitt 1980, Brotherson and Field 1987). However, *T. ramosissima* has spread into relatively undisturbed riparian ecosystems and smaller tributaries as well (Dudley *et al.* 2000). *Tamarix* species populations will likely continue to spread and invade areas where introduced, especially where anthropogenic alterations promote its colonization and growth

One of the primary source populations for *T. ramosissima* in the Santa Clara River is located just south of the Ventura/Los Angeles County Line (Stillwater Sciences and URS Corporation 2007). Since typical timing of seed production for *T. ramosissima* does not coincide with the last flood event in February 2005, phenology of this species should be reexamined along the Santa Clara River. It is uncertain how much *T. ramosissima* has become well established from this initial seedling establishment in 2005.

Life History and Timing

Tamarix ramosissima is a shrub or small tree that propagates by both sexual and vegetative reproduction. Typically, it produces seed from April through September, although seeds only remain viable for a few weeks (Duncan 1997). Germination of new seeds occurs only within the first 24 hours on saturated soils (Horton *et al.* 1960). Seedling growth is slow, but once established *T. ramosissima* plants grow faster than native plants (Johnson 1986). The tap root grows rapidly straight to the water table and then starts secondary branching in the capillary zone as well as the zone of saturation (Tomanek and Ziegler 1960, Schopmeyer 1974). Like native riparian plants, *T. ramosissima* can also spread vegetatively when branches are removed during flood events and land on bare, moist floodplains (Coffman, unpublished data).

Affected Habitats/Associated Vegetation and Ecological Interactions

Tamarix ramosissima has rapidly invaded moist floodplains of regulated rivers and streams in the arid and semi-arid Southwestern U.S. and is especially successful where salinity levels are elevated (Harris 1966, Zavaleta 2000, Shafroth *et al.* 2005, Coffman unpublished data). Seeds cannot germinate without sufficient soil moisture and young seedlings cannot tolerate a high frequency flood regime (Everitt 1980, Brock 1994). Since 1900 when dam building programs were initiated, scoured floodplains of regulated streams and rivers with infrequent flooding have provide the ideal conditions for *T. ramosissima* to establish (Harris 1966). Once established on moist floodplains *T. ramosissima,* plants are highly resistant to disturbance due to flood and fire (Graf 1978, Busch and Smith 1993).

Invasion of *Tamarix* in river systems of the southwestern U.S. have created both environmental and economic impacts. The extensive lateral root system of *T. ramosissima* makes it an extremely strong competitor with native riparian phreatophytes. Once plants have become well-established, they use prodigious amounts of water and can tolerate both long periods of inundation and drought (Duncan 1997). *Tamarix ramosissima* has a very high evapotranspiration (ET) rate under ideal site and climatic conditions (0.7 to 3.4 m yr^{-1}), however variable depending on technique used to measure and duration of measurement (Carmen and Brotherson 1982, Davenport *et al.* 1982, Shafroth *et al.* 2005). Although native riparian plants have a similar range of ET rates, leaf area index of *Tamarix* species is typically higher than native plants rendering overall higher stand ET rates (Shafroth *et al.* 2005). Leaves and stems of *T.*

ramosissima exude salt from special glands and deposit salt on riparian soils underneath its canopy, excluding other plant species intolerant of hypersaline environments (Duncan 1997). An increase in fire frequency in riparian corridors results in areas with large *Tamarix* infestations, since both green and senesced foliage are more highly flammable during the summer/fall months than native riparian vegetation (Busch and Smith 1993, Busch 1995). In general, wildlife habitat value is decreased in riparian ecosystems heavily invaded by *Tamarix* due to lower available food sources and altered structural characteristics (Shafroth *et al.* 2005). Insect and arthropod abundance tends to be lower in *Tamarix* infestations compared to native riparian vegetation, however, a few native generalist herbivore insects and pollinators have been reported to use *Tamarix* (Shafroth *et al.* 2005) Monotypic stands of *Tamarix* may provide only limited cover for larger mammals, nesting sites for birds, and herpetofauna in more southern latitudes due to lack of shading in mid- to late-summer (Hunter *et al.* 1988, Lovich and DeGouvenain 1998, Shafroth *et al.* 2005). Migratory birds prefer native vegetation to *Tamarix* for foraging substrates (Rosenberg *et al.* 1991, Yong and Finch 1997). Threatened and endangered bird species response to *Tamarix* invasions in riparian ecosystems is variable (Dudley *et al.* 2000, Dudley and DeLoach 2004). Both the endangered southern willow flycatcher (*Empidonax traillii extimus*) and the candidate for Federal endangered species list yellow-billed cuckoo (*Coccyzus americanus*) prefer native forests in some cases and but incorporate some habitat with *Tamarix* into their breeding territory (Shafroth *et al.* 2005).

The high ET rates associated with *T. ramosissima* infestations have led to a lowering of the water table in many areas in which water resources are already scarce (Shafroth *et al.* 2005). Due to its ability to adsorb water under drought conditions via its extensive root system, high ET rates continue into the dry summer months when water uptake by other phreatophyte species has ended. *Tamarix* species are thought to increase risk of flood damage when dense vegetation causes water to slow forcing flood flows upward and channel erosion where *Tamarix* encroaches on the channel edges and facilitates down-cutting of the channel (Graf 1978). Economic costs due to water loss and erosion control attributed to *Tamarix* infestations have been estimated at between $127-291 million per year (Zavaleta 2000).

Anthropogenic Watershed Disturbances that Promote Invasion

Anthropogenic alterations to natural riparian habitat, such as removal of riparian vegetation, land use conversion, regulation of water in rivers and streams, and groundwater pumping, have promoted invasion of *Tamarix* species (Harris 1966, Shafroth *et al.* 2005). By the mid-19th century, clearance of natural riparian vegetation from streams and rivers throughout the southwestern U.S. for building materials and fuel left watercourses open to invasion by *Tamarix*, with no competition from native plants (Harris 1966). Construction of dams and other water diversions throughout the western U.S. in the early to mid-twentieth century led to a decline in soil moisture availability, increased salinization of soils, and a reduced frequency of regular flushing flows that native riparian species require for survival (Jackson *et al.* 1990). Regulation of streams and groundwater pumping promotes conditions favorable to *Tamarix* species, which are able to withstand periods of very low soil moisture, high temperatures, and elevated soil salinity levels that native riparian plants cannot (Sala *et al.* 1996). Furthermore, *Tamarix* species are particularly well-adapted at rapidly colonizing open areas of fresh alluvial deposition created upstream of dams (Harris 1966). Once established, *Tamarix* can withstand fires, re-sprouting much more quickly from below-ground basal crowns than native riparian species (Busch 1995).

Control Methods

Mechanical, chemical and biological control methods have been used to control *T. ramosissima* (Shafroth *et al.* 2005). Mechanical removal was found effective only when both above- and below-ground biomass is removed, due to rapid re-sprouting of the root crown (Taylor and McDaniel 1998). Biomass may be burned effectively after removal from soil or to remove dead standing debris (Taylor and McDaniel 1998, Sprenger *et al.* 2002, McDaniel and Taylor 2003). Both small-scale manual foliar application of herbicides, such as imazapyr and triclopyr compounds, and aerial application of large infestations has proven effective (Duncan and McDaniel 1998, Duncan 2003, McDaniel and Taylor 2003). Several biocontrol agents have been tested and show potential for large-scale biocontrol of *T. ramosissima* (Tracy and DeLoach 1998, DeLoach (and 18 others) 2004). Specialist insects including the saltcedar leaf beetle (*Diorhabda elongata*) have been released throughout the western U.S. and results indicate sizable defoliation of *Tamarix* plants especially in the more northern study sites (DeLoach *et al.* 2004). Researchers at UCSB are testing ecotypes of the *Diorhabda elongata* from different latitudes in cages to see which is best suited for controlling *T. ramosissima* at the Santa Clara River latitude. Their response to day length as well as suitability of the genetic form of *T. ramosissima* present along the Santa Clara River will be tested. In spring 2007, open release of the most appropriate saltcedar leaf beetle ecotype will occur on sites along the Santa Clara River. On the Santa Clara River, several large source infestations should be removed to reduce the potential for seed dispersal.

The Ventura County Resource Conservation District (VCRCD) has completed the Upper Santa Clara River Watershed Arundo/Tamarisk Removal Plan (SCARP) and is initiating a similar plan for the lower watershed with the goal of implementing a long-term (20-years) removal plan for *A. donax* and *Tamarix* spp. (VCRCD 2006; N. Cabanting, pers. comm.). This plan includes a programmatic California Environmental Quality Act (CEQA) and National Environmental Policy Act (NEPA) document and related documentation for the implementation, maintenance, and monitoring of *A. donax* and *T. ramosissima* removal projects within riparian ecosystems (500-year floodplain) of the upper Santa Clara River watershed. The goal of this comprehensive document is to streamline the permitting process for any agency or organization to perform *A. donax* and *T. ramosissima* removal projects of any size within upper Santa Clara Watershed. From September 2005 to March 2006, the California Conservation Corps and the Los Angeles Agricultural Commissioner (for VCRCD) removed all *T. ramosissima* (1-2 acres) from approximately 75 acres of their 297-acre demonstration project at the confluence of the South Fork of the Santa Clara River and San Francisquito Creek in the City of Santa Clarita. Both the cut/paint and foliar application methods were used in removal of *T. ramosissima*. The VCRCD plans to monitor and maintain this removal area for the next 4-5 years and expand the area of removal each year. Official monitoring will be performed in fall 2006, before maintenance work and removal of *T. ramosissima* from other areas is initiated. According to visual inspection in May 2006, the cut/paint method performed the best with little regrowth. Some regrowth was observed with the foliar application.

Key Uncertainties

Outstanding questions regarding *T. ramosissima* ecology and management on the Santa Clara River are categorized and summarized below.

Ecology and Ecological Interactions:

- What is the phenology of *T. ramosissima* (particularly in relation to seed production and viability) along the Santa Clara River?
- What factors contribute to *T. ramosissima* germination and establishment success along a non-regulated watercourse like the Santa Clara River?
- How do elevated nutrient levels and light availability affect *T. ramosissima* growth and invasion success?

- What is the optimal nutrient uptake rate (and range) and efficiency of N, P and K for *T. ramosissima* compared to native riparian plant species?
- What is the relationship between biodiversity, *T. ramosissima* invasion and riparian ecosystem function in Southern California streams like the Santa Clara River?

Control and Monitoring:

- What restoration techniques and composition of native riparian species provide the best replacement vegetation after *T. ramosissima* removal under various conditions?
- What is the effectiveness of various biocontrol agents on *T. ramosissima* in the Santa Clara River?
- What are the most effective remote sensing approach and monitoring methods for *T. ramosissima* removal and spread?
- Where was *T. ramosissima* establishment success the highest after the 2005 flood events?

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