City of Ventura Special Studies

Estuary Subwatershed Study
Assessment of the Physical and Biological Condition of the Santa Clara River Estuary, Ventura County, California

Final Synthesis Report

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EXECUTIVE SUMMARY

Background

The City of San Buenaventura (City; also known as Ventura) owns and operates the Ventura Water Reclamation Facility (VWRF), which discharges tertiary treated municipal wastewater to the Santa Clara River Estuary (SCRE) just south of the City, near the mouth of the Santa Clara River. Under the California Enclosed Bay and Estuaries Policy, discharges of municipal wastewater to enclosed bays and estuaries are to be phased out except in circumstances "when the Regional Board finds that the wastewater in question would consistently be treated and discharged in such a manner that it would enhance the quality of receiving waters above that which would occur in the absence of the discharge." In 1976, the City submitted a plan for effluent utilization that included a "demonstration of enhancement" due to the VWRF discharge of freshwater to the SCRE. This plan indicated that some of the beneficial uses of the SCRE were enhanced by the presence of the discharge. Consequently, Order No. 77-100, adopted by the Los Angeles Regional Water Quality Control Board (LARWQCB) in May 1977, granted the City an exception to the discharge prohibition and allowed continued discharge of the VWRF effluent into the SCRE. However, due to a number of issues regarding the effects of existing discharge practices on existing beneficial uses, the LARWQCB ordered the City to complete three Special Studies under Order R4-2008-0011: an Estuary Subwatershed Study; a Treatment Wetlands Study; and a Phase 1 Recycled Water Market Study. This report addresses the Estuary Subwatershed Study (Study).

Objectives and Approach

The primary purpose of this Study is to confirm that VWRF effluent discharge to the SCRE provides an enhancement of existing beneficial uses as compared to the absence of discharge. This is accomplished by synthesizing findings from previous work within and adjacent to the SCRE and incorporating new monitoring data collected in 2009–2010 to establish technically defensible linkages between flow regimes, water quality, and habitat within the SCRE. The current and historical information is then incorporated into a discussion and analysis of alternative VWRF operations scenarios to determine how modifying VWRF discharge volume and water quality could affect future SCRE habitat conditions and subwatershed ecosystem dynamics under projected conditions of global warming. Scenario development was done in coordination with project stakeholders and was based on planned VWRF upgrades and likely management measures that could be applied to provide enhancement of beneficial uses of the SCRE and surrounding subwatershed.

Geomorphology

Historically, the SCRE contained an unconstrained river channel and main lagoon area that drained to the ocean towards the north. However, the combined impacts of continued encroachment of agricultural land, the establishment of levees for flood protection, and the completion of Harbor Blvd. bridge has decreased the overall area of the SCR by at least 75%, decreased the effective flow width during flood events, and essentially "locked" the SCRE into its present-day location and extent. Since the start of the current El Niño Southern Oscillation (ENSO) wet period in the late 1960s, the frequency and magnitude of storms that can cause substantial geomorphic change has increased. During these major storm events, geomorphic features (e.g., depositional bars, back channels) within and adjacent to the main lagoon are formed and reworked. A large amount of sediment passes through the SCRE and is discharged to the nearshore ocean, with coarser sediment contributing to the building of both offshore and

nearshore deltas, which in turn supply sediment for subsequent mouth berm building and downcoast beach replenishment. The January and February 2005 flood events were two of the largest on record and caused considerable geomorphic change within the SCRE. Since 2005, the lack of sizeable flood flows has enabled the SCRE morphology to rebound and temporarily stabilize. The present-day morphology will likely remain until the next large flood event (i.e., flow equal to or greater than the 5-year flood) occurs.

Hydrology and Hydraulics

Analysis of current and historical data shows that the relative rate of flows into and out of the SCRE varies inter-annually as well as over longer timescales, due to both natural and anthropogenic influences. In general, the Santa Clara River flow is the dominant inflow to the SCRE from the fall through the spring, causing a relatively high mouth breaching and subsequent open-mouth frequency from November through June (i.e., the mouth is open >50% of the time on average). As the SCRE and the rest of southern California are currently experiencing a "wet" ENSO cycle, average annual river inflow volume and open-mouth frequency are currently relatively high. During the summer months, the VWRF effluent discharge is the dominant inflow to the SCRE and the mouth breaching frequency is relatively low during most years (mouth closed >50% of the time on average). When the mouth remains closed for long periods during these dry conditions, the VWRF effluent discharge currently fills the SCRE to quasi-equilibrium "full" elevation that can be maintained for extended periods. The SCRE water balance developed for the study period (October 2009 and September 2010) showed that approximately 80% of the total inflow and outflow volume occurred during and in the days following storm events. The water balance also showed that approximately 90% of the flow into the SCRE during closed-mouth periods from March through September 2010 came from the VWRF (effluent and pond percolation), indicating the relatively large role the VWRF plays in maintaining SCRE volume during the spring and summer months.

Water Quality

Water quality conditions in the SCRE have both direct and indirect impacts on the selected focal species and both the VWRF discharge and ambient water quality have been intensively studied for the past decade. With TMDLs currently in development for nutrients and bacteria in the Santa Clara River, historical data indicates that water within the SCRE periodically exceeded Basin Plan objectives for several parameters including ammonia, biostimulatory substances, pesticides, toxicity, and bacteria. Currently, the VWRF effluent meets discharge effluent limitations set by the LARWQCB; however, variations in receiving water dissolved oxygen (DO) and pH as well as periodic exceedances of ammonia toxicity criteria within the SCRE may be caused by algal productivity and community (i.e., algal and bacterial cells) respiration resulting from nutrient enrichment (i.e., eutrophication). The SCRE nutrient balance shows that nutrient levels rapidly approach the levels in the largest flow source to the SCRE, which is the VWRF effluent during closed mouth conditions. Similarly, during open mouth conditions, nutrient levels in the SCRE rapidly approach the lower levels associated with nutrients in riverine sources or due to tidal exchanges with the nearshore ocean.

Vegetation

As with the SCRE geomorphology, there have been considerable changes to the vegetation assemblage over the past 150 years. General vegetation/habitat types currently comprising the SCRE in order of dominance include riparian, foredune, open water, open beach, disturbed, freshwater wetland, developed, upland, and aquatic vegetation. There has been a recent shift in vegetation successional stage upstream of Harbor Blvd., reflecting the recovery of riparian vegetation after the 2005 flood. In general, the main SCRE lagoon area has relatively little

aquatic vegetation due in large part to the regular occurrence of scouring flood flows. Comparing current vegetation conditions with historical record shows that there have been several key changes that are indicative of some of the primary impacts to the Santa Clara River and the SCRE including a loss of riparian shrublands and woodlands, changes in freshwater wetland distribution, and invasion by non-native plant species.

Focal Species

Four focal species were included in the Order R4-2008-0011, which required this study. The focal species are used to provide an understanding of linkages between species outcomes and ecosystem functioning so that management actions directed to benefit these species will also benefit the larger ecosystem.

- Southern California steelhead (*Oncorhynchus mykiss;* federally listed as endangered) habitat use of the SCRE habitats is currently not well documented. Several stranded *O. mykiss* were identified during stranding surveys following an unseasonal mechanical breach of the SCRE lagoon berm in October 2010. The SCRE provides a migratory corridor for upstream adult steelhead spawners and outmigrant smolts and SCRE conditions may have an important influence on anadromous fish survival. The potential water quality threats to steelhead juveniles in the SCRE include high water temperatures and high turbidity, along with a lack of cover to avoid predation and the prevalence of resident avian predators. DO conditions in the SCRE were frequently below suitable levels for steelhead during 2009–2010. Although the finding of the stranding study indicate that the steelhead are in fact using the SCRE.

- Tidewater goby (*Eucyclogobius newberryi;* federally listed as endangered) use aquatic habitats within the SCRE for their entire life-cycle, are the most common and widespread fish species in the SCRE, and appear to use the backwater habitat found in the VWRF outfall channel (Nautilus Environmental 2005). Major threats to goby in the SCRE include dispersal due to storm flows, dewatering of nests due to unseasonal mechanical breaches, as well as predation due to native and introduced species.

- Western snowy plover (*Charadrius alexandrinus nivosus;* federally listed as endangered) may be found near the SCRE during both the summer (nesting) and winter seasons. Open beach and foredune habitats are used by the species for nesting, which takes place on the ground above the high tide line on barren to sparsely-vegetated beaches. Western snowy plover uses the same open beach and foredune habitats for foraging, by gleaning for invertebrates such and insects and crustaceans found on the sand, in stranded seaweed on the beach, or from low-growing plants. Major threats to western snowy plover include habitat loss and degradation as a function of urban development and use of beaches.

- California least tern (*Sternula antillarum browni;* federally and state listed as endangered) is found near the SCRE only during the summer (nesting) season. California least tern forage in aquatic habitats where small bait fish are abundant, including shallow estuaries, lagoons, coastal ponds, or nearshore waters. Similar to western snowy plover, major threats to California least tern include habitat loss and degradation as a function of urban development and use of beaches.

Functionality of the Santa Clara River Estuary and Subwatershed

The functional linkages among SCRE ecosystem processes, resultant habitats, and biotic needs of the focal species is built on the assumption that natural watershed inputs drive physical processes

that, in turn, determine geomorphic attributes and physical habitat structure of the Santa Clara River, its floodplain, as well as the formation and stability of the SCRE lagoon and mouth berm. The following ecosystem functions were examined to assess VWRF impacts to focal species and human recreational uses of the SCRE and McGrath State Beach: 1) flow regulation; 2) sediment storage and beach building; 3) water quality regulation; 4) aquatic habitat maintenance; 5) wildlife habitat maintenance; and 6) recreational opportunities. The results of the analysis indicate that the current VWRF discharge to the SCRE provides an enhancement of aquatic and wildlife beneficial uses with respect to habitat area up to SCRE water stages near 10 ft NAVD88 under current conditions, with higher water levels bringing losses in rearing habitat for tidewater goby, losses in nesting and foraging habitat for western snowy plover, and a decrease in nesting habitat for California least tern. Current water quality conditions related to nutrient loading and algal growth may limit the suitability of the SCRE for over summering of steelhead; therefore it is determined that current water quality conditions in the SCRE are not enhanced by the VWRF discharge. Finally, current recreational opportunities (defined here as camping within McGrath State Park) may be limited by the current VWRF discharge regime due to increased SCRE stage associated with extended periods of berm closure.

Estimated Future Conditions Due to Climate Change

Potential effects of global climate change relevant to the region of the SCRE include increases in average annual air temperature, mean sea level (MSL), and storm frequency. For the most recent estimates of projected ranges of changes in atmospheric temperature, tide levels, and precipitation patterns/events due to global climate change, data were extracted from published scientific literature identified as being the most relevant or, in the case of the IPCC, internationally recognized analyses with implications for California. Analyses presented in this report indicate that changes in future sea levels are considered to be of greatest importance in determining changes to future SCRE outflows, with less consideration given to increases in air temperatures as well as peak runoff events associated with berm breaching and channel bed scour.

Alternatives Development and Analysis

Six discharge alternatives were developed based on Stakeholder input to represent the full range of potential management conditions of the SCRE, from existing conditions (No Action), to complete discharge elimination, to volume reductions, to additional treatment, including planned VWRF upgrades and potential implementation of constructed wetlands or further mechanical treatment for additional removal of nitrates (NO_3). Evaluation of these alternatives relied upon evaluation of SCRE stage vs. habitat relationships, the SCRE water balance, and simplified models of water quality to assess habitat conditions under current VWRF operations. The existing conditions were then compared to conditions in the absence of the VWRF discharge as well conditions associated with the various water treatment and reuse opportunities identified in the complementary *Treatment Wetlands Study* and *Recycled Water Market Study*.

Comparison of alternatives indicates that maintenance of current discharge will continue to be associated with periods of unsuitable DO levels attributable to excess nutrient loading to the SCRE and continued episodes of flooding as well as man-made breach events during the summer/fall period that may affect the focal species. Alternatives including treatment improvements to reduce nutrient loading to the SCRE were shown to provide benefits to steelhead and least tern. VWRF effluent flow reduction alternatives were shown to benefit focal species habitat and human recreational uses within and adjacent to the SCRE; however, decreases in SCRE stage have been shown to potentially limit rearing habitat for steelhead, suggesting that the current VWRF flows provide some degree of enhancement relative to the expected lower SCRE stage under a complete flow removal alternative.

1 INTRODUCTION

1.1 Background and Purpose

The City of San Buenaventura (City; also known as Ventura) owns and operates the Ventura Water Reclamation Facility (VWRF), which discharges treated municipal wastewater to the Santa Clara River Estuary (SCRE) just south of the City near the mouth of the Santa Clara River (Figure 1-1). With a design capacity of 14 million gallons per day (MGD), the VWRF began operations in 1958 and currently treats approximately 9 MGD of municipal wastewater to tertiary standards (i.e., partial denitrification and filtration) under waste discharge requirements contained in Order No. R4-2008-0011, adopted by the California Regional Water Quality Control Board, Los Angeles Region (Regional Board), on March 6, 2008 (National Pollution Discharge Elimination System [NPDES] Permit No. CA0053651).

Under the Water Quality Control Policy for the Enclosed Bays and Estuaries of California (Enclosed Bay and Estuaries Policy), originally adopted by the State Water Resources Control Board in 1974 and more recently updated as Resolution No. 95-84 on November 16, 1995, discharges of municipal wastewater to enclosed bays and estuaries are to be phased out except in circumstances "when the Regional Board finds that the wastewater in question would consistently be treated and discharged in such a manner that it would enhance the quality of receiving waters above that which would occur in the absence of the discharge." Under Water Quality Order No. 79-20, an evaluation of the degree to which further realization of beneficial uses is attained as a result of the discharge, as compared to the beneficial uses realized in the absence of the discharge, must be conducted. Whether further realization of beneficial uses is attained requires an examination of the degree to which chemical water quality and other aspects of beneficial uses are improved. For example, an examination of the biological benefits, such as additional wildlife habitat, provided by a discharge as compared to the absence of such discharge should be conducted to evaluate the degree to which wildlife related beneficial uses are further realized as a result of a particular discharge.

In 1976, the City submitted a plan for effluent utilization that included a "demonstration of enhancement" due to the VWRF discharge of freshwater to the SCRE. This plan indicated that some of the beneficial uses of the estuary, such as fish and wildlife habitat and non-contact water recreation, were enhanced by the presence of the discharge. Consequently, Order No. 77-100, adopted by the Regional Board in May 1977, granted the City an exception to the discharge prohibition and allowed continued discharge of the VWRF effluent into the SCRE. The demonstration of enhancement in Order No. 77-100 was based on maintaining a minimum daily discharge flow of 5.6 MGD.

The SCRE is currently designated to support eleven of the twenty four beneficial uses protected under the water quality control plan (Basin Plan) for the Los Angeles Region (LARWQCB 1994), including:

- Navigation (NAV)
- Water Contact and Non-Contact Recreation (REC-1, REC-2)
- Commercial and Sport Fishing (COMM)
- Marine Habitat (MAR)
- Estuarine Habitat (EST)

- Wetland Habitat (WET)
- Rare, Threatened, or Endangered Species (RARE)
- Spawning, Reproduction, and/or Early Development (SPWN)
- Migration of Aquatic Organisms (MIGR)
- Wildlife Habitat (WILD)

Although not all uses above are directly affected by discharges from the VWRF, a number of issues arose prior to the adoption of Order R4-2008-0011 regarding the effects of existing discharge practices on these uses. These included a desire to confirm the benefits the VWRF effluent discharge provides to the SCRE, and to the adjacent subwatershed under current conditions, as well as questions regarding the way in which discharge practices could be modified over time to further improve wildlife and habitat uses of those portions of the SCRE directly affected by the VWRF effluent discharge.

Due to these issues, the Heal the Bay and the City agreed that the City would prepare, and the Regional Board ordered the City to complete, a series of three Special Studies under Order R4-2008-0011:

- Estuary Subwatershed Study
- Treatment Wetlands Study
- Phase 1 Recycled Water Market Study

The City submitted work plans to the Regional Board for each of these studies on September 10, 2008 (Workplans) outlining the elements needed to address the information needs of the NPDES permit. The Workplans were approved on December 8, 2008 and form the foundation for the final reports on these three studies. The Treatment Wetlands Study and the Recycled Water Market Study were submitted on March 6, 2010 as required by the NPDES permit. The Estuary Subwatershed Study is required and will be submitted to the Regional Board by March 6, 2011.

This Estuary Subwatershed Study synthesizes findings from previous work within and adjacent to the SCRE and incorporates new monitoring data collected in 2009–2010 to establish technically defensible linkages between flow regimes, water quality, and habitat within the SCRE.

Figure 1-1. Study Area location (Santa Clara River Estuary [SCRE] and adjacent subwatershed near Ventura, CA).

1.2 Study Questions

The primary purpose of this Estuary Subwatershed Study (Study) is to confirm that VWRF effluent discharge to the SCRE provides an enhancement of existing beneficial uses as compared to the absence of discharge. In addition, though not mandatory for purposes of establishing enhancement, the Study additionally evaluates different discharge conditions, such as a decreased volume of discharge, elimination of discharge, or improved treatment of discharge to the SCRE, to determine management measures that might be applied to further improve beneficial uses of the SCRE and surrounding subwatershed. The specific tasks identified in the Study Workplan prepared by the City and submitted to the Regional Board aim to determine how to best manage discharge to maximize beneficial uses by focusing on the nexus between VWRF discharge and SCRE habitat conditions for certain species which currently occupy or use the SCRE and are listed under the state or federal Endangered Species Acts and/or are fully protected species under provisions of the California Fish and Game Code. Consideration is also given to the degree to which recreational uses can be improved, but such considerations should be considered secondary to the evaluation of enhancement provided by the VWRF and any additional improvements that might be made to SCRE wildlife and other beneficial uses. This report addresses the following study questions associated with Workplan tasks for existing conditions and current VWRF discharge operations:

(1) *What is the extent of the SCRE and adjacent subwatershed that is influenced by the VWRF discharge spatially, hydrologically, and functionally?*

(2) *What is the contribution of VWRF flows relative to other components of the overall seasonal and average monthly water balance for the SCRE (i.e., inflows from the lower Santa Clara River and adjacent shallow groundwater aquifer, SCRE discharge, and evapotranspiration)?*

(3) *What is the current relationship between inflows and SCRE stage and volume?*

(4) *What is the current breaching frequency of the SCRE, and what role does VWRF discharge play in breaching activity?*

(5) *What are the current aquatic and terrestrial vegetation assemblages within the SCRE and subwatershed?*

(6) *What is the influence of SCRE stage upon local groundwater levels, SCRE mouth breaching frequency, and SCRE inundation extent, vegetation dynamics, and habitat availability?*

(7) *What is the current eutrophic status of the SCRE?*

(8) *What is the contribution of VWRF flows to current water quality and associated habitat conditions in the SCRE compared to other inflows?*

(9) *Does the VWRF discharge currently enhance habitat conditions in the SCRE or could the discharge provide an enhancement under a different discharge scenario (i.e., overall decreased discharge)?*

1.3 Approach and Report Organization

As a necessary component of understanding the dominant processes maintaining the ecosystem and the specific influence of the VWRF effluent discharge on ecosystem function, this report provides a comprehensive assessment of current Estuary Subwatershed conditions and an overview of known changes to SCRE physical and biological conditions over the past 150 years. This current and historical information is then incorporated into a discussion and analysis of

alternative VWRF operations scenarios to determine how modifying VWRF discharge could affect future SCRE habitat conditions and subwatershed ecosystem functioning under projected conditions of global warming.

The following describes the structure of this report by major section heading, and when applicable, which specific study question number(s) from Section 1.2 is/are addressed:

- **Section 2: The Santa Clara River Estuary Setting**
  - Presents the physical setting of the Santa Clara River and Estuary, including a delineation of the subwatershed boundaries and a description of McGrath Lake (Study Question 1).
- **Section 3: Geomorphology**
  - Describes the morphology of the Santa Clara River channel and lagoon;
  - presents the current SCRE extent based on the equilibrium water surface elevation under low-flow, closed-mouth conditions (Study Question 1);
  - provides a detailed characterization of bed sediments, including their transport, deposition, and role in mouth berm formation;
  - explains historical geomorphic change; and
  - describes the development of the current SCRE stage vs. inundation area and SCRE stage vs. volume relationships (Study Question 3).
- **Section 4: Hydrology and Hydraulics**
  - Describes the process by which the SCRE subwatershed was delineated based on the interaction between VWRF discharge, SCRE stage, and extent of floodplain ground surface saturation (Study Question 1);
  - provides an overview of hydrologic (e.g., dominant water inflows including tides and waves, surface water, and groundwater/subsurface water) and hydraulic (e.g., mouth breaching and estuary stage/inundation dynamics) processes in the SCRE (Study Question 4); and
  - includes the development of contemporary estuary water balance (i.e., timing and volume of water inflows and outflows) from October 2009 to September 2010, which uses data collected for this study in conjunction with other data sources to determine the relative contribution of both monitored and unmonitored water sources (Study Question 2).
- **Section 5: Water Quality**
  - Describes SCRE water quality conditions based on historical data (1997–2008) and data collected as part of this study (2009–2010);
  - compares water quality data with water quality objectives outlined in the Los Angeles Regional Water Quality Control Board Basin Plan (LARWQCB 1994) and regional reference levels for nutrients; and
  - assesses of the trophic status and nutrient balance of the SCRE (Study Question 7).
- **Section 6: Vegetation**
  - Provides a description of contemporary and historical vegetation conditions including overall changes in vegetation (Study Question 5).
- **Section 7: Aquatic Habitats and Focal Species**
  - Describes current and historical aquatic habitat conditions (Study Questions 5 and 6); and

  - discusses species' status, distribution, life history information, and habitat requirements in the SCRE for the two focal fish species, steelhead and tidewater goby (Study Questions 5 and 6).
- **Section 8: Wildlife Habitats and Focal Species**
  - Describes contemporary and historical wildlife habitat conditions (Study Questions 5 and 6); and
  - discusses species' status, distribution, life history information, and habitat requirements in the SCRE for the two avian focal species, western snowy plover and California least tern (Study Questions 5 and 6).
- **Section 9: Functionality of the Estuary and Subwatershed**
  - Includes an overall conceptual model of the SCRE, which examines the functional linkages among SCRE ecosystem processes, resultant habitats, and biotic needs using a focal species approach to evaluate existing beneficial uses; and
  - describes ecosystem functions under current conditions to help evaluate effects of the VWRF discharge on the SCRE, focusing upon functions that are either directly affected by VWRF flows or that may be indirectly affected by management outcomes related to potential changes in these flows (Study Questions 1–8).
- **Section 10: Estimated Future Conditions Due to Climate Change**
  - Summarizes a technical memorandum by Carollo Engineers (2011) of potential effects of global climate change relevant to the region of the SCRE; and
  - includes ranges in future estimates of air temperatures, sea levels, and precipitation in 2050 and 2100.
- **Section 11: Alternatives Development and Analysis**
  - Assesses whether the VWRF effluent discharge enhances beneficial uses, relying upon evaluation of SCRE stage vs. habitat relationships, the SCRE water balance, and simplified models of water quality to assess habitat conditions under current VWRF operations as compared to in the absence of the VWRF discharge;
  - determines whether modifications to the VWRF effluent discharge might better optimize beneficial uses, primarily relying on evaluation of SCRE stage vs. habitat relationships, the SCRE water balance, and simplified models of water quality under several VWRF discharge alternatives including current VWRF discharge conditions and a no discharge scenario; and
  - addresses a range of discharge conditions in terms of annual or seasonal VWRF flow reductions and various water treatment and reuse opportunities identified in the complementary *Recycled Water Market Study* (Carollo Engineers 2010a) and *Treatment Wetlands Study* (Carollo Engineers 2010b) (Study Question 9).

1.4 Focal Species Selected for the Estuary Subwatershed Study

We are using a focal species approach to provide a logical basis for developing and understanding effects of VWRF flows on broader terrestrial and aquatic habitat uses within the SCRE. Focal species are often used to understand broader ecosystem linkages so that management actions directed to benefit these species will also benefit the larger ecosystem. Four focal species were selected during the study planning phase and were included in Order R4-2008-0011 to examine whether the VWRF effluent discharge to the SCRE provides an enhancement of existing beneficial uses and to determine how best to optimize beneficial uses—in this case prioritizing

evaluation of beneficial uses related to wildlife including fish and avian species and their habitat. The four selected focal species are:

- southern California steelhead (*Oncorhynchus mykiss*),
- tidewater goby (*Eucyclogobius newberryi*),
- western snowy plover (*Charadrius alexandrinus nivosus*[1]), and
- California least tern (*Sternula antillarum browni*).

As stated above, aquatic and wildlife habitat are not the only SCRE beneficial uses potentially affected by the VWRF. Further, this is by no means an exhaustive list of all species that are linked to habitats and ecological processes within the SCRE. However, the evaluation of discharge effects on the focal species is considered to be representative of effects on other sensitive species using or occupying the SCRE to yield a reliable assessment indicative of the degree to which wildlife related beneficial uses are enhanced, and might further be improved through different discharge management measures.

1.4.1 Southern California steelhead

The steelhead population found in the Santa Clara River is part of the southern California steelhead DPS and is federally listed as endangered. Steelhead have economic value because they support recreational fisheries and have a high public interest value in the Santa Clara River watershed since they represent one of the few remaining populations of southern California steelhead. Although steelhead must use the SCRE as a migratory corridor, little information regarding steelhead use of the SCRE as rearing habitat has been available. However, recent *O. mykiss* observations in the SCRE lagoon as well as detailed information on rearing in other similar coastal lagoons suggests that the SCRE currently provides rearing habitat for steelhead. A detailed description of this species' listing status, distribution, life history, habitat requirements, and use of the SCRE are discussed in Section 7.2.1.

1.4.2 Tidewater goby

The tidewater goby population found in the SCRE is federally listed as endangered. Tidewater goby use aquatic habitats of the SCRE for their entire life-cycle. This species is a weak disperser, and the SCRE population represents one of the few populations within this region of the southern California Coast. A great deal of information has been collected regarding tidewater goby use of the SCRE, which was used to determine habitat requirements and the identification of potential management actions to support species recovery. A detailed description of this species' listing status, distribution, life history, habitat requirements, and use of the SCRE are discussed in Section 7.2.2.

[1] There has been a proposal to revise the taxonomic status of *C. alexandrinus nivosus* to a full species, *C. nivosus* based on genetic analyses (Küpper et al. 2009), though the change has not been formally adopted by the American Ornithologist's Union.

1.4.3 Western snowy plover

Western snowy plover, federally listed as endangered, currently uses habitats associated with the SCRE for breeding, foraging, and overwintering. This species has narrow habitat requirements since it requires undisturbed beaches for nesting. Habitat loss and degradation as a function of urban development and use of beaches is a major threat to the species. Western snowy plover has a high public interest as a popular species to view by recreational birders. There are quantitative data about the abundance, distribution, and habitat preferences of this species in the SCRE because of surveys conducted near the SCRE for over 25 years (R. Smith, Ventura Audubon Society, pers. comm., 2010). A detailed description of this species' listing status, distribution, life history, habitat requirements, and use of the SCRE are discussed in Section 8.2.1.

1.4.4 California least tern

California least tern, federally and state listed as endangered, currently uses habitats associated with the SCRE for breeding and foraging. This species has narrow habitat requirements since it requires undisturbed beaches for nesting. Habitat loss and degradation as a function of urban development and use of beaches is a major threat to the species. California least tern has a high public interest as a popular species to view by recreational birders. There are quantitative data about the abundance, distribution, and habitat preferences of this California least tern in the SCRE because of surveys that have been conducted near the SCRE for over 25 years (R. Smith, Ventura Audubon Society, pers. comm., 2010). A detailed description of this species' listing status, distribution, life history, habitat requirements, and use of the SCRE are discussed in Section 8.2.2.

2 THE SANTA CLARA RIVER ESTUARY PHYSICAL SETTING

The SCRE is located at the interface between the Santa Clara River and the Pacific Ocean. Historically, the SCRE was an expansive ecosystem that included an open-water lagoon and a series of channels that supported intertidal vegetation; the current SCRE surface extent is approximately 75–90% of what it once was. For this project, the SCRE subwatershed is defined as the surrounding floodplain area where estuary infilling during closed-mouth, low river flow conditions is known to affect water-table elevation and influence sensitive habitat and human recreation. This includes areas where the ground surface is equal or less than an elevation of approximately 3 m (10 ft) NAVD88, or the maximum SCRE stage currently reached during closed-mouth, low-flow conditions. This area extends north to the VWRF and Ventura Harbor inlet, south into McGrath State Beach and to the southern edge of McGrath Lake, and east approximately 1.4 km (0.8 miles) upstream of Harbor Blvd. bridge (Figure 1-1).

The SCRE is a river mouth lagoon that lies atop the Oxnard Plain, the seaward edge of the Ventura Basin, a large structural trough that plunges westward into the Santa Barbara Channel (Orme 1982, as cited in O'Hirok 1985). The sedimentary layers beneath the Oxnard Plain make up the Mound groundwater sub-basin, which is part of the larger Santa Clara-Calleguas groundwater basin. The Mound sub-basin consists of several aquifers composed of sand and gravel separated by low permeability clay layers. The surface Semi-perched aquifer has a direct hydrologic connection with the SCRE, receiving or supplying water depending on differences in the estuary stage and local water table elevation.

To the south of the SCRE lies McGrath Lake, a natural surface expression of the Semi-perched aquifer water table (ESA 2003). McGrath Lake is the only remaining freshwater, coastal back-dune lake in Southern California (URS Corporation 2005) and currently has an area of approximately 11 ha (28 ac) and an average depth of approximately 0.6 m (2 ft) (Kennedy/Jenks Consultants 2002). Unlike the SCRE, the lake has no natural ocean outlet, although breaches can occur from wave overwash and artificial means (YCE 1999, as cited in Kennedy/Jenks Consultants 2002).

A detailed description of the contributing Santa Clara River watershed physical characteristics, climatic conditions, and subsequent flow and sediment transport dynamics to the SCRE and the Pacific Ocean is given in Appendix A.

3 GEOMORPHOLOGY

This section describes the morphology of the lower Santa Clara River channel and SCRE lagoon, both currently and historically. Of particular importance is that the network of channels and bars that comprise the main lagoon are formed and reworked during storm events and subsequent tidal exchange while the mouth remains open. This has been especially apparent over the past several years following 2005 storm events. Recent aerial photographs indicate that the SCRE total area has decreased substantially since 2005. The present-day morphology appears stable and will likely remain until the next large storm event.

This section also provides a detailed characterization of bed sediments, including their transport, deposition, and role in mouth berm formation. Primarily during high intensity, low recurrence storm events, the Santa Clara River discharges a considerable amount of sediment. In general, the coarser sediment that is delivered from the Santa Clara River during storm events contributes to the building of offshore and nearshore deltas, which in turn supplies sediment that builds the mouth berm and causes mouth closure during periods of low river discharge. Offshore delta deposits can vary considerably in volume and spatial extent as a function of storm magnitude and frequency.

Finally, a historical geomorphic change is discussed. During the mid-19th century, the SCRE had a much larger total area, or footprint, than currently exists, contained an unconstrained river channel and main lagoon area that drained to the ocean towards the north. Between 1953 and 1957, the continued encroachment of agricultural land and the establishment of levees for flood protection resulted in an approximate 75% decrease in historical channel and adjacent delta areas (Swanson et al. 1990). By 1968, an extensive flood protection levee was established along the north bank upstream of the main lagoon, and Harbor Blvd. bridge was completed across SCRE where the channel enters the main lagoon. Combined, these features decreased the overall area of the SCRE, decreased the effective flow width during flood events, and essentially “locked” the SCRE into its present-day location and extent. Since the start of the current El Nino Southern Oscillation (ENSO) wet period in the late 1960s, the frequency and magnitude of storms that can cause substantial geomorphic change (i.e, storms with a peak discharge of at least 100,000 cfs) has increased.

3.1 Current Conditions

3.1.1 Morphology

The SCRE is composed of a river channel that empties into a main lagoon impounded by a seasonally closed-mouth berm. Within this area, the most current bathymetric survey (2005 LiDAR surveys by both Ventura County and the United Stages Geologic Survey [USGS]) indicates that bed elevations range from 1.2 m (4.0 ft) NAVD88 in the current mouth channel location to 3.1 m (10.2 ft) NAVD88 at the upstream extent of inundation during low-flow, closed-mouth conditions, with a median bed elevation of approximately 1.9 m (6.5 ft) NAVD88 (Figure 3-1). The beach berm elevation currently varies between approximately 4 m (14 ft) and 5 m (17 ft) NAVD88, which is approximately between 2.6 and 3.5 m (8.7 and 11.7 ft) above mean higher high water (MHHW). On average, the SCRE bed is currently approximately 1 m (3 ft) below the adjacent floodplain that defines the southern and northern boundaries. Swanson et al. (1990) reported a ground surface elevation range between 0.3 m (1 ft) above mean sea level (MSL) (1.1 m [3.7 ft] NAVD88) on the bed at the SCRE mouth and 4 m (15 ft) MSL (5.3 m [17.7 ft]

NAVD88) on the mouth berm, indicating that the current elevation range is very similar to what existed almost 20 years ago.

The main lagoon contains a network of channels and bars that are formed and reworked during storm events and subsequent tidal exchange while the mouth remains open. The total inundated area, defined by the maximum inundation extent within the river channel and main lagoon under closed-mouth, low-flow conditions, is currently 81 ha (200 ac) and extends approximately 750 m (2,460 ft) upstream of Harbor Blvd. bridge (Figure 3-1). Approximately 16% of the area is occupied by tidal channels in the lower SCRE towards the mouth berm (bed elevation between 1.2–1.5 m [4.0–5.0 ft] NAVD88); the remaining area is made up of depositional bars, high flow channels, and the main SCRE channel. The overall SCRE bathymetry results show a stage-volume relationship with a fairly constant slope once the stage is above the tidal channels; however, the large depositional bar along the left bank upstream of Harbor Blvd. bridge causes the stage-area relationship to be steep (i.e., little area increase associated with an increase in stage) between approximately 2.1 and 2.7 m (7.0 and 9.0 ft) NAVD88 (Figures 3-2 and 3-3).

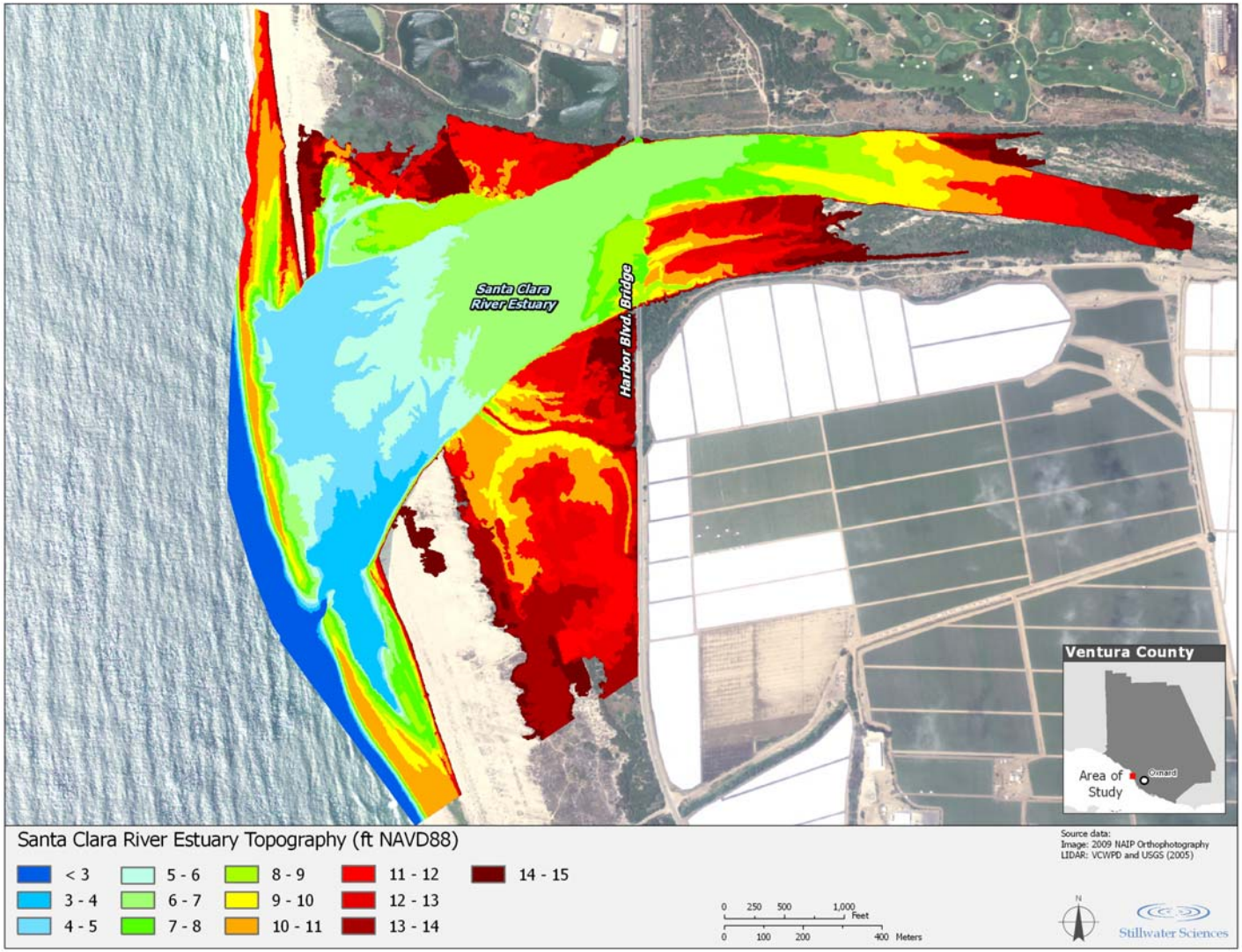


Figure 3-1. SCRE and surrounding floodplain topography (from 2005 LiDAR).

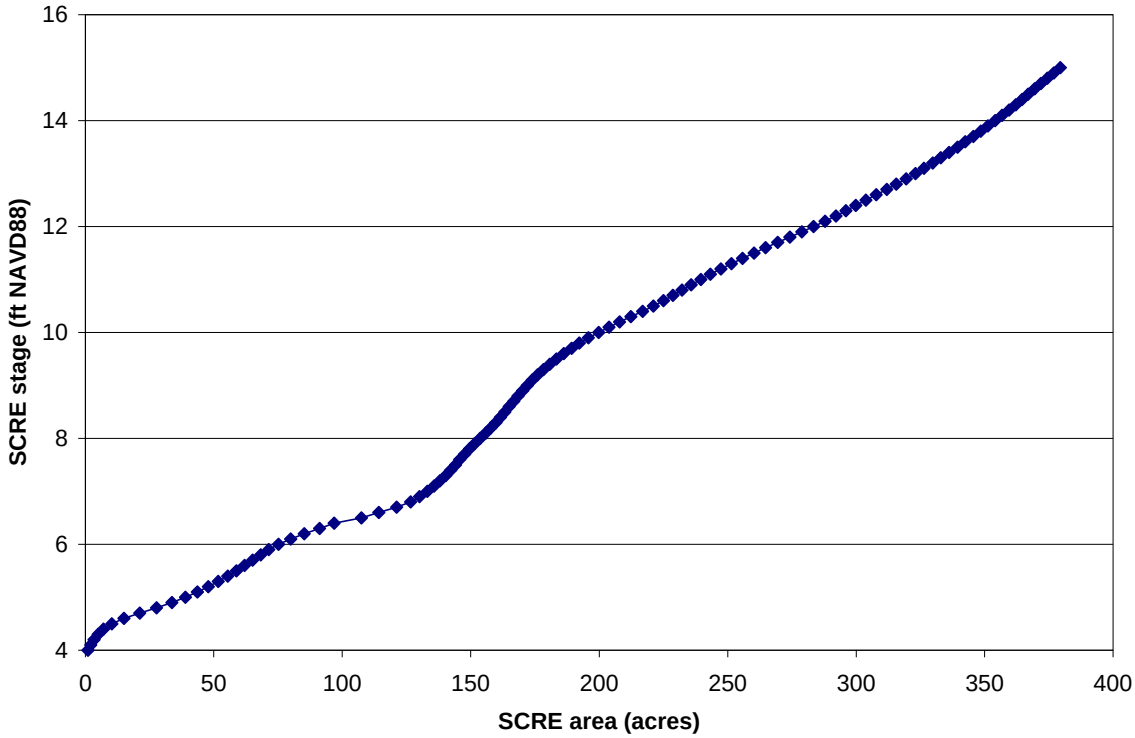


Figure 3-2. Relationship between SCRE stage and inundated area (derived from 2005 LiDAR).

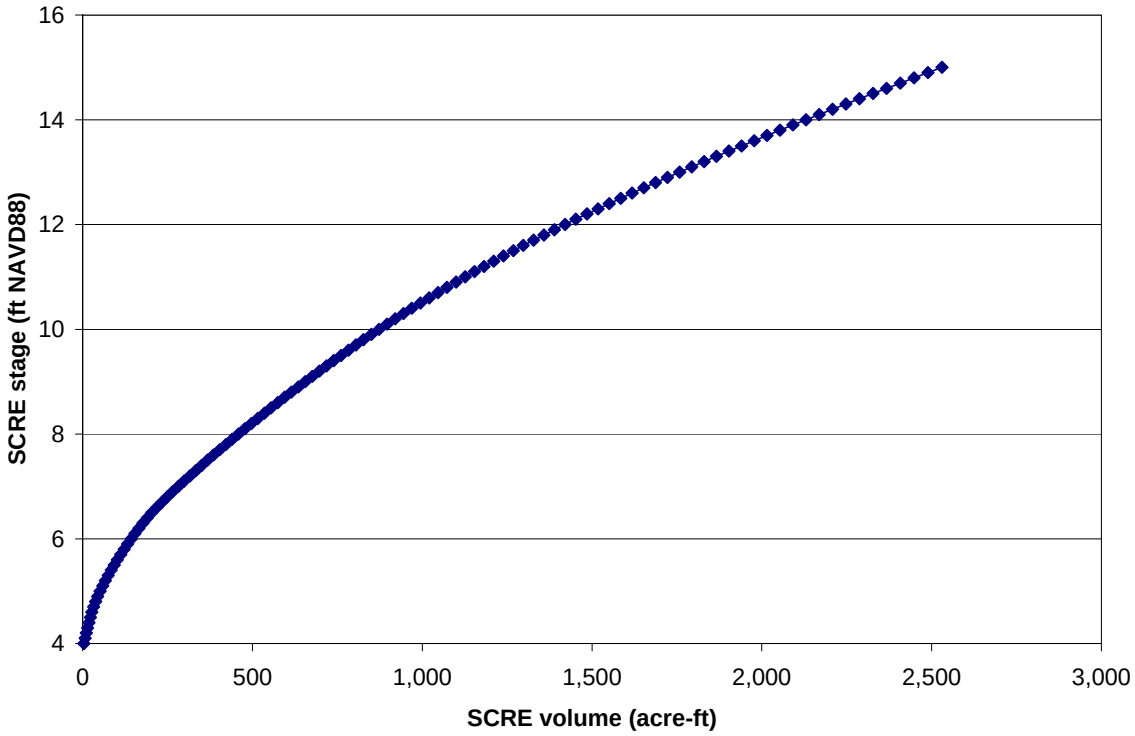


Figure 3-3. Relationship between SCRE stage and inundated volume (derived from 2005 LiDAR).

Over the past several years, the SCRE has been responding to morphologic changes induced by 2005 storm events. In January of 2005, a major storm hit the Santa Clara River watershed, delivering approximately 22 cm (9 in) of rain over 72 hours in Santa Paula, CA (Ventura County Watershed Protection District [VCWPD] Station 245A). The result was a peak discharge of approximately 136,000 cfs in the lower Santa Clara River at the Freeman Diversion (VCWPD Station 724), the second-largest flow recorded in the watershed. A subsequent storm in February 2005 delivered approximately 16.2 cm (6.4 in) of rain over 72 hours in Santa Paula, CA causing a peak discharge of approximately 82,000 cfs in the lower Santa Clara River. Comparison of 2000 SCRE bathymetry (Entrix, Inc. 2002a) with LiDAR-derived bathymetry collected after the February 2005 storm (from Ventura County and the USGS) downstream of Harbor Blvd. bridge shows the 2005 storms increased the SCRE volume and decreased the median bed elevation (Figures 3-4 and 3-5). However, recent aerial photographs indicate that the SCRE total area has decreased since 2005, which suggests that the volume has likely decreased to a pre-2005 state.

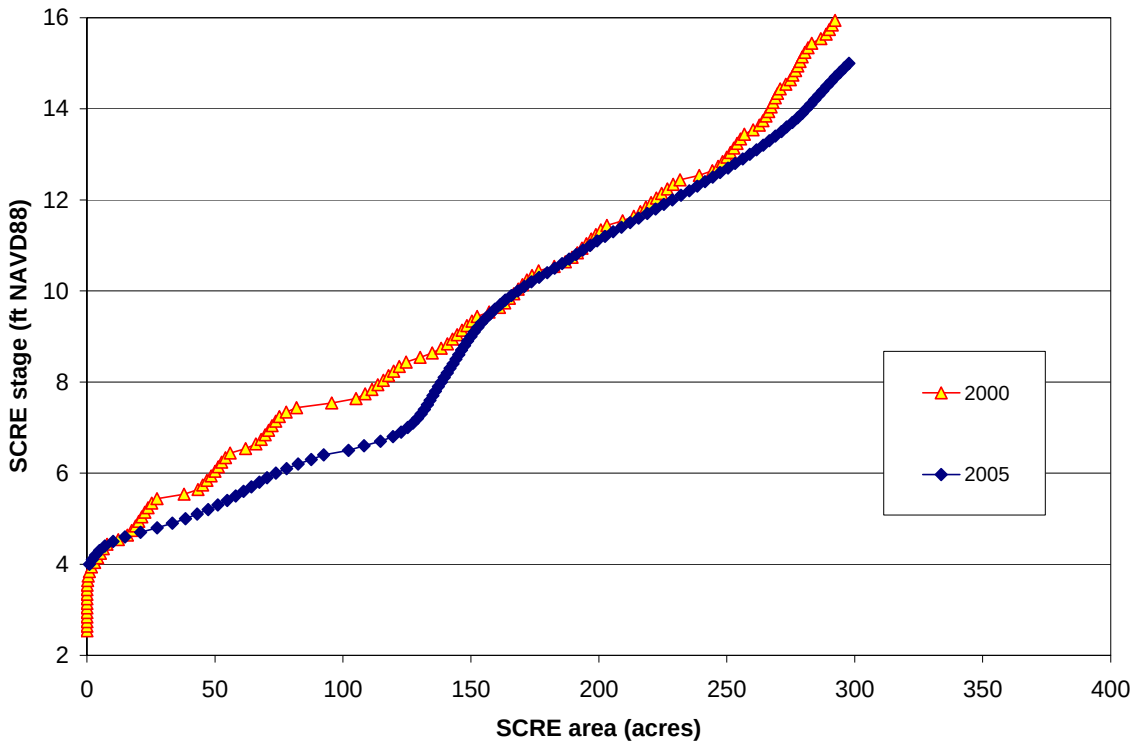


Figure 3-4. Comparison of SCRE inundated area downstream of Harbor Blvd. bridge with increasing stage for 2000 and 2005 estuary conditions.

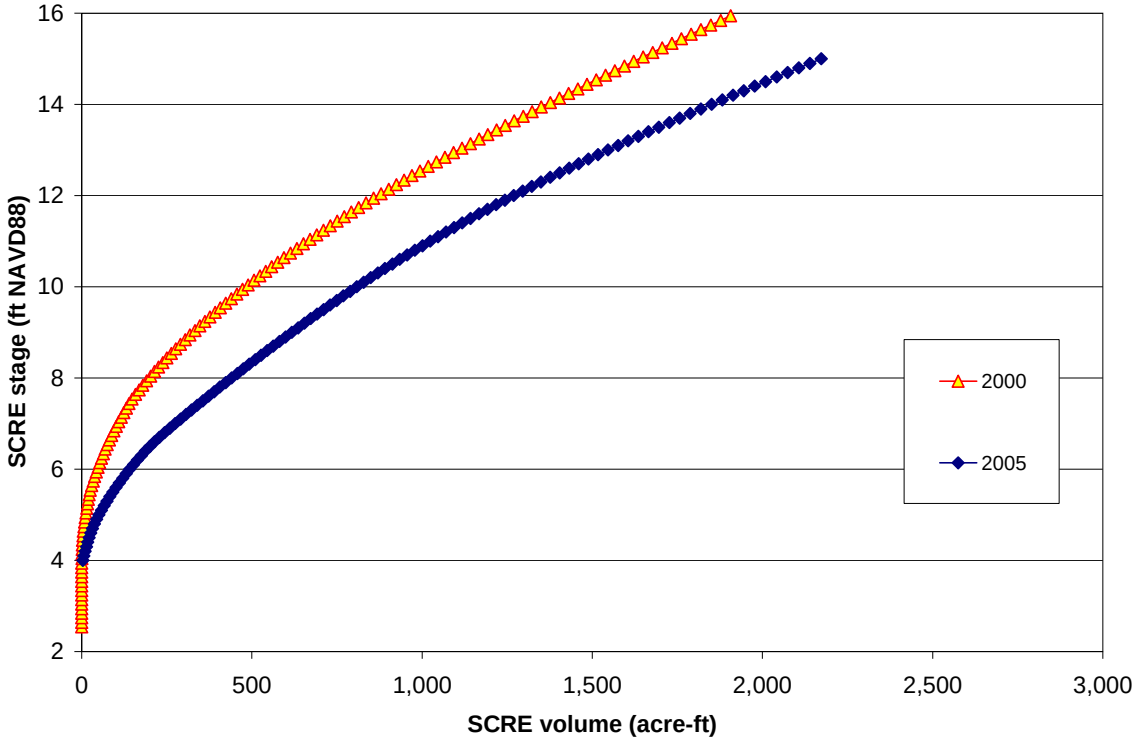


Figure 3-5. Comparison of SCRE inundated area downstream of Harbor Blvd. bridge with increasing stage for 2000 and 2005 estuary conditions.

3.1.2 Bed substrate

Bed sediments within the SCRE are characterized by stratified layers of coarse sand and range in size from clay- to boulder-sized particles. In general, the SCRE exhibits a pattern similar to most river-mouth lagoons where the surface bed particle size decreases moving downstream from the river-lagoon transition (i.e., zone where flow velocity decreases and larger sediment drops out of the transported load) to the lagoon-ocean interface. Estuary-wide and localized particle size distribution is strongly influenced by both storm-induced river flows and deposition during tidal exchange. During most storm flows, sediment ranging in size from silt/sand to gravel is delivered to the SCRE (see Section 3.1.3 for discussion of average annual sediment yield). During the peak discharge of larger storm events, flow velocity and associated shear stress in the dominant channel can be high enough to transport cobble and boulder-sized sediment far into the SCRE or through the estuary completely and out to the Santa Barbara Channel (O'Hirok 1985). Deposition of very fine sediment (i.e., silt and clay-sized material) on the surface above coarser sediment then occurs as storm flows recede (USFWS 1999) and also as a result of flocculation (aggregation of fine sediment) induced by river and ocean water mixing during open-mouth conditions (O'Hirok 1985). Therefore, between larger floods that cause significant bed scour and coarse sediment transport, lesser flood events result in transport and deposition of finer sediment above a coarser substrate.

Sediment data collected during a recent period between large storms (2002–2004) give an indication of the degree of inter- and intra-annual bed sediment size variability (Figure 3-6, Table 3-1). Surface sediment samples taken in spring of 2002 (Entrix, Inc. 2002a) downstream of

Harbor Blvd. bridge were composed, on average, of 13% gravel, 64% sand, and 22% silt and clay. Areas where bed sediment was greater than 12% gravel (i.e., the coarsest areas) were restricted to the region upstream of the outfall channel. The D_{50} (or, particle size for which 50% of the sediment is finer) of the samples ranged from 0.04 mm (coarse silt) in the main channel to 2.4 mm (fine gravel) towards the upstream extent of the SCRE, with a mean D_{50} value of 0.7 mm (coarse sand) for all samples (Entrix, Inc. 2002a). Particle size data from sites reoccupied over the following two years (spring 2003, fall 2003, and spring 2004) showed an overall increase in average gravel content towards Harbor Blvd. bridge and an overall increase in sand content towards the mouth berm. The bed at the station located in the lagoon towards the barrier beach transitioned from a mixture of half sand and half silt in spring 2002 and fall 2003 to mostly sand in spring 2004. The shift in median particle size at the station in the lagoon channel is attributed to normal seasonal flow dynamics (i.e., high winter flows caused scour and result in coarser bed sediment; quiescent conditions that followed resulted in fine sediment deposition and a relative decrease in the coarser sediment present).

Figure 3-6. Sediment sample locations from fall 2002 (Entrix, Inc. 2002a), fall 2003, and spring 2004 (Nautilus Environmental 2005) sampling efforts.

Table 3-1. SCRE bed particle size data for 2002–2004.

Location	Entrix (2002) sample ID	Spring 2002		Nautilus Environmental (2005) sample ID	Fall 2003	Spring 2004
		D_{50} (mm)	Grain size		Grain size	Grain size
1	B3	0.1	0% gravel 47% sand 53% silt+clay	A-1	1% gravel 51% sand 48% silt+clay	12% gravel 84% sand 4% silt+clay
2	B4	0.4	0.5% gravel 96% sand 4% silt+clay	A-2	16% gravel 81% sand 3% silt+clay	2% gravel 96% sand 3% silt+clay
3	B5	0.5	1% gravel 98% sand 1% silt+clay	A-3	<1% gravel 99% sand 1% silt+clay	<1% gravel 92% sand 8% silt+clay
4	B1	0.5	12% gravel 56% sand 31% clay	B-1	2% gravel 95% sand 3% silt+clay	2% gravel 97% sand 1% silt+clay
5	B6	1.0	35% gravel 63% sand 2% silt+clay	B-3	2% gravel 95% sand 3% silt+clay	9% gravel 88% sand 3% silt+clay
6	B7	0.04	7% gravel 32% sand 61% silt+clay	B-4	0% gravel 39% sand 61% silt+clay	<1% gravel 92% sand 8% silt+clay
7	B8	2.4	50% gravel 49% sand 1% silt+clay	C-3	13% gravel 84% sand 3% silt+clay	29% gravel 70% sand 1% silt+clay

More recent bed sediment data collected as part this study from fall 2009 through spring 2010 illustrate the current conditions (i.e., period between large storms that includes lesser, depositional storm events) within the SCRE. In general, the SCRE bed surface is still composed predominantly of sand throughout, with coarser sediment being present primarily towards the upstream estuary extent and within and adjacent to the main lagoon channel (i.e., dominant channel during high flow events). Upstream of Harbor Blvd. bridge, the main lagoon channel has large patches of coarse gravel (gravel with sand and cobbles) and coarse sand (sand with gravel and cobble) with adjacent high depositional bars containing coarse sediment (gravel and cobble) (Figure 3-7). Approximately 100 m (330 ft) upstream of the bridge, there is a distinct break in the main channel slope and associated transition from coarser to finer bed sediment going upstream to downstream. Downstream of Harbor Blvd. bridge, the main lagoon channel currently flows towards the south, resulting in coarser surface sediment in the southern portion of the SCRE and finer surface sediment towards the northern portion (Figure 3-8). Towards the mouth berm and in the outfall channel, surface sediment is predominantly sand. In addition to the presence of coarser sediment in the channel due to flow hydraulic effects, lateral erosion of the large depositional bar in the southern SCRE deposited during the 2005 flood events is currently exposing very coarse sediment (cobbles and boulders) below the surface sand layer. The relatively low-magnitude storms during the winter of 2010 resulted in fine sediment (silt and clay) deposition within the more quiescent portions of the SCRE, including the outfall channel. This fine sediment layer will remain until scoured by subsequent larger flows.

Figure 3-7. Representative bar sediment size along the main SCRE channel upstream of the Harbor Blvd. bridge (winter 2010).

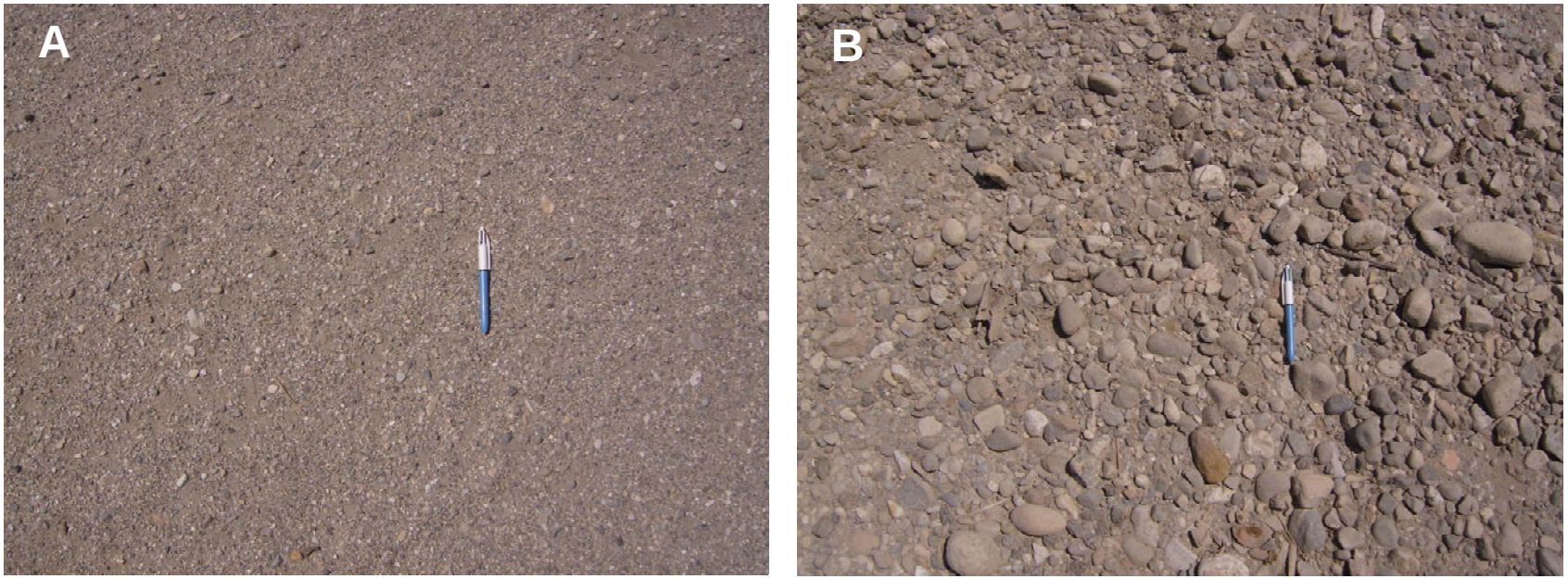


Figure 3-8. Representative bed sediment size in the northern portion (A) and southern portion (B) of the SCRE downstream of the Harbor Blvd. bridge (winter 2010).

3.1.3 Sediment transport, deposition, and mouth berm formation

The Santa Clara River discharges a considerable amount of sediment primarily during high intensity, low recurrence storm events (Section 3.1.2). In general, the coarser sediment (> 0.065 mm, or coarse sand and larger) that is delivered from the Santa Clara River during storm events contributes to the building of offshore and nearshore deltas, which in turn provide sediment for littoral transport and down-coast beach deposition and supplies sediment that builds the mouth berm and causes mouth closure during periods of low river discharge. Offshore delta deposits can vary considerably in volume and spatial extent as a function of storm magnitude and frequency. The largest recorded input of sediment from the Santa Clara River to the offshore delta occurred during the floods of 1969, where approximately 13.0 million yd^3 of sediment was deposited (Noble Consultants 1989), and in 2005 where approximately 6 million yd^3 of sediment was transported out of the SCRE (Barnard et al. 2009). High discharge events that deposit sediment to the offshore delta are dominated by hyperpycnal flows (i.e., sediment-laden flows that are more dense than ocean water) from the Santa Clara River that cause sediment to pass through the SCRE and nearshore zone (Warrick and Milliman 2003). Hyperpycnal events with the lowest exceedance probabilities (>100 year recurrence interval) have the potential to deposit sediment out of the adjacent littoral cell in offshore basins, essentially resulting in a net loss of sediment from the system (Warrick and Milliman 2003). This sediment routing can also lead to local bed erosion within the SCRE.

Nearshore deltas can form at the mouth of the Santa Clara River during both hyperpycnal and more frequently occurring hypopycnal (i.e., condition in which river discharge is less dense than ocean water) flows. These features are apparent at Santa Clara River mouth following the April 1958 and February 2005 storm events (Figure 3-9). Following flood events, nearshore delta features are subject to immediate wave impact and eroded sediment goes directly to longshore transport or is deposited on the offshore delta. The average wave height along in this area is approximately 1 m (3 ft) but ranges from 0.3 to 7 m (0.9 to 22 ft) (Orme 1982, as cited in O'Hirok 1985). Sediment that is immediately transported down-coast contributes to short-term beach building. Sediment that is stored in the offshore delta is eroded and transported down-coast eventually or is re-deposited at the mouth during low-flow conditions when the mouth berm forms.

Sediment availability from the adjacent deltas in concert with sediment from longshore transport, river discharge dynamics, and tidal dynamics combine to control the formation and maintenance of the Santa Clara River mouth berm. Owing to predominant deposition along this segment of the Pacific coast, the foreshore is rarely steepened enough to generate surging waves. The width of the surf zone is a function of the slope of the nearshore bottom, wave height, tidal stage, and discharge. During winter months, high-energy storms cause the mouth to remain open by both onshore wave action and increased offshore river discharge. The submerged delta formed off the SCRE following large winter river flows can create surf zones greater than 250 m (820 ft) wide. When tidal range decreases during periods of low river discharge, the sediment transport capacity decreases due to a decrease in tidal prism (or, volume of water exchanged between the SCRE and the ocean), wave action causes sediment to deposit at the mouth, and the mouth berm forms. Lower-intensity wave action and lower river discharges in the summer months facilitate onshore sediment transport and sediment deposition at the mouth, increasing mouth closure frequency and duration compared with the rest of the year (Swanson et al. 1990; Smith 1990, as cited in ESA, 2003). A decreased littoral sediment supply due to sediment trapping by dams and the dredging around Ventura Harbor has been suggested to have slowed the rate of mouth berm formation compared to historical conditions (Nautilus Environmental 2005).

Figure 3-9. Aerial photographs showing a nearshore delta deposit outside the SCRE mouth following 1958 (A) and 2005 (B) storm events.

3.2 Historical Geomorphic Change

The lower Santa Clara River and SCRE have undergone considerable geomorphic change over the past 150 years since European-American settlement due to a combination of land-use practices and climatic conditions. Historically, the SCRE was an expansive ecosystem that included an open-water lagoon and a series of channels that supported intertidal vegetation. Land development since the mid-19th century has resulted in a 75% (Swanson et al. 1990, ESA 2003) to 90% (Nautilus Environmental 2005) decrease in overall SCRE area and available habitat, and the confinement of flood flows by levees. Following the period of intensive development, a shift in precipitation patterns associated with the El Nino Southern Oscillation (ENSO) has resulted in a wet-period ENSO cycle in southern California over the past 40 years, resulting in a higher frequency and duration of large storms.

In an order to elucidate specific causes for geomorphic change and to help inform future geomorphic conditions, estuary morphologic change (location, extent, confinement, complexity) since the early 19th century was assessed in the context of major storm events and known development actions (e.g., onset of floodplain agriculture, construction and/or modification of levees, bridge building). Data sources included pre-existing descriptions of morphologic change in and around the SCRE (e.g., Swanson et al. 1990, Schwartzberg and Moore 1995, ESA 2003, Nautilus Environmental 2005, Barnard et al. 2009), orthorectified topographic maps and aerial photographs from 1855 through 2009 (provided by the U.S. Coastal and Geodetic Survey [USC&GS], the California Coastal Conservancy, the VCWPD, and Google Earth), and annual peak flow data for the lower Santa Clara River from 1815 through 2009. Annual peak flow data prior to 1950 was derived from a correlation with historical precipitation at Santa Paula, CA (Stillwater Sciences 2007); after 1950, the flow data come from gages monitored by the USGS (Station 11114000) and VCWPD (Station 723) (see Section 4 for a more detailed hydrology discussion). From this analysis, five distinct morphologic periods were identified (Table 3-2). These periods overlap somewhat with the five watershed land-use “impact periods” described in Appendix A; however, distinct storm and localized development impacts result in different bounding years for the various periods.

Table 3-2. Estuary morphologic periods (early 19th century to present).

Time period	Description	Major storms (≥Q 5-yr, or 50,000 cfs)[a,b,c]
Pre-settlement Early 1800s–1850s	Relatively pristine estuary ecosystem that supported extensive tidal and upland habitat.	1815 (78,406) 1825 (120,526) 1833 (52,796) 1840 (102,237)
Initial settlement 1850s–Early 1900s	Portions of the estuary were converted to agricultural land and channel infilling begins. Impact to estuary increased but estuary continued to maintained tidal and upland habitat.	1862 (111,132) 1884 (108,412) 1890 (82,690)
Agriculture Early 1900s–Late 1940s	Wide-spread land reclamation, channel infilling, and the start of levee building. Beginning of estuary confinement.	1907 (51,853) 1914 (55,522) 1928 (175,000)[d] **1938** (120,000) 1941 (131,552) 1943 (58,459)
Levees and Infrastructure Late 1940s–Early 1970s	Major development and infrastructure within and adjacent to estuary. Estuary becomes very confined by flood control levees.	**1958** (52,200) 1966 (51,900) **Jan 1969** (165,000) **Feb 1969** (150,000)
ENSO wet-period Early 1970s–2005	Development within the estuary subsides. Estuary very confined; levees contain 2005 floods.	**1973** (58,200) **1978** (102,200) **1980** (81,400) **1983** (100,000) **1992** (104,000) **1995** (110,000) **1998** (84,000) **2005** (136,000)
Present day Post-2005	Estuary in quasi-stable state following 2005 floods.	

[a] Known ENSO years (i.e., 1938–2005) are bold.
[b] The 5-year flood is used as the threshold flow because that flow's ability to cause vegetation scour and rework depositional bars (Swanson et al. 1990, as cited in Nautilus Environmental 2005).
[c] Estimated peak flood values from the correlation with Santa Paula precipitation are underlined (Stillwater Sciences and URS Corporation 2007).
[d] The St. Francis Dam failure; peak flood estimate from Begnudelli and Sanders 2007.

In the first half of the 19th century, the SCRE was a relatively pristine ecosystem that was characterized by a lagoon and extensive network of tidal sloughs and blind tidal channels that supported a mosaic of tidal, forested wetland, and upland vegetation. By the 1820s, forest clearing for agricultural purposes was prevalent in the upstream watershed but had not yet extended down to the SCRE. The only evidence of human impacts during the early- to mid-19th century is a road shown on the USC&GS 1855 map that crossed the lower Santa Clara River approximately 1.5 km (0.9 mi) upstream of the present-day Harbor Blvd. bridge location (Figure 3-1). During this time period, the estuary had a much larger total area, or footprint, than currently exists, contained an unconstrained mainstem river channel and main lagoon area that drained to the ocean towards the north. During extreme storm events, the entire SCRE complex was capable of moving across the Oxnard Plain as the mainstem channel banks eroded and the channel

migrated or avulsed (i.e., jumped to a completely new location). This resulted in the periodic creation of new, and abandonment of existing, estuary geomorphic features and associated tidal, wetland, and upland habitats. Geologic evidence suggests that the lower Santa Clara River channel and estuary may have moved locations over the past several thousand years (Clahan 2003). Although there is no evidence to suggest that the SCRE moved from its present-day location in the early 19th century, there are four storms that had the potential to cause substantial geomorphic change (Table 3-2). Of these, the 1825 and 1840 events are thought to have had peak discharges above the present-day 10-year flood in the lower Santa Clara River and therefore most likely had the greatest geomorphic impact.

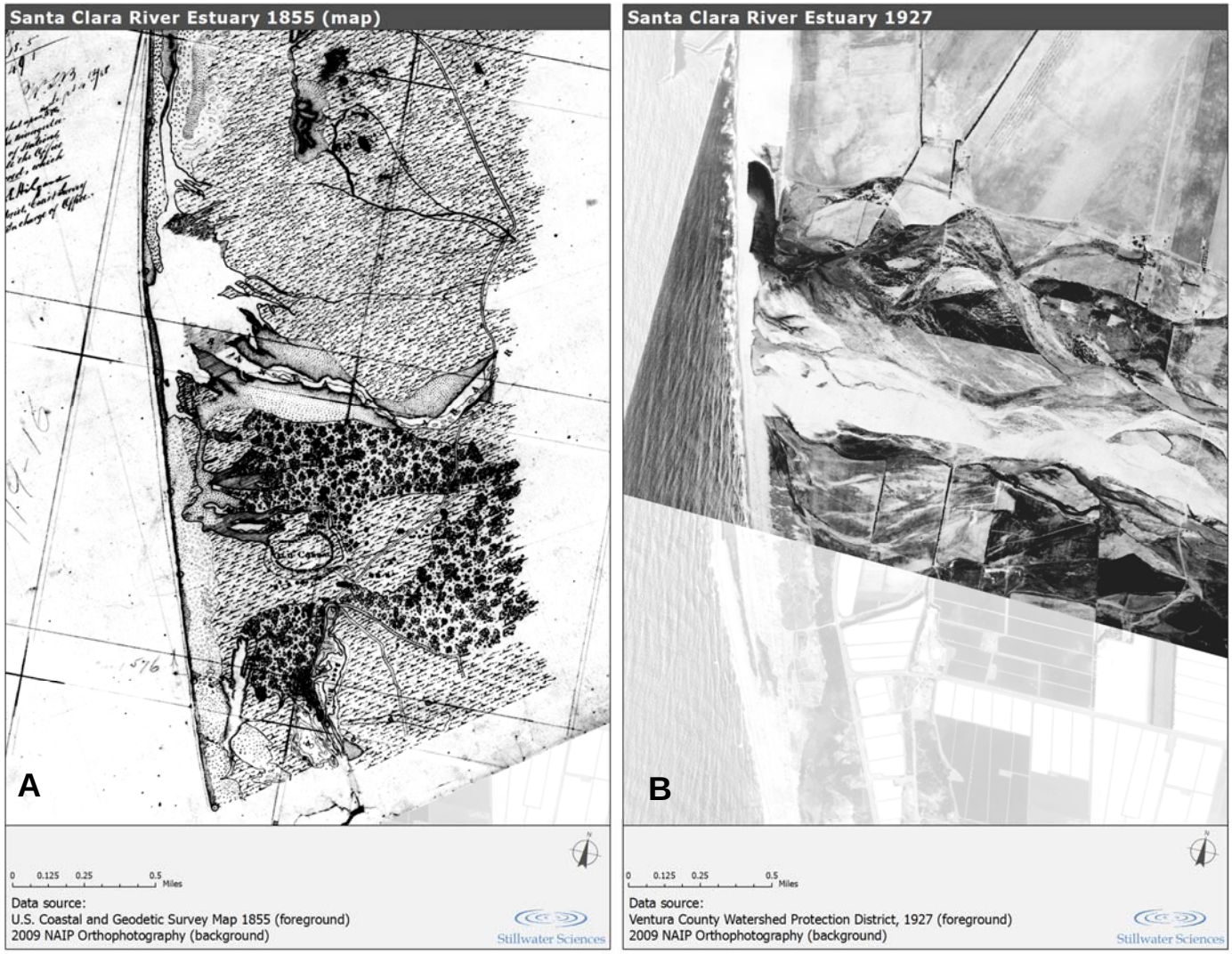


Figure 3-10. SCRE main channel, lagoon, and adjacent floodplain extent the Initial Settlement period (1855 [A] and 1927 [B]).

Between the mid-19th century and the early part of the 20th century, increased settlement in the lower watershed resulted in the onset of agricultural impacts to the SCRE. Similar to the previous period, the SCRE is thought to have had an unconstrained mainstem river channel and main lagoon area that drained to the ocean towards the north. However as settlement by European-Americans within the watershed continued to increase, there was a considerable shift in land use and the extent of inundated areas within the SCRE. In 1867, the estuary was still a "swamp" full of sloughs and lagunas (Schwartzberg and Moore 1995), but by 1900 development within the SCRE and adjacent floodplain had decreased the footprint and considerably reduced the amount of estuary habitat (Nautilus Environmental 2005). Habitat reduction was primarily caused by land clearing and channel infilling and redirecting in the southern portion of the SCRE. The two largest storms (1862 and 1884) most likely helped sustain and created tidal channel features and associated habitats, therefore intensive sustained land reclamation probably occurred after these storms towards the end of the time period. The 1884 storm in particular probably caused considerable geomorphic change as it dropped 24.3 cm (9.6 in) of rain on the City of Ventura over a short time period and produced a large flood that washed out bridges and caused major landsliding and debris flows in the upper watershed (Greely and Glassford 1891). A large secondary channel on the northern floodplain that is visible in a 1927 aerial photograph but not present in the 1855 USC&GS map may be a remnant of this storm.

Increased land reclamation for agricultural use through the first half of the 20th century continued to cause SCRE habitat fragmentation and reduction. This period is characterized by the continued planting of crops and the start of agricultural levee building. Although the SCRE had overall geomorphic characteristics similar to the previous time periods (e.g., unconstrained mainstem river channel and main lagoon area that drained to the ocean towards the north), the conversion of land within the SCRE to agricultural fields greatly decreased the overall footprint, removed geomorphic features, and simplified the habitat conditions. By the early 1930s, the main lagoon area decreased by approximately 50% compared to 1850s conditions and a majority of the side channels had been filled (Nautilus Environmental 2005). From the late 1920s to the late 1940s, aerial photographs show that the large secondary channel on the northern floodplain and smaller side channels to the south were being infilled and planted with crops (Figure 3-11). There also appears to have been small-scale levees on the north bank installed to help keep flood flows off of agricultural fields.

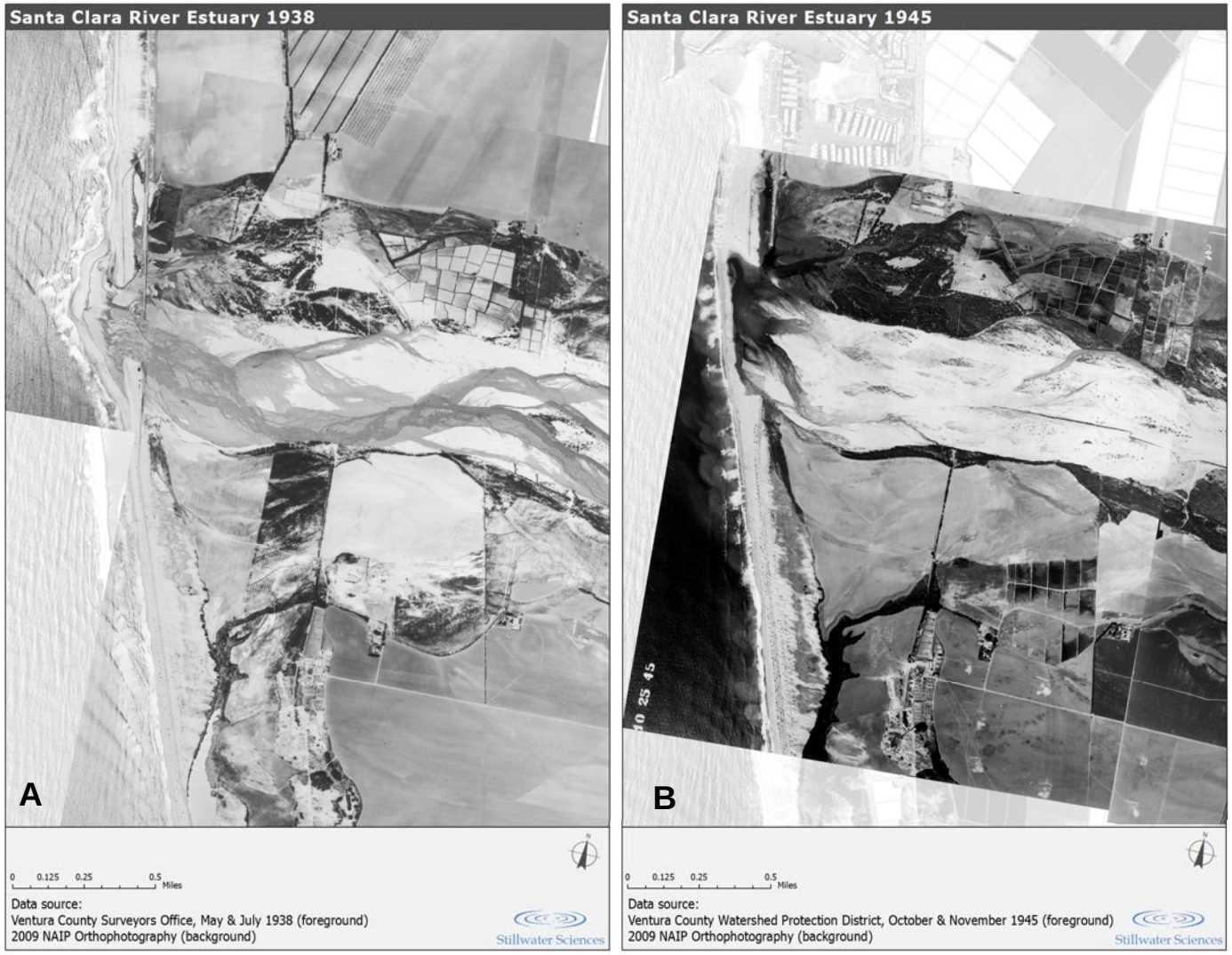


Figure 3-11. SCRE main channel, lagoon, and adjacent floodplain extent in the Agriculture period (Spring/Summer 1938 [A] and Fall 1945 [B]).

Several high flow events during this time period had the potential to cause geomorphic change, including the flood resulting from the St. Francis Dam failure on March 12, 1928. The peak flow estimate for the flood wave at the SCRE is approximately 175,000 cfs (Begnudelli and Sanders 2007), which is similar in magnitude to the January 1969 flood peak (165,000 cfs) and larger than a present-day 10-year flood event. Comparison of aerial photographs from the years before and after the dam failure suggests that the flood did not have that great a geomorphic impact on the SCRE, which may be due too the flood being able to spread out onto the Oxnard Plain and dissipate energy before reaching the estuary and ocean. Photographic evidence suggests that the smaller 1938 flood, however, did cause noticeable geomorphic change, including substantial bed scour, channel widening on the north side of the channel, and infilling of the wetland/backwater area to the north (Figure 3-11). This flood also deposited a considerable amount of sediment on the offshore delta that contributed to down-coast beach accretion between 1947 and 1955 (Inman 1950; Oceanographic Services Inc. 1977, as cited in O'Hirok 1985). An increase in upstream flow constraints by levees and other floodplain infrastructure between the 1928 dam failure and the 1938 flood event may have contributed to the differing geomorphic impacts.

By the early 1950s, geomorphic and habitat conditions were beginning to be affected by major levees and other infrastructure that were being established within the SCRE and lower Santa Clara River. In general, the main lagoon morphology remained similar to the previous time period but the mainstem channel upstream and overall footprint became more confined (Figure 3-12). By 1953, an extensive agricultural levee was built to reclaim land within the southern portion of the mainstem channel upstream of the main lagoon, essentially decreasing the channel area and effective flow width by one-third to one-half. Between 1953 and 1957, the continued encroachment of agricultural land and the establishment of levees for flood protection resulted in an approximate 75% decrease in historical channel and adjacent delta areas (Swanson et al. 1990). By 1968, the McGrath State Beach campground was established in the southern portion of the SCRE, Ventura Harbor was established in the historical northern backwater portion of the SCRE, the VWRF was established within the SCRE just north of the main lagoon, an extensive flood protection levee was established along the north bank upstream of the main lagoon, and Harbor Blvd. bridge was completed across SCRE where the channel enters the main lagoon. Combined, these features decreased the overall area of the SCRE, decreased the effective flow width during flood events, and essentially "locked" the SCRE into its present-day location and extent.

Figure 3-12. SCRE main channel, lagoon, and adjacent floodplain extent in the Levees and Infrastructure period (April 1958 [A] and February 1969 [B]).

Of the few high-flow events that occurred between the early 1950s and early 1970s, the January 1969 flood—the highest flow on record in the watershed—and the February 1969 flood both caused substantial geomorphic change. Comparison of aerial photographs from the years before and after the 1958 flood event suggest that the flow magnitude was not large enough to cause a noticeable and sustained change to the morphology of the lower Santa Clara River, the SCRE, or the adjacent coastline. The January and February 1969 storm events, however, were both larger than a present-day 10-year storm event, and both caused considerable damage in the lower Santa Clara River. The January flood overtopped the lower Santa Clara River mainstem channel just upstream of the Harbor Blvd. bridge and flowed over the northern floodplain, destroying the VWRF and other infrastructure before flowing through Ventura Harbor and out to the ocean (USACE 1970 and 1971, as cited in Nautilus Environmental 2005) (Figure 3-12). Photographic evidence suggests that the floods re-established many high-flow channels on both the northern and southern floodplains upstream of the SCRE and resulted in formation of a large depositional bar upstream of Harbor Blvd. that is still somewhat intact. These floods prompted the U.S. Army Corp of Engineers (USACE) to fortify and increase the height of the north bank levees to protect against future floods (Nautilus Environmental 2005).

Between the early 1970s and 2005, development impacts on the SCRE remained relatively constant and the overall morphology evolved within a fairly static, confined footprint. The completion of the VWRF treatment ponds and flood protection levees by 1974 marks the most recent major infrastructure establishment in the SCRE. Since then, the lower Santa Clara River and main estuary channel have migrated within a riverine/estuary corridor bound by the levee network on both the north and south banks, and the SCRE has lost approximately 50% of the remnant floodplain and riparian forest habitat (Swanson et al. 1990). As a result of the 1978 storm event, the main channel moved south towards its current location and depositional bars with side channels were established along the main SCRE channel downstream of Harbor Blvd. and in the mainstem channel upstream of Harbor Blvd. (Figure 3-13). Subsequent high flows during this time period caused considerable bar vegetation scour and bar morphologic evolution as the lower Santa Clara River mainstem channel and the main SCRE channel migrated. The long-term effects of this levee confinement on overall geomorphic character likely include an increase in bed/bar scour frequency due to levee-induced flow constriction and potentially from an increased hyperpycnal flow frequency associated with levee-induced decrease in floodplain sediment deposition.

Figure 3-13. SCRE main channel, lagoon, and adjacent floodplain extent in the ENSO wet period (May 1978 [A] and September 2005 [B]).

Compared to the previous time periods, geomorphic change appears to be most pronounced from the early 1970s to 2005 due in large part to the number of large storm events. Between 1978 and 2005, eight storm events had a peak flow greater than 50,000 cfs and five or those storm events had peak flows equaled or greater than 100,000 cfs. As would be expected from the earlier discussion in this section, all eight storms occurred during ENSO years. On average, a 50,000 cfs flow occurred once every 3 years and a 100,000 cfs flow occurred once every 5 years during this 27-year period. The most significant geomorphic changes came as a result of the January and February 2005 flood events. Aerial photographs taken in late February 2005 show that the floods scoured all bar vegetation in the channel upstream of the Harbor Blvd. bridge, scoured and widened the main SCRE channel downstream of Harbor Blvd. bridge, and deposited a large nearshore delta at the mouth (Figure 3-13). Topographic surveys show that the shoreline adjacent to the SCRE extended seaward approximately 130 m (430 ft) due to the large amount of sediment delivered from the 2005 flood events (Barnard et al. 2009). Unlike the 1969 floods, the levees in the lower Santa Clara River were capable of containing the 2005 flood flows and geomorphic change in the lower watershed was therefore confined to the Santa Clara River mainstem and SCRE.

Since 2005, the lack of large storm events has enabled the SCRE morphology to rebound and temporarily stabilize. In the months following the February 2005 event, wave action worked the large nearshore delta deposit into a mouth berm that extended approximately 200 m (700 ft) beyond the pre-2005 position, essentially increasing the SCRE area. Low river flows and small berm breaches resulted in muted tidal exchange and the formation of a small channel network towards the southern portion of the main lagoon. By 2006, vegetation was re-establishing on depositional bars and the continued inland movement of the mouth berm to the pre-2005 position caused the formation of northern and southern lagoon backwater areas (Figure 3-14). By 2007, the mouth berm position stabilized and the northern backwater area filled in. The southern backwater area remains intact and currently extends approximately 600 m (2,000 ft) south of the pre-2005 southeast corner of the main lagoon. The present-day morphology appears stable and will likely remain until the next large storm event.

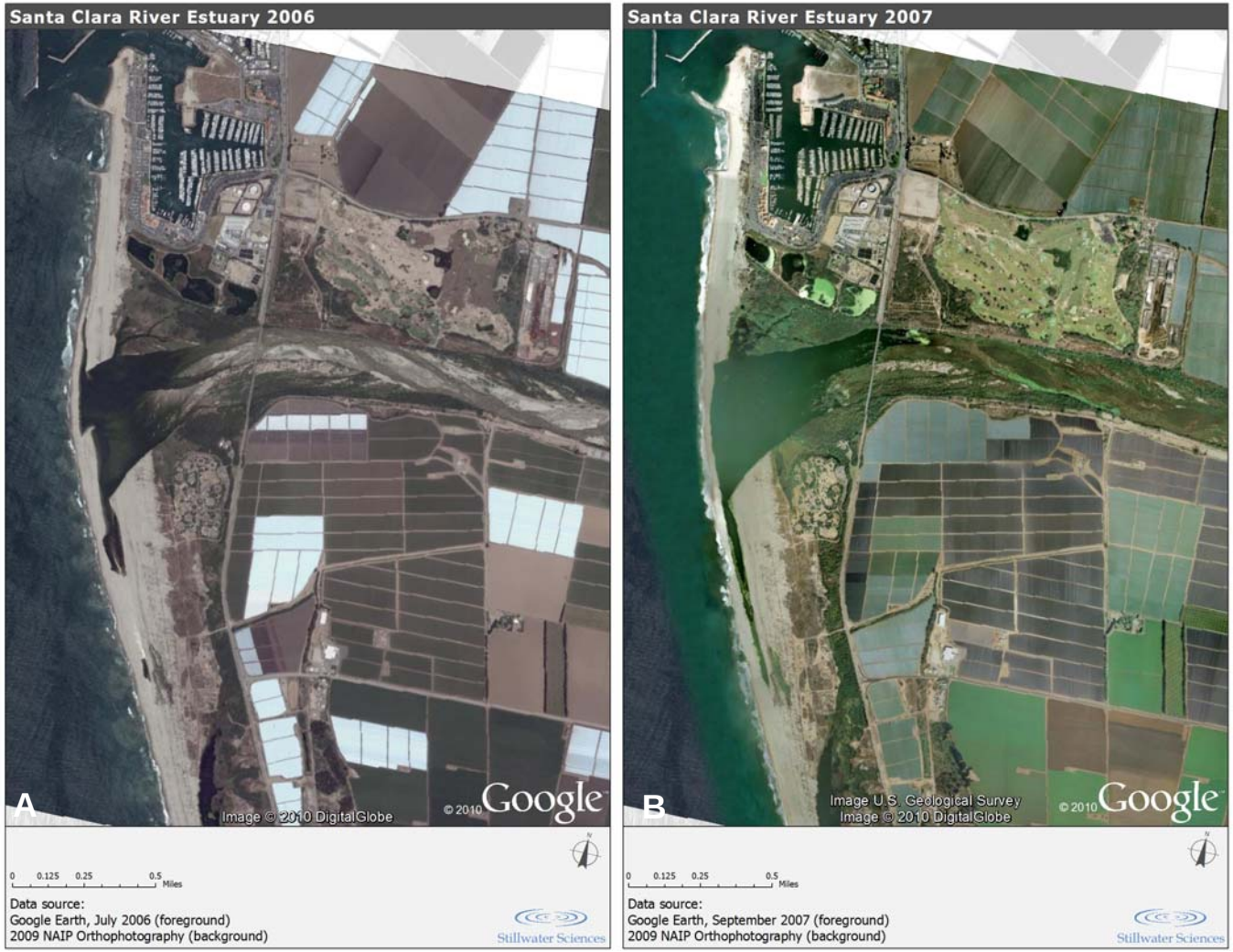


Figure 3-14. SCRE main channel, lagoon, and adjacent floodplain extent in the Present-day period following the 2005 flood events (July 2006 [A] and September 2007 [B]).

Viewed in total, the geomorphic changes associated with each discrete time period provide a comprehensive picture of geomorphic evolution within the SCRE since the early 19th century and provide an insight into the primary drivers for change. The extensive land reclamation and development along the lower Santa Clara River watershed and within the SCRE since 1855 has resulted in increased flow constriction and decreased overbank sediment deposition compared to pre-development conditions. These impacts act to increase the scour and sediment transport potential, thereby increasing an individual storm's geomorphic change potential. During the period of increased development-induced impacts, the frequency of large storm events within the Santa Clara River watershed also increased. Historical accounts and photographic evidence of storm-related impacts indicate that a minimum peak flow of approximately 100,000 cfs (around the peak flow for the 1884, 1978, and 1992 floods) is required to cause substantial and sustained geomorphic change to the SCRE. Prior to 1969 (around the start of current ENSO wet-period cycle), a flow of this magnitude occurred approximately once every 23 years. Since the start of this current wet-period, a flow of at least 100,000 cfs has occurred within an ENSO year approximately once every 7 years. Therefore, development has acted to increase the potential for geomorphic change during any given storm event, and recent changes in climatic forcing have then acted to increase the frequency and magnitude of storms that can cause substantial geomorphic change within the developed environment. For as long as the current wet-period ENSO cycle continues, the probability is thus about 15% in any given year that such a flow may occur; over the course of a decade, the likelihood of at least one such event is over 80%.

4 HYDROLOGY AND HYDRAULICS

This section provides an overview of hydrologic (e.g., surface and subsurface inflows and outflows) and hydraulic (e.g., mouth breaching and estuary stage/inundation) processes in the SCRE over the past several decades, and presents a contemporary SCRE water balance (i.e., timing and volume of water inflows and outflows) from spring 2009 through summer 2010. The analysis presented shows that the Santa Clara River flow is the dominant inflow from the fall through the spring, causing a relatively high mouth breaching and subsequent open-mouth frequency from November through June (i.e., the mouth is open >50% of the time on average). As the SCRE and the rest of southern California is currently experiencing a "wet" ENSO cycle, average annual river inflow volume and open-mouth frequency are relatively high compared to previous conditions. During the summer months, the VWRF effluent discharge is the dominant inflow and the mouth breaching frequency is relatively low during most years, as the mouth is closed >50% of the time on average. When the mouth remains closed for long periods during these dry conditions, the effluent discharge fills the SCRE to quasi-equilibrium "full" elevation. Within the current, post-2005 SCRE morphology, that "full" elevation can be maintained for months until the next storm causes mouth breaching.

The SCRE water balance clearly shows the relative contribution of each SCRE inflow and outflow under current conditions. When considering the entire water balance period (October 2009 to September 2010), the Santa Clara River dominated the surface inflows, and flow through an open-mouth dominated the outflows. The majority (approximately 80%) of that inflow and outflow volume occurred during and in the days following the four major storm events. During the dry season closed-mouth periods, discharge from the VWRF was a much larger portion of the total SCRE inflow. The combined VWRF effluent discharge and groundwater contribution from the VWRF wildlife ponds accounted for over 35% of the total SCRE inflow volume during the closed-mouth periods between October and December 2009 and 90% of the inflow volume during closed-mouth periods between March and September 2010. This seasonal difference illustrates the role that VWRF discharge plays in maintaining SCRE volume and wetted area during the spring and summer.

4.1 Surface Water

Surface water flow to the SCRE comes from three sources: Santa Clara River flows, VWRF effluent discharge, and tidal inflow. The relative contribution of each flow component to the total SCRE water volume varies both intra- and inter-annually due mostly to normal seasonal variations in precipitation dynamics and ENSO-driven annual differences in storm duration and intensity. These flow sources are discussed below in the context of seasonal and annual values and impacts to flow dynamics over the past several decades.

4.1.1 River flow

River flow into the SCRE is characterized by long durations of little to no flow punctuated by high flow events caused by short-duration, high-intensity precipitation events that travel quickly through the watershed. Over the past 80 years, mean daily discharge from the lower Santa Clara River into the SCRE (as measured at the Highway 101 bridge near Montalvo, CA and downstream at the Victoria Ave. bridge) ranged from 0 to over 90,000 cfs (Figure 4-1). Analysis of these data show that mean daily discharge over the course of a water year (WY), or the period from October through the following September, is less than 1 cfs approximately 70% of the time

and rarely exceeded 100 cfs (Figure 4-2). Monthly average daily mean discharge indicate that river flow into the SCRE is consistently low from May through December and reaches the annual maximum in February as a result of intense winter storms (Figure 4-3).

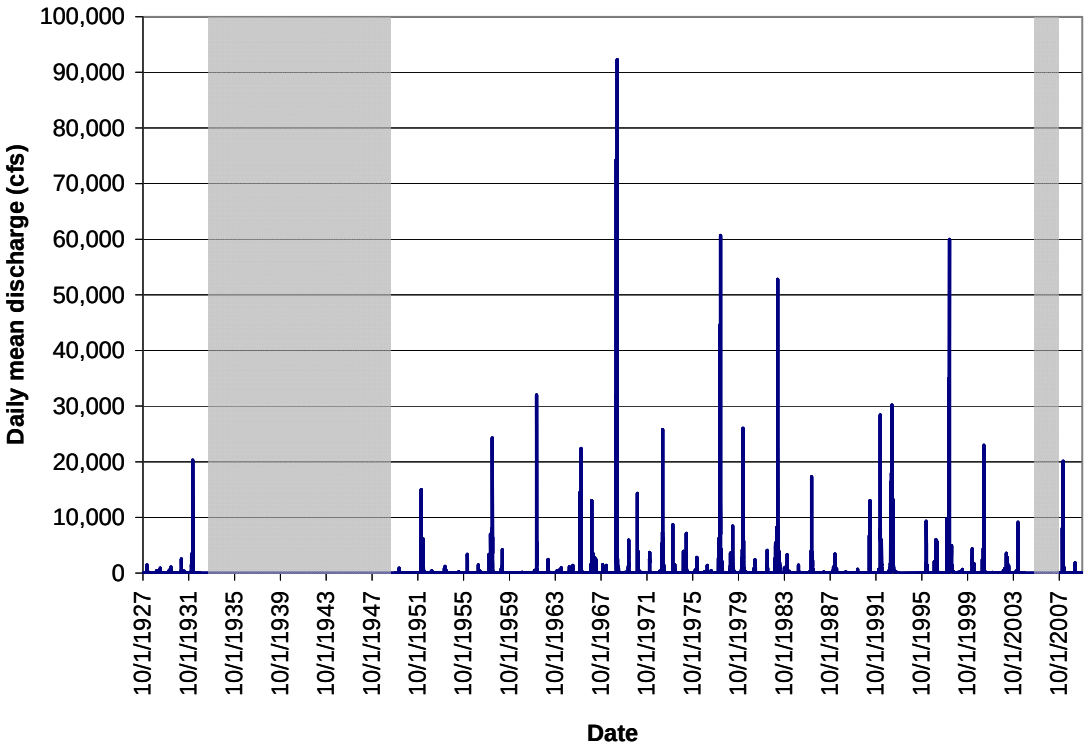


Figure 4-1. Lower Santa Clara River daily mean discharge measured at Montalvo, CA (USGS 11114000, WY 1927-2004) and at Victoria Ave. bridge (VCWPD 723, WY 2008-2009). Grey regions indicate time periods with no flow data.

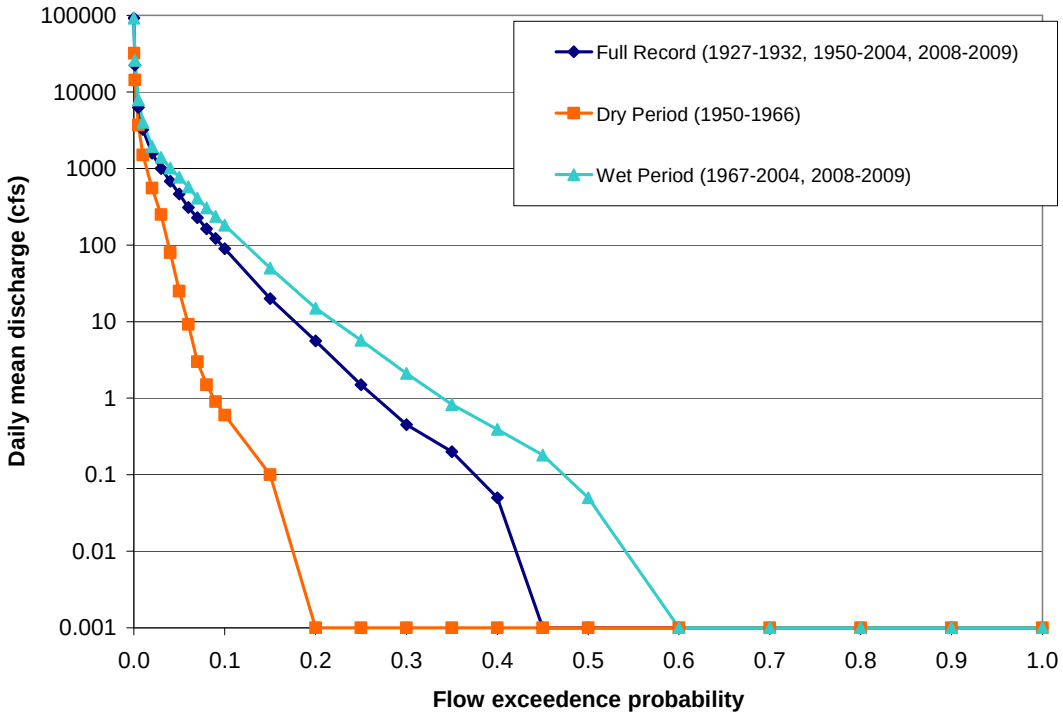


Figure 4-2. Lower Santa Clara River daily mean discharge exceedance probability at Montalvo, CA (USGS 11114000, WY 1927-2004) and at Victoria Ave. bridge (VCWPD 723, WY 2008-2009) illustrating the mid-20th century "dry" period and the recent "wet" period.

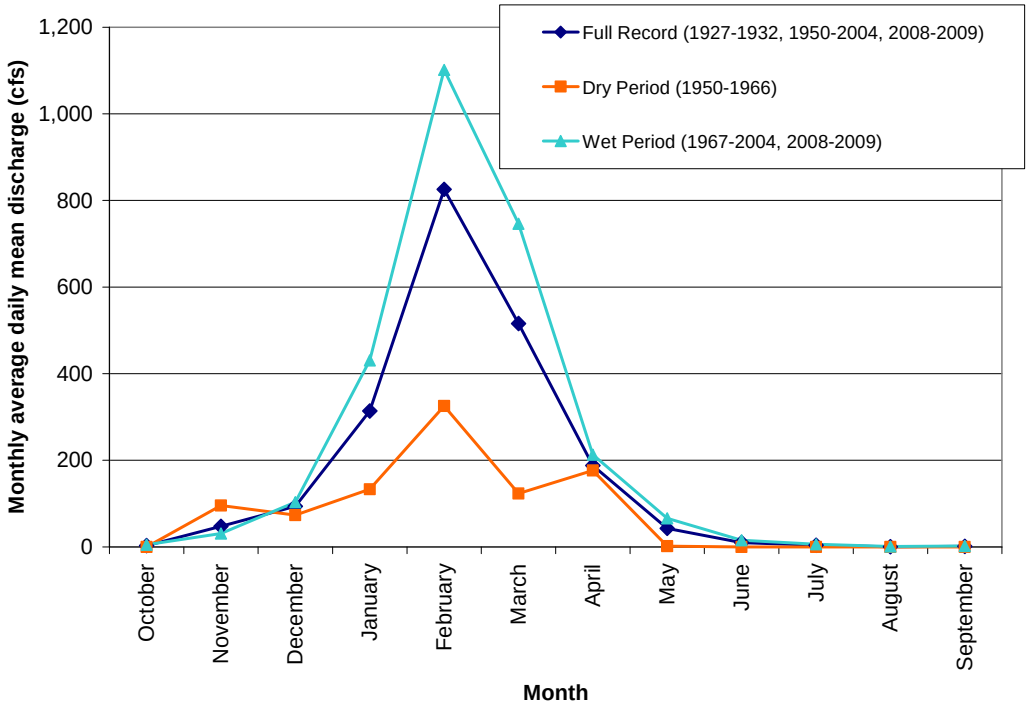


Figure 4-3. Lower Santa Clara River monthly average daily mean discharge at Montalvo, CA (USGS 11114000, WY 1927-2004) and at Victoria Ave. bridge (VCWPD 723, WY 2008-2009) illustrating the mid-20th century "dry" period and the recent "wet" period.

Comparing the daily mean flow records the ENSO-driven "dry period" (1950-1966) and "wet period" (1967-present) illustrates the impact that the difference in storm magnitude and frequency between the two periods has had on river flow into the SCRE. The "dry period" daily mean flows (between 1950 and 1966) exceeded 1 cfs less than 10% of the time and exceeded 100 cfs less than 5% of the time, while "wet period" daily mean flows (between 1967 and the present) exceeded 1 cfs approximately 35% of the time and exceeded 100 cfs approximately 15% of the time (Figure 4-2). Monthly average daily mean discharges shows similar "dry period" and "wet period" values for spring through fall (April through December), but a considerable difference in the wetter winter months. The current "wet period" monthly average daily mean flows are over three times higher than the "dry period" values for January through March, with February showing the greatest difference (Figure 4-3). The variability of daily mean flows during the "wet period" fall and winter months (October through April) has been lower than that of the fall and winter months during the "dry period," indicating the relative consistency of higher flows into the SCRE over the past 40 years. A hydrologic analysis that includes a more detailed examination of annual mean flow characteristics can be found in Nautilus Environmental (2005).

Within this period of shifting regional climatic conditions, there have been several anthropogenic influences that have also impacted surface flows to the SCRE. Reduction in the amount of flow to the SCRE has been caused by the installation of water-supply dams, agricultural practices within the Santa Clara River floodplain, and urbanization throughout the watershed. Flow regulation by tributary water supply dams has reduced flow to the Santa Clara River by approximately 25% (Warrick 2002, see Appendix A). This reduction can be most prominent during the dry months when water-supply dams impound flow that would otherwise be a significant component of total surface flow to the SCRE. On the mainstem Santa Clara River, the installation of Freeman Diversion Dam has resulted in the diversion up to 375 cfs of flow bound for the SCRE and an estimated 12% reduction of the freshwater volume to the SCRE during storm events (U.S. Bureau of Reclamation EIR, as cited in Swanson et al. 1990). The irrigation of agricultural fields along the Santa Clara River valley floor with local groundwater supplies has also likely contributed to the decrease in low-flow discharge to the SCRE by lowering the groundwater table adjacent to the mainstem Santa Clara River and decreasing groundwater flow gradients from the floodplain to the channel. The steady increase in watershed urbanization since the 1960s has resulted in a large amount of impervious area and an associated decrease in precipitation infiltration, groundwater recharge, and groundwater discharge to the Santa Clara River during low-flow conditions. In addition to decreasing low-flow discharge, watershed urbanization has also acted to increase the "flashiness" of flood flows that travel through the watershed and empty to the SCRE.

4.1.2 VWRF discharge

Effluent discharges from the VWRF to the SCRE have ranged from approximately 4 MGD (6 cfs) when the original trickling filter plant was placed in operation in 1958 (ESA 2003) to current peak flows reaching over 10 MGD (15 cfs) during high river discharge events. Since the facility upgrade to secondary (activated sludge) treatment following the 1969 flood event and implementation of the CWA, design flow capacity was increased to 14 MGD in 1972. Currently, treated effluent is discharged from the VWRF Effluent Transfer Station (ETS) to the Wildlife Ponds at a daily average rate of 5.8–8.4 MGD (9–13 cfs), with variations of 0.4–21.0 MGD (0.6–33 cfs) over the past 26 years of operation (Figure 4-4). Over this record, average flows for a given month have been similar from year to year (e.g., daily mean flows in January are similar for all years). Seasonal variability in ETS flows is possibly driven by infiltration and inflow of the collection system, since effluent flows are higher on average in the wetter winter months (January, February, and March) than the other months of the year (Figure 4-5).

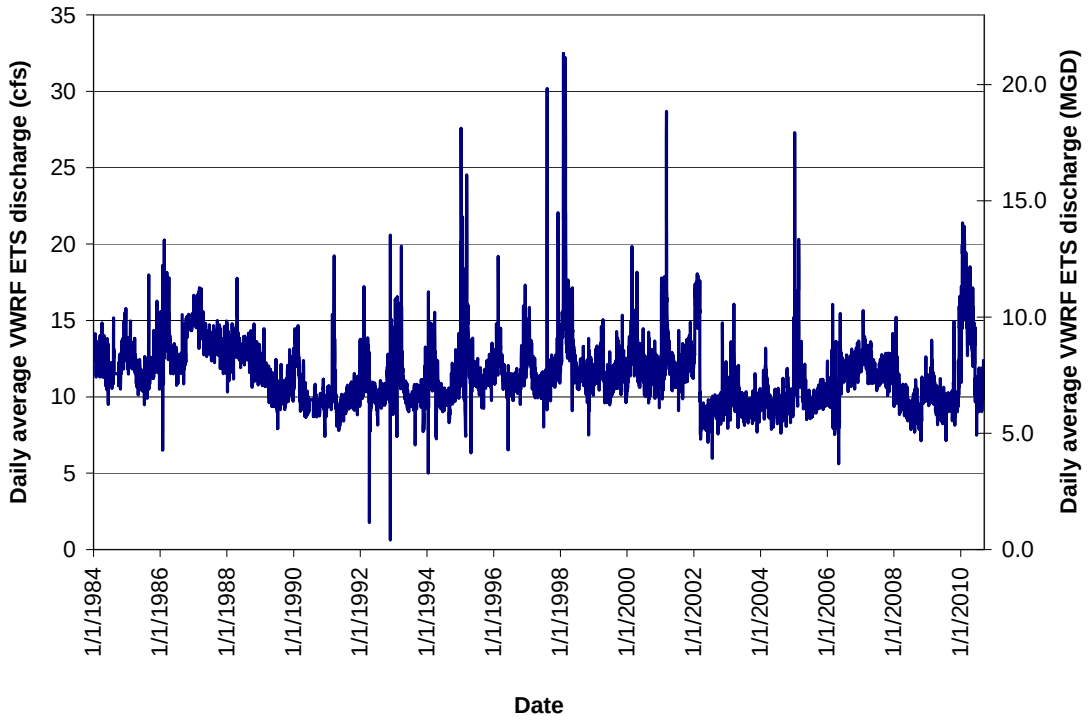


Figure 4-4. Daily mean VWRF Effluent Transfer Station (ETS) flow rate. Flow rate was derived from a total daily flow volume.

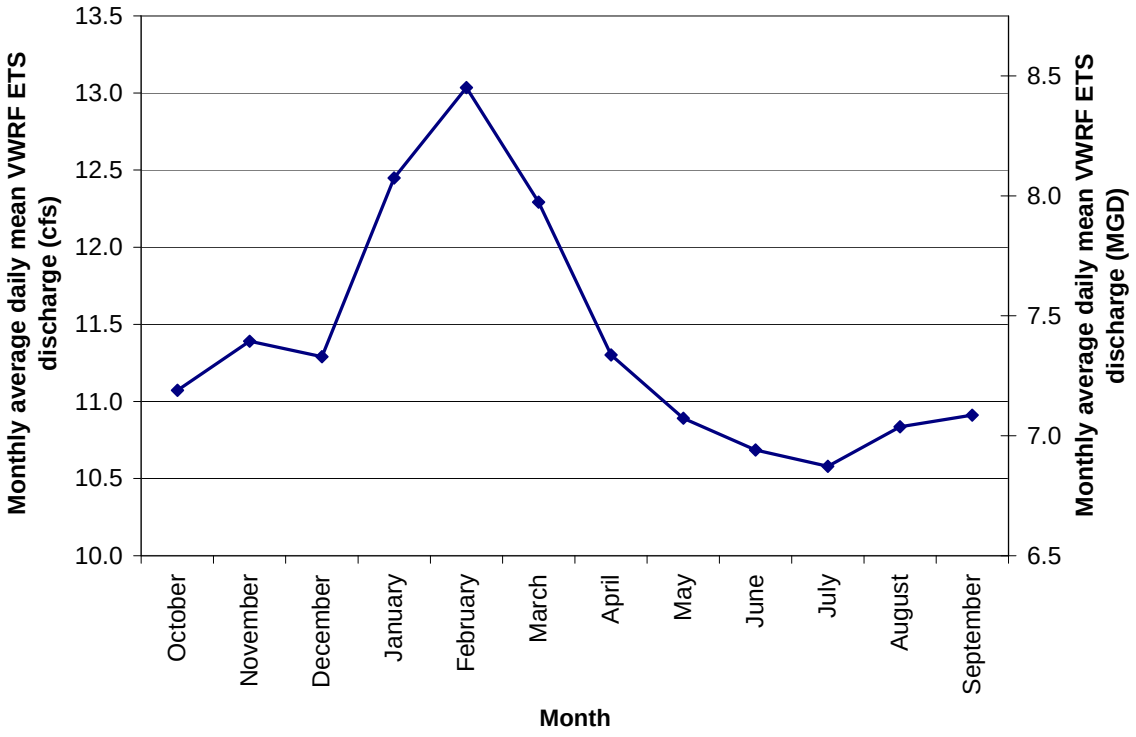


Figure 4-5. Monthly average daily mean ETS flow from 1984 to 2010.

The amount of VWRF flow discharged from the Wildlife Ponds to the SCRE is generally less than the ETS flows due to a combination of pond evaporation and recycled water diversions discussed in more detail in Section 4.2.2.8. During most of the year, the contribution of VWRF effluent to the SCRE total water volume is similar or much less than the volume of water delivered by the Santa Clara River. Comparison of average monthly ETS discharge volume with Santa Clara River discharge volume shows that the ETS discharge volume is up to an order of magnitude less than Santa Clara River volume during the winter and spring months (December–June) and that the two discharge volumes are similar for the fall (October-November) and early summer (July) (Figure 4-6). During the late summer (August-September), however, the average monthly ETS discharge volume is an order of magnitude greater than the Santa Clara River volume coming into the SCRE and is by far the dominant surface water source. This combined flow results in a summer and fall surface water flow into the SCRE that is likely greater than would be expected from an unregulated southern California river during closed-mouth conditions (ESA 2003).

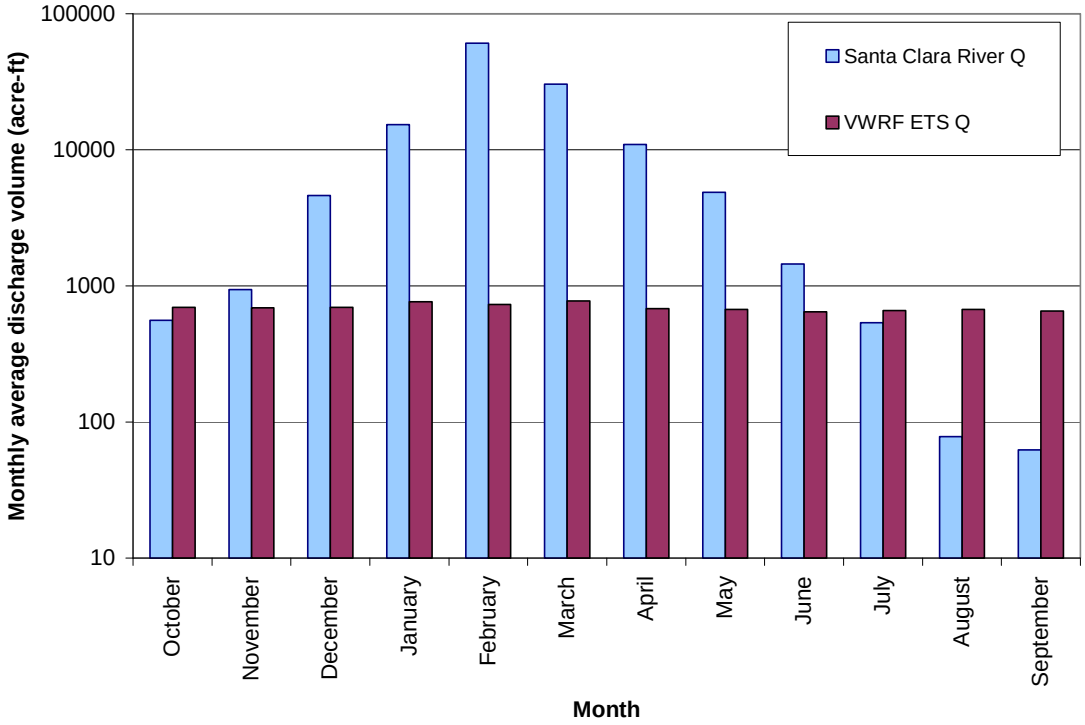


Figure 4-6. Comparison of monthly average flow volume delivered to the SCRE from the Santa Clara River and the VWRF (WY 1984-2004). Note the y-axis is a log scale.

4.1.3 Tidal flow

Tidal inflow to the SCRE during open-mouth conditions can account for a large portion of the total SCRE water volume when the estuary has drained following a storm or other breaching event. The tides adjacent to the SCRE are mixed semidiurnal (O'Hirok 1985), meaning that in general there are two high tides (a lower high and a higher high) and two low tides (a higher low and a lower low) per day, although there are days in which there is only one high tide and one low tide. The closest continuously operating tide gauge that can be used to reflect conditions adjacent to the SCRE is in Santa Barbara, CA (NOAA Station 9411340). At this tide gauge, mean

tide range (the difference between mean high water [MHW] and mean low water [MLW]) is 1.11 m (3.65 ft) and mean diurnal range (the difference between mean higher high water [MHHW] and mean lower low water [MLLW]) is 1.64 m (5.39 ft) (Table 4-1). In general, storm-induced increases in tidal elevation are relatively small (less than 0.3 m [1 ft]) in comparison with normal tidal fluctuations (Noble Consultants 1989). Mean sea level (MSL) adjacent to the SCRE has increased approximately 30 cm (12 in) since the onset of European colonization over 150 years ago (see Cayan et al. 2008 and references therein).

Table 4-1. Tidal elevations for Santa Barbara, CA (1990–2010).

Tidal datum	Elevation (m NAVD88)	Elevation (ft NAVD88)
Extreme high (observed January 1992)	+2.21	+7.26
Mean higher high water (MHHW)	+1.62	+5.30
Mean high water (MHW)	+1.38	+4.54
Mean diurnal tide level (MDTL)	+0.79	+2.60
Mean tide level (MTL)	+0.83	+2.71
Mean sea level (MSL)	+0.82	+2.69
Mean low water (MLW)	+0.27	+0.89
Mean lower low water (MLLW)	-0.03	-0.09

Source: NOAA Center for Operational and Oceanographic Products and Services website (www.co-ops.nos.noaa.gov)

Using MHHW (water surface elevation of 1.6 m [5.3 ft] NAVD88) as the maximum tidally-induced water surface elevation in the SCRE, the tide can currently extend approximately 400 m (1,300 ft) inland from the mouth berm and inundate approximately 22% of the SCRE area, filling approximately 7% of the total SCRE volume. Prior to the 2005 flood event, MHHW extended approximately 300 m (980 ft) inland, inundating approximately 13% of the area and filling approximately 3% of the volume (based on the 2000 SCRE bathymetry). Tidal flow into and out of the SCRE can last for days to weeks until the tide's ability to maintain an open channel is counteracted by strong wave action bringing sediment onshore, thereby causing mouth closure. The volume of tidal water exchanged during open-mouth conditions and the volume that remains in the SCRE when the mouth closes is a function of both tidal elevations and mouth elevation and geometry (i.e., the ability of the tide to crest the mouth during high tide and the drainage that the mouth allows during low tide).

4.1.4 Groundwater/Subsurface water

As previously discussed, the SCRE is hydrologically connected to the shallow, Semi-perched aquifer that lies at the top of the Mound groundwater sub-basin. The relatively shallow depth of the Semi-perched aquifer (< 30 m [100 ft]) results in a low storage capacity and leads to frequent saturation by infiltrated precipitation and agricultural irrigation in the floodplain area to the southeast of the SCRE. To the south, McGrath Lake is supplied primarily by groundwater from the contributing watershed and has relatively high recharge rates due to agricultural irrigation (URS Corporation 2005). Groundwater inflow can result in an increase in lake level of up to 3 in/day (YCE 1998, as cited in Kennedy/Jenks Consultants 2002). The relatively high local groundwater table can cause agricultural field adjacent to the lake to flood, with the flooding being exacerbated by groundwater flow out of the SCRE during closed-mouth, filling conditions (Nautilus Environmental 2005). Irrigation-induced saturation is offset by installation of tile drains below the crop root zone and by pumping of McGrath Lake to enhance the natural drainage of the fields (ESA 2003, URS Corporation 2005). Under normal operations, the lake is pumped so that

the water surface elevation is between 0.8–1.1 m (2.7–3.6 ft) above MSL (1.6–1.9 m [5.3–6.3 ft] NAVD88) (Kennedy/Jenks Consultants 2002). This pumping causes a local groundwater depression and flow southwards from the SCRE towards McGrath Lake when the estuary stage is above ~2.0 m (6.5 ft) NAVD88. The degree of floodplain saturation can be increased when the SCRE has a closed mouth and is filling, indicating the extent of hydrologic interaction between the SCRE and the Semi-perched aquifer to the south. Closer to the SCRE, groundwater elevation and floodplain saturation dynamics to the south have been impacted by the removal of the earthen levee protecting the northern boundary of McGrath State Beach in 1998. Levee removal and associated bank erosion has caused the northern boundary to migrate south and has resulted in an increased surface water and groundwater inundation extent within McGrath State Beach (Nautilus Environmental 2005).

Over the past 20 years, several studies have documented local hydraulic gradients, indicating the rate and direction of groundwater flow adjacent to the SCRE under typical conditions. Just upstream of the SCRE, monitoring of groundwater wells at the Bailard Landfill since 2004 have shown that the Semi-perched aquifer water table is consistently very close to the adjacent Santa Clara River bed elevation in spring and fall (Bailard Landfill Semi-Annual reports, 2004–2008), suggesting very little groundwater input to the river and SCRE at present except possibly during wet winter conditions. Water table measurements taken within McGrath State Beach along a transect extending 200 m (660 ft) south of the SCRE during closed-mouth conditions in summer and fall 1989 showed that water table elevations adjacent to the SCRE were approximately 1 m (3 ft) lower than the SCRE water surface at the onset of estuary filling and increased by approximately 0.6 m (1.9 ft) as the SCRE continued to fill, causing saturation of adjacent floodplain areas (Swanson et al. 1990). More recent data collected along a transect between the SCRE and McGrath Lake illustrate the dominant groundwater flow direction towards the SCRE during open-mouth conditions and the switch to a gradient directing flow towards McGrath Lake when the SCRE becomes full following mouth closure. As previously discussed, local hydraulic gradients and groundwater flow between the SCRE and McGrath Lake are strongly influenced by the management of the McGrath Lake water level.

In addition to groundwater exchange with the Semi-perched aquifer, the SCRE also exchanges subsurface water with the ocean across the mouth berm. The entire SCRE bed is above the adjacent mean tide level (MTL) and the majority (>80%) is above mean higher high water (MHHW), indicating that the subsurface flow direction is predominantly out of the SCRE towards the ocean when the mouth is closed and the SCRE is filling. In addition to hydraulic gradients and hydraulic conductivity, the length of the mouth berm is a key factor in controlling subsurface water exchange dynamics. All other factors being equal, a longer mouth berm results in more surface area and a higher rate and volume of subsurface flow. Recent photographic evidence indicates that the post-2005 SCRE mouth berm is longer than it has been for several decades, suggesting the current subsurface flow rates and volumes across the mouth berm are greater than pre-2005 storm values.

4.1.5 Mouth breaching dynamics

Breaching of the SCRE mouth berm and subsequent drainage of the SCRE occurs periodically as a result of both natural and anthropogenic influences. In many southern California estuaries, high river discharge during wetter months causes extended open-mouth periods, and low river discharge during drier months results in extended closed-mouth periods where the estuary fills with both surface river water and subsurface water (groundwater and ocean water seepage). Flow-induced breaching depends on the ability of the SCRE to store the storm-induced discharge and is therefore a function of flow magnitude, flow duration, and the SCRE water volume when the

storm-driven discharge enters the estuary. In addition to storm-induced wave impacts, wave overwash into the SCRE when the mouth is closed at any point in the year can initiate the formation of a mouth channel, causing rapid draining when the tide recedes and subsequent tidal exchange while the mouth stays open. During closed-mouth, low-river flow conditions, the constant VWRF discharge can cause the SCRE inflow volume to exceed outflow, thereby raising the SCRE water-surface above the mouth berm and initiating the formation and subsequent scouring of the mouth channel. Prior to 2005, this type of breaching was shown to occur when SCRE water surface elevation reached approximately 3 m (10 ft) NAVD88 (Nautilus Environmental 2005).

Mechanical breaching of the SCRE mouth berm has also occurred in the past as a means of alleviating the risk of flooding adjacent to the estuary. Known recent authorized breaches include an emergency breach as part of the 1994 McGrath Lake oil spill and occasional breaches associated with the Ventura Port District's annual winter dredging disposal operations (ESA 2003). The McGrath Beach State Park 1979 General Plan indicated that park personnel would routinely breach the estuary barrier to prevent flooding of the campground caused by high groundwater. Due to natural resource considerations, this practice ended by 1985 (ESA 2003).

The combination of breaching influences has resulted in an SCRE mouth that has been open more often than closed over the past 26 years. Daily observations of the SCRE mouth berm made by the City of Ventura from 1984 to 2010 indicate that Santa Clara River mouth has been open approximately 61% of days in the recent past, with 2009 having the lowest open-mouth frequency (16% of days in the year) and 1993 and 1995 having the highest open-mouth frequency (96% of days in each year) (Figure 4-7). On a seasonal basis, these daily observations show that the mouth has been open with the highest daily frequency during March (84% of time) and the lowest daily frequency during September (44% of time), and is open on average less than 50% of time from summer into the fall (July through October) (Figure 4-8). The variability in the amount of time the mouth is open is the lowest for February and March and highest for July and October, indicating that late winter months have a more consistently open mouth than the summer and fall months have a consistently closed mouth. This difference can be at least partly attributed to a combination of relatively high winter flows coming from the watershed, which have been very pronounced during this current "wet period," and the influence of VWRF discharge on mouth breaching during drier months of the year.

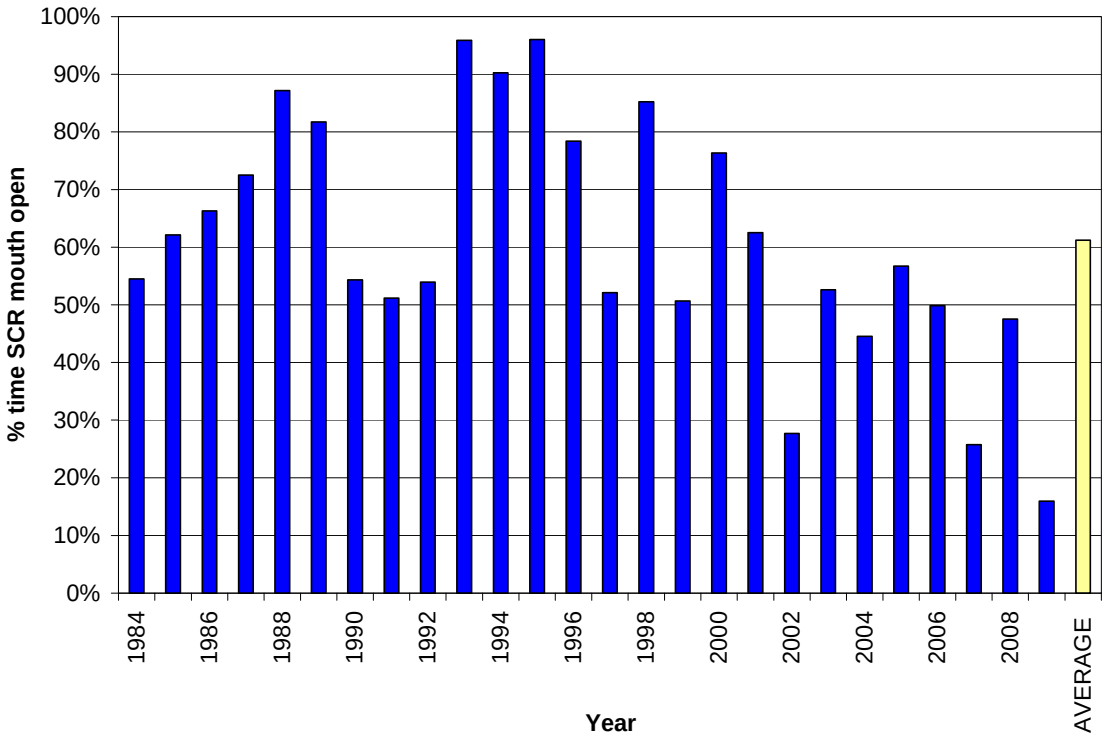


Figure 4-7. Percentage of time the Santa Clara River mouth was open on an annual basis (1984–2009).

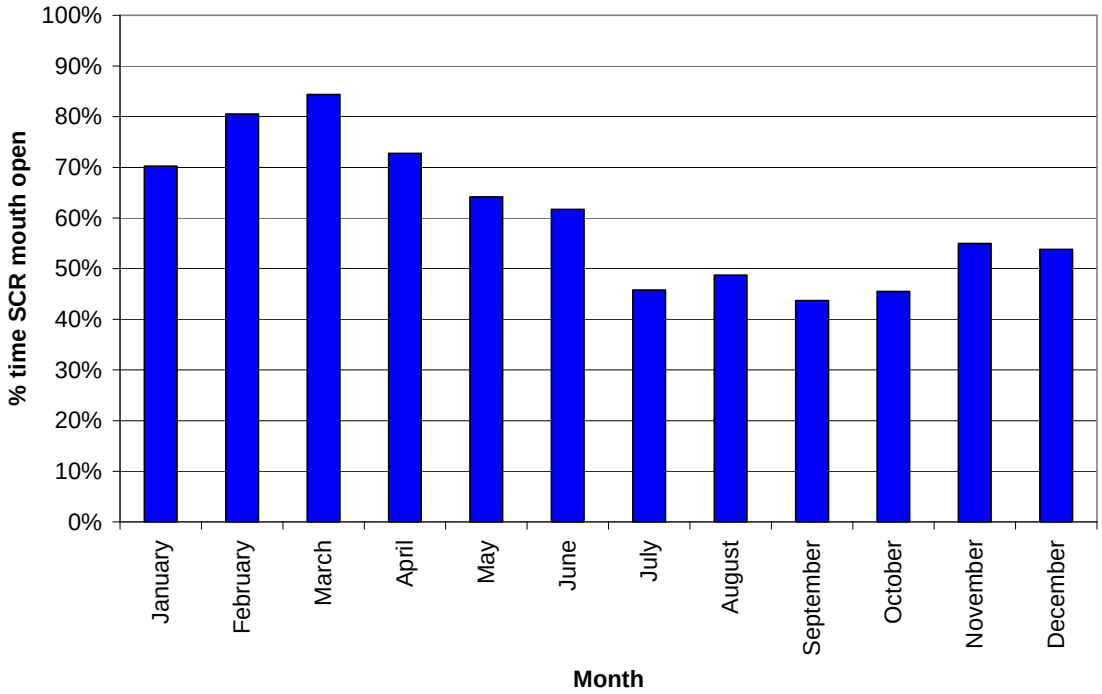


Figure 4-8. Percentage of time the Santa Clara River mouth was open on a monthly basis (1984–2009).

Examining the mouth observations from 1984 to 1998 (a wetter time interval that contained three ENSO years) and 1999 to 2010 (a drier time interval that contained one ENSO year) illustrates the impact of ENSO year flow characteristics on average annual and seasonal mouth status (Figure 4-9). Between 1984 and 1998, the mouth was open approximately 73% of the year, with March having the most open-mouth days of any month (91% of all days in March) and August having the least open-mouth days (56% of all days in August). Since 1998, the mouth has been open approximately 43% of the year on average, has been open a maximum 83% of the time in February and a minimum 23% of the time in October, and has been closed for long periods of time throughout the year. Extended storm duration during the ENSO years within the wetter time interval not only caused the mouth to stay open more during the winter months due to high river flows, it also likely contributed to more mouth breaching during drier parts of the year by increasing dry season baseflow. The average annual Santa Clara River flow volume measured in the lower Santa Clara River (USGS gage 11114000 and VCWPD gage 723) during these two time intervals indicates that the wetter time interval had both a higher average annual river flow volume and a higher average river flow volume during the summer months.

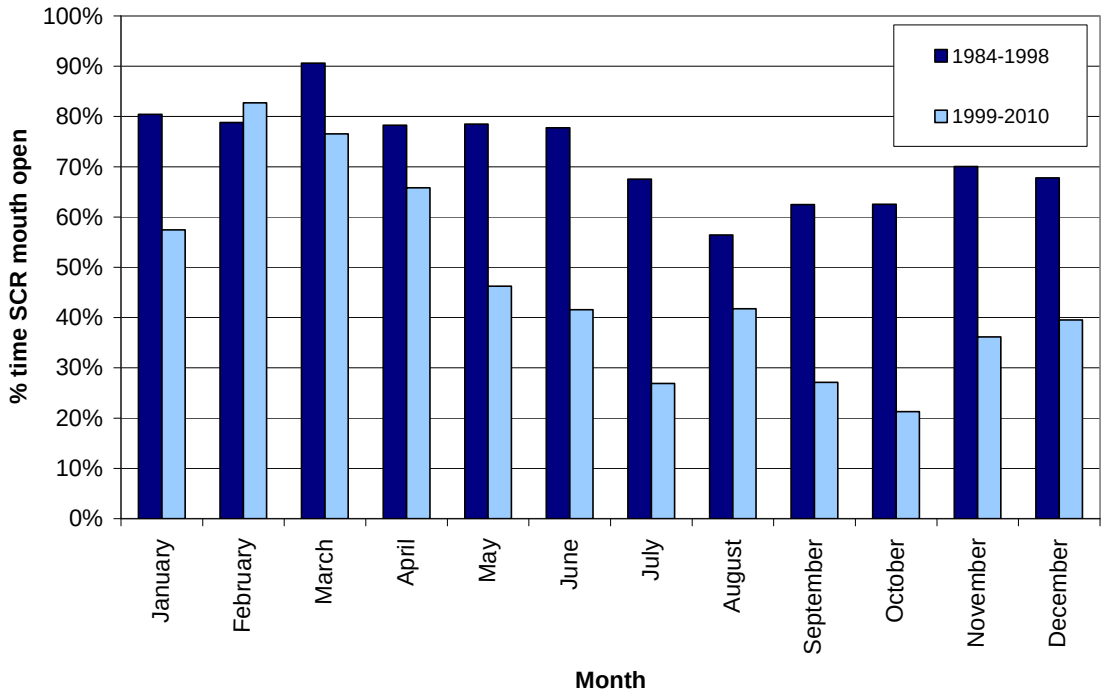


Figure 4-9. Comparison of the amount of time the Santa Clara River mouth was open on a monthly basis for the recent wetter interval (1984-1998) and drier interval (1999-2010).

Subsurface water seepage across the mouth berm acts to control SCRE volume during closed-mouth, low-flow conditions and therefore helps mediate mouth breaching frequency. As discussed previously, the mouth berm length is a key local control on the rate and volume of subsurface flow from the SCRE to the ocean. Comparison of the SCRE mouth berm in the 1855 USC&GS map with aerial photographs from the past 80 years shows that the mouth berm length has decreased by hundreds of meters since the onset of development with the SCRE and lower Santa Clara River (Section 3, Figures 3-10 through 3-14). This decrease in mouth berm length

suggests that the SCRE historically had a higher berm seepage rate and therefore likely breached less frequently during low-flow conditions in both drier months (periods of low river and groundwater discharge) and wetter months (baseflow conditions after storms but elevated groundwater discharge rates to the SCRE).

4.1.6 Estuary stage and inundation dynamics

The combination of seasonally variable SCRE inflow/outflow rates and mouth breaching frequency causes the SCRE stage to be highly dynamic over the course of a year. Recent data show that when the mouth is open, tidal exchange typically causes the SCRE stage to range between approximately 1.2 m (4 ft) NAVD88 (i.e., the minimum SCRE bed elevation) and 2.1 m (7 ft) NAVD88 (Nautilus Environmental 2005). For the most part, surface water flows freely through the open mouth and there is very little lag between a change in tidal elevation and associated change in SCRE stage during both filling and draining. When the mouth is closed, the SCRE has been shown to take over a month to fill and breach. During low-flow conditions, berm breaching has been shown to occur at approximately 3 m [10 ft] NAVD88 (Figure 4-10). When storm-driven river flows enter the SCRE during closed-mouth conditions, the filling rate is much more rapid and the stage that causes berm breaching can be higher than the breaching elevation during low-flow conditions, which is presumably caused by the high river flow volume overwhelming the relatively small mouth channel (Figure 4-10).

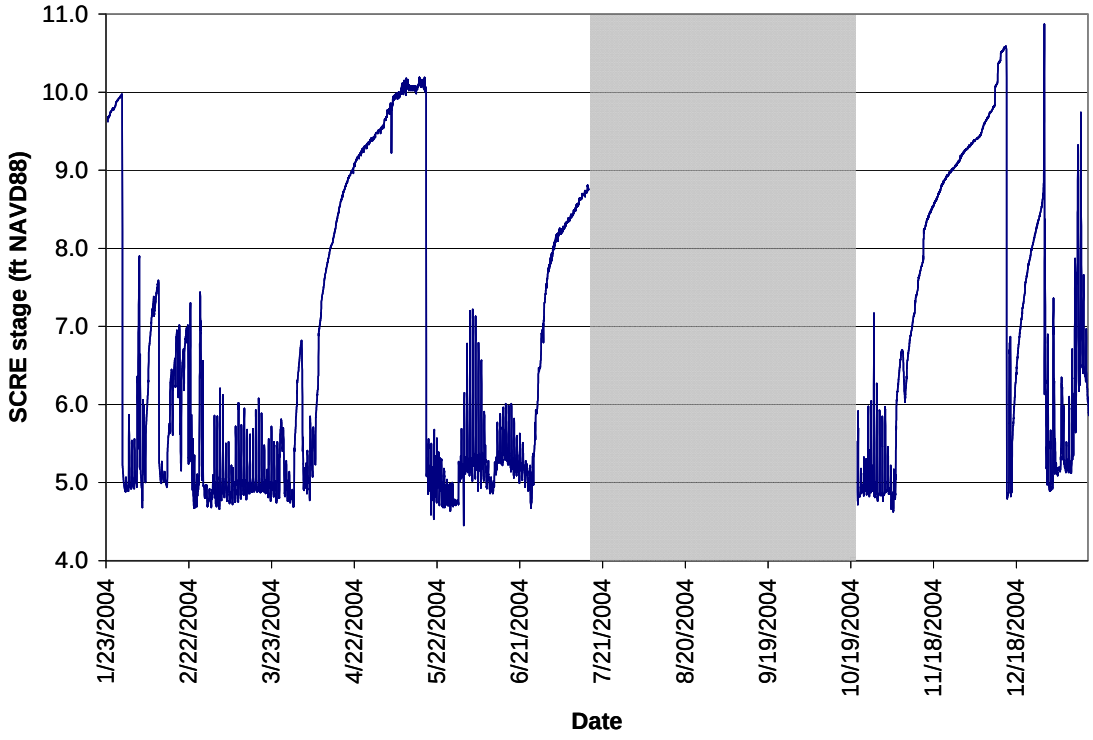


Figure 4-10. SCRE stage from January 2004 to January 2005 (from Nautilus Environmental 2005). The grey region indicates a time period with no stage data.

4.2 Current Estuary Water Balance

4.2.1 Introduction

The timing and volume of water inflows and outflows are important drivers of estuary ecological conditions. In the short-term, these factors strongly affect water surface area and water quality conditions. Over longer time periods, the water balance affects seasonal and annual hydroperiods that control vegetation assemblages and zonation, seasonal habitat availability for aquatic organisms, and salmonid in- and out-migration patterns. Within estuary systems like the SCRE, where the hydrology is impacted by watershed development (e.g., upstream water diversion, groundwater extraction, and year-round external water inflow), the duration and frequency of open- and closed-mouth periods and the dominant sources of water to the estuary throughout the year can be somewhat different than more natural systems due to altered inflow and outflow rates. Specific to the SCRE, the near-constant VWRF discharge is a major inflow that influences filling rates, volume, inundation extent, mouth breaching, and ecological conditions during low-flow, closed-mouth conditions.

In order to better understand the overall connection between estuary hydrology and ecology within the SCRE, a comprehensive water balance was developed. The goals of the water balance were to: (1) describe the relative contribution of each inflow and outflow component on total SCRE volume during open- and closed-mouth conditions, and (2) determine the impact that changing VWRF discharge can have on SCRE volume and associated habitat and recreational opportunities during low-flow, closed-mouth conditions. The water balance includes data collected specifically for this project as well as data collected concurrently by others, and is based on a combination of measured and interpolated information and literature-derived empirical values. In this section, we present the SCRE water balance for current (WY 2010) conditions. Later in the report, the water balance is used as a primary tool to help assess the impact of developed VWRF discharge alternatives on SCRE volume, inundated area, and associated conditions for the selected focal species and human recreation (Section 11).

4.2.2 Methods and results

A water balance is a method for accounting for the inflows, outflows, and change in water stored in a specific reservoir over a given time period (Hornberger et al. 1998). The change in water volume stored over a given time period can be expressed by the equation

$$\frac{dV}{dt} = I - O$$

where V is the reservoir volume, t is time, I is the volume of water input for the time step dt, and O is the volume of water output for the time step dt. The volume of water stored at a given point in time can then be expressed by the equation

$$V(t+1) = V(t) \pm \frac{dV}{dt}$$

Essentially, the reservoir volume at any given time is equal to reservoir volume from the previous time step plus the amount of water put in minus the amount of water taken out over that time step. When reservoir volume is a measured quantity, the volume difference between successive time steps gives the total water volume gained or lost. Combining this information with measured rates

of water input and output between successive time steps can help constrain discrete rates of water input and output for which little or no data are available. Compiling these data over long time periods (i.e., many successive time steps) allows for determining average seasonal and annual contributions of individual water balance components.

The inflows and outflows to the SCRE are composed of many different components (Figure 4-11). Water inflows include the following:

- Direct precipitation
- VWRF discharge
- Santa Clara River discharge
- Tidal inflow when the mouth is open
- Subsurface inflow across the berm when the mouth is closed
- Groundwater inflow from the Semi-perched aquifer

Water outflows include the following:

- Evaporation
- Tidal outflow when mouth is open
- Subsurface outflow across berm when the mouth is closed
- Groundwater outflow to the Semi-perched aquifer

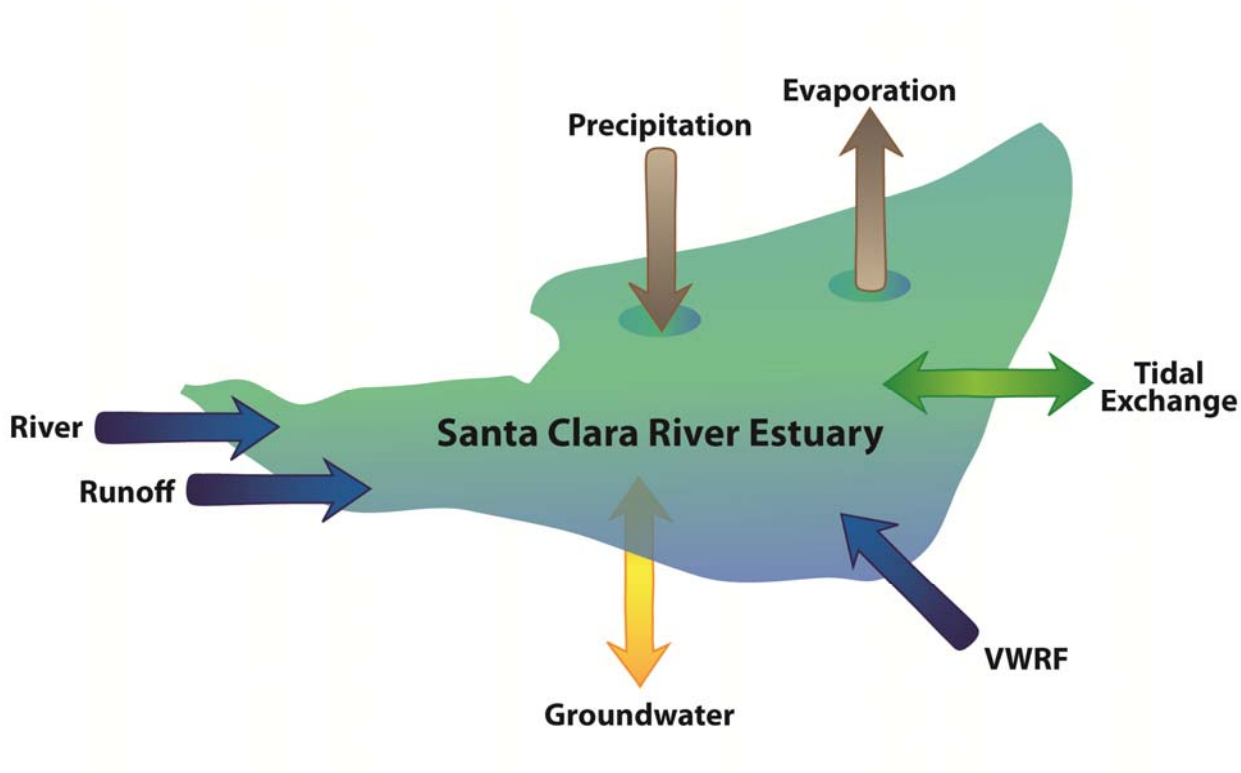


Figure 4-11. Conceptual model of the dominant SCRE inflows and outflows.

The SCRE volumetric change over time and the contribution of individual inflow and outflow components on the SCRE water balance were determined combining SCRE volume measurements with calculated inflow and outflow rates. SCRE volume was determined from a time series of SCRE stage combined with a SCRE stage-volume relationship derived from recent bathymetric data. Precipitation inflow rates and evaporation outflow rates were calculated from time series of precipitation and evaporation depths measured near the SCRE combined with the SCRE stage time series and an SCRE stage-area relationship developed from the SCRE bathymetry. Groundwater flow rates between the SCRE and the subwatershed areas to the north and south were determined by combining flow estimates based on hydraulic conductivity, flow area, and measured local hydraulic gradients with flow estimates derived from the residual of the measured estuary volume and the combination of all other measured flow rates. Subsurface flow rates across the mouth berm were derived from estimates of berm hydraulic conductivity, flow area, and the hydraulic gradient between the SCRE and the ocean. VWRF and Santa Clara river inflow rates came from flow gage measurements, with river flows being augmented by calculated run-off downstream of the gage during storm events. Tidal inflow and outflow rates and unmeasured groundwater flows were derived from differences between measured SCRE volume and SCRE volume calculated from the individual water balance components. A detailed description of data collection techniques can be found in the project Monitoring Plan (Stillwater Sciences 2009).

The project monitoring period began in the spring of 2009 and extended into late summer 2010. The water balance period covers the majority of WY 2010, beginning in late October 2009 at the start of a filling period when all inflow and outflows that could be monitored were being monitored and ending in mid-September 2010 when the SCRE breached after a long filling period. When compared to water years over the past several decades, WY 2010 was a “wet” water year, meaning that the total annual river discharge and precipitation volumes in the lower Santa Clara River watershed were well above median values. This “wetter than average” condition was primarily driven by a few short-duration storm events in fall 2009 and winter 2010. During the spring and summer 2010 when the mouth was closed, the river discharge and precipitation volumes were very similar to long-term average conditions. These data collected during WY 2010 are well-suited for the purposes of understanding the impact of the VWRF discharge on SCRE ecology during low flow, closed-mouth conditions between storm events, a primary focus of this project.

In order to develop a water balance of the SCRE (Section 4.2.3) data on SCRE bathymetry, stage and associated inundation are discussed below to illustrate annual conditions and seasonal trends. Estimation methods for each water balance component are presented for the 2009–2010 water balance period, with a discussion of the compiled water balance for both the entire water balance period, as well as for closed-mouth filling periods. These closed mouth conditions represent periods when VWRF flows are affecting SCRE volume and area, and thus habitat availability for focal fish and bird species.

4.2.2.1 Current bathymetry

The SCRE bathymetry for current conditions was developed by modifying high-resolution 0.3 m (1 ft) LiDAR data collected by Ventura County in January 2005 and by the USGS in October 2005 (see Stillwater Sciences 2009 for more details on combining these data sources). The January and February 2005 flood events caused considerable morphologic change within and adjacent to the SCRE. The most noticeable change was the formation of a large nearshore delta and seaward migration of the mouth berm by over 100 m (330 ft) (Section 3.1.3). These features were captured in the both 2005 LiDAR datasets. By 2009 (the beginning of the water balance

time period), the SCRE morphology rebounded and the mouth berm position moved inland to the pre-2005 location. Therefore, the 2005 LiDAR datasets were modified to reflect current 2009/2010 conditions by removing topographic data beyond the current mouth berm location. This modification resulted in relatively modest changes to the SCRE stage-area and stage-volume relationships (Figures 4-12 and 4-13).

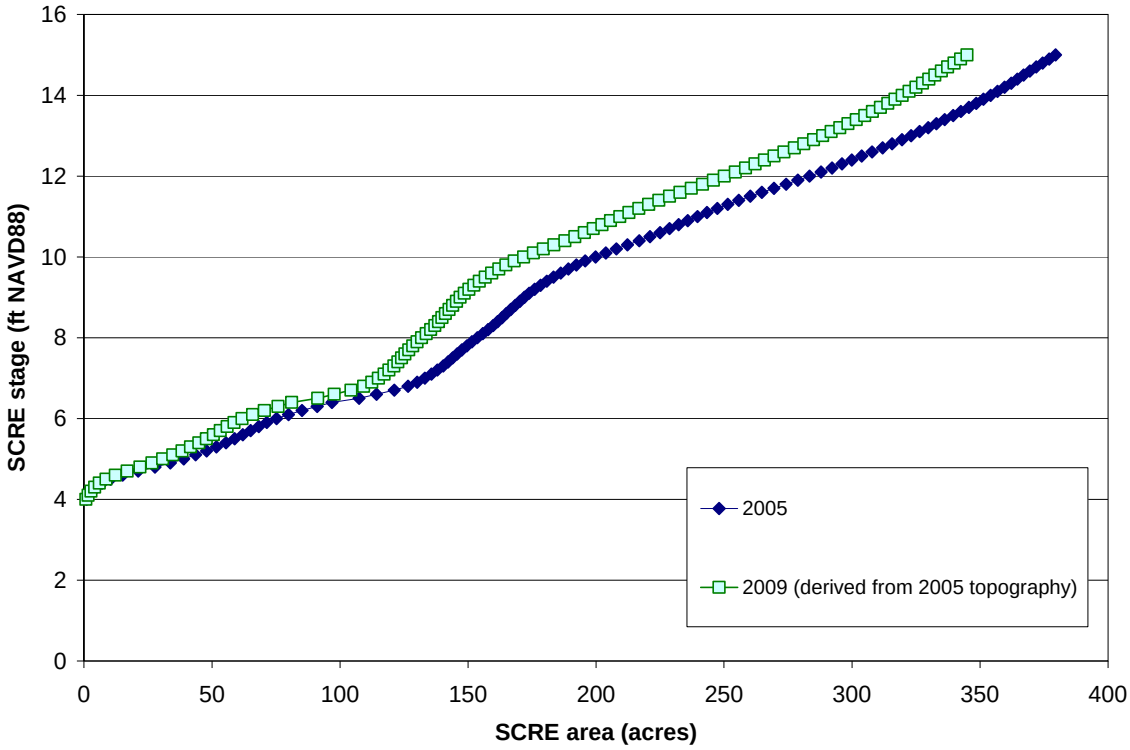


Figure 4-12. Comparison of SCRE inundated area with increasing stage for 2005 and 2009 estuary conditions.

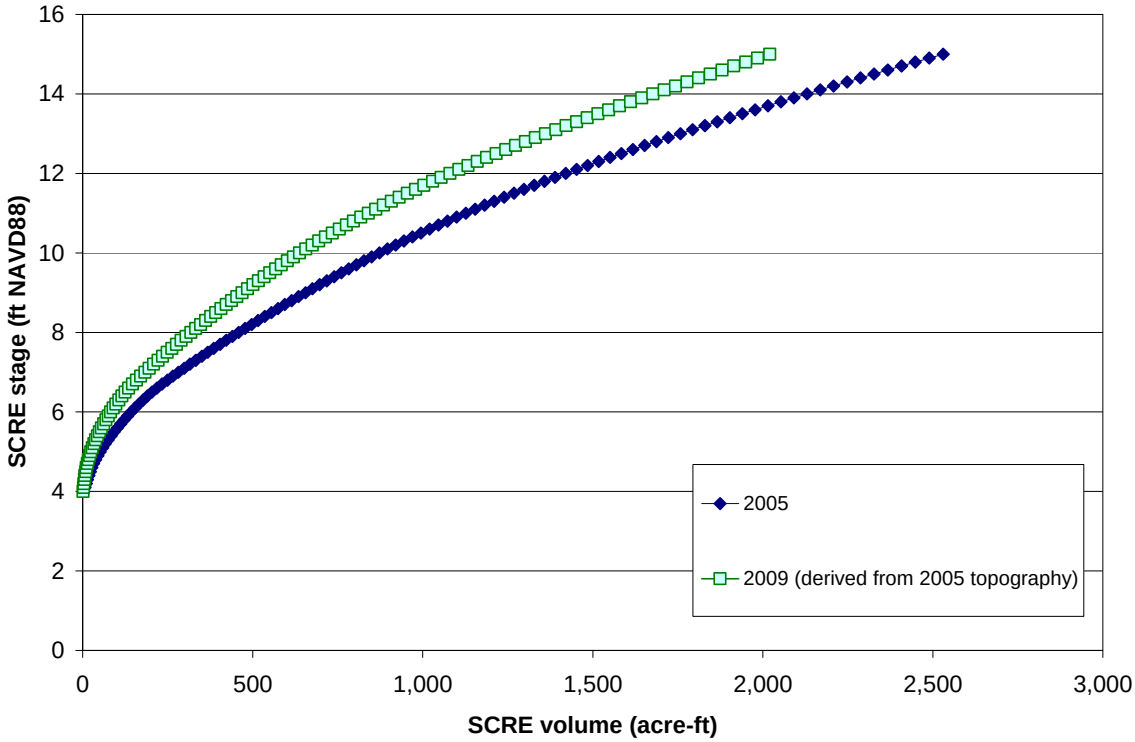


Figure 4-13. Comparison of SCRE inundated volume with increasing stage for 2005 and 2009 estuary conditions.

4.2.2.2 Monitored stage, area, and volume

Estuary stage data used for developing the SCRE water balance were compiled from two sources: data collected explicitly for this study and data from the City's on-going SCRE water quality monitoring efforts. SCRE stage monitored specifically for this study occurred at two locations along the SCRE south bank (Figure 4-14). The downstream location (SR-1) was in the lower SCRE adjacent to the channel on the south bank that originates in the McGrath State Beach campgrounds. The upstream location (SR-2) was in the middle SCRE also along the south bank approximately 200 m (660 ft) upstream of the downstream gage. These stage recorders were installed at the SCRE bed surface in sheltered locations and began monitoring stage on a 30-minute time step on September 1, 2009. For the period before the logger installation (May 2009–August 2009) and during periods when the SCRE stage was below the installed loggers (i.e., when just the lower SCRE downstream of the loggers is inundated), water depth measurements from the City's water quality sondes were used to fill in the SCRE stage data time series. The City has two sondes installed in the SCRE: one in the northern portion of the middle SCRE in the outfall channel (North sonde, or SCR1) and one in the southern portion of the middle SCRE approximately 300 m (984 ft) upstream from the mouth berm (South sonde, or SCR2). Water depth measurements at the South sonde have been correlated to SR-1 stage measurements so that the sonde-recorded depths can be easily converted to SR-1 stage. During times when the stage was below the installed loggers or when the City sondes are removed for data downloading or other purposes, the SCRE stage was inferred based on stage data for previous or subsequent time periods and recorded tide elevation data at Santa Barbara, CA (NOAA Station 9411340). A more complete description of SCRE stage data collection methodology can be found in the project Monitoring Plan (Stillwater Sciences 2009).

Figure 4-14. Location of SCRE stage recorders and groundwater monitoring wells.

The compiled SCRE stage data at SR-1 and adjacent tide elevation data from May 20, 2009 to September 15, 2010 are shown in Figure 4-15. During this time period, the mouth was closed for nearly twice as long as it was open (the mouth was closed over 70% of the time) and the SCRE stage ranged from approximately 1.5 m (5.0 ft) NAVD88 to 4 m (14 ft) NAVD88. Several mouth berm breaching events occurred during this time period due to different factors: (1) as a result of unauthorized channel trenching (September 11, 2009); (2) as a result of storm-induced increase in river discharge (October 14, 2009; December 13, 2009; January 18, 2010); and (3) as a result of wave overwash (April 7, 2010). Following mouth berm breaching, the mouth remained open and for 8 to 63 days. The amount of time the mouth remained open depended on both river discharge and tidal elevation. The maximum SCRE stage during open-mouth conditions (2 m [9 ft] NAVD88 on February 28, 2010) occurred directly after a storm event due to higher river discharge and elevated tidal elevation. Following berm closure after all breaching events, the SCRE filled rapidly for the first several days (over 0.9 m [3.0 ft] stage increase in 7–17 days), then either breached soon after due to natural means, or approached a quasi-equilibrium "full" stage of approximately 3 m (10 ft) NAVD88. Comparing these data to the SCRE stage data for 2004–2005 reported by Nautilus Environmental (2005) shows that the "full" stage is similar between the two time periods; however, the SCRE can currently maintain the "full" stage during low river flow conditions whereas in the past the mouth would consistently breach once the "full" stage was reached (see Figure 4-10).

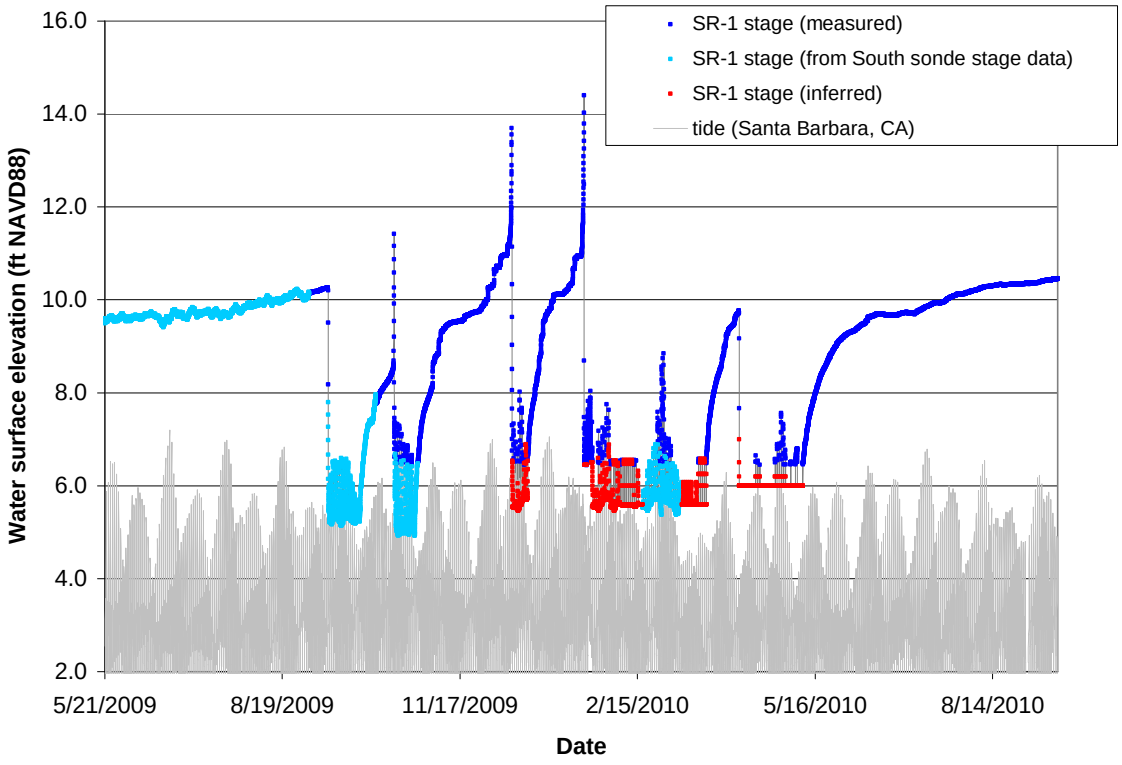


Figure 4-15. SCRE stage and adjacent tidal elevation for the entire study period (May 20, 2009 to September 15, 2010).

Analysis of the amount of time a given stage value was equaled or exceeded (i.e., the time a given bed elevation was inundated) from May 2009 to September 2010 provides an indication of SCRE inundation dynamics on a seasonal basis (Figure 4-16). The SCRE bed was inundated above 1.5

m (5.0 ft) NAVD88 for the entire time period, above 2.8 m (9.4 ft) NAVD88 approximately half the time (i.e., the average inundation elevation), and inundated above the "full" elevation (3.0 m [10 ft] NAVD88) less than 10% of the time. During the spring (April–June) and summer (July–September) months when river discharge was low and the mouth berm was closed more often than not, the average inundation elevation was close to the average for the entire time period (2.9 m [9.5 ft] NAVD88 for spring and 3.0 m [9.8 ft] NAVD88 for summer). The SCRE was not inundated to the "full" elevation during the spring; however the SCRE bed was inundated above this elevation approximately 5% of the time during the summer due to the continued SCRE filling following mouth closure in the spring. The fall (October–December) and winter (January–March) months were characterized by more open-mouth periods relative to the rest of the year, resulting in a relatively lower average inundation elevation. The extended SCRE filling periods and storm-induced increases in SCRE stage during the fall and winter resulted in the "full" elevation being exceeded approximately 15% and 10% of the time, respectively.

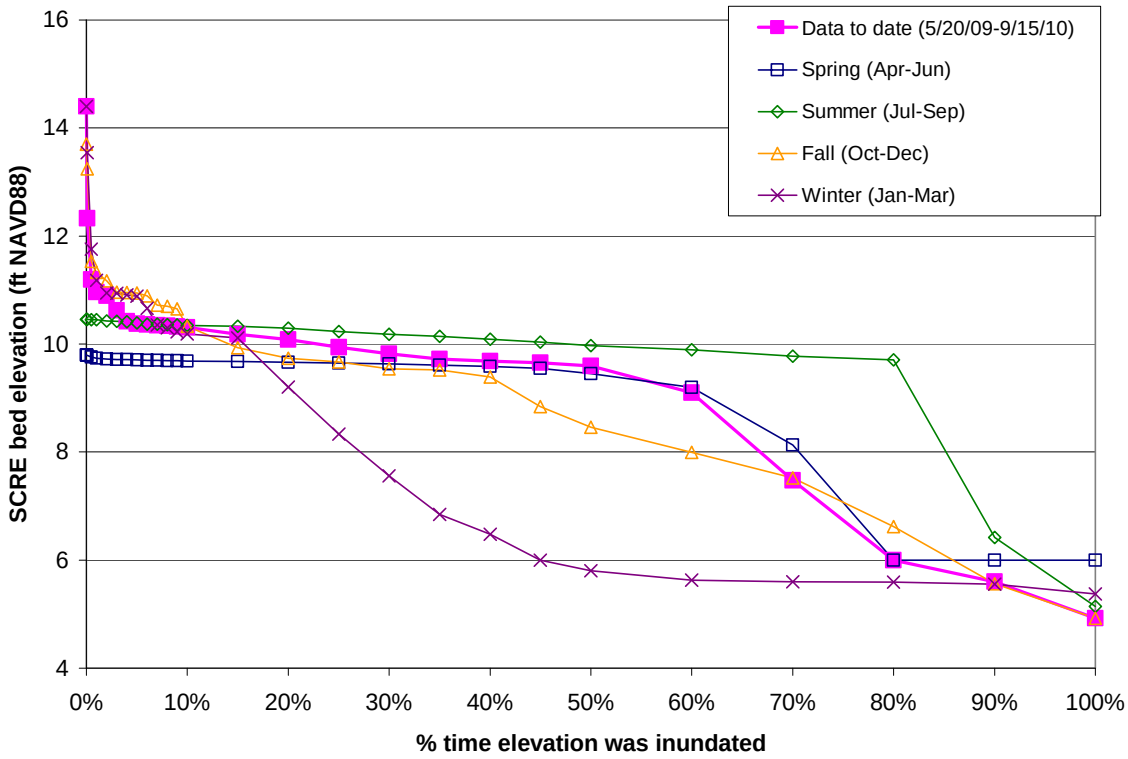


Figure 4-16. SCRE inundated elevation exceedance probability for the entire study period and by season.

Combining the SCRE stage data from May 2009 to September 2010 with the stage-area and stage-volume relationships derived from the 2009 bathymetry illustrates area and volume dynamics during open- and closed-mouth conditions. During this monitoring period, the SCRE inundated area ranged between 10 and 133 ha (27 and 330 ac) ("full" area of 72 ha [179 ac]) (Figure 4-17) and SCRE volume ranged from 20 acre-ft to 1,800 acre-ft ("full" volume of 680 acre-ft) (Figure 4-18). As would be expected, the time that individual area or volume values were exceeded have similar trends to the SCRE bed inundation analysis (i.e., the time that a given stage value was exceeded) shown above. Analysis of the area exceedance time shows that the average SCRE wetted area for the entire time period was approximately 87% of the "full" area.

The wetted area exceedance was the maximum value during summer (94% of the "full" area), and the lowest value during winter (31% of the "full" area). Similarly, analysis of volume exceedance time shows that the average SCRE volume for the entire time period was 82% of the "full" volume, with the maximum value during summer (92% of the "full" volume), and the lowest value during winter (10% of "full" volume).

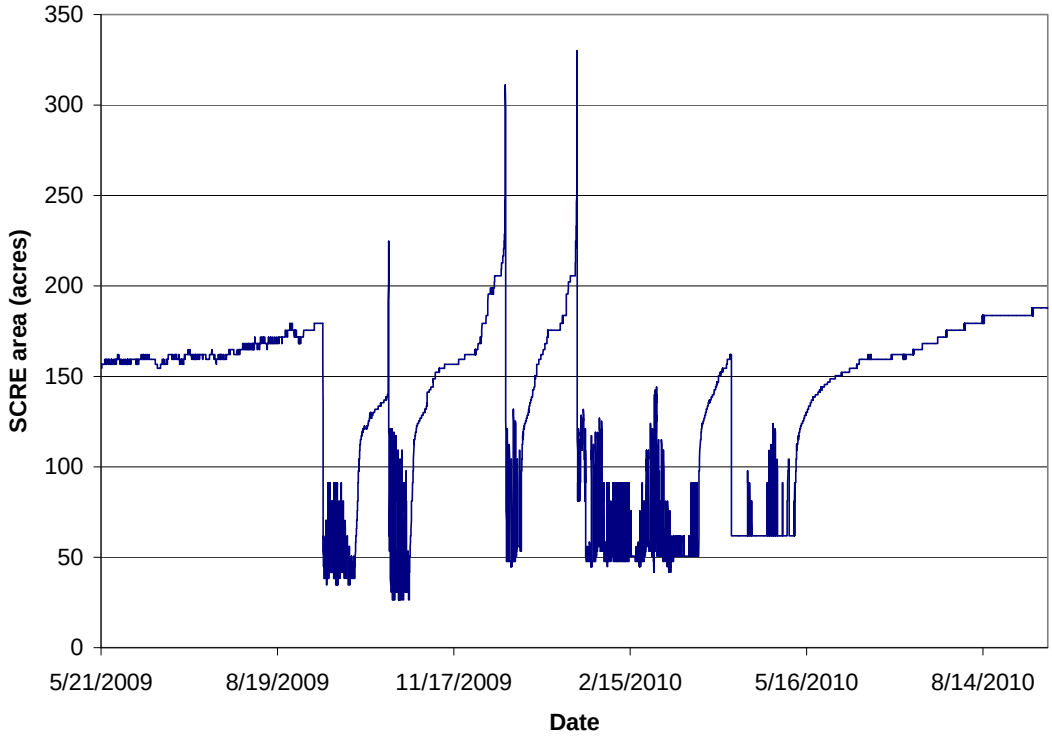


Figure 4-17. SCRE inundated area for the entire study (May 20, 2009 to September 15, 2010).

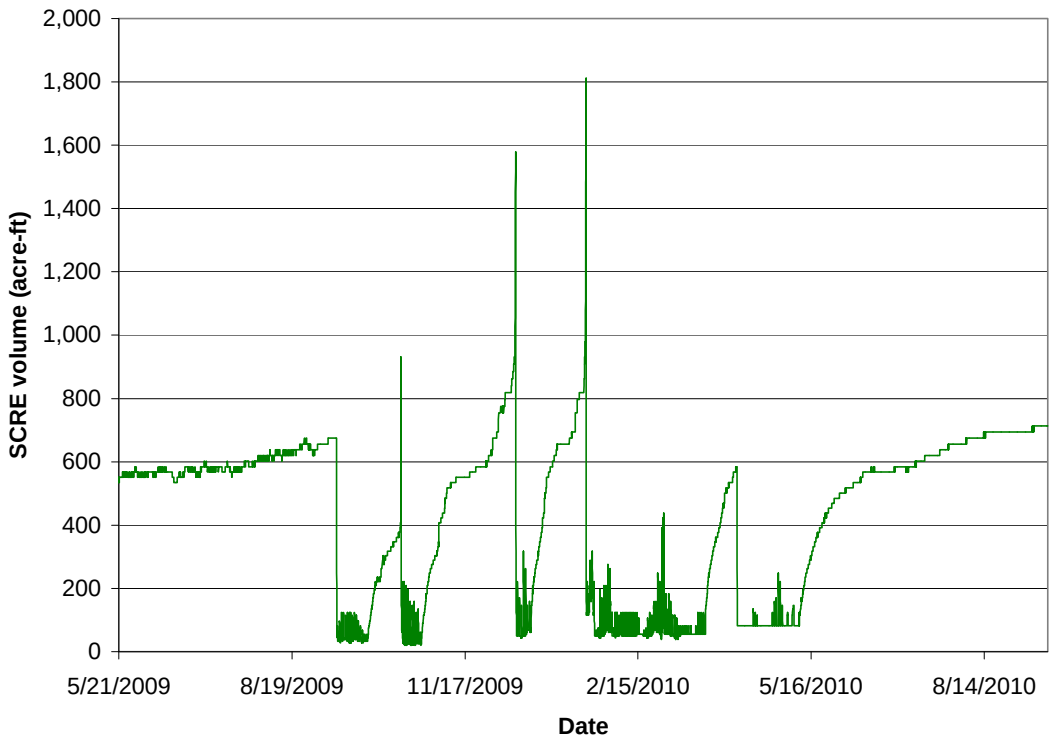


Figure 4-18. SCRE inundated volume for the entire study period (May 20, 2009 to September 15, 2010).

4.2.2.3 Evaporation

The volume of SCRE water lost to evaporation was derived from a combination of hourly pan evaporation (i.e., depth of water evaporated from an open volume of water) at the El Rio-UWCD Spreading Grounds (VCWPD Station 239), approximately 9 km (6 mi) east of the SCRE, and the SCRE inundated area time series. Pan evaporation is a function of air temperature, humidity, solar radiation, and wind, all of which can vary with increasing distance from the ocean and increase in topography. For the purposes of this study, the relatively close proximity of the SCRE and the El Rio-UWCD spreading grounds allows for the assumption that these conditions are the same for both sites. Pan evaporation rates are usually higher than actual evaporation rates due to the heating of the metal pan throughout the day. Therefore, pan evaporation was reduced by 30% to get actual evaporation rates (see Linacre 1994 for more information). Hourly evaporation estimates were then converted to a 30-minute time step assuming a constant rate of evaporation throughout any given hour, which is deemed appropriate for the purposes of constructing the water balance. In general, this method ensured the accurate assessment of the volume of SCRE water loss to the atmosphere as the SCRE expands and recedes during open- and closed-mouth conditions.

The evaporation volume time series for October 2009 to September 2010 is shown in Figure 4-19. As expected, evaporation rates were the highest during the spring and summer months when the mouth was closed and the SCRE was filling (corresponding to time periods with relatively high air temperature, solar radiation, and water availability). The maximum evaporation rate for this time period was 0.21 acre-ft, which is equivalent to a constant 5 cfs flow of water out of the SCRE.

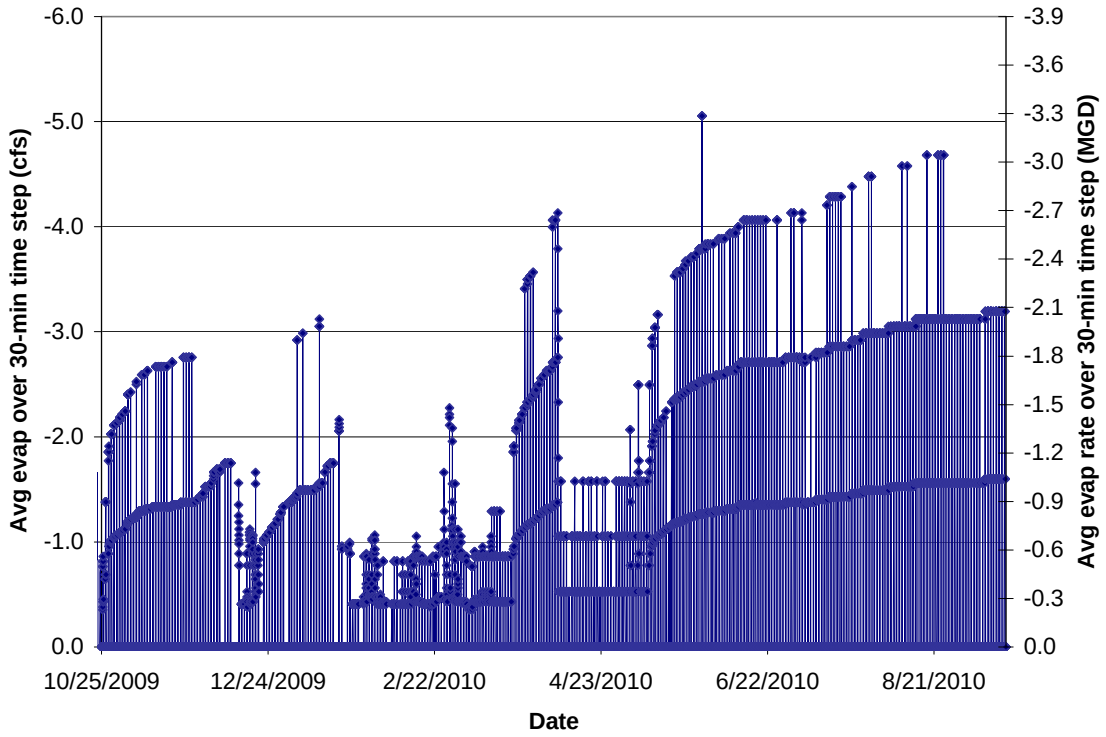


Figure 4-19. Evaporation volumetric loss rate from the SCRE during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.4 Precipitation

The volume of direct precipitation onto the SCRE was derived from a combination of hourly precipitation in downtown Ventura (VCWPD gage 66E) and the SCRE inundated area time series. Similar to the evaporation time series, hourly precipitation estimates were converted to a 30-minute time step assuming a constant rate of precipitation within any given hour. In general, this method ensured that the volume of precipitation that fell directly onto the SCRE during storm events was accurately assessed.

The precipitation volume time series from October 2009 to September 2010 is shown in Figure 4-20. There were four storms during this time period that delivered a considerable amount of precipitation (>2.5 cm [1 in] of water) to the SCRE and upstream watershed and caused increased Santa Clara River discharge and mouth breaching. The maximum rate of precipitation input occurred on January 18, 2010 during a storm that caused mouth breaching. Approximately 3.7 acre-ft of rain fell onto the SCRE over a 30-minute time period on that day, which is equivalent to a constant 90 cfs flow into the SCRE.

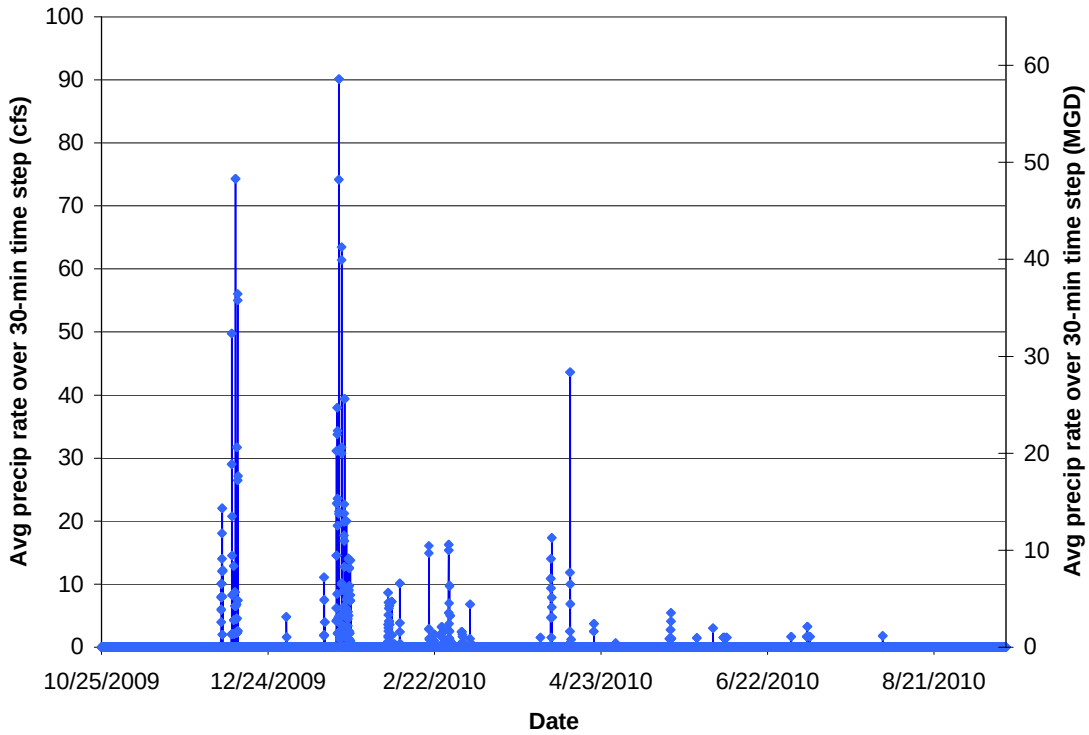


Figure 4-20. Precipitation volumetric inflow rate to the SCRE during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.5 Monitored groundwater flow across south bank

In order to assess groundwater flow rates between the SCRE and the subwatershed area to the south, water table elevation was monitored along a transect extending south through McGrath State Beach (Figure 4-14). The transect was comprised of three monitoring wells located approximately 260 m (870 ft) (GW-1), 550 m (1,820 ft) (GW-2), and 1,040 m (3,400 ft) (GW-3) from the SCRE south bank within the foredune complex between the SCRE and McGrath Lake. All wells were installed within the Semi-perched aquifer to a depth of 7 m (25 ft) near well locations from previous hydrologic studies (ESA 2003 and URS Corporation 2005). The monitoring wells were installed and water table elevation began being monitored on a 30-minute time step on October 26, 2009. A more complete description of water table elevation data collection methodology can be found in the project Monitoring Plan (Stillwater Sciences 2009).

The water-table elevation at the three monitoring wells and the SCRE stage from October 2009 to September 2010 is shown in Figure 4-21. These data illustrate the interplay between SCRE stage and the water table elevation during closed-mouth, filling conditions and show the groundwater gradient between the floodplain to the south and the SCRE during closed- and open-mouth conditions. During the first two closed-mouth, filling periods (October 26, 2009–December 13, 2009 and December 21, 2009–January 18, 2010), the groundwater gradient was initially towards the SCRE but as the SCRE continued to fill, the gradient switched within a matter of several days and the water table elevation at GW-1 increased to essentially the same elevation as the SCRE stage. As SCRE stage and GW-1 water table elevation continued to rise, the water table at GW-2 and GW-3 also increased due to groundwater flow from the SCRE and from precipitation

infiltration. Following the mouth breaching on December 13, 2009, the water table responded quickly and a groundwater gradient towards the SCRE was re-established. After the January 18, 2010 breach, the rapid draining of groundwater towards the SCRE caused the water table elevation along the entire transect to decrease in concert. The increase in SCRE stage during the closed-mouth period between March 21, 2010 and April 7, 2010 caused the GW-1 water table elevation to increase by over 0.6 m (2.0 ft); however, post-breach groundwater drainage caused the water table to decrease within three weeks of the mouth breach. The closing of the mouth on May 10, 2010 caused the water table to rise and resulted in a groundwater gradient away from the SCRE; however the water table at GW-1 increased at a rate much lower than during the previous three closed-mouth periods. This is likely due to the fact that the previous closed-mouth periods occurred during wetter periods when a higher groundwater table caused more ground saturation adjacent to the SCRE.

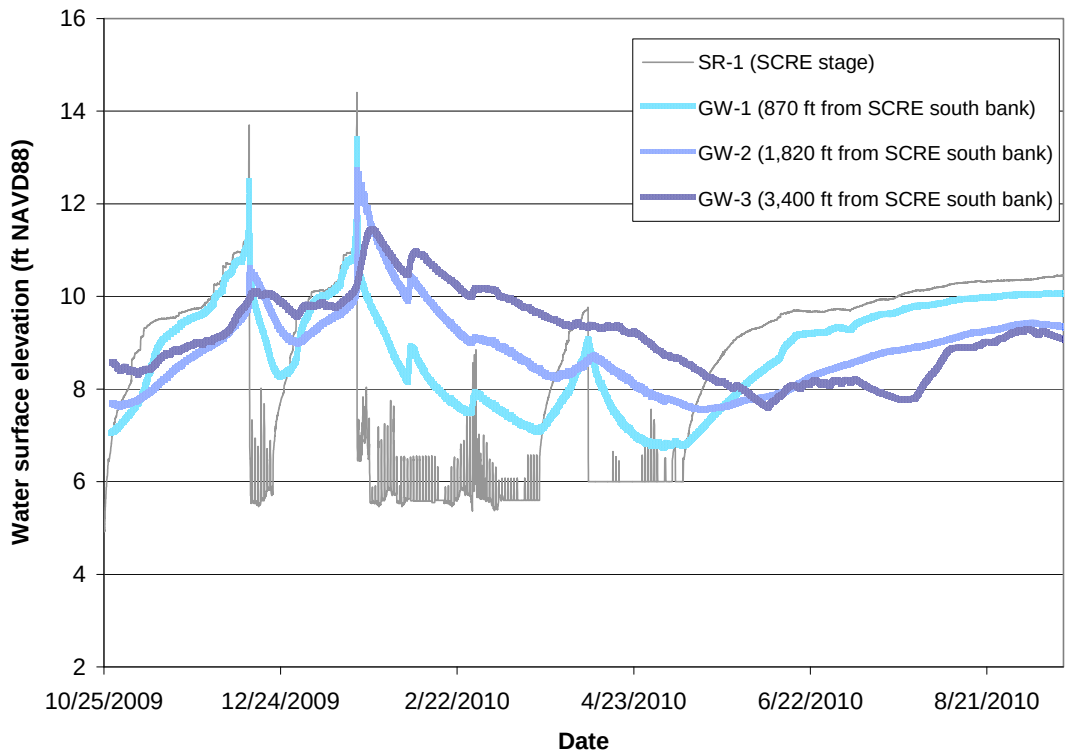


Figure 4-21. SCRE stage and adjacent groundwater elevation for the water balance period (October 25, 2009 to September 15, 2010).

Estimates of groundwater flow volumes were made using Darcy's Law and groundwater flow area. Darcy's Law combines hydraulic conductivity, or the ability of sediment to transmit water, and hydraulic gradient to calculate flow velocity, and is expressed as:

$$V = -K\left(\frac{dh}{dl}\right)$$

where V is flow velocity; K is hydraulic conductivity; and dh/dl is hydraulic gradient, or the ratio of the difference in elevation between the adjacent water table and the SCRE (dh) and the distance over which that gradient applies (dl). The hydraulic gradient was calculated on a 30-

minute time step using a combination of SCRE stage data and the water table elevation measurements at GW-1, GW-2, and GW-3. Hydraulic conductivity estimates were derived from the particle size data derived from samples taken at various depths at each monitoring well location and the Hazen equation for determining the hydraulic conductivity of sand, which can be expresses as:

$$K = C(D_{10})^2$$

where C is an empirical coefficient based on sand type (a value of 100 was used here) and D_{10} is the particle size for which 10% of the sample is finer. Analysis of the sediment samples provided an upper zone (0–7 m [0–25 ft] depth) D_{10} of 0.29 mm and a lower zone (>7 m [25 ft] depth) D_{10} of 0.10 mm. These particle values result in a K of 70 m/day (230 ft/day) and 10 m/day (33 ft/day), respectively. The McGrath Lake Watershed Management Study (URS Corporation 2005) calculated an average K value of 30 m/day (100 ft/day) for the area adjacent to McGrath Lake using measured data, suggesting our K values are of appropriate magnitude.

The groundwater flow velocity was combined with flow area to determine groundwater flow volume on a 30-minute time step. The flow area was derived from the flow length and the depth of groundwater flow. The flow length used in the analysis was 1.6 km, or half the distance from the SCRE mouth to the Bailard Landfill (the area where groundwater flow rate is presumably very low). The base of the groundwater flow field was the bottom of the Semi-perched aquifer, which is assumed to be at approximately -18 m (-60 ft) NAVD88, or approximately 23 m (75 ft) below the southern floodplain ground surface (estimate taken from Turner 1975). The top of the flow field therefore determined the flow depth at each time step and is calculated from adjacent water table elevation at GW-1.

The groundwater flow rate from the southern floodplain between October 2009 and September 2010 is shown in Figure 4-22. The maximum rate of groundwater flow out of the SCRE (i.e., negative flow rate) occurred just before the storm-induced breach on December 13, 2009 when the SCRE volume was above “full” capacity (a constant 2 cfs flow rate over 30 minutes). Conversely, the maximum groundwater flow rate into the SCRE (i.e., positive flow rate) occurred just after the storm-induced breach on January 19, 2010 when the SCRE was draining rapidly (a constant 2.7 cfs flow rate over 30 minutes). During the extended open-mouth period between January 19, 2010 and March 21, 2010, the average groundwater flow rate into the SCRE was approximately 0.4 cfs over 30 minutes. During the periods when the mouth was closed and the SCRE was filling, the groundwater flow rate out of the estuary stabilized at approximately 0.2 cfs.

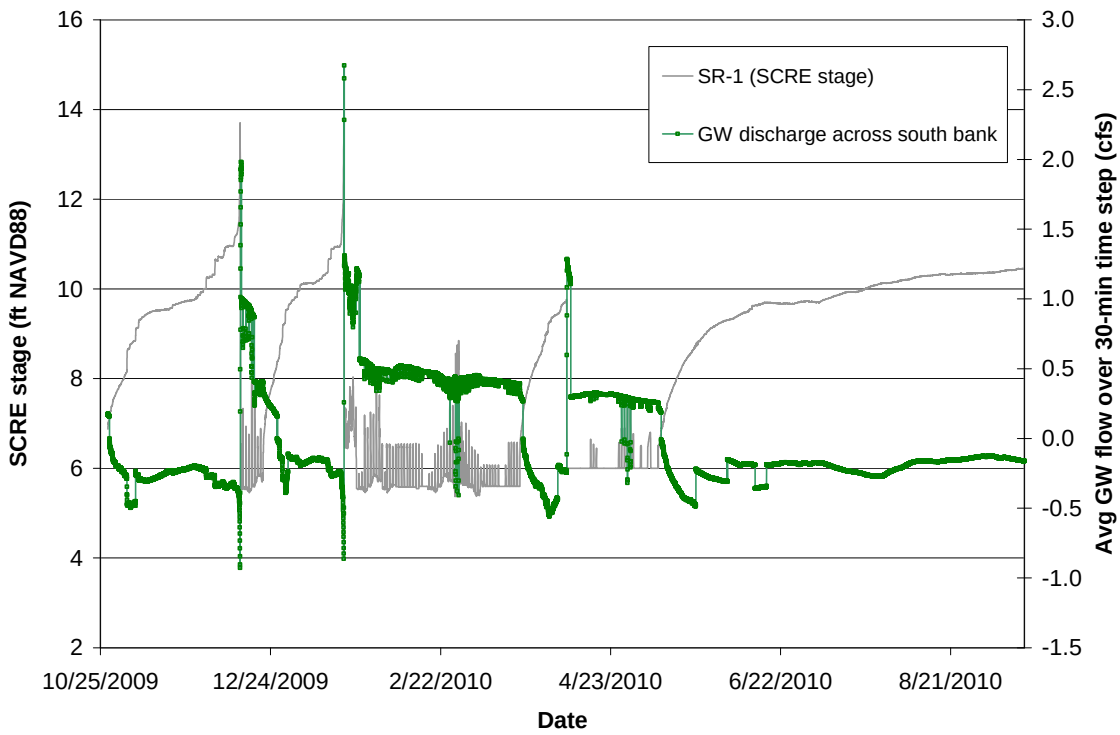


Figure 4-22. SCRE stage and groundwater volumetric inflow (positive flow rate) and outflow (negative flow rate) across the south bank during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.6 Groundwater flow from the VWRF ponds

As with the groundwater flow across the south bank, groundwater contribution to the SCRE from the VWRF Wildlife Ponds was calculated using Darcy's Law combined with flow area estimates. The overall approach was to use data collected at and adjacent to the ponds to estimate the input variables and use the reported average annual pond percolation rate of 1 MGD (Carollo Engineers 2010b) to constrain the groundwater flow rate. As the Wildlife Ponds are unlined, they readily supply water to the Semi-perched aquifer and the SCRE. The hydraulic gradient between the ponds and the SCRE, therefore, was calculated from the difference between the Wildlife Ponds average water surface elevation (5.8 m [19 ft] NAVD88, as recorded in the 2005 LiDAR dataset) and the SCRE stage, and an average distance between the ponds and the SCRE of 60 m (200 ft). The flow area was calculated from the length of the bank along the ponds (875 m [2,870 ft]) and the local depth of groundwater flow, which was calculated as the difference between the local aquifer base elevation (assumed to be -18 m [-60 ft] NAVD88) and average of the pond and SCRE water surface elevation (the presumed average water table elevation). The hydraulic conductivity was then calculated as a fitted parameter that resulted in the average groundwater flow rate over the water balance period being similar to the reported 1 MGD, or average 1.6 cfs, pond percolation rate. This process yielded a hydraulic conductivity of 4 m/day (13 ft/day), which although considerably less than the hydraulic conductivity estimates for the floodplain south of the SCRE, is within the range of hydraulic conductivity for a well-sorted sand or sand and gravel aquifer (3-300 m/day [10-1,000 ft/day], from Bear 1972).

The rate of groundwater flow between VWRF Wildlife Ponds and the SCRE between October 2009 and September 2010 is shown in Figure 4-23. Because the ponds' water surface elevation was higher than the maximum SCRE stage over the water balance period, the direction of groundwater flow was consistently from the ponds to the SCRE. During open-mouth periods, the groundwater flow rate into the SCRE ranged from 1.5 cfs to 1.9 cfs. The maximum rate of groundwater flow occurred during low-tide when SCRE was draining and the hydraulic gradient between the ponds and the SCRE was therefore the greatest. During the closed-mouth periods, the groundwater flow rate ranged from 0.8 cfs (occurring just before storm-induced mouth breach on January, 18 2010) to 1.9 cfs (occurring just after mouth closure). During the extended closed-mouth period from May 9, 2010 to September 15, 2010, the groundwater flow rate from the ponds to the SCRE stabilized at 1.4-1.5 cfs.

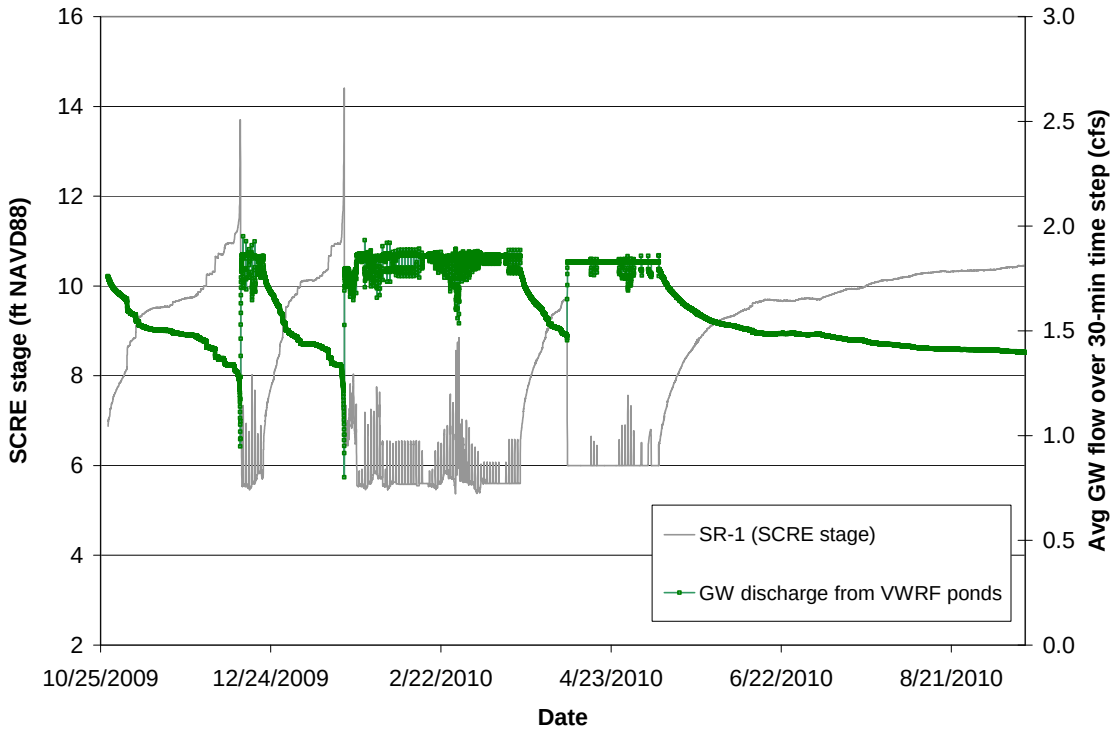


Figure 4-23. SCRE stage and groundwater volumetric inflow (positive flow rate) from the VWRF Wildlife Ponds during the water balance period (October 25, 2009 to September 15, 2010)

4.2.2.7 Subsurface flow across mouth berm

Similar to the methodology described above for determining groundwater flow from the adjacent floodplains, subsurface flow volumes across the mouth berm were calculated using Darcy's Law with measured hydraulic gradients, hydraulic conductivity, and flow area. Hydraulic gradients were calculated on a 30-minute time step using a combination of measured SCRE stage data, tide elevation measured at Santa Barbara, CA (NOAA Station 9411340), and an assumed mouth berm width. The mouth berm width used in this analysis was 90 m (300 ft), which was derived from recent aerial photographs and is considered the average width that takes into account variability due to tidal elevation and SCRE stage fluctuations. Hydraulic conductivity estimates were developed using the Hazen equation and a D_{10} value derived from sediment samples taken at

depth along the mouth berm and from the samples taken at the monitoring well locations. Analysis of the sediment samples provided an upper zone D_{10} of 0.28 mm, which equates to a hydraulic conductivity that is essentially the same as the upper zone in the floodplain to the south of the SCRE. It was assumed that the sediment deposits below the mouth berm were the same as below the floodplain to the south of the SCRE, resulting in a 7-m (25-ft) thick upper zone with a hydraulic conductivity of 70 m/day (230 ft/day) and a 15-m (50-ft) thick lower zone conductivity of 10 m/day (30 ft/day).

As with the groundwater volume calculations, the subsurface flow velocity was combined with flow area to determine flow volume on a 30-minute time step. The flow area was derived from the mouth berm length (variable with stage, reaching a maximum value of 2,440 m [8,000 ft] when the SCRE stage reaches 3.7 m [12 ft] NAVD88) and the depth of flow. The bottom of the Semi-perched aquifer was assumed to be the same elevation as was used to determine groundwater flow across the southern SCRE bank (-18 m [-60 ft] NAVD88), and the top of the subsurface flow field across the mouth berm was set by the SCRE stage.

The subsurface flow volume across the mouth berm from October 2009 and September 2010 is shown in Figure 4-24. The flow direction was almost exclusively out of the SCRE (i.e., negative flow rate) over this entire period. The subsurface flow across the mouth berm was directed into the SCRE for less than 2 hours and had a maximum flow rate of 0.04 cfs. The maximum rate of flow out of the SCRE (i.e., negative flow rate) occurred just before the storm-induced breach on January 18, 2010 when the SCRE volume was at its maximum value for the entire water balance period (a constant 18 cfs flow rate). During the longest open-mouth period between January 19, 2010 and March 21, 2010, the average flow direction was out of the SCRE at a rate of approximately 1.3 cfs. The flow rate out of the SCRE during the extended filling period between May 9, 2010 and September 15, 2010 ranged from approximately 0.5 cfs just after mouth closure to approximately 9.2 cfs when the SCRE reached the "full" stage.

In an effort to corroborate the estimates of subsurface flow through the mouth berm during closed-mouth conditions, we performed a simple SCRE water residence time analysis. Following mouth closure, the SCRE salinity decreases due to the dilution effect of the incoming river, groundwater, and VWRF effluent flow. The rate at which the salinity decreases gives the residence time for that period, or the amount of time water the flows into the SCRE stays in the SCRE. The measured residence time can be used to determine the rate of outflow, which can then be compared to the outflow rate through the closed mouth berm from the water balance approach as an independent check on calculated outflow values.

Using the assumptions that that the South sonde salinity readings (recorded as specific conductivity) were indicative of the average SCRE salinity and that the mouth berm flow rate was the only unconstrained flow rate, we used the longest closed-mouth period with high quality South sonde salinity data (June 29, 2009 to August 28, 2009) for determining residence time. During this period, the SCRE specific conductivity decreased from approximately 4 mS/cm to 2.5 mS/cm as the SCRE volume increased from approximately 570 acre-ft to 640 acre-ft. Because we did not begin monitoring groundwater flow across the south bank until October 2009, considering mouth berm flow the only unconstrained variable during this closed-mouth period was not appropriate. We therefore had to combine the rate of salinity decrease with the inflow and outflow rates from a later closed-mouth period with the same rate of SCRE volume increase (June 20, 2010 to August 20, 2010).

The residence time analysis output an average water loss rate through the closed mouth berm of approximately 8 cfs. The average berm outflow rate calculated from measured hydraulic

gradients, hydraulic conductivity, and flow area for June 20, 2010 to August 20, 2010 was also approximately 8 cfs, suggesting that the method and variables used to determine the flow rate through the mouth berm were appropriate.

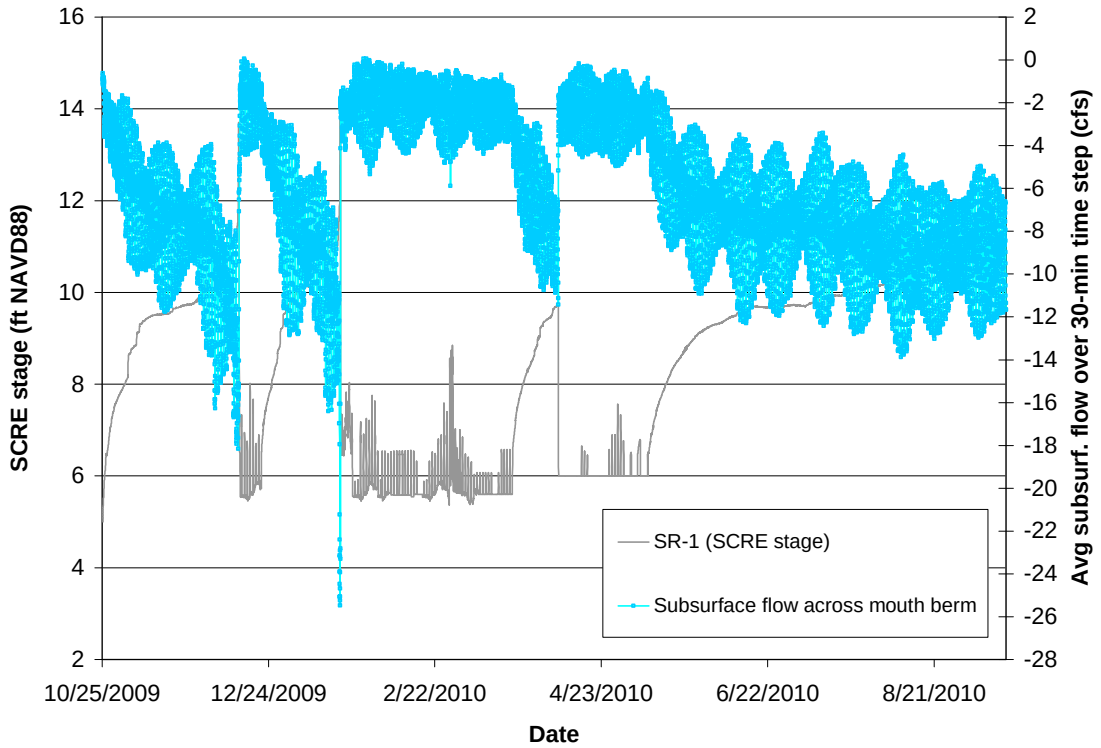


Figure 4-24. SCRE stage and volumetric inflow (positive flow rate) and outflow (negative flow rate) of subsurface water across the mouth berm during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.8 VWRF discharge

The VWRF effluent discharge rate and volumes delivered to the SCRE from the Wildlife Pond system outfall channel were determined from a combination of the daily ETS flow volume estimates made by the City combined with estimates of evaporation loss from the ponds, estimates of percolation loss, and estimates of effluent diverted for recycled water uses. Evaporation loss was estimated on a 30-minute time step from the reduced El Rio-UWCD spreading grounds pan evaporation estimate (Section 4.2.2.3) combined with the total pond area (7.7 ha [19.1 ac]). Pond percolation loss was determined by distributing the estimated annual average 1 MGD loss (Carollo Engineers 2010b) evenly over the course of each day in the water balance period. Prior to the ETS flow meter relocation in January 2010, recycled water diversions also had to be considered. Approximately 0.3 MGD (0.4 cfs) of recycled water diverted for use within the VWRF is not discharged to the Wildlife Ponds and is considered lost (J. Willis, pers. comm., 2011). The amount of recycled water that leaves the VWRF and is used for local irrigation purposes varies throughout the year, reaching a monthly maximum during the summer when irrigation water is in high demand. From January through September 2010, the total daily volume of recycled water diverted for use within the VWRF and for local irrigation ranged from approximately 0.3 to 1.0 MGD (0.4 to 1.5 cfs).

The VWRF surface flow volume from October 2009 to September 2010 is shown in Figure 4-25. Over this time period, the discharge rate ranged from approximately 3 MGD (4 cfs) to approximately 13 MGD (20 cfs), with an average value of approximately 7 MGD (11 cfs). On average, discharge values were lower in the summer and fall months (approximately 6 and 4.5 MGD [9 cfs and 7 cfs], respectively) and higher in the winter and spring months (approximately 9.5 and 8 MGD [15 cfs and 12 cfs], respectively).

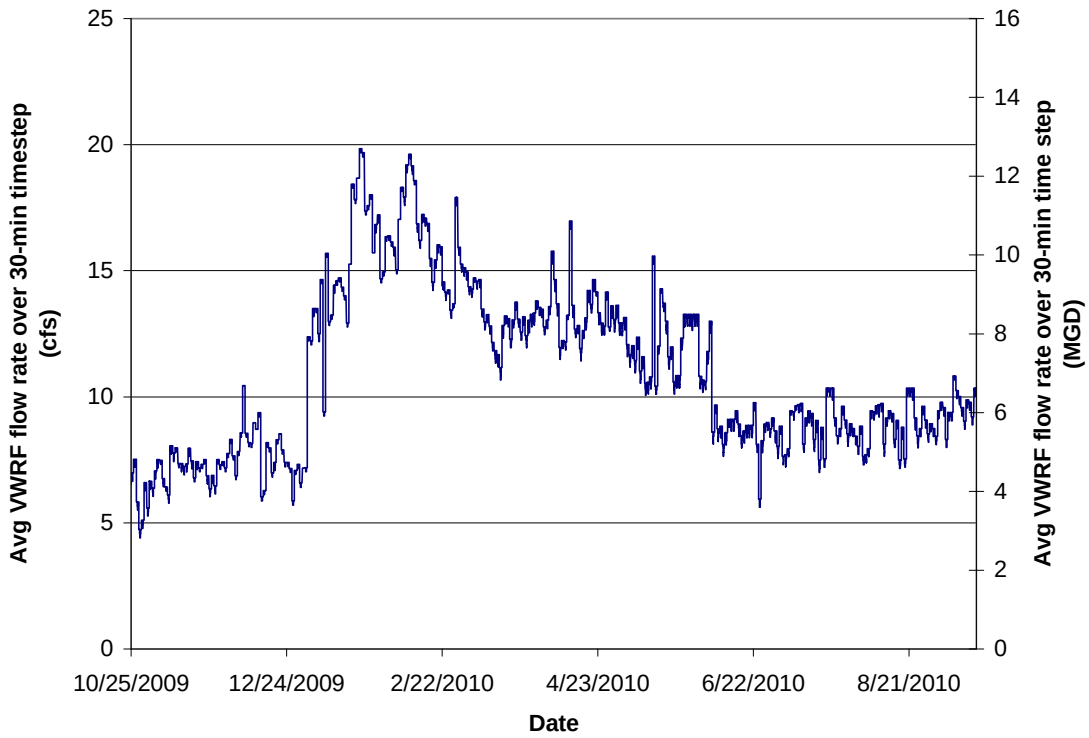


Figure 4-25. VWRF effluent discharge rate into the SCRE during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.9 Santa Clara River discharge

The volume of Santa Clara River discharge flowing into the SCRE was determined primarily from daily mean discharge estimates approximately 2.4 km (1.5 mi) upstream of the SCRE eastern boundary at the Victoria Ave. bridge (VCWPD Station 723). Daily mean discharge rate was converted directly to discharge over 30-minute time step by assuming a constant discharge rate. Each 30-minute time step discharge rate was then converted to discharge volume.

During storm events, the run-off downstream of Victoria Ave. bridge that flows into the SCRE was estimated using the Rational Method. The Rational Method is a simple equation for calculating storm-induced discharge from small areas as a function of contributing watershed and precipitation characteristics, and can be expressed as:

$Q_{peak} = CIA$

where Q_{peak} is the storm peak discharge (cfs), C is a dimensionless run-off coefficient based on contributing watershed land use/vegetation cover, I is rainfall intensity (in/hr), and A is contributing drainage area (ac).

The total contributing drainage area for both the northern floodplain (1,018 ha [2,517 ac]) and southern floodplain (353 ha [874 ac]) was determined from a variety of sources including the 2005 LiDAR dataset collected by Ventura County, Moreland Ditch Hydrology Study (provided by the VCWPD), the McGrath Lake Watershed Management Study (URS Corporation 2005), and City of Ventura Stormwater System Atlas (VPW 2008). A GIS feature dataset of the total contributing area was created and subdivided into discrete parcels based on runoff characteristics. Each unique parcel was assigned a runoff coefficient designation based on degree of development and impervious area extent (Dunne and Leopold 1978). The area draining into the Santa Clara River downstream of Victoria Ave. is primarily cultivated agricultural land (C value between 0.15 and 0.4) and urban development (C value between 0.6 and 0.85). An area-weighted C value was then computed for the total floodplain area to the north (C=0.452) and total floodplain area to the south (C=0.444). Rainfall intensity was calculated using the hourly rainfall records for downtown Ventura (VCWPD gage 66E) and dividing cumulative storm rainfall volume by the storm duration.

The calculated peak discharge was then used to develop simple synthetic, symmetrical storm event hydrographs using the following assumptions:

- no runoff from contributing areas prior to and following storm;
- the time from peak discharge to zero discharge is twice storm duration and is essentially the total hydrograph duration;
- the rising and falling limbs of the storm hydrographs are linear from zero discharge to peak discharge; and
- flow from north and south contributing areas joins lower Santa Clara River at one single confluence point.

Table 4-2 shows the storm duration, intensity, and calculated peak discharge values for the four "significant" storms (storms that delivered more than 2.5 cm [1 in] of rain) that occurred during the water balance period.

Table 4-2. Storm intensity and calculated peak runoff (Q_{peak}) for storm events during the water balance period (October 25, 2009 to September 15, 2010).

Storm event	Storm duration (hr)	Storm total (in)	Storm intensity (in/hr)	Hydrograph duration (hr)	Q_{peak} for northern floodplain (cfs)	Q_{peak} for southern floodplain (cfs)
December 10–15, 2009	57	1.67	0.03	114	33	12
January 17–23, 2010	133	5.81	0.04	266	49	17
February 5–8, 2010	35	1.95	0.06	70	65	22
April 10–13, 2010	13	1.23	0.10	26	93	32

The time series of river discharge volume from the Victoria Ave. bridge gage data combined with the calculated run-off volumes from the floodplain areas downstream of Victoria Ave. bridge from October 2009 to September 2010 is shown in Figure 4-26. The maximum discharge during the storm events ranged from 875 cfs and 9,670 cfs. The daily mean discharge at the beginning of

the second storm event on January 18, 2010 was reported as 2,040 cfs. The water balance analysis presented in Section 4.2.3 suggests that the daily mean flow is approximately one-quarter of this value and that reduction is reflected in Figure 4-26. Error associated with this and other data sources used in this analysis is discussed later in the report.

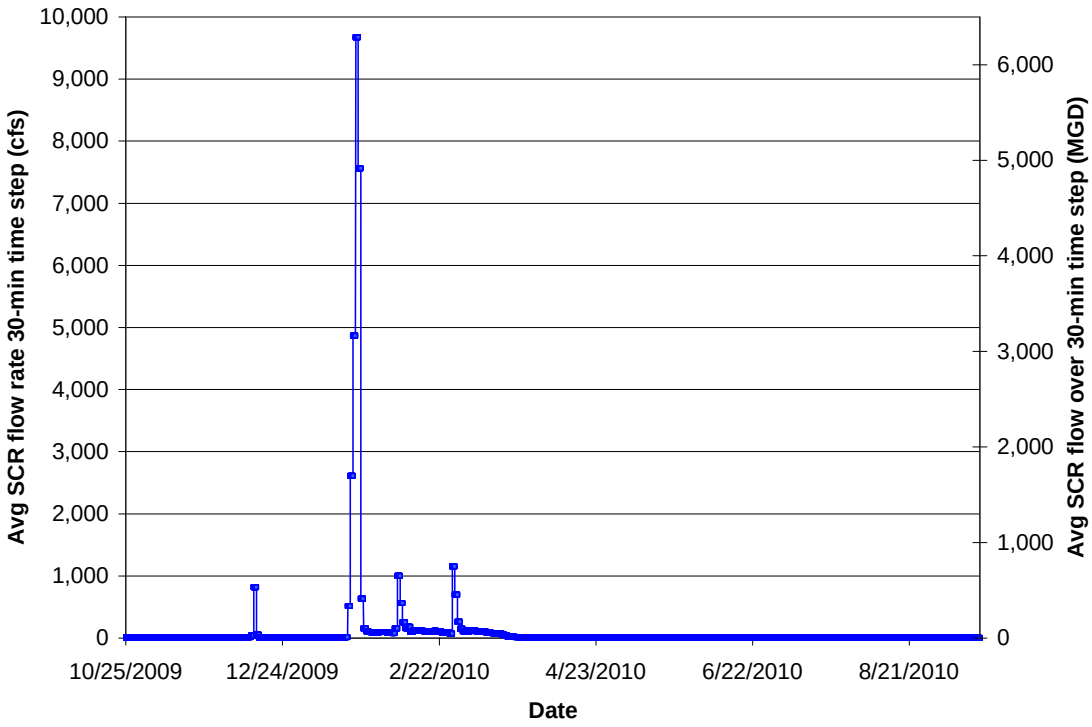


Figure 4-26. Lower Santa Clara River discharge rate into the SCRE during the water balance period (October 25, 2009 to September 15, 2010).

4.2.2.10 Unmeasured groundwater flow

Due to several project constraints, it was not possible to monitor and account for the groundwater flow contribution from the northern floodplain area upstream of the VWRF Wildlife Ponds or the groundwater upwelling along the mainstem Santa Clara River channel between the Harbor Blvd. bridge and Victoria Ave. bridge. As these unmonitored areas are relatively large and the flows are gradient-driven, the flow volumes can be a considerable portion of the total flow contribution during those times when the water table is higher and the SCRE stage is low. Therefore, constraining this combined unmeasured groundwater flow volume was necessary for completing the SCRE water balance.

The inflow and outflow rates during the four closed-mouth periods were used to develop relationships between the unmeasured groundwater flow and SCRE stage (Table 4-3). For each of these periods, all monitored flow components were summed on a 30-minute time step. Beginning at the first time step following mouth closure, a flow volume was then added (presumed to be unmeasured groundwater flow) in an iterative fashion to the sum of all monitored flow components so that the following conditions were met: (1) the total calculated volume increase including the unmeasured groundwater flow was within 25% of the measured SCRE volume increase (i.e., the volume increase derived from the stage-volume relationship) over each 30-minute time step during the first few days of rapid filling; and (2) the total calculated volume

increase including the unmeasured groundwater flow was within 10% of the measured SCRE volume increase for each 30-minute time step once the filling rate and SCRE stage stabilized. This approach resulted in linear relationships between the unmeasured groundwater flow and SCRE stage for each closed-mouth period. As would be expected, unmeasured groundwater flow contribution was highest right after mouth closure and decreased rapidly as the SCRE filled. For this analysis, only flow directed towards the SCRE (i.e., positive flow) was considered. These relationships differed by season, presumably due to differences in local water table elevations (i.e., for the same SCRE stage, more unmeasured groundwater was added to the SCRE during the wetter months because the water table was higher).

Table 4-3. Unmeasured groundwater flow relationships with SCRE stage and calculated maximum discharge rates during the four closed-mouth periods

Closed-mouth period	Period duration	Developed relationship between unmeasured groundwater discharge and SCRE stage	Average discharge rate (cfs)
1	10/25/09–12/13/09	Unmeasured GW discharge = -8,000(SCRE stage) + 81,400	4
2	12/21/09–1/18/10	Unmeasured GW discharge = -18,200(SCRE stage) + 186,000	7
3	3/22/10–4/7/10	Unmeasured GW discharge = -8,300(SCRE stage) + 75,000	3
3	5/9/10–9/15/10	Unmeasured GW discharge = -8,700(SCRE stage) + 74,000	0.3

These relationships were also used for adding unmeasured groundwater flow to the SCRE when the mouth was open. The relationship for closed-mouth period #1 was used for the open-mouth period between December 13, 2009 and December 21, 2009 and the relationship for closed-mouth periods #3 and #4 was used for the open-mouth periods between January 18, 2010 and March 21, 2010, and April 7, 2010 and May 9. 2010.

Field observations and daily mean discharge information were used to help corroborate the unmeasured groundwater flow discharge estimates for dry, baseflow conditions. The daily mean river discharge at the Victoria Ave. bridge (VCWPD Station 723) on November 17, 2009 was reported as 1.7 cfs. Approximately 3.7 km (2.3 mi) downstream at Harbor Blvd. bridge, the average mid-day river discharge was estimated to be approximately 3.8 cfs (average flow velocity = 1 cfs, average flow width = 2.3 m [7.5 ft], and average flow depth = 0.2 m [0.5 ft]). These data are from a dry period before the first rain of the season (i.e., no surface input from adjacent drainage channels) and there was every indication that the 2.1 cfs increase in river flow was primarily from groundwater input between the two bridges. The developed relationship between unmeasured groundwater discharge and SCRE stage shown in Table 4-3 yielded a groundwater contribution of 2.8 cfs for this day. The similarity between the river discharge increase and calculated groundwater discharge provides confidence in the unmeasured groundwater discharge calculated for this day and the approach to developing the relationships between unmeasured groundwater discharge and SCRE stage in general.

4.2.2.11 Surface flow through the mouth channel

The flow volume in and out of the SCRE during open-mouth conditions was calculated on a 30-minute time step from the difference between the measured volume (derived from the SCRE stage-volume relationship) and the sum of all of the other water balance components described above. Therefore, when the sum of all the water balance components was less than the measured SCRE volume, there was surface flow into the SCRE and when the sum of all components was

greater than the measured SCRE volume, there was surface flow out through the open mouth. Flow was directed into the SCRE during those times when the tidal elevation was greater than the SCRE stage and flow was out of the SCRE when tidal elevation was less than the SCRE stage.

4.2.3 Santa Clara River Estuary water balance (WY 2010)

The comparison of measured SCRE volume (i.e., volume derived from stage measurements and the stage-volume relationship) and the SCRE volume calculated by summing all volume components is shown in Figure 4-27. As previously discussed, the volume values are identical when mouth was open and there was tidal exchange due to the method using the volume residual to calculate the volume of surface flow into and out of the SCRE through the mouth. During the closed-mouth, filling periods, the calculated SCRE volume time series tracks the measured volume time series very well; however the calculated SCRE volume was initially much greater than the measured volume on January 18, 2010 when a storm increased river discharge and cause rapid SCRE filling. All other SCRE inflow and outflow volumes for this period were well-constrained, so it was assumed that the disagreement was primarily due to the daily mean flow value from the Victoria Ave. bridge gage (VCWPD Station 723) being over-estimated. Therefore, the river discharge rate into the SCRE on January 18, 2010 was decreased by a factor of 4 (from 2,040 cfs to 510 cfs) to ensure the calculated SCRE volume better matched the measured volume as the SCRE filled.

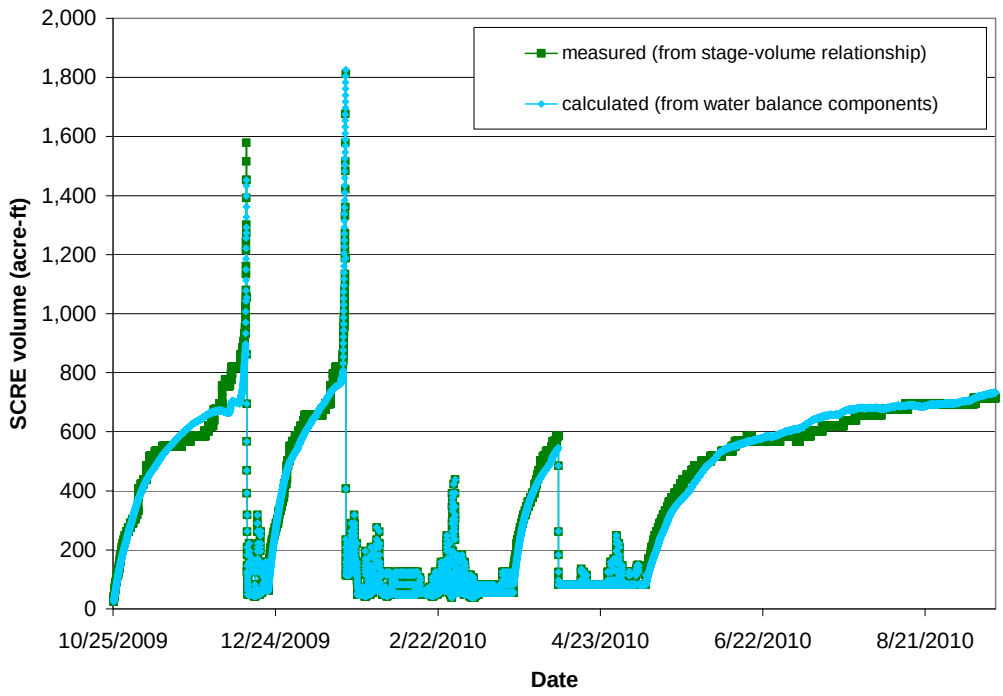


Figure 4-27. Comparison of measured SCRE inundated volume and inundated volume derived from compiling inflow and outflow rates during the water balance period (October 25, 2009 to September 15, 2010).

Compiling the total inflow and outflow volume from each source clearly illustrates the relatively small contribution from the VWRF effluent discharge when considering the entire range of hydrologic conditions during the full water balance period (open- and closed-mouth conditions, high and low river flow) (Figure 4-28). Between October 25, 2009 and September 15, 2010, the

volume of water coming into the SCRE was dominated by Santa Clara River flow (approximately 73,000 acre-ft total volume) and the volume of water leaving the SCRE was dominated by surface flow through an open mouth (approximately 82,000 acre-ft total volume). The majority of the river flow into the SCRE (approximately 84%) occurred during and in the days following the four major storm events. It follows that a large portion of the surface flow leaving SCRE (approximately 80%) occurred during these storm events after the mouth had breached. Overall, Santa Clara River flow comprised approximately 80% of the total inflow volume, VWRF effluent discharge comprised approximately 8%, tidal flow into the SCRE through an open mouth comprised approximately 7%, unmeasured groundwater flow comprised approximately 4%, and all other inflow volume components each comprised ≤ 1% of the total inflow volume. With regards to outflows, surface flow through an open mouth comprised approximately 95% of the total volume leaving the SCRE, subsurface flow across the closed-mouth berm comprised approximately 4%, and all other outflow volume components each comprised less than 1% of the total outflow volume. It should be noted that the relative contribution of both river flow to the total inflow volume and flow through the mouth to total outflow volume is high because these data were collected during a "wet" water year (i.e., total river discharge volume is approximately 3 times greater than the value for a "normal" water year). Combined, these data fit into an overall understanding that storm events drive the dominant sources of both water inflow (from increased river discharge) and water outflow (from sustained open-mouth periods), and that VWRF effluent discharge dominates water inflow to the SCRE only during low-flow conditions.

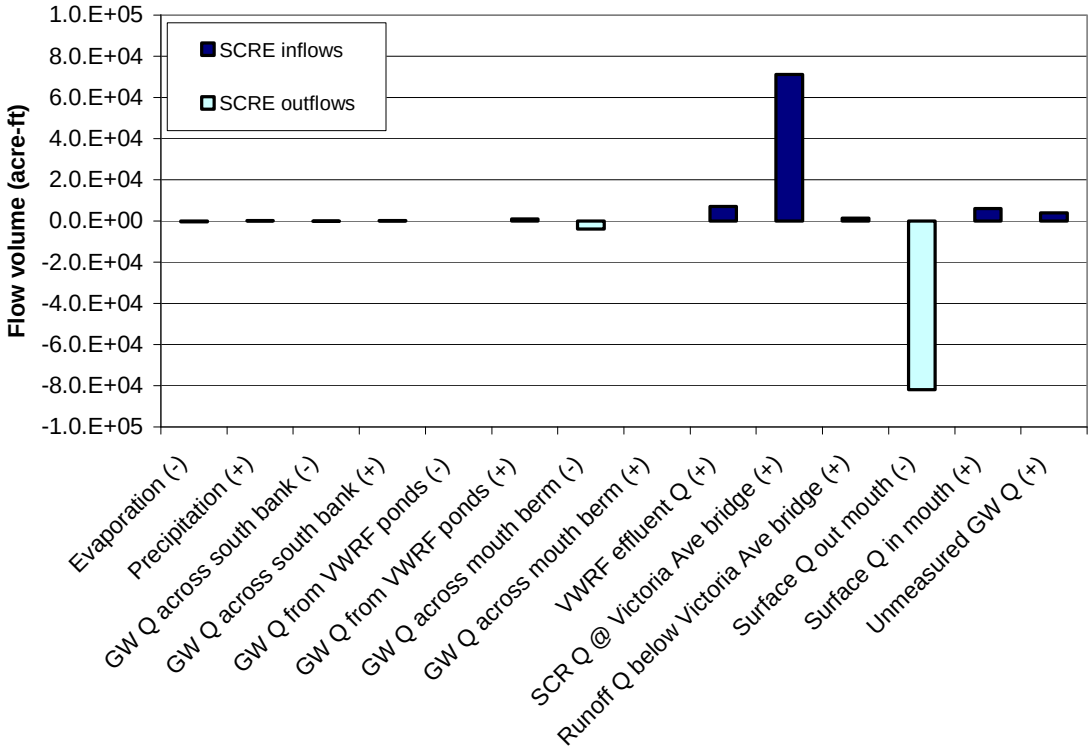


Figure 4-28. Total SCRE inflow (positive) and outflow (negative) volume by source during the water balance period (October 25, 2009 to September 15, 2010).

Examination of component flow volumes during only closed-mouth periods when river flow was low and SCRE was filling shows how the contribution from the VWRF varies by season. For the fall/winter period (closed-mouth conditions between October 25, 2009 and January 18, 2010), the

volume of water entering the closed SCRE was approximately 3 times greater that the volume leaving (Figure 4-29). As expected during the wetter months of the year, the volume of water coming into the SCRE was predominantly from the Santa Clara River (approximately 45% of the total inflow volume), although VWRF effluent discharge combined with groundwater flow from the VWRF Wildlife Ponds was not that much less, accounting for approximately 35% of the total inflow volume. The combined measured groundwater flow from the southern floodplain area and the unmeasured groundwater flow, which is presumed to be dominated by groundwater flow from upstream of the Harbor Blvd. bridge, had a combined contribution of approximately 15% of the total inflow volume. Under "normal" water year hydrologic conditions, both river flow and groundwater flow to the SCRE would presumably be lower on average and the relative flow contribution from the combined VWRF flow sources (effluent discharge and groundwater flow from the Wildlife Ponds) would therefore be greater.

During the closed-mouth periods within the spring/summer 2010 (closed-mouth conditions between March 22, 2010 and September 15, 2010), flow from the VWRF by far dominated the total flow volume into the SCRE (Figure 4-30). Overall, the relatively high outflow rate across the closed-mouth berm caused the total outflow volume for these periods combined to be approximately 1.5 times higher than the inflow volume. The inflow was dominated by flow from the VWRF, with effluent discharge and groundwater flow from the Wildlife Ponds combined contributing approximately 90% of the total inflow volume. The remaining 10% of the inflow volume came from an equal contribution of unmeasured groundwater flow from upstream of the Harbor Blvd. bridge and Santa Clara River flow. Although these data were collected during a "wet" water year, the fact that river flows during the spring/summer 2010 were similar to average flow conditions suggests that both the river and groundwater flow volumes are similar to what would be expected for a "normal" water year.

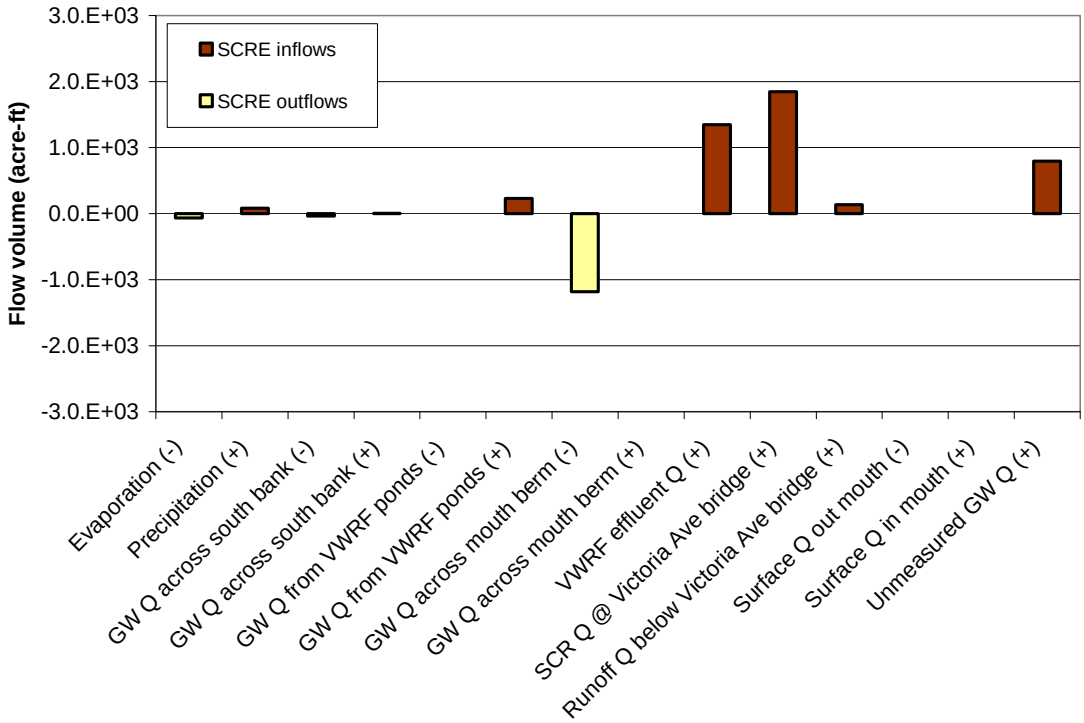


Figure 4-29. Combined total inflow (positive) and outflow (negative) volume by source during the fall and winter closed-mouth, filling periods (October 25, 2009 to December 13, 2009 and December 21, 2009 to January 18, 2010).

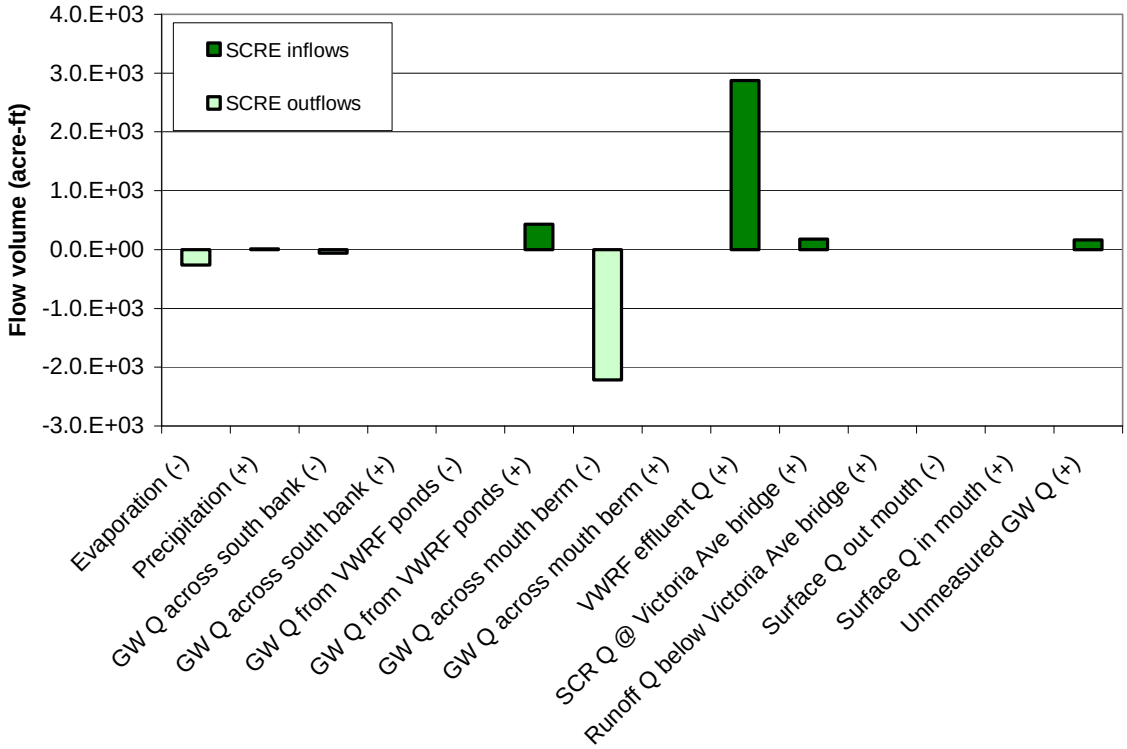


Figure 4-30. Combined total inflow (positive) and outflow (negative) volume by source during the spring and summer closed-mouth, filling periods (March 22, 2010 to April 7, 2010 and May 9, 2010 to September 15, 2010).

4.2.3.1 Uncertainty and error

Because of the range in quality of data sources used and the need to develop estimates for components not directly measured, water balances inherently contain a high degree of uncertainty (Winter 1981, LaBaugh 1985, Kondolf and Matthews 1991). Uncertainty, or error, associated with each flow component used to develop a water balance can range dramatically, from a small percentage to over an order of magnitude of the actual flow contribution. Further, because multiple sources of flow are mathematically combined in a water balance, the resulting model predictions will contain uncertainties at least as large as the most uncertain component. Following error propagation theory, total uncertainty increases whenever quantities are added, subtracted, multiplied, divided or operated in any other fashion (Taylor 1982). In the simplest case of addition of independent source terms, propagation of errors is estimated as a root mean square of these terms. However, for cases where the contributing terms are not independent, additional error terms must be developed quantifying the co-variance in the two terms. For example, estimates of evaporation are based on a stage-area relationship that is in turn based upon direct stage measurements and bathymetric data (each with their own degree of error). Because of this extensive interdependence of the various source terms, no attempt was made to construct an overall uncertainty estimate of the SCRE water balance. However, the dominant sources of uncertainty were assessed and are discussed below.

Within the water balance compiled for this study, error exists within estimates for both SCRE inflow and outflow volumes and the measured total SCRE volume. Much of the estimates of area

and volume are dependent upon LiDAR data from 2005 to develop estimates of the current (2009–2010) SCRE bathymetry. Using a modified version of the 2005 LiDAR to develop a stage-volume relationships use in the water balance analysis likely induces error in the volume estimates due to differences due to instrument error (on the order of millimeters), measurement error (some portions of the SCRE were inundated and represented by the water surface), and errors due to actual changes in the bed morphology from sediment accumulation and scour occurring between 2005 and 2009–2010. Several variables used in the water balance were estimated rather than measured, including the semi-perched aquifer depth within and adjacent to the SCRE, water table elevation between the SCRE and GW-1, mouth berm width with changing tide and SCRE stage, hydraulic conductivity, as well as the effective width and depth of the mouth berm seepage face. No uncertainty estimate can be made for some of these parameters. Furthermore, uncertainty is large even for more direct measurements such as the VWRF effluent flow, which was estimated at the effluent transfer station rather than at the actual outfall into the SCRE.

Perhaps the three largest sources of uncertainty in the SCRE water balance relate to the river inflow, berm outflow, and the estimate of 'unmeasured groundwater flow.' The Santa Clara River discharge is based upon a stage vs. flow rating curve over a large width with a seasonally variable cross-section due to sediment mobilization. In addition to periodic stage measurement equipment malfunctions, the Santa Clara River discharge rating curve is inaccurate at low flows (i.e., when water depth is below the lowest rating curve value) and during periods after storms between rating curve adjustments. As mentioned above, effective cross-sectional area of the berm face as well as the range in hydraulic conductivity throughout the berm cannot be readily determined. Lastly, the 'unmeasured groundwater flow' term in the water balance is essentially determined as the volume of water that makes the sum of all water balance components come close to matching the 'measured' volume. Essentially, this is similar to calculating the residual, or determining the difference between the 'measured' SCRE volume for a given time step and the sum of all other flow components. When there are no error estimates for the water balance components or the 'measured' volume, the residual term effectively aggregates the error of other terms in the SCRE water balance to fit to the observed stage vs. volume relationship.

Although there is error within the water balance analysis, and it was developed during a relatively wet water year, the water balance can be a useful tool for determining the relative volume and area change associated with modifying SCRE inflow and outflow rates. Later in the report, the water balance is used to assess how decreasing the VWRF effluent discharge impacts SCRE inundation dynamics and subsequent focal species habitat area and recreational opportunities at McGrath State Beach.

5 WATER QUALITY

To improve understanding of the functioning of the SCRE, this section includes: contemporary SCRE water quality conditions based on recent data and comparisons to water quality objectives outlined in the Los Angeles Regional Water Quality Control Board Basin Plan (LARWQCB 1994), a discussion of historical water quality conditions, and an assessment of the trophic status and nutrient balance of the SCRE. Although the VWRF has been associated with historical NPDES violations with respect to trace metals and bacteria, no recent violations have occurred. In the SCRE, historical samples periodically exceed Basin Plan objectives for several parameters including ammonia, biostimulatory substances, pesticides, toxicity, and bacteria. While the VWRF currently meets RWQCB discharge effluent limitations, variations in receiving water dissolved oxygen (DO) and pH—as well as periodic exceedances of ammonia toxicity criteria within the SCRE—may be attributed to algal productivity and community (i.e., algal and bacterial cells) respiration resulting from nutrient enrichment (i.e., eutrophication). Implementation of planned unit process improvements at the VWRF, including denitrification, will reduce nitrate (NO_3) concentrations in the VWRF discharge.

5.1 Contemporary water quality conditions

Water quality within the SCRE is highly variable on an annual basis due to the combination of meteorological variations coupled with the seasonal variability in the SCRE mouth conditions and ocean exchanges. Contemporary water quality data for the SCRE were compiled from a variety of sources at locations within the SCRE (Table 5-1)—including 2009–2010 data collected as part of this study (Stillwater Sciences 2010)—to characterize current conditions both seasonally and in association with SCRE mouth closure status. In addition, the City has deployed two recording *in situ* water quality sondes (Yellow Springs Instruments, Yellow Springs, Ohio) for extended periods within the outfall channel and along the south side of the SCRE, allowing for the evaluation of physical water quality (temperature, DO, pH, conductivity) on a diurnal basis at various times of year.

Table 5-1. Water quality data sources.

Data source	Data type	Dates of record	*In situ* data					Analytical data				
			Temperature	pH	Dissolved Oxygen	Specific Conductivity	Salinity	Turbidity	Nutrients	Metals	Chlorophyll-a	Bacteria
USFWS 1999	multi-probe	1997–1999	X	X	X	X	X					
Entrix, Inc. 2002a	multi-probe, surface grab	2001–2002	X	X	X	X	X	X		X		
Nautilus Environmental 2005	multi-probe	2003–2005	X	X	X	X	X					
Ventura (2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000)	multi-probe, surface grab	2000–2010	X	X	X		X		X	X	X	X
Stillwater Sciences 2010	multi-probe, surface grab	2009–2010	X	X	X	X		X	X		X	
City of Ventura (unpublished)	Continuous multi-probe	2009–2010	X	X	X	X						

Appendix B provides a compilation of all water quality data shown in Table 5-1 in electronic form, with the contents of the accompanying data CD included as part of this report. The VWRF annual reports that detail monthly receiving water quality data collected from 2000–2008 are included in Appendix B. USFWS (1999) provides an overall assessment of *in situ* water quality and biological conditions in the SCRE. The City completed a *Metals Translator Study* (Entrix, Inc. 2002b) to determine site-specific criteria within the SCRE that includes analytical metals data for copper, nickel, lead and zinc. Nautilus Environmental (2005) evaluated whole water copper toxicity for VWRF effluent and receiving water samples collected from 1999–2003 as well as sediment toxicity testing. Stillwater Sciences (2010) provides a summary of methods and water quality data collected in 2009–2010 with individual records for each *in situ* parameter (DO, pH, temperature, conductivity, turbidity) and analytical parameter (nutrients, chlorophyll-a, bacteria, and trace metals) provided as Appendix C.

For the purposes of the analyses in this report, all data were grouped by their general location within the SCRE (Figure 5-1). Sites that were located in the Santa Clara River or in the upstream extent of the SCRE were designated as "Riverine." Samples collected adjacent to the "Estuary Mouth" were given this designation. Samples collected within the outfall channel locations were designated as being in either the "Upper Outfall" channel adjacent to the VWRF, or within the "Lower Outfall" channel near the transition between the channel and the SCRE. The "Upper Estuary" designation is for sites generally located between the eastern end of the McGrath State Beach campground and the open water areas east of Harbor Blvd. The "Middle Estuary" designation was used for sites located between the mouth of the outfall channel and the east end of the McGrath State Beach campground. The "Lower Estuary" designation was used for sites located downstream of the outfall channel, including all sampling locations in this area except those specifically located near the SCRE mouth.

Figure 5-1. Approximate water quality sampling locations in the SCRE and adjacent floodplain.

5.2 *In situ* water quality in grab samples

Numerous factors may influence SCRE water quality conditions including Santa Clara River and VWRF flows, changes in SCRE mouth conditions, and seasonal changes in weather and air temperatures. Long-term seasonal averages of *in situ* water quality in-grab samples, or spot measurements, monitored during daylight hours since 1997 (Table 5-1) are shown in Tables 5-2 through 5-5 for both open- and closed-mouth conditions during winter, spring, summer, and fall time periods. Water temperatures in the SCRE vary from wintertime lows near 13ºC (55ºF) regardless of whether the SCRE mouth is open or closed, to summertime highs of near 23ºC (73ºF) with the mouth open (Table 5-2), and nearer 25ºC (77ºF) with the mouth closed. Specific conductivity and salinity measurements were generally higher at locations nearest the ocean under both open-mouth and closed-mouth conditions, but were highly variable due to ocean exchanges of saltwater and wind mixing[2]. Under open-mouth conditions, salinity ranged from near 15–16 ppt near the mouth to approximately 1 ppt upstream of the SCRE. Although mixing of the lower portions of the SCRE was evident under closed-mouth conditions, historical samples show that salinity near the estuary mouth location ranges from 5–11 ppt depending upon the time since mouth closure, with salinity at the upstream (Riverine) location remaining in the range of 1 ppt. Profiles collected for this study from 2009–2010 (Appendix C) show that only the upper and lower outfall locations exhibited any signs of density stratification in some sampling events due to fresh water flows from the VWRF.

Tables 5-2 through 5-5 show that with the exception of locations within the outfall channel and in riverine sampling locations upstream of the SCRE that were sometimes below saturation, DO levels were well above saturation during daytime sampling periods in all seasons. For open- and closed-mouth conditions, average DO reached levels near 18.4 mg/L and 14.4 mg/L, respectively. As discussed further in Section 5.1.2, this "supersaturation" of DO is consistent with high levels of algal productivity, consistent with photosynthetic uptake of dissolved CO_2 by algae in the SCRE during daylight hours. For example, under closed-mouth conditions, average summertime pH exceeds 9.3. Seasonal pH variations in the SCRE for open-mouth conditions are relatively minor on an annual basis (7.8–8.3), likely due to ocean exchanges (Table 5-4). The VWRF lower outfall location generally exhibits lower pH levels (7.1–7.9) than open water SCRE sites (7.8–9.3). Turbidity measurements were generally highest in the locations in the lower SCRE, with the highest levels occurring during summer and fall and lowest levels occurring during spring.

[2] Ocean water contains approximately 35 ppt salinity, corresponding to specific conductance levels on the order of 50 mS/cm

Table 5-2. Average *in situ* water quality in grab samples of the SCRE by mouth closure status during winter (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Temperature (°C)	Mean	13.4	13.1	13.5	13.1	16.6	12.6	13.3	13.3	14.5	12.6	14.9	13.4	17.2	17
	Range	8–20.2	9.6–16.2	7.9–20	9.6–16.7	14.1–19.4	11.4–13.7	7.5–20	9.1–16.7	12.2–18.3	11.5–13.3	8.1–19.5	9.2–17.8	16.7–17.7	NA
Dissolved Oxygen (mg/L)	Mean	9.8	11.5	9.4	11	10.6	11.3	9.5	10.4	9.3	9.9	9.3	11.2	7.5	11.8
	Range	1.9–21	3.4–21.9	1–20	1.4–22.1	4.8–16	10.6–11.9	1.8–21	0.2–21	5.2–12.1	6–12.5	1.7–21	1.8–22.8	5–10.1	NA
Dissolved Oxygen (%)	Mean	104	117	98	111	113	122	94	104	92	98	97	114	80	124
	Range	18–229	37–211	9–221	16–211	50–176	112–129	17–220	2–216	49–122	58–130	16–212	17–221	52–108	NA
Salinity (ppt)	Mean	15	10	13	11	3	21	4	7	1	6	9	8	1	1
	Range	0–37	1–32	0–39	0–31	1–11	15–24	0–27	0–26	1–2	1–17	0–33	1–31	NA	NA
Specific Cond. (ms/cm)	Mean	24.7	16.4	21	17.6	5.8	32.7	7.2	11.8	2.4	10.2	14.3	13.3	2.5	2.7
	Range	0.8–58.6	1.6–49.9	1.2–60.5	1–48.3	2.2–18.1	23.8–37.6	0.8–42.3	1.3–41.3	1.7–3.3	2.7–27.4	1–51.4	1.5–49.4	2.2–2.7	NA
pH (s.u.)	Mean	7.8	8.8	7.9	8.2	7.9	7.9	7.8	8.2	8.2	7.8	7.6	8.1	7.6	7.3
	Range	7.4–8.3	8–9.3	6.9–8.8	7.5–9.3	7.1–8.5	7.9–8	7–8.9	7.3–9.1	7.8–8.5	7.5–8.7	6.9–8.3	7.4–9.4	7.4–7.9	NA
Turbidity (ntu)	Mean	95.2	NA	34	NA	47.9	NA	66.9	NA	33.1	5	42.3	NA	14.9	NA
	Range	NA	NA	8–79.2	NA	9–86.9	NA	9–130	NA	4–90.2	5–5	7–77.6	NA	7.8–22	NA

Sources: (USFWS 1999; Entrix, Inc. 2002a; Nautilus Environmental 2005; Ventura [2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000]; Stillwater Sciences 2010)

Table 5-3. Average In situ water quality in grab samples of the SCRE by mouth closure status during spring (1997–2010).

	Location	**Near mouth**		**Lower estuary**		**Middle estuary**		**Upper estuary**		**Riverine**		**Lower outfall**		**Upper outfall**	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Temperature (°C)	Mean	16.5	18.7	17.1	19	21.8	19.5	17	18.6	20	16.8	18.8	19.4	23.1	21.6
	Range	10.6–25	13–23.6	9.7–26.5	14–29.1	18.6–24.4	18–21.1	10.2–25.8	12.7–24.6	16.4–24.2	16.2–17.8	11.8–25.5	13.9–29.1	NA	21.3–21.9
Dissolved Oxygen (mg/L)	Mean	9	11.7	7.9	11.7	13.5	17.6	8.7	9.9	12.6	6.8	9	9.8	8.8	9.2
	Range	1.6–19.8	0.1–27.1	0.1–19.8	0.9–23	8.8–21	16.7–18.5	0.4–16.8	0.2–20	9–21	1.2–11.7	0.8–14.6	0.1–20	NA	8.2–10.1
Dissolved Oxygen (%)	Mean	97	131	86	131	157	190	91	110	142	71	98	109	104	104
	Range	15–211	1–337	1–205	12–255	100–255	177–203	4–184	2–232	96–254	13–124	10–166	1–217	104–104	93–116
Salinity (ppt)	Mean	8	6	7	6	2	2	2	3	1	1	2	3	1	1
	Range	0–35	0–29	0–37	0–29	1–5	2–2	0–17	1–15	1–2	NA	0–14	0–28	NA	NA
Specific Cond. (ms/cm)	Mean	12.9	9.5	12.3	9.8	3.8	4	3.4	5.6	2.5	2.8	4.3	5.8	2.2	2.2
	Range	0.8–54.3	0.8–46.2	1.2–57.9	0.8–45.2	1.8–8	3.9–4.1	0.8–26.6	1.5–24.1	1.8–3	2.5–2.9	1.3–22.2	1.3–43.4	NA	1.9–2.4
pH (s.u.)	Mean	8.1	9.3	7.8	9.1	8.1	9.0	7.9	8.6	7.9	7.9	7.8	8.6	7.6	7.9
	Range	7.8–8.8	9–9.8	7.1–8.6	7.4–9.9	7.1–9	8.4–9.5	7.4–8.2	7.7–9.4	7.6–8.1	7.5–8.6	7.2–8.4	7.6–9.4	NA	7.6–8.2
Turbidity (ntu)	Mean	NA	19.9	10	19.1	10	16.8	10	18.4	10	4.2	NA	4.1	10	1
	Range	NA	NA	NA	NA	NA	16.8–16.8	NA	17.8–18.9	NA	2.5–7	NA	NA	NA	NA

Sources: (USFWS 1999; Entrix, Inc. 2002a; Nautilus Environmental 2005; Ventura [2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000]; Stillwater Sciences 2010)

Table 5-4. Average *In situ* water quality in grab samples of the SCRE by mouth closure status during summer (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Temperature (°C)	Mean	19.8	21.8	20.3	22.7	22.9	24.6	22.1	21.8	21.2	23.1	22.7	22.5	NA	23.9
	Range	14.1–26	16.2–26.6	15.6–25	17.5–27.1	20.9–24.9	21.1–26.9	15.3–33.3	16.9–29.7	21.2–21.2	18.3–29	15.9–26.6	17.9–28.3	NA	22.4–25.2
Dissolved Oxygen (mg/L)	Mean	8.8	9.6	7.6	10.3	14.9	12.5	9.9	7.8	9.7	13.7	8.4	8.6	NA	6.6
	Range	1.5–18.9	0.3–23.3	0–21	0.2–21	8.9–21	3–21	3.2–20	0.4–21	9.7–9.7	8.2–21	3.3–15.6	1.9–21	NA	5.7–7.1
Dissolved Oxygen (%)	Mean	104	113	89	123	180	150	114	92	110	164	97	101	NA	78
	Range	19–233	4–301	0–258	3–271	101–258	38–255	39–236	5–273	110–110	90–277	40–178	22–258	NA	69–84
Salinity (ppt)	Mean	16	5	12	4	2	2	2	2	1	2	3	2	NA	1
	Range	1–33	1–34	0–32	1–32	1–2	1–4	0–7	0–11	NA	1–3	1–12	0–11	NA	NA
Specific Cond. (ms/cm)	Mean	26.2	8.8	19.9	6.8	3.3	3.7	4.4	4.4	2.1	3.1	4.8	4.3	NA	2.2
	Range	2.4–52.4	1.9–52.8	0.8–50	2.6–50.5	2.6–4	2.2–6.7	0.8–12	1.3–17	2.1–2.1	2.2–4.9	1.8–19.9	0.8–17.1	NA	NA
pH (s.u.)	Mean	7.8	8.8	7.9	8.6	8.3	8.1	7.9	8.4	8.1	8.2	7.8	8.2	NA	7.1
	Range	7.8–7.8	7.8–9.8	7.3–8.6	7–9.8	8.1–8.4	7.1–9.1	7.8–8.1	7.6–9.5	8.1–8.1	7.7–9	7.6–7.8	7.1–9.4	NA	7.1–7.2
Turbidity (ntu)	Mean	NA	NA	NA	15.8	NA	6.0	NA	11.3	NA	2.3	NA	7.0	NA	5.0
	Range	NA	NA	NA	3–52	NA	2–10	NA	1–23	NA	0–5	NA	NA	NA	0–10

Sources: (USFWS 1999; Entrix, Inc. 2002a; Nautilus Environmental 2005; Ventura [2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000]; Stillwater Sciences 2010)

Table 5-5. Average In situ water quality in grab samples of the SCRE by mouth closure status during fall (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Temperature (°C)	Mean	16.8	19	17.6	19.3	21.6	20.8	16.9	18.7	19.9	18.5	19.2	19.5	NA	21
	Range	12.7–24.2	9.7–25.4	11.6–24	11–25.3	20.1–26.1	16.3–24.7	9.5–24.6	10.5–26.2	19–20.9	16–21.8	13–24.7	11.2–25.8	NA	18.4–24.5
Dissolved Oxygen (mg/L)	Mean	8.8	10.1	8.7	10.9	12	13.7	9.1	8	9.7	12	8.2	8.4	NA	6.3
	Range	3.5–19	0.5–19.1	4.5–19.5	2–21	9–15	7–21.1	0.9–24.5	0.2–21.2	9.7–9.7	6.4–21	0.2–18.6	0.3–19.8	NA	4.9–9.3
Dissolved Oxygen (%)	Mean	99	111	99	122	145	159	92	88	110	133	94	93	NA	71
	Range	42–188	6–219	45–242	22–246	102–188	73–257	10–213	2–259	110–110	67–246	2–182	3–203	NA	54–108
Salinity (ppt)	Mean	16	4	14	4	4	3	5	3	1	2	8	3	NA	1
	Range	1–32	1–26	1–33	1–25	1–12	1–6	0–27	1–10	1–2	1–4	1–32	1–22	NA	NA
Specific Cond. (ms/cm)	Mean	25.4	6.2	22.4	6.7	7.1	5.9	8.4	4.8	2.9	3.8	13.4	5.8	NA	2.4
	Range	2.6–50.5	1.8–41.3	2.1–51.4	2.1–38.7	2.4–18.8	2.5–9.7	1.2–42.3	1.9–15.9	2.6–3.2	2.9–6.6	1.9–50.6	2.1–34.7	NA	2.2–3
pH (s.u.)	Mean	7.8	9.2	7.8	9.0	7.9	8.8	7.8	8.6	8.0	8.4	7.6	8.1	NA	8.0
	Range	7.4–8.2	8.1–9.6	7.3–8.3	7.1–9.6	7.6–8.7	8.4–9.6	7.3–8.4	7.2–9.7	7.7–8.2	7.5–9.2	7.4–7.8	7.5–9	NA	7.0–8.0
Turbidity (ntu)	Mean	NA	16.7	NA	17.00	NA	21.3	NA	18.6	NA	2.4	NA	4.5	NA	6.0
	Range	NA	15.1–18.3	NA	16.5–17.5	NA	20.5–22.1	NA	14.8–22.3	NA	0–4	NA	3.8–5.3	NA	0–12.0

Sources: (USFWS 1999; Entrix, Inc. 2002a; Nautilus Environmental 2005; Ventura [2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000]; Stillwater Sciences 2010)

5.2.1 Continuous *in situ* water quality records

The City employs two continuous recording multi-parameter water quality instruments ("sondes"), one on the north side of the estuary adjacent to the outfall channel (SCR1) and another on the south side near the McGrath State Beach campgrounds (SCR2). The sondes record *in situ* water quality parameters (DO, pH, temperature, conductivity, turbidity) and estimates of chlorophyll-a at approximately 15-minute intervals to provide a near-continuous data record from approximately February 2009 to July 2010 (Appendix B).

Figures 5-2 and 5-3 show recorded depth (m) and *in situ* water quality data from the sondes including water temperature (°C), DO saturation (%), and specific conductivity (mS/cm) between May 2009 and July 2010 with periods corresponding to open- and closed-mouth conditions shown by blue or white shading.[3] Water temperatures follow expected seasonal increases from winter to summer. Developing ranges for in *in situ* water quality data required extensive data review and validation. Closer examination of Figures 5-2 and 5-3, however, reveals periods of missing or unrepresentative data associated with dewatering of the sondes, removal for maintenance or high flows, wave overwash events, and the loss of one of the sondes due to vandalism in Fall 2009 (data gap from 10/8/2009 to 2/18/2010 for SCR1 in Figure 5-2). For example, under open-mouth conditions, large daily temperature variations and higher daily maxima suggests both sondes were regularly dewatered and exposed to air during the lower portion of the tidal cycle (i.e., end of ebb and beginning of flood tides). In other circumstances, individual sensors reporting constant zero or negative values indicated they were not functioning. In order to properly represent water quality conditions when the sondes were submerged and operational, we have trimmed the data shown in Figures 5-2 and 5-3 to periods when the units were submerged. In addition, to ensure outlier data is excluded, we have characterized the variations in *in situ* water quality using order statistics (median, lower and upper quartiles) by season and mouth closure status (Table 5-6).

[3] For additional sensor data (pH, chlorophyll-a, and turbidity), see accompanying data CD (Appendix B)

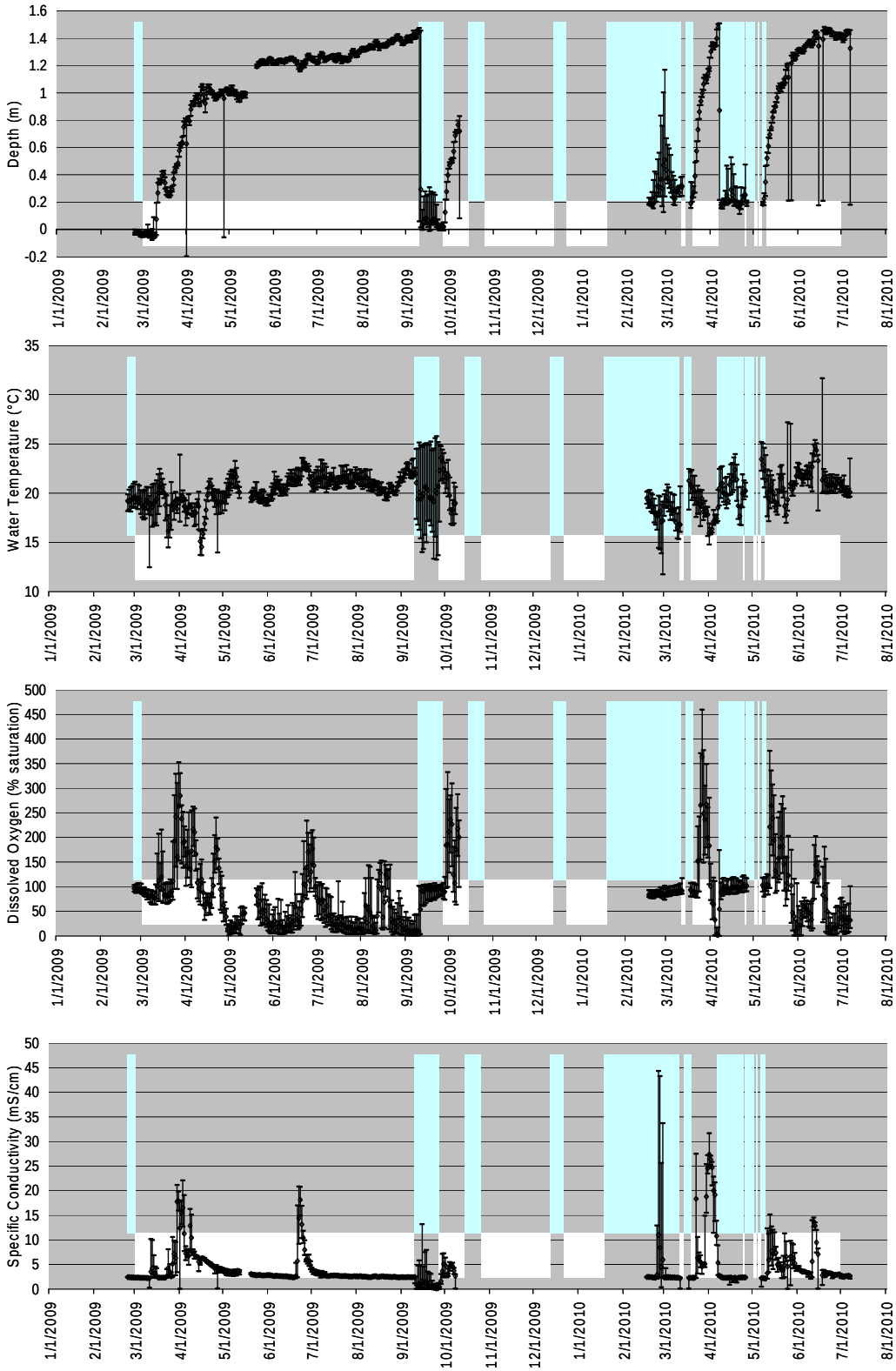


Figure 5-2. 2009–2010 *in situ* water quality for selected parameters (SCR1 North) in the SCRE during closed-mouth (white shading) and open-mouth (blue shading) conditions.

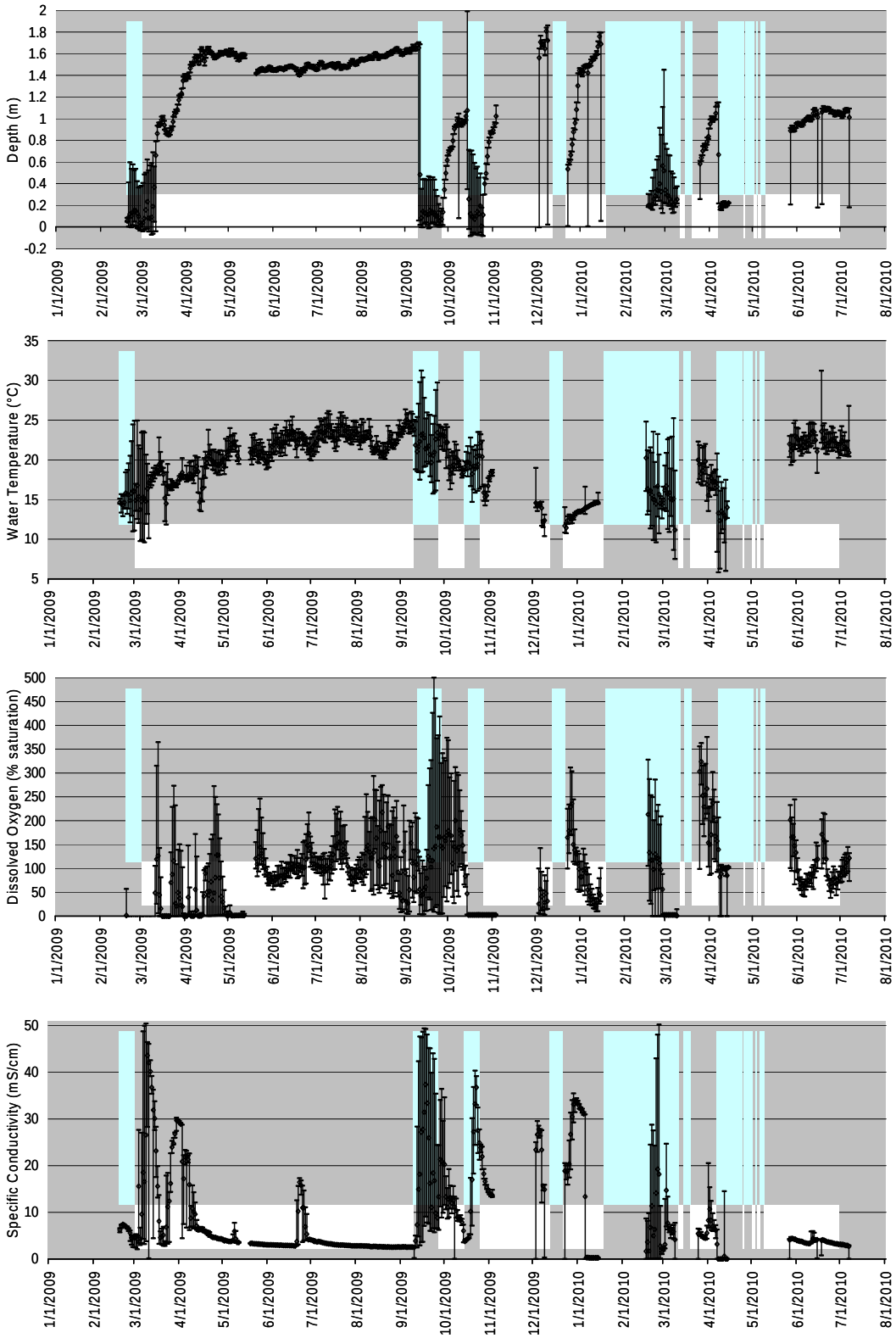


Figure 5-3. 2009-2010 *in situ* water quality for selected parameters (SCR2 South) in the SCRE during closed-mouth (white shading) and open-mouth (blue shading) conditions.

Table 5-6. Summary of *in situ* water quality in the SCRE grouped by season and mouth closure status (2009–2010).

Parameter by season		Estuary mouth open						Estuary mouth closed					
		North sonde (SCR1)			South sonde (SCR2)			North sonde (SCR1)			South sonde (SCR2)		
		Median	1st Quartile	3rd Quartile	Median	1st Quartile	3rd Quartile	Median	1st Quartile	3rd Quartile	Median	1st Quartile	3rd Quartile
Temperature (°C)	Winter	18.5	11.7	21.0	15.3	9.9	24.5	NA	NA	NA	13.5	10.4	14.8
	Spring	19.1	12.5	22.7	15.3	7.5	25.3	19.0	13.7	23.3	18.8	11.9	24.0
	Summer	NA	NA	NA	NA	NA	NA	21.1	18.8	23.6	22.5	19.0	26.1
	Fall	23.6	19.3	25.6	20.9	14.7	31.2	21.5	16.8	24.8	20.9	14.3	26.3
pH (s.u.)	Winter	7.4	7.3	8.1	7.2	6.6	8.4	NA	NA	NA	8.4	6.6	9.2
	Spring	7.5	7.3	14.0	6.8	5.5	9.6	8.5	6.8	9.8	7.7	5.8	10.0
	Summer	NA	NA	NA	NA	NA	NA	7.7	7.0	9.3	8.8	7.7	9.7
	Fall	7.8	7.1	8.2	7.5	6.6	9.3	8.1	6.2	9.9	8.6	6.6	10.3
Dissolved Oxygen (mg/L)	Winter	7.9	6.4	9.8	9.8	1.3	28.0	NA	NA	NA	7.2	0.3	27.9
	Spring	8.0	5.0	11.3	7.6	0.1	17.5	9.4	0.0	41.1	10.3	0.0	32.4
	Summer	NA	NA	NA	NA	NA	NA	4.1	0.1	21.0	9.9	0.7	28.7
	Fall	7.6	0.8	10.0	9.6	0.3	37.1	8.4	0.1	31.6	11.0	0.4	32.3
Dissolved Oxygen (%)	Winter	87.3	73.4	106.8	111.9	12.3	287.2	NA	NA	NA	81.8	2.7	311.5
	Spring	88.3	57.6	117.6	83.6	0.9	187.1	105.6	0.1	459.7	118.8	0.0	375.4
	Summer	NA	NA	NA	NA	NA	NA	45.8	1.5	233.9	113.7	8.2	293.4
	Fall	84.0	9.6	107.4	110.8	3.2	500.0	94.6	1.0	332.4	127.1	4.5	374.1
Specific conductivity (us/cm)	Winter	3,670	323	44,324	8,387	101	50,171	NA	NA	NA	27,120	293	35,419
	Spring	2,339	170	4,213	11,564	110	50,405	6,729	100	31,664	10,232	2,774	42,513
	Summer	NA	NA	NA	NA	NA	NA	3,377	2,147	20,818	3,597	2,380	17,270
	Fall	1,985	101	13,159	18,503	2,431	49,374	3,105	184	7,251	10,996	197	36,388
Turbidity (NTU)	Winter	35.5	1.2	1199.0	48.5	0.2	1096.1	NA	NA	NA	34.7	0.3	440.6
	Spring	15.5	0.4	1198.9	260.7	0.2	1110.8	12.8	0.5	797.3	173.7	0.2	1138.8
	Summer	NA	NA	NA	NA	NA	NA	15.3	0.1	453.4	20.4	4.0	324.4
	Fall	55.0	6.3	412.4	72.8	0.2	1130.8	11.7	1.4	291.5	105.3	0.2	1108.0
Chl-a (ug/L)	Winter	10.0	2.0	33.4	17.6	0.5	500.0	NA	NA	NA	34.1	0.0	500.0
	Spring	7.7	0.4	48.2	116.6	5.5	352.3	48.2	0.6	432.8	85.6	0.0	500.0
	Summer	NA	NA	NA	NA	NA	NA	24.0	0.0	194.7	43.8	2.1	223.8
	Fall	61.4	8.3	173.6	30.9	0.1	500.0	63.2	0.1	232.3	76.8	0.0	216.0

As with the *in situ* grab sample summary (Tables 5-2 though 5-5), water temperatures and DO were generally higher under closed-mouth conditions relative to open-mouth conditions (Table 5-6). Specific conductivity at these locations was generally higher during open- mouth conditions that would allow greater exchanges of ocean water. Table 5-6 shows that average DO levels are generally higher for the North sonde (SCR1) adjacent to the outfall channel during cooler weather in late fall and early spring than during warm weather, whereas the South sonde (SCR2) exhibited elevated DO during summer and early fall. Although there were episodes of high turbidity during winter and spring (potentially associated with runoff), turbidity and chlorophyll-a levels were elevated into the summer and fall and higher under closed-mouth conditions. As discussed above,

these patterns are consistent with algal productivity in the SCRE, further supported by the increasing pH levels and their variation during spring and summer that would be expected to result from photosynthetic uptake of dissolved CO_2 by algae during periods of higher solar insolation.

In order to confirm that the VWRF discharge was not elevating water temperatures in the SCRE, a re-analysis of the continuous temperature data was performed for the two sondes. As represented by SCR1, water temperatures exiting the wildlife ponds at the VWRF are generally lower than SCRE water temperatures (SCR2) during summer months, but can be warmer during winter months (Figures 5-4 and 5-5). Larger differences between the two sonde locations were observed during open mouth condtions that correspond to reduced mixing of the outfall channel with other portions of the SCRE. Although confirmation of these observations would require deployment and monitoring of a network of thermographs distributed across the SCRE, it is likely that the temperature differences exhibited between the outfall channel and the open water location of the sonde SCR2 would not be observable between open water sites within a short distance of the outfall channel terminous. In addition, during open mouth conditions, the areas of the SCRE nearest the mouth likely experience a cooling effect due to colder ocean water entering the SCRE.

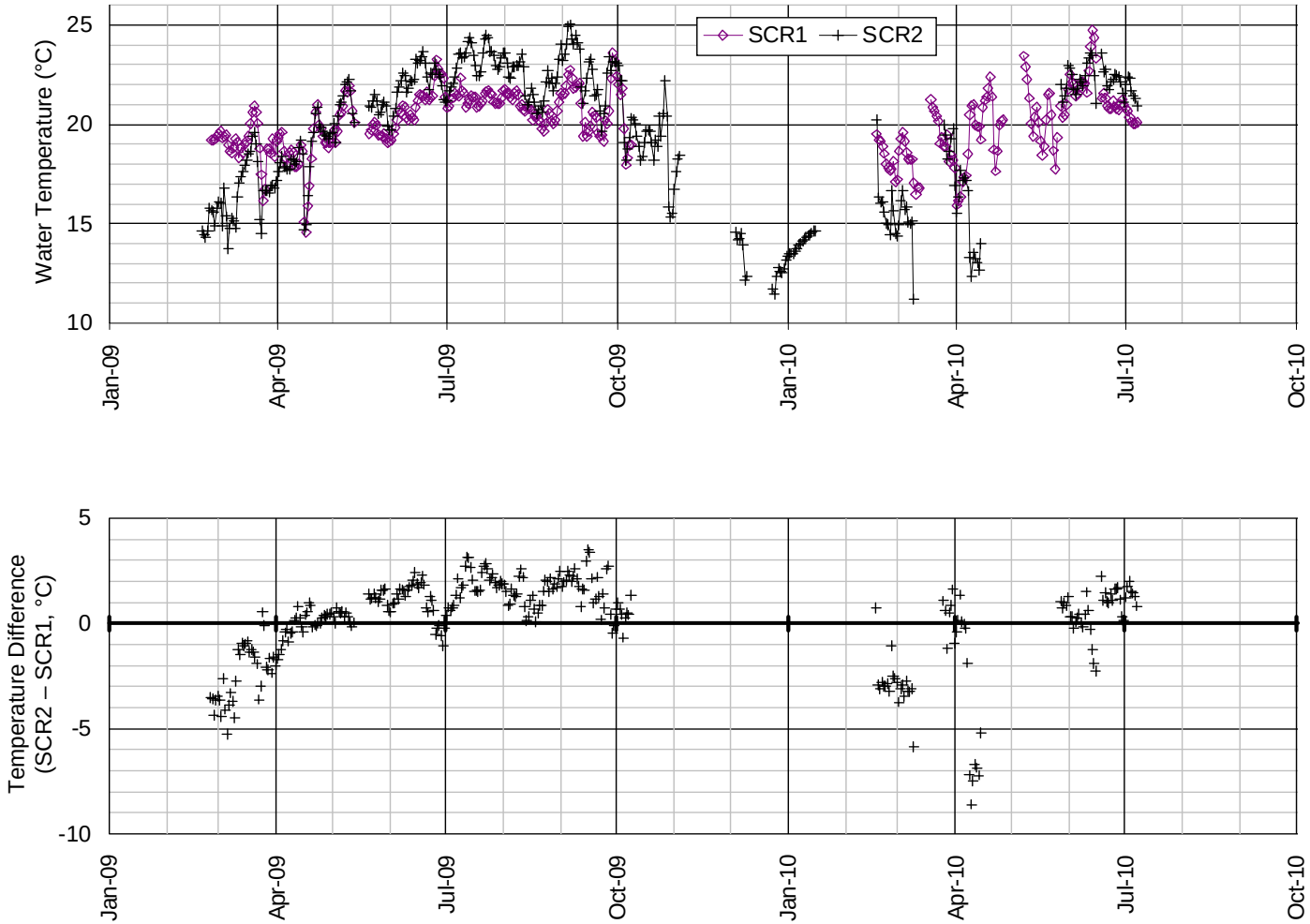


Figure 5-4. Continuous water temperature records for the lower outfall channel (sonde SCR1) and lower estuary (sonde SCR2) sites**.**

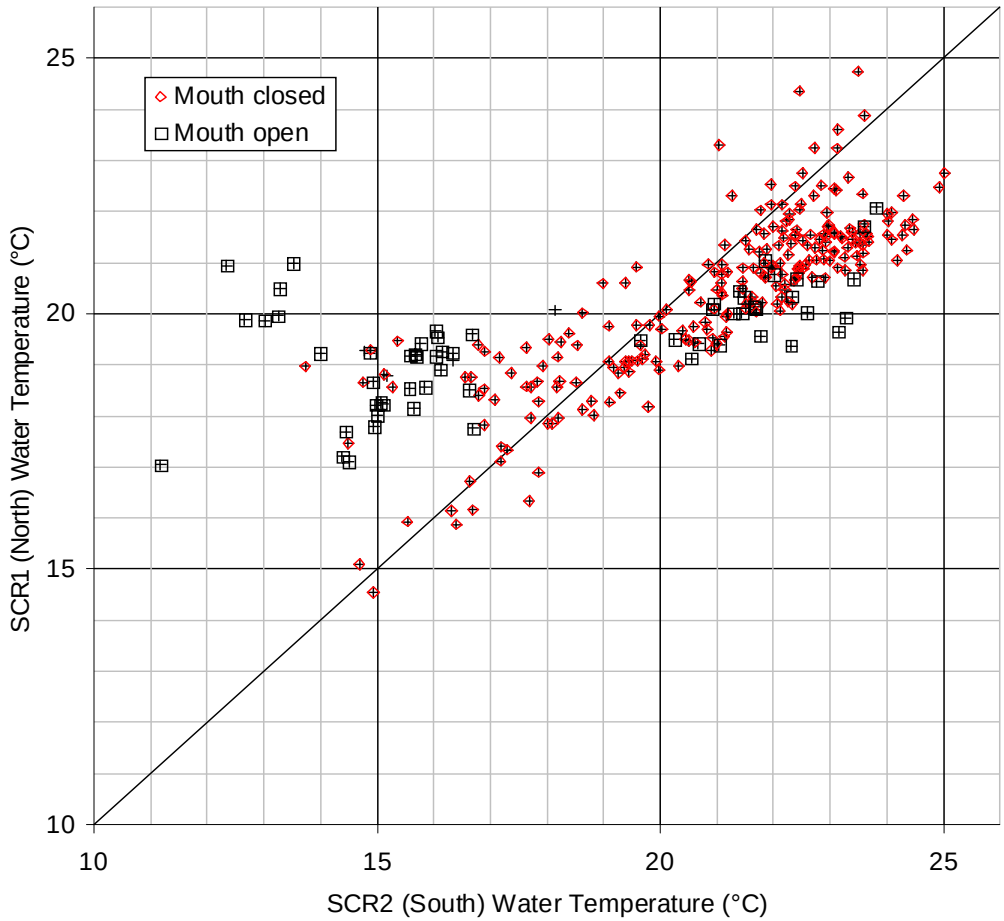


Figure 5-5. Water temperatures measured at the lower outfall channel (sonde SCR1) and lower estuary (sonde SCR2) sites under closed and open-mouth conditions.

Lastly, a re-analysis of the sonde data was performed to examine the effects of algal productivity more closely. Diurnal variations in DO for the North and South sonde locations are provided in Figure 5-6 for three representative time periods when both units were operational. Under closed-mouth conditions, DO at the North sonde (SCR1) is well below saturation and lower than at the open water site at the South sonde (SCR2) location. This is possibly due to a combination of residual biochemical oxygen demand (BOD) entering from the VWRF outfall channel, with the reduced wind mixing provided in the reed stands along the north side of the SCRE. During several periods, elevated DO is shown for the South sonde (SCR2), reaching peaks several hours after solar maximum, as would be expected from excess algal productivity. At both locations, DO is below suitable levels for aquatic organisms (< 7mg/L) for some portions of the day in both cooler and warmer times of year (Figures 5-2 and 5-3; Table 5-6), although this effect is more pronounced at the North sonde (SCR1)[4].

[4] Daily DO minima of 1 mg/L or less were observed for both locations during some portions of the data record (Appendix B), although developing exceedance statistics was not possible due to the inability to separate periods of sensor malfunctions from periods of near zero DO.

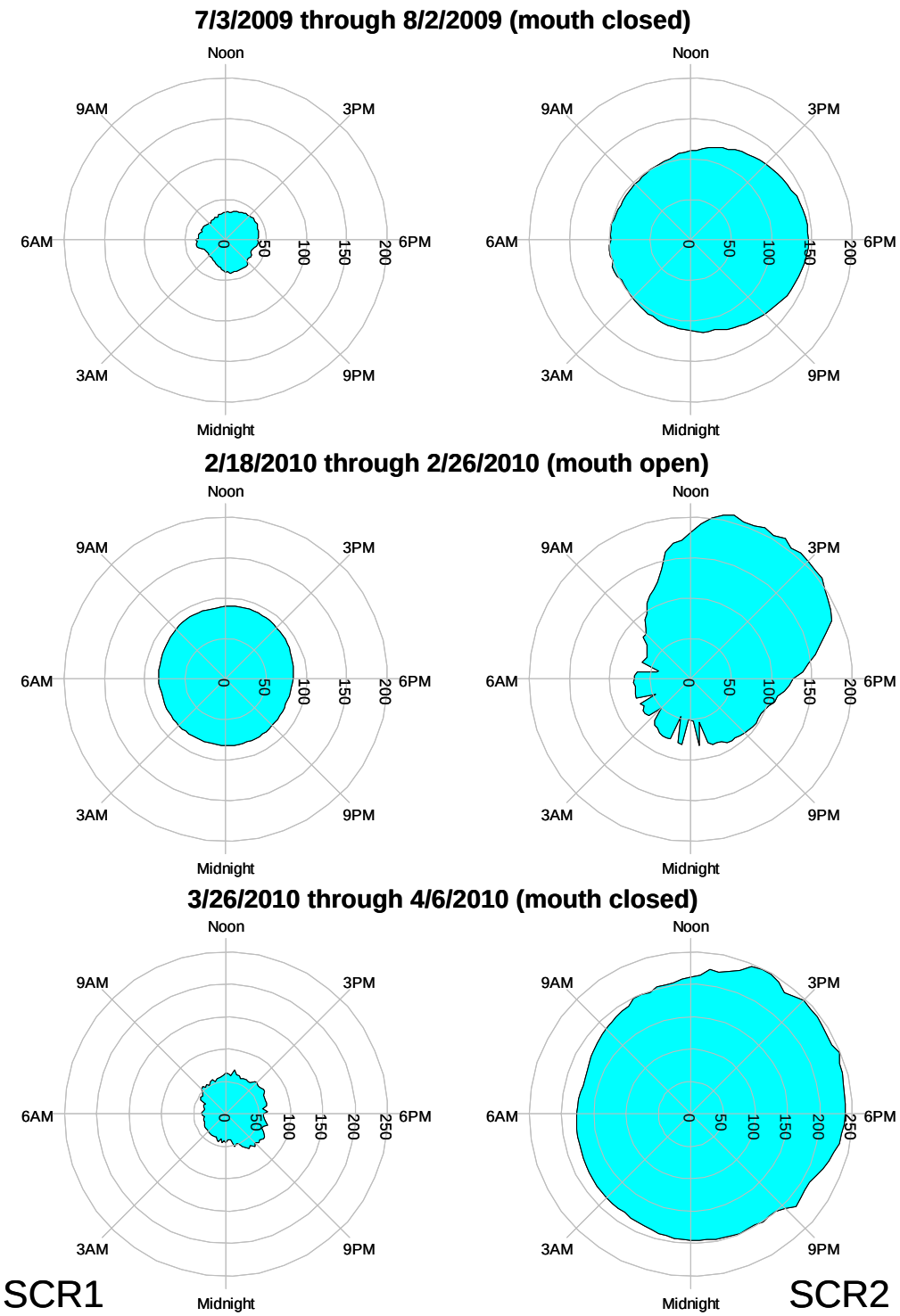


Figure 5-6. Diel variations in DO saturation for three representative periods at the North sonde (SCR1) and South sonde (SCR2) locations in the SCRE.

5.2.2 Analytical water quality

Analytical water quality sampling has been conducted by the City and its consultants since 2000 (Table 5-1). Summary statistics of grab sample analysis results from various locations and under open- and closed-mouth conditions are shown in Tables 5-7 through 5-10 for winter, spring, summer, and fall. We provide a brief discussion of analytical water quality conditions under different seasonal conditions are below.

Nutrient levels and related chlorophyll-a concentrations reflect seasonal changes in water temperature and sunlight as well as runoff and ocean exchanges associated with SCRE mouth conditions Although no consistent seasonal or spatial patterns in ammonia concentrations are apparent, low levels of ammonia were detected throughout the SCRE at all times of year and under both open- and closed-mouth conditions (Tables 5-7 through 5-10). In contrast, total Kjeldahl nitrogen (TKN) was generally higher for closed-mouth than for open-mouth conditions from spring through fall, consistent with algal contributions to observed nitrogen levels. Nutrient levels varied by season and location, with nutrients at seasonal highs during spring and fall under open-mouth conditions, slightly lower in the summer and fall, and lowest during winter (Tables 5-7 through 5-11). Excluding the Upper Outfall site, levels of nitrate + nitrite were highest in the winter (8.6–9.3 mg-N/L), lower during spring (5.1–8.5 mg-N/L), and fall (5.3–7.2 mg-N/L), and lowest in summer (2.1–6.6 mg-N/L) for closed-mouth conditions. The Upper Estuary and Riverine locations were consistently in excess of other locations during winter and spring under open-mouth conditions. However, nitrate levels measured in the both outfall channel sites were higher than most other locations in the SCRE under all mouth conditions and seasons (Tables 5-7 through 5-10). Figure 5-7 shows these data for open- and closed-mouth during winter and spring for illustration. Total phosphorus levels were also consistently higher in the outfall channel under all conditions (Tables 5-7 through 5-10). As discussed further in Sections 5.4 and 5.5, these observations of higher phosphorus under closed-mouth relative to open-mouth conditions in the SCRE suggests the VWRF is contributing measurable amounts of phosphorus to the SCRE, which in the presence of even small amounts of nitrogen, may contribute to algal growth. Although observed chlorophyll-a levels were variable, they were consistently higher during spring and summer. Chlorophyll-a and pH levels were also higher under closed-mouth conditions than open-mouth conditions (Figure 5-8).

For bacterial samples collected in the SCRE, total and fecal bacteria counts are generally higher when the mouth of the SCRE is open than under closed-mouth conditions (Tables 5-7 through 5-10). This occurs more often during winter and spring, perhaps due to increases in storm runoff entering the SCRE from upstream locations, or due to increased populations of over-wintering birds.

For some trace metal samples collected in the SCRE (e.g., aluminum [Al], copper [Cu], lead [Pb], and zinc [Zn]), analytical concentrations were generally higher during winter and spring (Tables 5-7 and 5-8) as well as during open-mouth conditions relative to closed-mouth conditions. Because of the strong association of metals with mineral-oxide surfaces (Brown et al. 1999), these observations are consistent with increases in runoff-derived total suspended solids (TSS) under these conditions. Although trace metals concentrations at outfall sampling locations were not elevated with respect to other locations within the SCRE (Table 5-8), historically elevated copper levels in the VWRF effluent prompted improvements in source control and treatment processes (Dan Pfeifer, pers. comm., 2010). In addition, the City implemented a *Resident Species Study* (Entrix, Inc. 2002a) and *Metals Translator Study* (Entrix, Inc. 2002b) to better characterize the effects of copper discharges on fresh and saltwater organisms.

Table 5-7. Analytical water quality parameters in grab samples from the SCRE by mouth closure status during winter (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Ammonia (mg-N/L)	Mean	0.4	0.3	0.4	0.5	0.1	NA	0.6	1.0	0.1	NA	0.8	0.4	0.1	NA
	Range	0.1–0.7	0.1–0.6	0.1–0.7	0.1–1.7	0.1	NA	0.1–2.6	0.1–9.0	0.1	NA	0.1–2.4	0.1–1.8	0.1	NA
TKN (mg-N/L)	Mean	1.5	1.4	1.6	1.7	1.1	NA	2.5	1.6	1.0	NA	2.8	1.6	0.6	NA
	Range	0.9–6	0.4–2.3	0.2–6.2	0.2–2.9	1.1	NA	0.1–11.7	0.3–2.9	1.0	NA	0.2–6.3	0.2–3.1	0.6	NA
Nitrates+Nitrites (mg-N/L)	Mean	3.6	8.6	3.0	7.5	4.5	NA	3.9	4.7	1.4	NA	8.8	9.3	7.4	NA
	Range	0.4–8.0	1.8–15.3	0.4–9.8	0.5–15.2	4.5–4.5	NA	0.8–2.6	0.3–15.7	1.4–1.4	NA	1.2–19.0	1.8–14.2	7.4	NA
Phosphorus, Total (mg/L)	Mean	0.6	1.5	0.5	1.2	0.5	NA	0.6	1.2	0.2	NA	1.3	1.6	0.4	NA
	Range	0.7–1.8	0.1–4.9	0.6–1.2	0.2–3.2	0.5	NA	0.7–2.5	0.1–5.4	0.2	NA	0.1–3.3	0.3–4.5	0.4	NA
Chlorophyll-a (µg/L)	Mean	24	51	85	80	36	NA	42	40	2	NA	14	37	81	NA
	Range	2–250	2–140	2–900	2–340	36	NA	2–310	2–130	2	NA	2–31	2–98	81	NA
Aluminum (µg/L)	Mean	1953	NA	NA	NA	NA	NA	1495	NA	NA	NA	NA	NA	NA	NA
	Range	66–6520	NA	NA	NA	NA	NA	46–5360	NA	NA	NA	NA	NA	NA	NA
Tin (µg/L)	Mean	100	NA	NA	NA	NA	NA	100	NA	NA	NA	NA	NA	NA	NA
	Range	100	NA	NA	NA	NA	NA	100	NA	NA	NA	NA	NA	NA	NA
Antimony (µg/L)	Mean	8.3	NA	NA	NA	NA	NA	1.0	NA	NA	NA	NA	NA	NA	NA
	Range	1.0–51.3	NA	NA	NA	NA	NA	1.0	NA	NA	NA	NA	NA	NA	NA
Arsenic (µg/L)	Mean	5.3	NA	NA	NA	NA	NA	2.0	NA	NA	NA	NA	NA	NA	NA
	Range	0.2–16.0	NA	NA	NA	NA	NA	2.0	NA	NA	NA	NA	NA	NA	NA
Barium (µg/L)	Mean	68	NA	NA	NA	NA	NA	68.9	NA	NA	NA	NA	NA	NA	NA
	Range	28.8–110.0	NA	NA	NA	NA	NA	37.4–117.0	NA	NA	NA	NA	NA	NA	NA
Beryllium (µg/L)	Mean	0.2	NA	NA	NA	NA	NA	0.2	NA	NA	NA	NA	NA	NA	NA
	Range	0.2	NA	NA	NA	NA	NA	0.1–0.2	NA	NA	NA	NA	NA	NA	NA
Cadmium (µg/L)	Mean	4.0	NA	NA	NA	NA	NA	4.0	NA	NA	NA	NA	NA	NA	NA
	Range	4.0	NA	NA	NA	NA	NA	4.0	NA	NA	NA	NA	NA	NA	NA
Chromium (µg/L)	Mean	8.2	NA	NA	NA	NA	NA	7.1	NA	NA	NA	NA	NA	NA	NA
	Range	7.0–14.8	NA	NA	NA	NA	NA	7.0–9.5	NA	NA	NA	NA	NA	NA	NA
Cobalt (µg/L)	Mean	7.2	NA	NA	NA	NA	NA	4.8	NA	NA	NA	NA	NA	NA	NA
	Range	1.0–40.8	NA	NA	NA	NA	NA	1.0–44.6	NA	NA	NA	NA	NA	NA	NA

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Copper (µg/L)	Mean	8.0	2.0	4.7	7.8	NA	NA	6.3	5.2	3.0	3.2	NA	2.0	5.9	7.4
	Range	2.0–15.0	2.0	2.7–6.7	7.8	NA	NA	0.5–16.5	2.0–7.1	1.9–4.0	3.2	NA	2.0	5.0–6.7	7.4
Lead (µg/L)	Mean	5.1	NA	0.5	0.5	NA	NA	20.1	0.5	0.5	0.5	NA	NA	0.5	0.5
	Range	5.0–5.5	NA	0.5	0.5	NA	NA	0.5–100.0	0.5	0.5	0.5	NA	NA	0.5	0.5
Mercury (µg/L)	Mean	0.	0.2	NA	NA	NA	NA	0.2	0.2	NA	NA	NA	0.2	NA	NA
	Range	0.2	0.2	NA	NA	NA	NA	0.2	0.2	NA	NA	NA	0.2	NA	NA
Molybdenum (µg/L)	Mean	10.0	NA	NA	NA	NA	NA	11.7	NA	NA	NA	NA	NA	NA	NA
	Range	6.7–14.1	NA	NA	NA	NA	NA	2.1–21	NA	NA	NA	NA	NA	NA	NA
Nickel (µg/L)	Mean	15.4	NA	3.2	4.7	NA	NA	12.7	5.4	5.8	8.7	NA	NA	3.5	4.9
	Range	15.4	NA	3.2	4.7	NA	NA	12.7	5.4	5.8	8.7	NA	NA	3.5	4.9
Selenium (µg/L)	Mean	12.9	NA	NA	NA	NA	NA	10.4	NA	NA	NA	NA	NA	NA	NA
	Range	2.0–69.7	NA	NA	NA	NA	NA	2.0–81.0	NA	NA	NA	NA	NA	NA	NA
Silver (µg/L)	Mean	1.9	0.2	NA	NA	NA	NA	3.8	0.2	NA	NA	NA	0.2	NA	NA
	Range	0.2–12.4	0.2	NA	NA	NA	NA	0.2–12.6	0.2	NA	NA	NA	0.2	NA	NA
Thallium (µg/L)	Mean	1.0	NA	NA	NA	NA	NA	1.0	NA	NA	NA	NA	NA	NA	NA
	Range	1.0	NA	NA	NA	NA	NA	1.0	NA	NA	NA	NA	NA	NA	NA
Vanadium (µg/L)	Mean	14.7	NA	NA	NA	NA	NA	8.9	NA	NA	NA	NA	NA	NA	NA
	Range	4.0–48.1	NA	NA	NA	NA	NA	4.0–19.2	NA	NA	NA	NA	NA	NA	NA
Zinc (µg/L)	Mean	19.9	9.3	17.85	36.5	NA	NA	12.96	13.27	3.5	26.3	NA	15.4	27.8	59.7
	Range	2.0–39.6	9.3	13.7–22.0	36.5	NA	NA	0.5–35.9	12.7–14.3	0.5–5.6	26.3	NA	15.4	24.2–31.5	59.7
Fecal coliform (MPN)	Mean	1,143	666	1,154	666	NA	NA	1,066	293	NA	NA	898	667	NA	NA
	Range	2–33,000	2–5,400	2–16,000	2–9,000	NA	NA	2–16,000	7–3,500	NA	NA	17–16,000	8–5,000	NA	NA
Total coliform (MPN)	Mean	12,210	4,135	9,900	3,009	NA	NA	9,479	4,064	NA	NA	5,992	4,002	NA	NA
	Range	4–54,0000	2–24,000	26–24,000	2–24,000	NA	NA	49–16,000	23–24,000	NA	NA	79–13,000	40–16,000	NA	NA

Sources: Entrix, Inc. 2002a; Ventura (2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000)

Table 5-8. Analytical water quality parameters in grab samples from the SCRE by mouth closure status during spring (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Ammonia (mg-N/L)	Mean	0.5	0.3	0.5	0.4	NA	0.1	0.4	0.3	NA	0.1	0.7	0.3	NA	0.3
	Range	0.1–0.8	0.1–0.5	0.1–0.8	0.1–0.9	NA	0.1	0.1–1.3	0.1–1.4	NA	0.1	0.1–2.1	0.1–0.6	NA	0.1–0.4
TKN (mg-N/L)	Mean	1.6	3.6	1.4	3.9	NA	1.5	1.8	4.1	NA	0.4	2.5	2.9	NA	0.7
	Range	0.3–5.5	0.9–11.9	0.4–4.3	1.3–13.2	NA	0.9–2.0	0.2–6.4	0.7–16.9	NA	0.4	0.6–4.7	0.8–10.7	NA	NA
Nitrates+Nitrites (mg-N/L)	Mean	3.9	5.1	3.6	5.7	NA	7.7	4.2	4.7	NA	5.8	8.8	8.5	NA	15.5
	Range	0.4–1.7	0.8–8.8	0.4–11.5	3.3–8.4	NA	6.2–9.1	0.4–20.8	0.8–21.3	NA	5.5–6.1	0.4–21.4	4.0–17.0	NA	13.0–18.0
Phosphorus, Total (mg/L)	Mean	0.8	1.9	0.4	0.7	NA	1.0	0.5	0.7	NA	0.1	1.4	2.4	NA	2.5
	Range	0.1–5.9	0.2–10.3	0.1–1.4	0.1–1.1	NA	0.9–1.0	0.1–6.4	0.6–1.8	NA	0.02–0.2	0.1–5.1	0.5–10.0	NA	2.4–2.6
Chlorophyll-a (µg/L)	Mean	41	64	26	68	NA	44	10	53	NA	8	22	33	NA	2
	Range	2–500	2–140	2–110	2–250	NA	44	2–47	2–270	NA	4–11	2–140	2–100	NA	NA
Aluminum (µg/L)	Mean	1039	587	NA	NA	NA	NA	139	337	NA	NA	553	763	NA	NA
	Range	443–3510	35–1810	NA	NA	NA	NA	75–358	18–1730	NA	NA	540–566	763	NA	NA
Tin (µg/L)	Mean	100	100	NA	NA	NA	NA	100	100	NA	NA	100	100	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Antimony (µg/L)	Mean	2.9	7.2	NA	NA	NA	NA	3.5	1.0	NA	NA	1.0	1.0	NA	NA
	Range	1–5.4	1–31.8	NA	NA	NA	NA	1–7.7	NA	NA	NA	NA	NA	NA	NA
Arsenic (µg/L)	Mean	21.9	18.7	NA	NA	NA	NA	2.0	3.5	NA	NA	2.0	5.9	NA	NA
	Range	2–81.4	2–81.4	NA	NA	NA	NA	2.0	2–9.5	NA	NA	NA	NA	NA	NA
Barium (µg/L)	Mean	66.2	47.8	NA	NA	NA	NA	73.5	49.4	NA	NA	28.5	4.6	NA	NA
	Range	36.5–98.4	25.2–77.8	NA	NA	NA	NA	47.1–97	27.6–85.7	NA	NA	27.6–29.4	4.6	NA	NA
Beryllium (µg/L)	Mean	0.2	0.2	NA	NA	NA	NA	0.2	1.9	NA	NA	0.2	0.2	NA	NA
	Range	NA	NA	NA	NA	NA	NA	0.2	0.2–8.2	NA	NA	NA	NA	NA	NA
Cadmium (µg/L)	Mean	4.0	4.0	NA	NA	NA	NA	4.0	4.0	NA	NA	4.0	4.0	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Chromium (µg/L)	Mean	7.0	8.1	NA	NA	NA	NA	7.0	7.4	NA	NA	7.0	7.0	NA	NA
	Range	NA	7–12.6	NA	NA	NA	NA	NA	7–9.5	NA	NA	NA	NA	NA	NA
Cobalt (µg/L)	Mean	1.0	1.2	NA	NA	NA	NA	1.1	1.3	NA	NA	1.0	1.0	NA	NA
	Range	NA	1–2.8	NA	NA	NA	NA	1.7–1.4	1.0–2.5	NA	NA	NA	NA	NA	NA

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Copper (µg/L)	Mean	4.8	5.9	4.7	8.8	NA	NA	5.5	4.8	4.6	2.4	5.2	3.9	2.5	10.0
	Range	2.0–6.0	2.0–8.8	NA	5.2–12.4	NA	NA	2.0–8.2	2.0–9.4	4.6	2.3–2.5	3.6–6.9	NA	NA	8.9–11.1
Lead (µg/L)	Mean	8.6	8.8	0.5	0.5	NA	NA	4.4	3.9	0.5	0.5	9.7	13.8	0.5	0.5
	Range	5.0–15.0	5.0–24.0	NA	NA	NA	NA	0.5–5.0	0.5–5.0	NA	NA	8.1–11.3	13.8	0.5	0.5
Mercury (µg/L)	Mean	0.1	0.2	NA	NA	NA	NA	0.1	0.2	NA	NA	0.02	0.2	NA	NA
	Range	0.02–0.2	0.02–0.2	NA	NA	NA	NA	0.02–0.2	0.02–0.2	NA	NA	0.02	0.02	NA	NA
Molybdenum (µg/L)	Mean	18.4	17	NA	NA	NA	NA	14.9	18.4	NA	NA	2.2	18.9	NA	NA
	Range	14.7–21	13–18.9	NA	NA	NA	NA	9.2–18.9	14.3–20.0	NA	NA	19–21.4	18.9–18.9	NA	NA
Nickel (µg/L)	Mean	13.9	11.8	3.1	6.5	NA	NA	1.3	9.9	1.5	9.8	18.6	23.9	2.2	5.6
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Selenium (µg/L)	Mean	7.2	4.3	NA	NA	NA	NA	3.5	2.8	NA	NA	4.4	82.1	NA	NA
	Range	2.0–12.7	2.0–13.5	NA	NA	NA	NA	2.0–10.2	2.0–8.4	NA	NA	2.0–6.8	82.1	NA	NA
Silver (µg/L)	Mean	2.4	3.7	NA	NA	NA	NA	2.9	5.6	NA	NA	0.2	0.2	NA	NA
	Range	0.2–8.9	0.2–8.9	NA	NA	NA	NA	0.2–7.0	0.2–25.8	NA	NA	NA	NA	NA	NA
Thallium (µg/L)	Mean	1.0	1.0	NA	NA	NA	NA	1.0	1.0	NA	NA	1.0	1.0	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Vanadium (µg/L)	Mean	8.7	9.1	NA	NA	NA	NA	6.7	8	NA	NA	4.2	5.6	NA	NA
	Range	4.0–21.7	4.0–21.8	NA	NA	NA	NA	4.0–12.8	4.0–2.8	NA	NA	4.0–4.3	NA	NA	NA
Zinc (µg/L)	Mean	31.8	17.5	2.4	27.2	NA	NA	7.2	9.2	0.5	3.1	31.0	2.5	31.9	32.6
	Range	7.3–95.6	7.3–27.2	NA	24.1–30.2	NA	NA	2.0–12.9	2.0–21.4	0.5–0.5	2.4–3.9	25.0–37.0	2.5	31.9	29.7–35.5
Fecal coliform (MPN)	Mean	1,345	977	1,982	995	NA	NA	978	374	NA	NA	926	490	NA	NA
	Range	8–16,000	2–9,000	17–16,000	2–16,000	NA	NA	2–9,200	2–3,500	NA	NA	14–5,400	5–5,400	NA	NA
Total coliform (MPN)	Mean	5,372	3,185	5,250	2,083	NA	NA	4,868	3,234	NA	NA	3,767	4,556	NA	NA
	Range	49–24,000	11–24,000	24–24,000	7–16,000	NA	NA	16–24,000	14–24,000	NA	NA	49–24,000	11–24,000	NA	NA

Sources: Entrix, Inc. 2002a; Ventura (2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000)

Table 5-9. Analytical water quality parameters in grab samples from the SCRE by mouth closure status during summer (1997–2010).

	Location	**Near mouth**		**Lower estuary**		**Middle estuary**		**Upper estuary**		**Riverine**		**Lower outfall**		**Upper outfall**	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Ammonia (mg-N/L)	Mean	0.4	0.4	0.7	0.5	NA	0.1	0.5	0.4	NA	0.1	0.6	0.3	NA	0.2
	Range	0.1–1.4	0.1–0.9	0.1–1.5	0.1–1.2	NA	0.1–0.1	0.1–1.0	0.1–1.5	NA	0.1	0.1–1.4	0.1–0.6	NA	0.2
TKN (mg-N/L)	Mean	1.4	3.2	1.3	3.2	NA	1.5	1.7	3.2	NA	0.1	1.4	2.9	NA	0.4
	Range	0.5–2.5	1.0–8.2	0.4–2.1	1.4–7.1	NA	1.5	0.1–1.5	0.9–8.6	NA	0.1	0.3–1.9	0.8–7.6	NA	0.4
Nitrates+Nitrites (mg-N/L)	Mean	3.1	3.6	3.4	2.7	NA	3.7	2.4	2.2	NA	6.4	9.4	6.7	NA	13.5
	Range	0.4–6.3	0.4–12.1	0.8–5.4	0.4–5.6	NA	3.7	0.6–1.2	0.8–8.3	NA	6.4	5–13.4	0.8–16.2	NA	13.5
Phosphorus, Total (mg/L)	Mean	0.4	1.4	0.3	1.0	NA	1.0	0.3	1.2	NA	0.5	1.4	2.2	NA	4.0
	Range	0.3–1.1	0.1–6.8	0.1–1.4	0.5–3	NA	1.0	0.3–1.2	0.1–8.5	NA	0.5	0.1–5.8	0.1–9.9	NA	4.0
Chlorophyll-a (µg/L)	Mean	19	107	42	100	NA	42	33	2	NA	3	35	100	NA	2
	Range	2–60	2–300	2–130	2–290	NA	42	2–260	2	NA	3	7–80	5–310	NA	2
Aluminum (µg/L)	Mean	42.4	291.7	NA	NA	NA	NA	81.3	172.7	NA	NA	NA	205	NA	NA
	Range	3.0–81.7	78.6–832	NA	NA	NA	NA	70.0–92.6	12.0–378	NA	NA	NA	48.7–284	NA	NA
Tin (µg/L)	Mean	100	100	NA	NA	NA	NA	100	100	NA	NA	NA	100	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Antimony (µg/L)	Mean	1.0	1.0	NA	NA	NA	NA	2.0	1.3	NA	NA	NA	2.8	NA	NA
	Range	NA	NA	NA	NA	NA	NA	1.0–3.0	1.0–4.9	NA	NA	NA	1.0–6.3	NA	NA
Arsenic (µg/L)	Mean	4.0	2.5	NA	NA	NA	NA	2.0	3.1	NA	NA	NA	22.3	NA	NA
	Range	2.5–5.5	1–7.7	NA	NA	NA	NA	2.0	1.0–9.1	NA	NA	NA	1.0–61.8	NA	NA
Barium (µg/L)	Mean	32.2	41.7	NA	NA	NA	NA	83.0	65.4	NA	NA	NA	33.4	NA	NA
	Range	27.1–37.2	24.5–87.0	NA	NA	NA	NA	82.4–83.5	22.0–272	NA	NA	NA	26.2–42.2	NA	NA
Beryllium (µg/L)	Mean	0.2	0.2	NA	NA	NA	NA	0.2	0.2	NA	NA	NA	0.2	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Cadmium (µg/L)	Mean	4.0	4.0	NA	NA	NA	NA	4.0	4.0	NA	NA	NA	4.0	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Chromium (µg/L)	Mean	8.7	7.0	NA	NA	NA	NA	7.0	7.0	NA	NA	NA	7.0	NA	NA
	Range	7.0–9.1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Cobalt (µg/L)	Mean	1.5	1.2	NA	NA	NA	NA	1.0	2.2	NA	NA	NA	1.0	NA	NA
	Range	1.0–2.0	1.0–1.7	NA	NA	NA	NA	NA	1.0–11.5	NA	NA	NA	NA	NA	NA
Copper (µg/L)	Mean	6.0	3.7	NA	2.6	NA	NA	6.0	6.2	NA	3.4	NA	2.0	NA	2.9
	Range	NA	2–6.0	NA	0.5–6.8	NA	NA	NA	0.5–19.6	NA	0.5–9.3	NA	NA	NA	0.5–6
Lead (µg/L)	Mean	5.0	6.5	NA	0.5	NA	NA	5.0	4.8	NA	0.5	NA	8.4	NA	0.5
	Range	NA	5.0–15.5	NA	0.5–0.5	NA	NA	5.0	0.5–14.7	NA	0.5–0.5	NA	5–15.3	NA	0.5–0.5
Mercury (µg/L)	Mean	0.2	0.1	NA	NA	NA	NA	0.2	0.4	NA	NA	NA	0.1	NA	NA
	Range	NA	0.02–0.2	NA	NA	NA	NA	0.2	0.2–4.1	NA	NA	NA	0.02–0.2	NA	NA
Molybdenum (µg/L)	Mean	9.9	17.1	NA	NA	NA	NA	21.6	15.2	NA	NA	NA	15.4	NA	NA
	Range	9.4–1.3	14.1–23.4	NA	NA	NA	NA	21.2–22.0	8.1–21.0	NA	NA	NA	13.0–18.0	NA	NA
Nickel (µg/L)	Mean	2.0	19.7	NA	2.1	NA	NA	1.0	8.1	NA	2.0	NA	8.9	NA	1.9
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Selenium (µg/L)	Mean	2.0	10.1	NA	NA	NA	NA	2.0	4.0	NA	NA	NA	9.0	NA	NA
	Range	NA	2.0–52.1	NA	NA	NA	NA	2.0	2–11.6	NA	NA	NA	7.3–11.6	NA	NA
Silver (µg/L)	Mean	8	2.3	NA	NA	NA	NA	28.0	3.0	NA	NA	NA	0.2	NA	NA
	Range	0.2–15.8	0.2–14.9	NA	NA	NA	NA	24.3–31.7	0.2–14.7	NA	NA	NA	NA	NA	NA
Thallium (µg/L)	Mean	1	1	NA	NA	NA	NA	1	1	NA	NA	NA	1	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Vanadium (µg/L)	Mean	17.4	7.8	NA	NA	NA	NA	4.0	7.9	NA	NA	NA	4.0	NA	NA
	Range	4.0–30.8	4.0–30.8	NA	NA	NA	NA	NA	4.0–27.4	NA	NA	NA	4.0	NA	NA
Zinc (µg/L)	Mean	25.2	30.7	NA	17.2	NA	NA	11.8	22.8	NA	3.47	NA	17.4	NA	23.97
	Range	4.1–46.2	3.1–97.4	NA	12.6–23.0	NA	NA	8.3–13.8	2.0–96.5	NA	0.5–9.4	NA	13.2–20.4	NA	11.0–30.9
Fecal coliform (MPN)	Mean	908	1,070	1,820	859	NA	NA	1,866	745	NA	NA	1,814	1,216	NA	NA
	Range	2–3,500	2–16,000	8–24,000	2–9,200	NA	NA	9–16,000	2–24,000	NA	NA	30–24,000	2–16,000	NA	NA
Total coliform (MPN)	Mean	2,395	3,507	5,665	2,801	NA	NA	2,944	4,123	NA	NA	4,738	3,586	NA	NA
	Range	5–9,200	2–24,000	17–24,000	2–24,000	NA	NA	33–24,000	2–24,000	NA	NA	33–24000	4–24,000	NA	NA

Sources: Entrix, Inc. 2002a; Ventura (2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000)

Table 5-10. Analytical water quality parameters in grab samples from the SCRE by mouth closure status during fall (1997–2010).

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Ammonia (mg-N/L)	Mean	0.3	0.3	0.7	0.3	NA	0.3	0.5	0.4	NA	0.1	0.4	0.3	NA	0.4
	Range	0.1–0.6	0.1–0.8	0.1–1.1	0.1–0.9	NA	0.3	0.1–1.6	0.1–1.5	NA	0.1	0.2–0.7	0.1–0.7	NA	0.1–0.9
TKN (mg-N/L)	Mean	1.5	3.5	4.7	4.6	NA	2.6	3.7	3.5	NA	0.9	3.0	4.3	NA	1.9
	Range	0.7–3.1	0.6–10.1	1.6–14.7	1.6–25.6	NA	1.9–3.3	0.1–11.2	0.9–9.8	NA	0.1–1.4	0.2–5.9	1.4–3.3	NA	1.5–2.6
Nitrates+Nitrites (mg-N/L)	Mean	5.4	5.3	6.3	4.7	NA	6.5	2.1	3.5	NA	4.3	7.7	7.3	NA	15.3
	Range	0.4–12.2	0.7–14.0	0.4–14.3	0.5–12.0	NA	3.4–11.0	0.4–8.9	0.8–1.8	NA	3.3–5.0	0.8–14.5	3.4–16.0	NA	11.0–19.0
Phosphorus, Total (mg/L)	Mean	0.9	1.4	1.5	0.8	NA	0.8	0.5	0.6	NA	0.3	2.3	3.1	NA	3.1
	Range	0.3–1.8	0.9–9.3	0.6–3.0	0.1–2.9	NA	0.5–1.4	0.6–2.1	0.5–1.9	NA	0.2–0.5	0.5–4.2	0.8–34	NA	2.8–3.4
Chlorophyll-a (µg/L)	Mean	13	12	48	97	NA	59	29	82	NA	8	36	86	NA	49
	Range	2–31	2–290	1–120	2–250	NA	3–150	2–150	2–370	NA	5–13	4–95	2–590	NA	8–130
Aluminum (µg/L)	Mean	760	161	NA	NA	NA	NA	424	343	NA	NA	361	NA	NA	NA
	Range	361–114	18–295	NA	NA	NA	NA	92–1330	33–1350	NA	NA	361	NA	NA	NA
Tin (µg/L)	Mean	100	100	NA	NA	NA	NA	100	100	NA	NA	100	NA	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Antimony (µg/L)	Mean	35.7	30.3	NA	NA	NA	NA	7.6	1.0	NA	NA	1.0	NA	NA	NA
	Range	1.0–100.0	1.0–113.0	NA	NA	NA	NA	1.0–15.1	1.0	NA	NA	1.0	NA	NA	NA
Arsenic (µg/L)	Mean	3.6	2.0	NA	NA	NA	NA	2.0	2.0	NA	NA	2.0	NA	NA	NA
	Range	2.0–6.7	2.0	NA	NA	NA	NA	2.0	2.0	NA	NA	2.0	NA	NA	NA
Barium (µg/L)	Mean	26.4	22.6	NA	NA	NA	NA	62.5	41.1	NA	NA	45.0	NA	NA	NA
	Range	14.0–39.7	4.8–33.0	NA	NA	NA	NA	37.0–91.9	23.7–65.6	NA	NA	45.0	NA	NA	NA
Beryllium (µg/L)	Mean	0.2	0.2	NA	NA	NA	NA	0.2	0.2	NA	NA	0.2	NA	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Cadmium (µg/L)	Mean	4.0	4.0	NA	NA	NA	NA	4.0	4.0	NA	NA	4.0	NA	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Chromium (µg/L)	Mean	7.0	7.0	NA	NA	NA	NA	7.0	7.0	NA	NA	7.0	NA	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

	Location	Near mouth		Lower estuary		Middle estuary		Upper estuary		Riverine		Lower outfall		Upper outfall	
	Mouth	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**	**Open**	**Closed**
Cobalt (µg/L)	Mean	1.0	1.0	NA	NA	NA	NA	2.5	1.2	NA	NA	1.0	NA	NA	NA
	Range	1.0	1.0	NA	NA	NA	NA	1.0–3.4	1.0–1.8	NA	NA	1.0	NA	NA	NA
Copper (µg/L)	Mean	4.4	6.1	NA	2.2	NA	NA	7.7	3.8	NA	2.8	8.9	5.0	NA	2.5
	Range	2.0–6.0	2.0–15.7	NA	0.5–5.0	NA	NA	2.0–18.4	0.5–6.5	NA	0.5–6	3.2–13	5.0	NA	0.5–6.0
Lead (µg/L)	Mean	5.0	5.0	NA	0.5	NA	NA	5.0	3.9	NA	0.5	5.0		NA	0.5
	Range	5.0	5.0	-	0.5	-	-	5.0	0.5–5.0	NA	0.5–0.5	5.0	NA	NA	0.5
Mercury (µg/L)	Mean	0.16	0.13	NA	NA	NA	NA	0.13	0.16	NA	NA	0.2	0.2	NA	NA
	Range	0.02–0.2	0.02–0.2	NA	NA	NA	NA	0.02–0.2	0.02–0.2	NA	NA	0.2	0.2	NA	NA
Molybdenum (µg/L)	Mean	9.5	17.5	NA	NA	NA	NA	13.9	12	NA	NA	13.5	NA	NA	NA
	Range	7.5–12.5	12.4–25.1	NA	NA	NA	NA	9.4–15.9	1.0–19.0	NA	NA	13.5	NA	NA	NA
Nickel (µg/L)	Mean	1.7	6.7	NA	3.8	NA	NA	1.0	6.9	NA	4.8	7.3	NA	NA	7.5
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Selenium (µg/L)	Mean	5.3	3.4	NA	NA	NA	NA	3.4	9.5	NA	NA	2.0	NA	NA	NA
	Range	2.0–11.1	2.0–7.5	NA	NA	NA	NA	2.0–6.1	2.0–62.8	NA	NA	2.0	NA	NA	NA
Silver (µg/L)	Mean	0.2	0.2	NA	NA	NA	NA	5.9	1.5	NA	NA	0.2	0.2	NA	NA
	Range	0.2	0.2	NA	NA	NA	NA	0.2–12.5	0.2–11.7	NA	NA	0.2	0.2	NA	NA
Thallium (µg/L)	Mean	1.0	1.0	NA	NA	NA	NA	1.0	1.0	NA	NA	1.0	NA	NA	NA
	Range	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Vanadium (µg/L)	Mean	24.7	13.3	NA	NA	NA	NA	4.2	16.2	NA	NA	4.0	NA	NA	NA
	Range	4.0–40.0	4.0–41.0	NA	NA	NA	NA	4.0–4.9	4.0–40.0	NA	NA	4.0	NA	NA	NA
Zinc (µg/L)	Mean	15.5	21.4	NA	21.7	NA	NA	22.1	12.2	NA	8.2	62.2	2.7	NA	27.33
	Range	2.0–30.2	5.0–40.3	NA	11.0–42.0	NA	NA	2.0–62.3	2.2–30.0	NA	0.5–22.0	46.5–78.0	2.7	NA	15.0–48.0
Fecal coliform (MPN)	Mean	1,404	1,266	1,460	848	NA	NA	1,878	649	NA	NA	1,329	1,446	NA	NA
	Range	2–16,000	2–24,000	34–16,000	2–24,000	NA	NA	17–16,000	2–24,000	NA	NA	5–16,000	4–24,000	NA	NA
Total coliform (MPN)	Mean	4,126	3,182	4,617	2,607	NA	NA	5,271	3,674	NA	NA	5,282	6,151	NA	NA
	Range	2–16,000	20–24,000	11–24,000	13–24,000	NA	NA	220–24,000	33–24,000	NA	NA	12–16000	23–24,000	NA	NA

Sources: Entrix, Inc. 2002a; Ventura (2009, 2008, 2007, 2006, 2005, 2004, 2003, 2002, 2001, 2000)

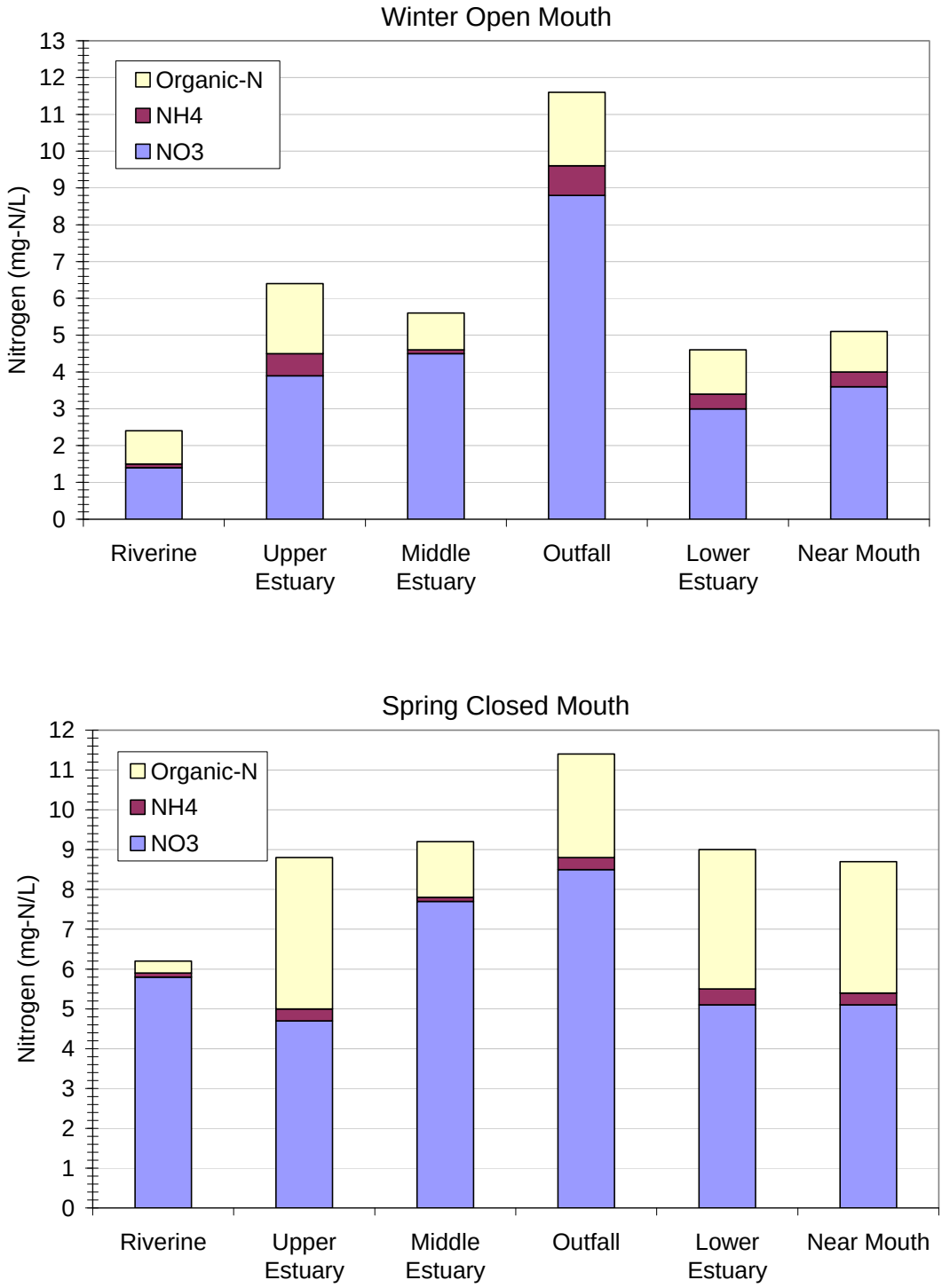
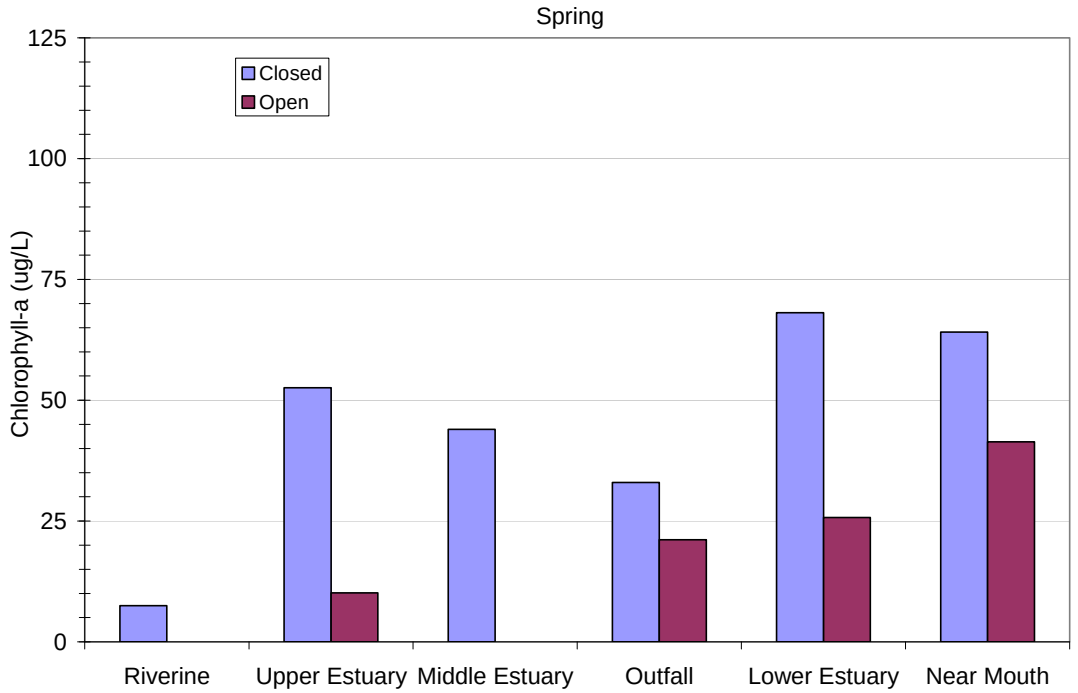


Figure 5-7. Average nitrogen levels in grab samples collected between 2000 and 2010 for open- (winter) and closed-mouth (spring) conditions.[5]

[5] See Tables 5-7 through 5-10 for other seasons and mouth closure conditions.



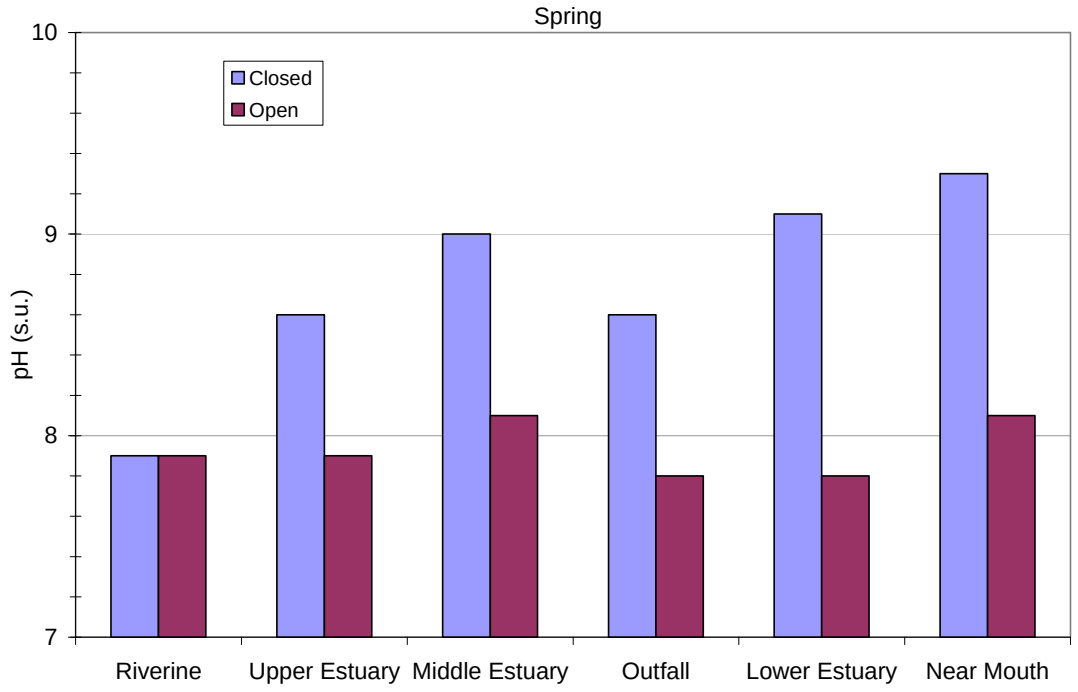


Figure 5-8. Average chlorophyll-a and pH levels in grab samples collected between 2000 and 2010 for open- and closed-mouth conditions during spring.[6]

[6] See Tables 5-7 through 5-10 for other seasons and mouth closure conditions.

5.3 Comparisons to Existing Water Quality Criteria

Water quality objectives for the SCRE are outlined in the Los Angeles Regional Water Quality Control Board Basin Plan (LARWQCB 1994). Although upstream discharges, non-point source runoff, and regional geology contribute to water quality conditions in the SCRE, we provide a brief comparison of existing water quality data with Basin Plan criteria to provide an indication of the suitability of the SCRE to support designated beneficial uses listed in Section 1.1. Based upon contemporary water quality conditions, aquatic beneficial uses (COMM, EST, MAR, MIGR, RARE, SPWN) may not be supported at all times due to exceedances of water quality objectives for ammonia, biostimulatory substances, DO, and pH. It should be noted that the VWRF is in compliance with existing discharge limitations and later discussion (Chapter 9 and 11) is limited to conditions most directly affecting focal fish species. Further, historical exceedances of water quality objectives in the SCRE for ammonia, bacteria, DO, pesticides, and toxicity may not be reflective of current conditions or directly attributed to ongoing VWRF operations. Details of water quality comparisons are provided below.

5.3.1 Ammonia

Because un-ionized ammonia (NH_3) is highly toxic to fish and the relative portioning of un-ionized ammonia to total ammonia ($NH_3 + NH_4^+$) is controlled by pH and water temperature, amendments to the Basin Plan Resolution No. 2004-022 establish water quality objectives for ammonia in inland surface waters not characteristic of freshwater based on the more stringent of the freshwater or saltwater criteria[7]. Available grab sampling data (ammonia, pH, salinity and temperature) from both the City of Ventura Annual Reports and 2009–2010 water quality monitoring data (Appendix C) were used to identify applicable receiving water objectives (acute value of 0.192 mg-N/L, chronic value of 0.029 mg-N/L). Of the 101 sample results for which free ammonia could be calculated, 23 (23%) were in exceedance of the chronic criterion (Table 5-11) on nine different days. Although it should be noted that historical exceedances of water quality objectives for ammonia may not be reflective of current conditions, five samples collected as part of this study (approximately 5%) contained free ammonia in excess of the final acute criterion. In general, all exceedances were observed under closed-mouth conditions with relatively high pH levels, with the majority of these exceedances occurring from spring though fall. These conditions are accompanied by reduced tidal flushing that would be expected to remove ammonia and accumulated algae from the SCRE.

[7] Resolution No: 2004-022 established one-hour (acute) and 4-day average (chronic) objectives for saltwater as fixed concentrations of un-ionized ammonia, whereas the freshwater criteria continue to be determined as a function of temperature as pH. In all cases, saltwater criteria were found to be lower than freshwater criteria.

Table 5-11. Analytical samples found in excess of water quality objectives for ammonia.

Date	Mouth	Location	Temp	pH	Total ammonia (mg-N/L)	Free ammonia (mg-N/L)	Data source[a]
10/2/2007	Closed	Upper Estuary	18.6	9.21	0.2	0.06	1
10/2/2007	Closed	Lower Estuary	19.7	9.42	0.2	0.09	1
10/2/2007	Closed	Near Mouth	19.9	9.51	0.1	0.05	1
11/6/2007	Closed	Lower Estuary	16.2	9.34	0.3	0.10	1
11/6/2007	Closed	Lower Outfall	16.3	8.83	0.3	0.04	1
5/6/2008	Closed	Lower Outfall	21.8	9.10	0.3	0.09	1
5/6/2008	Closed	Upper Estuary	22.2	9.54	0.1	0.06	1
7/1/2008	Closed	Upper Estuary	21.8	9.49	0.1	0.05	1
8/5/2008	Closed	Upper Estuary	24.5	8.33	0.9	0.08	1
10/7/2008	Closed	Near Mouth	20.5	9.58	0.1	0.06	1
10/7/2008	Closed	Lower Outfall	21.0	9.60	0.1	0.06	1
12/9/2008	Closed	Upper Estuary	11.3	8.08	9.0	0.16	1
12/9/2008	Closed	Upper Estuary	12.3	8.05	2.9	0.04	1
9/1/2009	Closed	Upper Estuary	25.4	9.70	0.1	0.07	2
9/1/2009	Closed	Upper Outfall	25.0	9.70	0.9	0.63	2
9/1/2009	Closed	Lower Outfall	24.7	9.57	0.7	0.44	2
9/1/2009	Closed	Near Mouth	23.9	9.45	0.6	0.33	2
9/1/2009	Closed	Lower Estuary	24.0	9.50	0.4	0.23	2
5/27/2010	Closed	Near Mouth	21.1	9.78	0.1	0.07	3
5/27/2010	Closed	Lower Estuary	21.4	9.62	0.4	0.24	3
5/27/2010	Closed	Middle Estuary	21.1	9.48	0.1	0.05	3
5/27/2010	Closed	Upper Estuary	20.5	9.33	0.1	0.04	3
5/27/2010	Closed	Upper Estuary	20.3	9.43	0.1	0.05	3

[a] Source:
1 City of Ventura Annual Reports (Ventura 2007, 2008)
2 Stillwater Sciences 2010
3 Stillwater Sciences (in progress)

5.3.2 Bacteria, coliform

Total and fecal coliform bacteria are used as indicator organisms to determine the likelihood that pathogenic bacteria may be present in surface waters used for recreational purposes. The LARWQCB (1994) established water quality objectives for waters designated for contact (REC-1) and non-contact (REC-2) recreational uses, with recent amendments for marine waters (Resolution No. 2001-018) as follows for single sample collections: (a) total coliform density shall not exceed 10,000/100 ml, (b) fecal coliform density shall not exceed 400/100 ml, (c) *Enterococcus* density shall not exceed 104/100 ml, and (d) total coliform density shall not exceed 1,000/100 ml, if the ratio of fecal-total coliform exceeds 10%. Bacterial levels in the SCRE (Tables 5-7 through 5-10) are well above the coliform objectives. Although it should be noted that historical exceedances of water quality objectives for bacteria may not be reflective of current conditions and no intensive sampling data was available to construct the multi-sample log-mean concentrations specified in the Basin Plan, approximately 12% of total coliform samples reviewed as part of this study exceeded the 10,000/100 ml criteria and 35% of fecal coliform samples exceeded 400/100 ml. A detailed bacterial source tracking study would be required to confirm linkages between the influences of migratory and resident bird populations and other sources, including the VWRF. Nevertheless, the frequency of exceedances was relatively consistent across sampling locations and seasons, suggesting non-point source origins of bacteria. For example, during open-mouth periods, the proportion of samples in excess of the water quality objectives was slightly increased, consistent with runoff sources from upstream locations. The SCRE was placed on the CWA 303(d) list for coliform bacteria in 1998 (LARWQCB 2009) and a bacterial TMDL for the Santa Clara River and upstream reaches was approved under LARWQCB Resolution R10-006 in July 2010.

5.3.3 Bioaccumulation

The LARWQCB (1994) determined that toxins may not be present at levels that may bioaccumulate and pose a threat to aquatic life or human health. Although a number of substances are present in the SCRE that could lead to potential bioaccumulation (e.g., mercury, selenium), no tissue data for sportfish or other organisms is available to evaluate this narrative water quality objective.

5.3.4 Biochemical Oxygen Demand (BOD)

Surface waters may not contain substances that result in increased BOD in quantities sufficient to adversely affect beneficial uses. The City regularly reports effluent BOD for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit (e.g. Ventura 2009, 2008, etc.), detailing any exceedances of its permit as part of its regular reporting requirement. Because no recent violations of permit requirements have been documented, no further discussion of this water quality objective is provided here.

5.3.5 Biostimulatory substances

Biostimulatory substances are defined in the Basin Plan as substances (such as excess nitrogen or phosphorus) that stimulate aquatic growth. Biostimulatory substances shall not promote aquatic growth to nuisance levels or to the point that the growth negatively affects beneficial uses (LARQWB 1994). The Basin Plan does not specify particular water quality objectives for biostimulatory substances and nutrient levels in the SCRE arrive from multiple sources including the VWRF. Nevertheless, based upon variations in DO and pH (as well as elevated levels of chlorophyll-a [Tables 5-7 through 5-10]), the SCRE exhibits signs of nutrient enrichment, or

"eutrophication". Measured nutrient concentrations in the outfall channel and all other sampling stations are well above regional reference levels (TN=0.5 mg-N/L, TP=0.030 mg-P/L as 25th percentile) for the Southern and Central California Chaparral and Oak Woodlands (Level III) ecoregion (USEPA 2000, 2001). Additionally, the SCRE is currently on the CWA 303(d) list for nitrate (LARWQCB 2009). A separate assessment of trophic status of the SCRE is provided in Section 5.5.

5.3.6 Chemical constituents

The Basin Plan (LARWQCB 1994) establishes a narrative water quality objective for surface waters so that they do not contain chemical constituents in amounts that adversely affect designated beneficial uses. However, these protections are generally established to limit a variety of chemical constituents in drinking water supplies, such as inorganic compounds, volatile organic compounds (VOCs), and non-volatile synthetic organic compounds (SOCs) such as herbicides and pesticides. The City has conducted receiving water sampling and analysis at three sites since 2000 (Florence Jay, pers. comm. 2011), with low levels of trihalomethane compounds detected in samples analysis reports reviewed for this study (ASL 2009 and 2010a). Additional sampling in 2010 of the three groundwater wells adjacent to the SCRE found one semi-volatile organic compound (1,2-dichloropropane) above detection limits and low levels of several trace metals including copper, lead, zinc, and nickel in the samples analyzed (ASL 2010b). No other chemical constituents were detected in the surface water or groundwater samples.

5.3.7 Chlorine, total residual

The Basin Plan states that total residual chlorine may not exceed 0.1 mg/L in effluent and may not persist in receiving waters at any concentration (LARWQCB 1994). The City regularly reports effluent chlorine residual for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit and details any exceedances of its permit as part of its regular reporting requirement. Data from the City of Ventura Annual Reports (2000–2008) shows that total residual chlorine has not been detected in the SCRE and no further discussion of this water quality objective is provided.

5.3.8 Color

The Basin Plan prohibits coloration that adversely affects designated beneficial uses (LARWQCB 1994). Since no known discharges have been attributed to discoloration of the SCRE and there is no known SCRE data regarding color, no further discussion of this water quality objective is provided.

5.3.9 Exotic vegetation

Introduced exotic vegetation proximal to streams poses a potential threat to native vegetation and often provides little value to native fish and wildlife species. As such, the Basin Plan prohibits the introduction of exotic vegetation that would negatively impact designated beneficial uses (LARWQCB 1994). Giant reed (*Arundo donax*), a non-native invasive plant, is widespread throughout the riparian area of the SCRE (Section 6.1). However, because giant reed establishment is widespread throughout California, it is unclear whether the stands found within the SCRE are related to the VWRF discharge or mouth closure status.

5.3.10 Floating material

Floating material including solids, liquids, foams, and scum that create an aesthetic nuisance or provide substrate for undesirable organisms is prohibited under the Basin Plan (LARWQCB 1994). Although the City is required to report any "upset" conditions at the VWRF that results in the discharge of foams and other floating materials, no recent incidences of this type have occurred (Dan Pfeifer, pers. comm. 2010). Due to this and the lack of historical data for the SCRE, no further discussion of this water quality objective is provided.

5.3.11 Methylene Blue Activated Substances (MBAS)

Methylene blue activated substances (MBAS) include detergents which can interfere with the function of aquatic organisms' gills. Under the Basin Plan (LARWQCB 1994), MBAS concentrations cannot exceed 0.5 mg/L in drinking water. The City regularly reports MBAS for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit, detailing any exceedances of its permit as part of its regular reporting requirement. Typically, MBAS in the VWRF discharge to the Wildlife Ponds is below 1 mg/L year-round. Due to the relatively low concentrations of MBAS in the VWRF effluent prior to discharge to the SCRE and the lack of existing MBAS data at receiving water stations within the SCRE, no further discussion of this water quality objective is provided.

5.3.12 Mineral quality

Mineral quality is determined by the geology of the surrounding watershed. However, polluted discharges such as industrial process water and water softening brines can degrade the mineral quality of surface waters. The Basin Plan defines the water quality objectives for six mineral quality indicators at various locations in the region (LARWQCB 1994). The City regularly reports mineral and total dissolved solids (TDS) levels for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit, detailing any exceedances of its permit as part of its regular reporting requirement. In general, TDS from the VWRF is below other sources of water to the SCRE (Carollo Engineers 2010b). Because no agricultural (AGR), potable (DOM, MUN), or industrial process water (IND) uses of the SCRE are defined, no specific mineral quality objectives are defined for the SCRE are included and no further discussion of this water quality objective is provided.

5.3.13 Nitrogen (nitrate, nitrite)

Excess nitrogen may result in excessive aquatic growth (biostimulatory substance) and pose a human health risk. Although beneficial uses of the SCRE do not include potable water supply (DOM, MUN), the current effluent limitation for the VWRF is 10 mg-N/L (Order No. R4-2008-0011). It should be noted that nitrogen in the SCRE arrives from multiple sources including the VWRF and upstream locations. For example, the SCRE and several reaches of the Santa Clara River were placed on the CWA 303(d) list for nitrate in 2008 (LARWQCB 2009), suggesting that upstream sources may contribute to observed levels of nitrates in the SCRE. Although riverine samples from the Santa Clara River upstream of the SCRE are generally elevated with respect to NO_3(Tables 5-7 through 5-10), of 467 SCRE water samples analyzed for this study (Appendix B), 72 (15%) had a concentration of NO_3 + NO_2 greater than 10 mg-N/L. However, it should be noted that historical exceedances of water quality objectives for nitrogen may not be reflective of current conditions, and ongoing improvements to the VWRF process performance are expected to meet this requirement (Carollo Engineers 2010b). Nitrogen is discussed further in Section 5.2.5

under biostimulatory substances and a separate assessment of trophic status of the SCRE is provided in Section 5.5.

5.3.14 Oil and grease

As defined in the Basin Plan, waters may not contain oils, greases, waxes or other materials that create a film on the surface of the water that adversely affects beneficial uses or becomes a nuisance. The City regularly reports oil and grease levels well below its discharge permit level of 10 mg/L for the VWRF discharge to the Wildlife Ponds, detailing any permit exceedances as part of its regular reporting requirements. No known oil and grease discharges have occurred within the SCRE and due to the lack of existing SCRE data regarding oil and grease, no further discussion of this water quality objective is provided.

5.3.15 Dissolved oxygen (DO)

For the protection of aquatic life, the Basin Plan states that DO in receiving waters must be greater than 7 mg/L as an annual mean and that no single sample shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations (LARWQCB 1994). The City regularly reports concentrations of DO at the ETS and in receiving waters as a condition of its NPDES permit, detailing any exceedances of its permit as part of its regular reporting requirements. Although historical results of DO sampling at the ETS are on the order of 8 mg/L, analysis for this study shows that DO levels frequently do not meet the Basin Plan water quality objectives above. The frequency of exceedance in grab samples collected since 1997 is presented below in Table 5-12. It should be noted that historical exceedances of water quality objectives for DO may not be reflective of current conditions. However, although SCRE DO levels may not be directly attributed to ongoing VWRF operations, DO levels found during continuous *in situ* water quality monitoring conducted by the City in the lower outfall channel and at the south side of the SCRE during 2009–2010 were below the 5 mg/L water quality objective for more than 25% of the monitoring period in all seasons but winter (Table 5-6).

Table 5-12. Frequency of analytical samples found below the 5 mg/L water quality objective for DO (1997–2010).

Location	Winter		Spring		Summer		Fall	
	Open	Closed	Open	Closed	Open	Closed	Open	Closed
Near Mouth	3 of 70 (4%)	1 of 38 (3%)	4 of 71 (6%)	3 of 36 (8%)	5 of 33 (15%)	9 of 73 (12%)	1 of 31 (3%)	10 of 82 (12%)
Lower Estuary	8 of 82 (10%)	3 of 43 (7%)	18 of 85 (21%)	3 of 36 (8%)	8 of 35 (23%)	15 of 93 (16%)	4 of 41 (10%)	6 of 90 (7%)
Middle Estuary	1 of 7 (14%)	0 of 7	0 of 8	0 of 2	0 of 2	3 of 9 (33%)	0 of 5	0 of 4
Upper Estuary	14 of 127 (11%)	9 of 68 (13%)	19 of 130 (15%)	16 of 71 (22%)	4 of 47 (9%)	53 of 140 (38%)	3 of 40 (7%)	52 of 153 (34%)
Riverine	0 of 7	0 of 5	0 of 5	2 of 4 (50%)	0 of 1	0 of 7	0 of 2	0 of 7
Lower Outfall	6 of 74 (8%)	1 of 42 (2%)	5 of 74 (7%)	5 of 36 (14%)	2 of 34 (6%)	14 of 76 (18%)	2 of 31 (6%)	15 of 82 (18%)
Upper Outfall	0 of 2	0 of 1	0 of 1	0 of 2	NA	0 of 3	NA	1 of 5 (20%)

5.3.16 Pesticides

Because pesticides represent a hazard to aquatic organisms and plants, the Basin Plan (LARWQCB 1994) defines maximum allowable concentrations for specific pesticides. The City regularly reports pesticide screening analysis results for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit and has no documented exceedances (Dan Pfeifer, pers. comm. 2010). The City has conducted receiving water sampling and analysis for VOCs, Pesticides, and PCBs at three sites since 2000 (Florence Jay, pers. comm. 2011), with no detection of pesticides in analysis reports reviewed for this study (ASL 2009 and 2010a). Although the SCRE was placed on the CWA 303(d) list for toxaphene in 1998 (LARWQCB 2009), neiother the surface water data or additional sample data taken in 2010 from the three groundwater wells adjacent to the Estuary (ASL 2010b) exhibited detectable levels of pesticides.

5.3.17 pH

Changes in pH other than those caused by natural processes can negatively affect aquatic organisms. Elevated pH is of particular concern due to the pH control of free ammonia levels, which can be toxic to fish and other aquatic organisms. Basin Plan (LARWQCB 1994) water quality objectives for bays and estuaries only allow a pH range of 6.5 to 8.5 as a result of waste discharge and pH levels may not be altered by more 0.2 units from as a result of waste discharge. On an annual basis, pH in the SCRE was between 6.5 and 8.5 in approximately 61.5% of discrete samples collected from 1997–2010 (Appendix B), and a large number of samples (38.5%) were in excess of 8.5. Although it should be noted that historical exceedances of water quality objectives for pH may not be reflective of current conditions or directly attributed to ongoing VWRF operations, elevated pH may be the result of increased photosynthetic activity in the SCRE, decreasing levels of dissolved carbon dioxide in the water column and increasing the proportion of free (unionized) ammonia.

5.3.18 Polychlorinated Biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) are a highly toxic group of compounds and the LARWQCB (1994) sets a maximum daily average of 30 ng/L in order to protect estuarine environments. The City regularly reports PCB analysis results for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit, detailing any exceedances of its permit as part of its regular reporting requirement. The City has conducted receiving water sampling and analysis for VOCs, Pesticides, and PCBs at three sites since 2000 (Florence Jay, pers. comm. 2011), with no detection of PCBs in analysis reports reviewed for this study (ASL 2009 and 2010a). Analysis of samples taken in 2010 from the three groundwater test wells adjacent to the Estuary did not show any detectable levels of PCBs (ASL 2010b). Although no direct tissue data for sportfish or other organisms was available, available results suggests little or no PCBs within the SCRE and no linkages to continued VWRF operations.

5.3.19 Radioactive substances

The Basin Plan (LARWQCB 1994) prohibits the presence of any radioactive substances that are harmful to organisms or that result in radioactive substances entering the food web to the extent they become harmful to organisms. However, since no known discharges of radioactive substances exist in or near the SCRE and no direct water or tissue data for sportfish or other organisms is available, no further discussion of this water quality objective is provided.

5.3.20 Solid, suspended, or settleable materials

The Basin Plan (LARWQCB 1994) established that waters shall not contain suspended and settleable materials that cause nuisance or adversely affect beneficial uses. The City regularly reports suspended and settleable solids analysis results for the VWRF discharge to the Wildlife Ponds as a condition of its NPDES permit, detailing any exceedances of its permit as part of its regular reporting requirements. Available TSS data from the *Metals Translator Study* (Entrix, Inc. 2002b) ranges from 0.05 mg/L to 87 mg/L. Natural background variations in TSS in the SCRE is also very large due to sediment transport from the upper watershed (Section 3.1.3). Further, since sediment deposition within the SCRE is dynamic and strongly influenced by runoff events, it is unlikely that discharges from the VWRF or other local sources have an appreciable effect upon sediment in the SCRE. No further discussion of this water quality objective is provided.

5.3.21 Taste and odor

Undesirable tastes and odors are prohibited in concentrations that negatively impact edible aquatic resources, cause nuisance, or adversely affect beneficial uses (LARWQCB 1994). Although no potable water (DOM, MUN) uses are defined for the SCRE, commercial and sportfishing uses (COMM) are included. No occurrences of taste and odor complaints in sportfish were identified as part of this evaluation and no further discussion of this water quality objective is provided.

5.3.22 Temperature

Although no specific water temperature objectives are established for the SCRE under the Basin Plan (LARWQCB 1994), temperature objectives for discharges to enclosed bays and estuaries are specified in the Thermal Plan (SWRCB 1975). The Thermal Plan imposes the following restrictions upon elevated water temperature discharges: (a) maximum discharge temperature shall not exceed the natural receiving water temperature by more than 11.1ºC (20°F), (b) discharges either individually or combined with other discharges shall not create a zone, defined by water temperatures of more than 0.6ºC (1°F) above natural receiving water temperature, which exceeds 25% of the cross-sectional area of a main river channel at any point, (c) no discharge shall cause a surface water temperature rise greater than 2ºC (4°F) above the natural temperature of the receiving waters at any time or place, and (d) additional limitations shall be imposed when necessary to assure protection of beneficial uses. Although no comprehensive water temperature monitoring program has evaluated the effluent temperatures of the VWRF or within the SCRE, Figures 5-4 and 5-5 indicate that the VWRF discharge is generally cooler than receiving waters in all but winter months. For this reason, it is unlikely that the VWRF discharge affects water temperatures within the SCRE and no further discussion of this water quality objective is provided.

5.3.23 Toxicity

The Basin Plan (LARWQCB 1994) states that all waters are required to be free from toxins at concentrations that are detrimental to human, plant, animal, or aquatic life. Furthermore, no acute toxicity is permitted in ambient waters, including mixing zones, and chronic toxicity is prohibited in ambient water outside of mixing zones. The acute toxicity water quality objective is 90% survival in at least three consecutive 96-hr bioassays, with no less than 70% survival in any single test. Although ammonia toxicity is discussed in Section 5.2.1, we provide a summary of direct toxicity monitoring conducted by the City below.

In response to prior concerns regarding discharges of copper to the SCRE, the City previously completed a number of investigations, including a source control study for copper that identified corrosion of domestic plumbing as a major copper source to discharges from the VWRF (Nautilus Environmental 2005). In 2002, the City completed several studies related to the development of a site-specific copper criterion, including a *Resident Species Study* and a *Metal Translator Study* (Entrix, Inc. 2002a, b). These studies concluded that metal concentrations entering the study area from upstream closely approximated those found in the treated VWRF effluent. These studies also found that a freshwater criterion for copper was appropriate due to the predominance of freshwater and euryhaline species found in the SCRE (Nautilus Environmental 2005) and that salinity in the estuary was predominantly below the 10 ppm level for saltwater.

Although the VWRF has generally met discharge limits for all metals except for periodic exceedances of copper criteria discussed above, in reviewing historical water quality data compiled for this study (Table 5-7 through 5-10), analytical results for copper, nickel, lead, zinc in the SCRE were above the USEPA (2000) California Toxics Rule (CTR) criteria for these metals, and ammonia levels were periodically above toxicity criteria established under the Basin Plan (Section 5.2.1). In 2003, Regional Board staff and fishery resource agencies (California Department of Fish and Game [CDFG], National Marine Fisheries Service [NMFS], U.S. Fish and Wildlife Service [USFWS]) concluded that: (a) The City should conduct toxicity tests on the effluent using non-freshwater species, (b) The City should analyze copper accumulation in the sediments of the estuary, (c) the freshwater quality criteria of the CTR are inappropriate for the SCRE due to the presence of marine and estuarine species, and (d) in addition to addressing non-compliance with the copper limits, the fundamental question of whether the VWRF should be allowed to continue discharging into the estuary should be evaluated. Nautilus Environmental (2005) provides the results of these investigations. The City regularly reports toxicity testing results for the VWRF outfall junction structure and several receiving water stations in the SCRE. Tests included acute (LC50) toxicity for fathead minnow (*Pimephales promelas*), and chronic toxicity tests for growth (fathead minnow, green algae [*Selenastrum capricornutum*]), *Ceriodaphnia dubia* reproduction, and survival.

The SCRE was placed on the CWA 303(d) list for toxicity in 2008 (LARWQCB 2009). Toxicity in both marine and freshwater species has been observed in routine tests of SCRE receiving waters (Nautilus Environmental 2005, VWRF Annual Reports). Despite exceedances of copper criteria in the VWRF discharge, the source of episodic toxicity events in the SCRE appears to be unrelated to the VWRF discharges. Nautilus Environmental (2005) reports the results of over 100 toxicity tests of water samples collected from SCRE receiving water stations, with approximately 10% of the samples exhibiting toxicity. Copper concentrations were generally low in all water samples tested, and no relationships were observed between aquatic toxicity and concentrations of copper or other contaminants of concern (e.g., Ni, Zn). Although no evaluation of ammonia or pesticide levels was included in these tests, the toxicity events identified by Nautilus Environmental (2005) generally occurred during stormwater or dry weather outflow events, suggesting that upstream and groundwater sources are of concern. For example, no toxicity was observed during a dry weather period when the SCRE mouth was closed and VWRF discharge had the greatest potential to influence water quality in the lagoon. Nautilus Environmental (2005) concluded that sediment toxicity was observed only on an intermittent basis and not related to concentrations of copper, nickel, or zinc, suggesting that sediment levels across the estuary were not adversely impacted by the discharge.

As a condition of its NPDES Permit, the City has completed annual Benthic Macroinvertebrate (BMI) studies to assess the biological integrity of the SCRE using a multiple lines of evidence (MLOE) approach that includes assessment of BMI community structure, sediment chemistry,

and direct sediment toxicity testing (ABC Laboratories 2004–2009). Sediment contaminants were measured in conjunction with BMI monitoring in Fall 2008: copper and nickel were the only heavy metals measured that exceeded NOAA (Long and Morgan 1990) Effects Range Low (ERL)[8] thresholds, while none of the 187 individual organic constituents measured was above the detection limit (ABC Laboratories 2009). Sediment toxicity was also tested, using the amphipod *Hyalella azteca* in 10-day toxicity exposure survival trials. None of the sediments tested were acutely toxic to the amphipod, suggesting that any contaminants present in SCRE sediments are tightly bound and are not biologically available (ABC Laboratories 2009). In addition, high sediment loads from the upper Santa Clara River watershed (Section 3.1.3) and frequent flood-scour events in the SCRE reduce the likelihood of accumulation of natural organic matter in the sediments along with any potential for the accumulation of sediment bound contaminants.

5.3.24 Turbidity

The water quality objective for turbidity in surface waters states that turbidity may not cause nuisance or adversely affect beneficial uses (LARWQCB 1994). The Basin Plan limits turbidity increases to below 20% above background levels between 0 and 50 Nephelometric Turbidity Units (NTU). Turbidity levels in the SCRE vary within this range during winter (Table 5-2 and Table 5-6), with increased summer level associated with increased algal chlorophyll-a. However, no direct turbidity discharges from the VWRF or other facilities were identified as part of this evaluation, and no further discussion of this water quality objective is provided.

5.4 Historical Water Quality Conditions

Little information exists to provide a comparison of current and historical water quality in the SCRE as a means of establishing a relevant reference condition. However, since human settlement and land-use changes of the Santa Clara River watershed and SCRE pre-date the establishment of modern water quality assessment techniques, it is unlikely that any such information represents a pristine or undisturbed state. Nonetheless, examining historical land use changes and other factors influencing water quality before and after the VWRF began operations in 1958 may increase the understanding of the current functioning of the SCRE.

In addition to historical changes in geomorphology and hydrology summarized in Sections 3 and 4 of this report, Nautilus Environmental (2005) provides a summary of historical land use changes in the surrounding watershed. Time periods discussed include the Agrarian Era (1782–1870s) (associated with grazing beginning with the arrival of Spanish and Mexican settlers), the Commercial Era (1870–1920) (characterized by more intensive settlement and agricultural practices, increased oil extraction, and rising competition for water), and the current Industrial Era (1920–present) (which led to even more intensive development and utilization of natural resources throughout the watershed). The most direct indications of the influences of water diversions and land development are the large reductions in SCRE extent and the loss of freshwater wetlands and salt marshes that once characterized the SCRE (Section 6.2). Although current conditions in the SCRE generally correspond to a freshwater system, water quality changes associated with historical development and water diversion from the Santa Clara River prior to the completion of the VWRF likely included the development of increasingly saline

[8] The ERL (Effects Range Low) and ERM (Median) sediment quality guidelines (Long and Morgan, 1990) are the 10th and 50th percentiles, respectively, on an ordered list of concentrations in sediment found in the literature that co-occur with any biological effect. It should be noted that the ERL should not be interpreted as a toxicity threshold, but rather "… a concentration below which effects are rarely observed".

conditions within the lagoon portion of the SCRE. Entrix (2002b) provides a review of benthic macroinvertebrate communities in other similarly-sized estuaries within coastal southern California, including: Mugu Lagoon, Malibu Lagoon, Santa Margarita Estuary, Batiquitos Lagoon, San Dieguito Lagoon, Los Pensaquitos Lagoon, and the Tijuana Estuary. Although there are large differences in the morphology of these estuaries and water quality conditions, the SCRE includes a greater proportion of freshwater species under current conditions, indicative of the freshwater influence of the current VWRF discharge regime.

5.5 Trophic Status Assessment

Based upon observations of high algal concentrations during the past ten years (2000–2010) and recent (2009–2010) data indicating extended periods of DO below Basin Plan criteria (Section 5.1), there is a strong indication that the SCRE is currently impacted by excess nutrient inputs. Eutrophication in estuaries has traditionally been assessed using freshwater approaches such as the Carlson (1977) Trophic State Index (TSI), which is based upon relationships between water column transparency (Secchi depth), algal biomass (chlorophyll-a), and nutrients (total phosphorus). More recently, however, it has been recognized that eutrophication in estuaries is not solely controlled by nutrients such that high and low nutrient concentrations do not necessarily indicate the presence or absence of eutrophic conditions (Bricker et al. 2003). For this reason, we have evaluated the trophic state of the SCRE using functional comparisons to other estuaries based upon an approach adapted from the United States National Estuarine Eutrophication Assessment (NEEA) (Bricker et al. 1999). The NEEA relies upon the assessment of sixteen factors that may be broadly grouped into changes in algal conditions, nutrients, DO and ecosystem community response. Table 5-13 provides a subset of these factors with established threshold values that were assessed from available for this study.

Table 5-13. Eutrophication potential associations with water quality parameters.

Parameter	Low potential	Moderate potential	High potential	Hypereutrophic
Chlorophyll-a (ug/L)	0-5	5-20	20-60	>60
TIN (mg/L)	<0.1	0.1-1.0	>1	
PO_4 (mg/L)	0-0.01	0.01-0.1	>0.1	
DO (mg/L)	>5	5-2	2-0	

Adapted from Bricker et al. (1999)

In addition to the abundance of suspended algae—which is assessed by chlorophyll-a as an indicator—the NEEA used a number of measures of algal conditions including water clarity (e.g., Secchi depth and suspended sediments), the frequency of occurrence and duration of blooms, and coverage of epiphytes and macro-algae such as *Enteromorpha and Ulva spp*. Other qualitative measures include losses in submerged aquatic vegetation (SAV), shifts in planktonic and benthic community structure (e.g., changes in abundance and diversity relative to undisturbed conditions). However, the dynamic nature of the SCRE, the lack of suitable reference sites for comparison, and the short duration of monitoring conducted for this study limits the applicability of these measures. Table 5-14 presents the NEEA metrics with established threshold values using long-term averages (2000–2010) at open water sites for chlorophyll-a and TIN (sum of inorganic nitrogen including NH_4^+, NO_3^-, and NO_2^-) (Tables 5-7 through 5-10), DO (Tables 5-2 though 5-5), and 2009–2010 averages for PO_4 (Appendix B).

Table 5-14. Trophic status assessment based upon observations of algal conditions, nutrients, and DO at open water sites in the SCRE.

Season	Mouth status	Chlorophyll-a (ug/L)	TIN (mg-N/L)	PO_4 (mg-P/L)	DO (mg/L)
Winter	Open	20–80	3.4–4.6	0.5–2.5	1–21
		High Potential	High Potential	High Potential	High Potential
	Closed	40–80	5.7–8.9	0.1–5.4	0.2–22.1
		High Potential	High Potential	High Potential	High Potential
Spring	Open	10–40	3.5–4.6	0.1–6.4	0.1–21
		High Potential	High Potential	High Potential	High Potential
	Closed	30–70	5.0–7.7	0.2–10.3	0.1–27
		High Potential	High Potential	High Potential	High Potential
Summer	Open	20–40	2.9–3.7	0.1–1.4	0–21
		High Potential	High Potential	High Potential	High Potential
	Closed	100–110	2.5–3.9	0.1–8.5	0.3–23.3
		Hyper-eutrophic	High Potential	High Potential	High Potential
Fall	Open	10–50	2.6–7.0	0.3–3.0	0.9–24.5
		High Potential	High Potential	High Potential	High Potential
	Closed	80–110	4.3–7.5	0.1–9.3	0.5–21.2
		Hyper-eutrophic	High Potential	High Potential	High Potential

Long-term averages of chlorophyll-a, nutrients, and DO suggest highly eutrophic conditions in the SCRE (Table5-14), generally higher for closed-mouth conditions and in summer and fall when algal growth potential is at its highest. High TIN levels (which are dominated by NO_3) along with PO_4 levels accompany large variations in DO (Table 5-14) during diel monitoring conducted in 2009–2010 (Figure 5-6).

Although historical nutrient conditions in the SCRE likely did not approach the nutrient levels shown in Table 5-14, or even the background levels established for the Southern and Central California Chaparral and Oak Woodlands ecoregion (EPA 2001), the degree to which the VWRF facility has contributed to changes in historical water quality cannot be readily determined. Algal populations typically go through large cycles and the balance of several limiting factors (nutrients, light, depth, temperature) underlies observed blooms and die-offs in the population (Horne and Goldman 1995). In terms of nutrient resources, the ratio of nitrogen to phosphorus has been used as an indication of nutrient balance in aquatic systems based upon observations by Redfield (1958), who found a consistent elemental composition of ocean algae species approaching 106:16:1 (molar) ratios for carbon, nitrogen, and phosphorus (C:N:P). Conceptually, the use of this "Redfield ratio" as an indication of nutrient limitation is based upon the assumption that proximate elemental composition in algae reflects the nutrient composition in the surrounding water column and that deviations from the Redfield ratio indicate an over- or under-abundance (i.e., nutrient limitation) of one nutrient or another. Practically, a molar N:P ratio of 16:1 corresponds to a mass ratio of approximately 7.2:1 and indicates a balance of algal nutrients

in aquatic ecosystems on a theoretical basis, with the transition from nitrogen to phosphorus limitation at TIN:PO_4 mass ratios suggested by Boynton et al. (1982) below 10:1 molar (4.5:1 by mass) and above 20:1 molar (9:1 by mass) ratios, respectively. Using the Redfield ratio estimates above as a guide, Table 5-15 presents TIN:PO_4 ratios for the SCRE during 2009–2010 using data collected for this study (Appendix B).

Table 5-15. Average mass ratio of nitrogen to phosphorus (TIN:PO4) in the SCRE by season and mouth closure status (2009–2010).

Season	Mouth	Near mouth	Lower estuary	Middle estuary	Upper estuary	Riverine	Lower outfall	Upper outfall
Winter	Open	6.1	6.1		16.6		4.2	
	Closed	17	14.5	8.4	15.1	190	10.7	7.1
	Avg.	14.4	12.5	8.4	15.4	190	9.2	7.1
Spring	Open	19.5	16.9		74.7		15.6	
	Closed	6.4	16.8	9.4	33.5	196.7	8.1	7.5
	Avg.	14.8	16.9	9.4	59.9	196.7	12.9	7.5
Summer	Open	12.2	52.4		26.3		25.4	
	Closed	6.8	16.3		8.6		9.5	
	Avg.	9.4	32.7		16.8		17.1	
Fall	Open	17.4	15.2	11.3	17.5	11.8	18.9	24.5
	Closed	19.8	11.2		19.5		42.7	
	Avg.	18.4	13.5	11.3	18.3	11.8	29.1	24.5

Overall, Table 5-15 suggests that the SCRE is phosphorus limited for almost all locations, seasons, and under both open- and closed-mouth conditions. This has important management implications in relation to this study in that measurable reductions of algal biomass in the SCRE may not occur until the TIN:PO_4 ratio approaches 4.5:1 by mass. Since PO_4 levels ranged from 0.3–1.0 mg-P/L at open water sites during 2009–2010 under closed-mouth conditions, TIN levels would need to be below 1.5–4.5 mg-N/L in the SCRE to effectively limit algal growth. Several VWRF discharge and onsite treatment alternatives developed in Section 11 will reduce TIN levels in the SCRE. However, since typical saturation conditions for algal growth (200 ug/L nitrogen and 20–80 ug/L for PO_4 as cited by Horne and Goldman [1994]) are well below levels typically found in riverine samples at the inlet of the SCRE (Tables 5-7 through 5-10), even aggressive discharge limits for NO_3 may not reliably limit algal growth in the SCRE at all times of year.

5.6 Nutrient Balance for the Santa Clara River Estuary

As a means of understanding the relative contributions of nutrients from the local watershed (VWRF and upstream sources) under current or alternative future condtions, we have developed a simplified nutrient balance for the SCRE. A mass balance of algal nutrients such as nitrogen (NH_4, NO_3+NO_2, PO_4) may be used to account for the mass or stock flows of these nutrients using reactor theory as:

Accumulation rate = Input rate – Output rate ± generation (decay) rate

In order to model this mass balance in the SCRE, we assume that the total of all material inflows and outflows balance over the course of a day (i.e., mass in equals mass out), and further assume

that the lagoon is well mixed due to the shallow SCRE lagoon depth and consistent onshore winds. Although there are a number of internal transfers between inorganic and organic nutrients, sediment water exchanges, and atmospheric exchanges, this latter assumption of complete mixing over a relatively short daily time scale means that the concentration in the mixed lagoon is the same as in the outflow (e.g., to the ocean or shallow groundwater). This is represented in the differential expression below:

$$\frac{dC_E V_E}{dt} = Q_R C_R + Q_{VWRF} C_{VWRF} + Q_G C_G + Q_O C_O - \left(Q_R + Q_{VWRF} + Q_G + Q_O\right) C_E - Rxn$$

Here the volumetric flows (Q) and mass concentrations (C) refer to the lower Santa Clara River (R), VWRF, shallow groundwater (G), Estuary (E), and tidal exchanges from the ocean (O). A separate reaction term (*Rxn*) may be included to represent mass removal by physical processes other than advective flow transport (e.g., sedimentation, burial, and volatilization) such as biologically mediated processes for nutrients (e.g., algal uptake, nitrification/denitrification). Since the compiled water balance from October 2009 through July 2010 (Section 4.2) shows that the volume of the SCRE is non-steady for large portions of the 2009–2010 monitoring period, we can not use a steady assumption for water quality conditions, and including the reaction term, the relationship above simplifies to the ratio of the net mass loadings to the summation of flow and volume change in the SCRE:

$$C_E \approx \frac{\sum_{i=1}^{n} Q_i C_i - Rxn}{\sum_{i=1}^{n} Q_i + \frac{dV_E}{dt}}$$

Initially, we present results assuming no reactions, which is valid for conservative elements such as salts, and for nutrients when rates of biological removal are low. Based on these assumptions, the relationship above shows that equilibrium concentrations in the SCRE will be largely determined by the relative magnitude of the contributing flows, principally Santa Clara River flow and tidal exchanges relative to VWRF flows. As a means of testing this modeling assumption, available flow data from the compiled water balance and water quality data for the Santa Clara River, VWRF, and Estuary Mouth sites were used to construct mass loading and outflow estimates for salts (using Conductivity as a proxy), total inorganic nitrogen (TIN), and phosphate (PO_4).

Table 5-16. Predicted and observed conductivity and nutrient concentrations for 2009–2010 sampling period.

Sampling date	Estuary (measured)			Estuary (predicted)			Prediction error		
	Cond (mS/cm)	TIN (mg-N/L)	PO_4 (mg-P/L)	Cond (mS/cm)	TIN (mg-N/L)	PO_4 (mg-P/L)	Cond (%)	TIN (%)	PO_4 (%)
9/1/2009	NA	NA	NA	NA	NA	NA	NA	NA	NA
11/17/2009	10.1	11.3	1.1	30.4	11.0	1.6	-70%	-3%	-48%
1/27/2009	13.8	2.2	0.1	13.3	2.6	0.3	-4%	17%	302%
3/24/2010	7.0	8.9	0.5	3.0	9.1	1.0	-57%	2%	118%
5/27/2010	4.4	5.8	0.2	2.4	15.3	1.9	-46%	164%	972%
7/27/2010	2.5	3.2	0.4	2.3	15.5	2.1	-5%	390%	439%
10/12/2010	NA	NA	NA	NA	NA	NA	NA	NA	NA

Extending the mixing model assumptions to salinity resulted in underestimates of measured conductivity during the 2009–2010 sampling period and it is clear that this approximation does not fully capture ocean exchanges (e.g., partial breaches, wave overwash) during open-mouth conditions. Observed salinity was generally higher than modeled (Table 5-16), with close agreement on only two dates, January 27, 2010 during open-mouth conditions, and July 27, 2010 during closed-mouth conditions.

The modeling approach used here generally resulted in overestimates of observed nutrient levels, suggesting that biological uptake should be included in the assessment of future conditions. For TIN, over-estimates occurred during spring and summer conditions (Table 5-16), perhaps due to unaccounted algal uptake or denitrification occurring in the estuary. Estimates of areal reaction rates for TIN uptake needed to account for the discrepancy between modeled and observed TIN levels for the May 27, 2010 and July 27, 2010 events would be on the order of 400–500 mg-N/m^2-day, within rates that have been observed in other systems (Seitzinger 1988, as cited by Horne 1995). The apparent overestimates in PO_4 levels would also be reduced through inclusion of removal mechanisms such as algal uptake and sedimentation.

Based upon the discussion above, it is apparent the SCRE is not well mixed at all times and biological uptake of nutrients is occurring during periods of algal blooms or due to a combination of suspended and epiphytic algal growth and denitrification in the SCRE. The modeling approach used here allows for the assessment of the relative magnitude of nutrient loadings to the SCRE from contributing sources. However, it should be noted that the dynamic nature of the SCRE may result in large departures of estimated concentrations from actual conditions due to a number of factors related to hydraulics (e.g., rapid changes in SCR flows, ocean exchanges), mixing (e.g., wind, salinity), and seasonal algal growth dynamics discussed above. Nevertheless, the approach appears to capture seasonal and flow-related changes well enough to represent relative changes in nutrient levels associated with alternative discharge and treatment scenarios developed in later sections of this report. Lastly, it should be noted that under almost all conditions, nutrient levels rapidly approach concentrations arriving from the largest flow components of the water balance (such as VWRF flows under closed-mouth conditions and SCR and/or ocean flows under open-mouth conditions).

6 VEGETATION

This section provides a description of contemporary and historical vegetation conditions including overall changes in vegetation. General vegetation/habitat types currently comprising the SCRE in order of dominance include riparian, foredune, open water, open beach, disturbed, freshwater wetland, developed, upland, and aquatic vegetation. Comparing current vegetation conditions with historical records, there have been several key changes that are indicative of some of the primary impacts to the Santa Clara River and the SCRE including a loss of riparian shrublands and woodlands, changes in freshwater wetland distribution, and invasion by non-native plant species.

6.1 Contemporary Vegetation Conditions

Prior to the vegetation mapping and classification conducted specifically for the City of Ventura in 2009 as part of this study, vegetation conditions around the SCRE had been mapped and described in 1989 (Swanson et al. 1990), 2002 (ESA 2003), and 2005 (Stillwater Sciences and URS Corporation 2007). In addition, California Department of Pesticide Regulation surveyed a number of vegetation transects in the McGrath State Beach and McGrath Lake areas in 2007/2008. Together, these mapping efforts provide a reliable indication of contemporary riparian and wetland vegetation conditions around the SCRE, and a description of the inter-annual variation that can occur.

The following description of vegetation conditions in and around the SCRE is based primarily on the 2009 vegetation mapping effort, which mapped 202 ha (500 ac) of vegetation, 54 ha (134 ac) of disturbed and/or developed lands associated with McGrath State Beach campground, related facilities, and access roads, 51 ha (127 ac) of open water (the SCRE was filled at the time of the 2009 vegetation mapping), 38 ha (95 ac) of open beach, and 4 ha (10 ac) of unidentified aquatic vegetation in McGrath Lake. This mapping is presented in Figure 6-1 and summarized in Table 6-1. Appendix D is a comprehensive list of the plant species observed during the vegetation mapping effort.

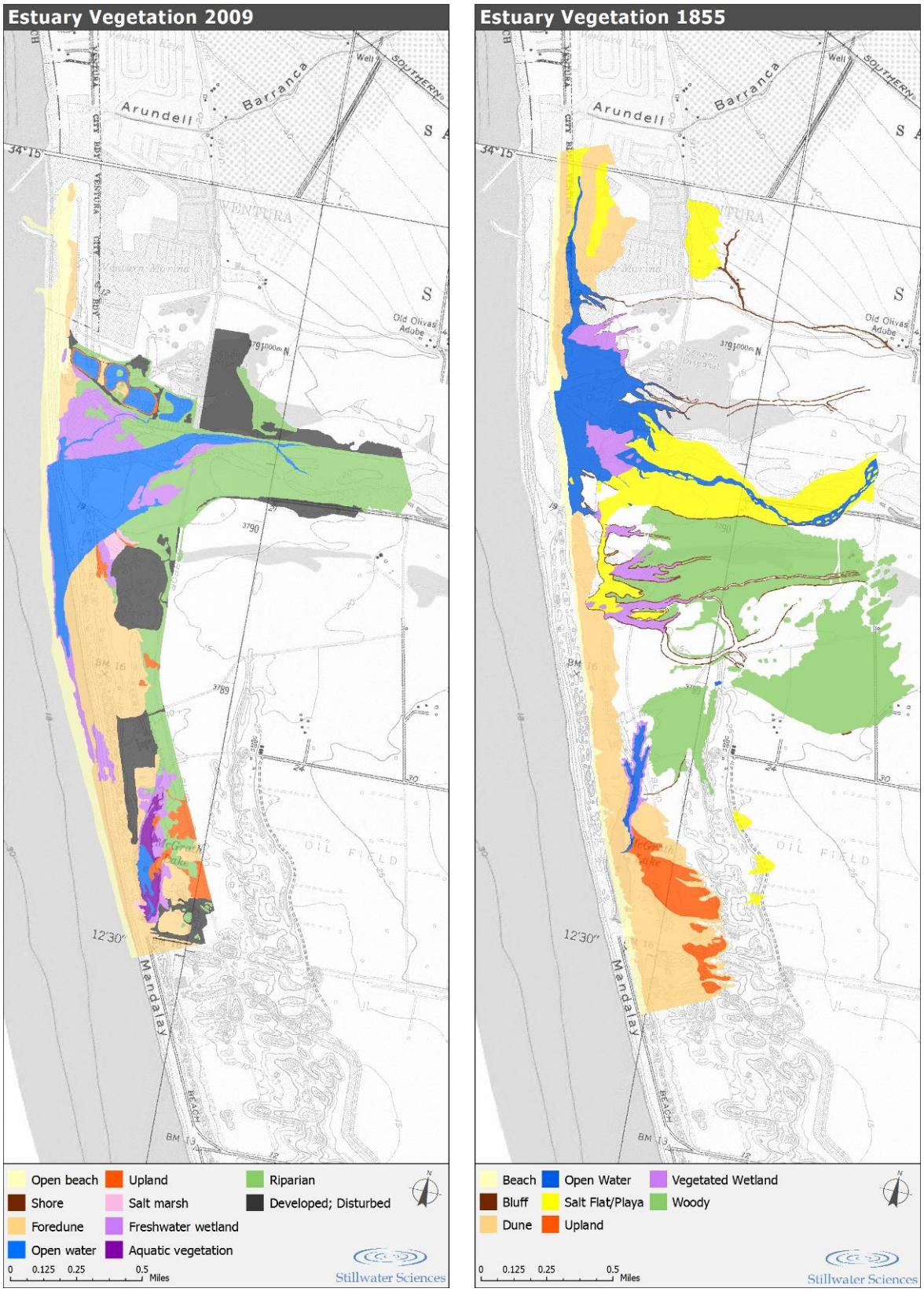


Figure 6-1. SCRE and adjacent floodplain vegetation in 2009 (Stillwater Sciences 2009) and 1855 (Grossinger et al. 2011).

Table 6-1. Vegetation mapping units around the SCRE in 2009.

General vegetation type	General vegetation type (acres)	Map unit	Classification level[1]	Vegetation form[2]	Map unit (acres)
Foredune	175.6	*Abronia* spp. - *Ambrosia chamissonis*	Alliance	Herbaceous	127.7
		Carpobrotus spp.	Semi-Natural Stands	Herbaceous	47.8
Freshwater wetland	69.4	*Arundo donax*	Semi-Natural Stands	Herbaceous	16.9
		Persicaria hydropiperoides–Melilotus alba	Provisional Alliance	Herbaceous	1.2
		Schoenoplectus californicus	Alliance	Herbaceous	40.1
		Typha domingensis	Alliance	Herbaceous	9.3
		Typha latifolia	Alliance	Herbaceous	2.0
Riparian	225.4	*Baccharis salicifolia*	Alliance	Shrubland	23.7
		Riverwash	Super Alliance	Herbaceous	1.6
		Salix lasiolepis	Alliance	Shrubland	28.4
				Woodland	155.6
		Salix lucida	Alliance	Woodland	16.0
Salt marsh	4.7	*Argentina egedii*	Alliance	Herbaceous	1.3
		Distichlis spicata	Alliance	Herbaceous	1.9
		Leymus triticoides	Alliance	Herbaceous	1.5
Upland	24.9	*Ambrosia psilostachya*	Provisional Alliance	Herbaceous	2.1
		Baccharis pilularis	Alliance	Shrubland	11.8
		Myoporum laetum	Semi-Natural Stands	Shrubland	2.6
		Non-native grasses and forbs	Provisional Alliance	Herbaceous	7.1
		Urtica dioica	Provisional Alliance	Herbaceous	1.1
Aquatic vegetation			Map unit	NA	10.3
Developed			Map unit	NA	65.5
Disturbed			Map unit	NA	68.6
Open beach			Map unit	NA	95.2
Open water			Map unit	NA	126.7
Total					**866.3**

[1] Vegetation classification follows the second edition of *A Manual of California Vegetation* (Sawyer et al. 2009).
[2] Herbaceous vegetation types are dominated by annual or perennial aquatics, forbs, or grasses and generally lack above-ground woody stems. Shrublands are dominated by shrubs—short woody plants commonly with two or more stems. Woodlands are dominated by trees—tall woody plants commonly with one stem, or trunk.

Overall, the composition and extent of upland and tidal vegetation mapped in 2009 is very similar to or the same as that mapped in 1989 (Swanson et al. 1990), 2002 (ESA 2003), and 2005 (Stillwater Sciences and URS Corporation 2007). All four mapping efforts describe:

1. a strip of predominately native foredune vegetation along the coastal strand on both sides of the SCRE, with some large stands of non-native invasive iceplant (*Carpobrotus* spp.);

2. a ring of freshwater wetland vegetation in and around the perimeter of the SCRE that appears to die back during winter in response to the draining of the SCRE, more saline water conditions when the sandbar is breached, and/or natural seasonal die back;
3. several patches of salt marsh vegetation on the south edge of the SCRE, adjacent to McGrath State Beach (this vegetation is likely maintained by the northeast trending levee north of the campground [Swanson et al. 1990]);
4. a corridor of riparian vegetation of varying successional stages on the left and right banks of the upstream end of the SCRE; and
5. large, dense patches of non-native invasive giant reed (*Arundo donax*) in the perimeter wetlands, and high percent cover of giant reed in portions of the riparian corridor.

The USFWS concluded that the vegetation present in the SCRE was typical of other southern California river mouth estuaries (Greenwald et al. 1999).

Some small discrepancies occur between the three vegetation maps. In a few instances, alliance names were changed to conform to recent changes in plant species nomenclature (e.g., *Potentilla anserina* to *Argentina egedii*, and *Scirpus* spp. to *Schoenoplectus californicus*) and the vegetation alliance and association names in the recently published second edition of *A Manual of California Vegetation* (Sawyer et al. 2009) (e.g., Mixed Willow Forest to *Salix lucida* Woodland). (Swanson et al. [1990] classified the SCRE by wetland type, but the plant species compositions they recorded conform to those of the later mapping efforts.) In other instances, the composition of vegetation mapping units did not change, but the boundaries of the mapping units were modified slightly based on the field mapping or photo-interpretation of the USDA (2009) orthophotography.

The primary, substantive changes in contemporary vegetation composition or distribution occurred on the lower floodplain of the Santa Clara River, upstream of the Harbor Blvd. Bridge. In this area, there has been a shift in vegetation successional stage, reflecting the recovery of riparian vegetation from the 2005 flood. The vast majority of this area was mapped as Riverwash Herbaceous Super-Alliance in 2006, defined by Stillwater Sciences and URS Corporation (2007) as:

> The majority of the area mapped under this super-alliance was bare riverwash following the large floods in January and February 2005. At the time of the field surveys in summer 2005 and 2006, these areas were typically dominated by white sweetclover (*Melilotus alba*), generally ranging from 10–35% cover. In many areas, seedlings of willows (*Salix exigua, S. laevigata, S. lucida* ssp. *lasiandra,* and *S. lasiolepis*) and mulefat (*Baccharis salicifolia*), and occasionally Fremont cottonwood (*Populus fremontii*), were common (at up to 10% cover). In addition, low levels of giant reed (*Arundo donax*) were often present (at up to 10% cover). This suggests that the dominance of these stands by herbaceous vegetation is likely to be short-lived, with transformation into willow-dominated or *Arundo*-dominated stands likely occurring within a few years unless another large flood scours out the established vegetation and resets these areas.

This predicted transformation was demonstrated in 2009: floodplain areas previously mapped as Riverwash Herbaceous Super-Alliance have transitioned to shrub-dominated *Salix lasiolepis* and *Baccharis salicifolia* alliances (Stillwater Sciences 2009). Repeat sampling by USFWS documented a similar pattern of succession following winter flows in 1998 and 1999, with

recolonization of the scoured river channel by April of the following year (Greenwald et al. 1999).

In their description of the Riverwash Herbaceous Super-Alliance, Stillwater Sciences and URS Corporation (2007) noted that:

> Many stands have wetter margins near the summer low flow channel that support a greater diversity of herbaceous species more common in the Floodplain Wetland Super-Alliance, including barnyard grass (*Echinochloa crus-galli)*, sprangletop (*Leptochloa uninervia*), knotweed (*Polygonum* spp. [now *Persicaria* spp.]), annual rabbitsfoot grass (*Polypogon monspeliensis*), watercress (*Rorippa nasturtium-aquaticum* [now *Nasturtium officinale*]), and water speedwell (*Veronica anagallis-aquatica*).

This same high-diversity herbaceous vegetation along the low-flow channel margin was also observed in 2009, but was generally separated into discrete mapping units such as the *Schoenoplectus californicus*, *Typha domingensis*, *Typha latifolia*, and *Persicaria hydropiperoides-Melilotus alba* herbaceous alliances (Table 6-1).

Free-floating or rooted floating-leaved aquatic vegetation is uncommon in the SCRE. None of the contemporary vegetation mapping efforts noted any significant stands of aquatic vegetation. Approximately 4 ha (10 ac) of aquatic vegetation has been mapped in McGrath Lake (Figure 6-1, Table 6-1), but the species composition was not identified (ESA 2003, Stillwater Sciences 2009). Watercress, which is farmed farther upstream and is a component of the Riverwash Herbaceous Super-Alliance and occurs in a few small patches within the SCRE, was the only aquatic plant species observed in 2009 and 2010 (Stillwater Sciences 2009; Stillwater Sciences, field observation, 2010). During repeat surveys of the SCRE from 1997–1999, USFWS found that aquatic vegetation was sparse and observed only three aquatic plant species. Sparse duckweed (*Lemna* sp.) was documented on one occasion on the north edge of the SCRE, upstream of the effluent discharge channel, and a few clumps of ditch-grass (*Ruppia* sp.) were observed twice at the end of the effluent discharge channel (Greenwald et al. 1999). Floating and submerged green algae (*Enteromorpha* sp.) was, and continues to be, the only notable aquatic vegetation, with the highest percent covers documented by USFWS on the south edge of the SCRE, closest to McGrath State Beach campground (Greenwald et al. 1999, Stillwater Sciences, field observation, 2010). While aquatic vegetation distribution is likely influenced by a number of factors, it is hypothesized that high turbidity levels during the growing season may suppress submersed aquatic vegetation establishment and growth in the SCRE.

6.2 Historical Vegetation Conditions

Historical conditions in and around the SCRE were summarized by Swanson et al. (1990) and Nautilus Environmental (2005) and have been mapped and assessed in detail by the San Francisco Estuary Institute (SFEI), the Southern California Coastal Water Research Project (SCCWRP), and California State University Northridge (CSU) as a part of the Ventura and South Coast Wetlands Historical Ecology Project (Grossinger et al. 2011). All of these efforts interpreted and/or digitized the 1855 USC&GS topographic map (t-sheet) (#683) covering the lower Santa Clara River as a means of understanding historical conditions in and around the SCRE. The Historical Ecology Project map, which is more complete and has been geo-referenced, is presented in Figure 6-1 and summarized in Table 6-2.

Table 6-2. Vegetation mapping units around the SCRE in 1855.

General vegetation type	Acres
Beach	42.5
Bluff	33.7
Dune	225.3
Open Water	117.7
Salt Flat/Playa	168.0
Upland	41.0
Vegetated Wetland	47.7
Woody	313.4
Total	989.20

Source: Grossinger et al. 2011

The 1855 map, like the contemporary vegetation maps, showed a strip of dune along the coastal strand on both sides of the SCRE. To the north of the SCRE, which was located farther north at the time, backwater extended for nearly 1 mi behind the sand dunes and patches of vegetated wetland and salt flat/playa occurred (Figure 6-1). To the south, a number of side-channels or sloughs, relicts of abandoned river channels, supported vegetated wetlands and created what is now McGrath Lake. While the Grossinger et al. (2011) interpretation of the 1855 map does not identify any salt-marsh vegetation specifically (Figure 6-1)[9], Swanson et al. (1990) found that vegetation and topographic patterns were suggestive of salt marsh and Nautilus Environmental (2005) identified approximately 6 ha (16 ac) of salt marsh associated with the side-channels, although they ultimately concluded that there was little potential for salt marsh in the SCRE. Large swaths of woody vegetation occurred behind the dunes, paralleling the active Santa Clara River channel and several abandoned channels (Figure 6-1). A large expanse of salt flat/playa is mapped just upstream of the SCRE and bordering the river channel on both sides, but based on this distribution is more likely to be recently scoured, unvegetated floodplain. The more typical distribution of salt flats/playas as isolated features along the coast can be seen to the north and south of the SCRE (Figure 6-1). Patches of upland vegetation occurred south of what is now McGrath Lake (Figure 6-1).

Aerial photography from 1927 and 1938 supports this interpretation of the 1855 map. While vegetation had already been cleared and farming occurred on much of the floodplain, particularly to the south of the channel, no substantive levees had yet been built along the lower Santa Clara River at the time these photographs were taken. Multiple, braided channels are visible, both within the wide, sandy mainstem channel and on the floodplains. A large tract of riparian vegetation, ½-mi wide in some places, occurs on the northern floodplain. Nautilus Environmental (2005) noted that salt marsh was not evident during this time period, neither in aerial photographs from 1929 nor a USGS map from 1933, supporting the conclusion that there has been little potential for salt marsh in the SCRE. In their analysis of a 1957 aerial photograph, Swanson et al. 1990 noted that at the time that the Harbor Blvd. Bridge had been constructed, the Santa Clara River was generally confined to its present width along with the northeast trending levee at McGrath State Beach that protects and maintains the existing salt marsh vegetation. By 1978,

[9] It is worth noting that the Salt Flat/Playa cover type in the Grossinger et al. (2011) map is not likely to be salt marsh; its distribution suggests that it is recently scoured, unvegetated floodplain, and the more typical distribution of salt flats/playas as isolated features along the coast can be seen to the north and south of the estuary (Figure 3-51).

when the next available set of aerial photographs had been taken, levees had been completed on both sides of the SCRE and both floodplains were either developed or being farmed.

While a one-to-one comparison between these historical sources and 2009 vegetation conditions is not feasible since detailed vegetation types are not apparent and/or the mapping efforts covered different areas, there have been several key changes in vegetation between these time periods that are indicative of some of the primary impacts to the Santa Clara River and the SCRE. These include: (1) a loss of riparian shrublands and woodlands; (2) changes in freshwater wetland distribution; and (3) invasion by non-native plant species, particularly giant reed and iceplant. Other physical changes in the SCRE since 1855, such as the reduction in estuary size and loss of backwater-channels, were described previously in Section 3.2.

6.2.1 Loss of riparian shrublands and woodlands

The formation and maintenance of riparian shrublands and woodlands in the SCRE area are strongly influenced by the dynamic patterns of flooding, scour, and sediment deposition exhibited by the lower Santa Clara River (Stillwater Sciences and URS Corporation 2007). Prior to high-flow events, flow is conveyed primarily in a single low-flow channel and vegetation heavily encroaches into the active channel (Stillwater Sciences 2006). During a high-flow event, the channel widens and forms a primarily single meandering channel that scours away nearly all vegetation in the active channel (Stillwater Sciences 2006). Following the high-flow event, flows recede into a braided channel and vegetation re-established in the active channel, although the active channel may now have a dramatically different plan-form than before (Stillwater Sciences 2006, Greenwald et al. 1999).

Historically, perennial stream flow, higher channel and groundwater elevations, and an unconfined floodplain likely supported a more-or-less continuous and broad riparian forest in the vicinity of the SCRE (a portion of which is shown in Figure 6-1). It is likely that, prior to water diversion and groundwater pumping, many rivers in southern California experienced perennial stream flow except in the driest years and that the channel would have supported higher groundwater elevations and more frequent floodplain inundation under the historical flow regime (Boughten et al. 2006). These channel conditions would have facilitated the recruitment and establishment of large tracts of riparian vegetation. Simons, Li & Associates (1983) report that the Santa Clara River floodplain was, historically, as much as 3.2 km (2 mi) wide in its lowermost reaches. The riparian area likely supported dense, multi-stored stands of broadleaf trees, including *Populus* (cottonwood), *Platanus* (sycamore), and *Salix* spp. (willows) that rely on shallow groundwater. Based on a review of numerous historical resources, Grossinger et al. (2011) conclude that the "west grove" near the SCRE was one of the three largest stands of riparian forest along the lower Santa Clara River.

Stillwater Sciences (2007) calculated a 70% reduction in the size of the riparian corridor in the lower 11 km (7 mi) of the river, based on flood extent in 1938 and 2005, while Nautilus Environmental (2005) calculated a 91% reduction in willow riparian woodlands around the SCRE, based on vegetation in 1929 and 2002. Currently, levees and other channel constrictions limit the lateral extent of flooding and reduce the area where riparian vegetation can become established and maintained (Stillwater Sciences and URS Corporation 2007). In addition, historical accounts of central and southern California coastal rivers describe extensive efforts to clear the "monte"—the cottonwood and willow forests that covered the lower reaches of the large rivers (Boughton et al. 2006). Vegetation clearing occurred primarily to prepare the land for ranching and farming and for fuel. More recently, urban development has begun replacing the farmland that replaced riparian forests. Major changes in land use and hydrology in the Santa

Clara watershed upstream of the SCRE, along with increasing demands on ground and surface water for agricultural and urban uses, have lowered the groundwater table, beyond the reach of tree roots in many areas, and have contributed to increased flood intensity (Stillwater Sciences and URS Corporation 2007). Although the water table immediately adjacent to the SCRE is influenced by SCRE water levels, vegetation dominated by species capable of growing in areas with little access to the groundwater and/or colonizing quickly after high intensity flood events, such as the riverwash herbaceous alliance and giant reed, have increased in extent relative to historical conditions (Stillwater Sciences and URS Corporation 2007).

In addition to decreasing the overall extent of riparian vegetation, these impacts have narrowed the riparian corridor and eliminated the transition zone between SCRE vegetation and upland land uses (Figure 6-1). Narrow riparian corridors without a transition zone provide lower quality habitat, with increased introduction of non-native species and greater edge effects for riparian species, and fail to perform ecosystem services such as filtering runoff and preventing soil erosion (Gregory et al. 1991, Naiman and Decamps 1997, Patten 1998, Groffman et al. 2003). The significant loss of riparian vegetation and transition zone in the lower Santa Clara River equates with a similarly significant loss in habitat quality, quantity and ecosystem functioning.

6.2.2 Changes in freshwater wetland distribution

Comparison of the 1855 and 2009 vegetation maps suggests that the overall extent (i.e., total acreage) of freshwater wetland vegetation in the SCRE area has not changed dramatically (Tables 6-1 and 6-2), but that the way it is distributed in the landscape has (Figure 6-1). In 1855 wetland vegetation, and potentially salt marsh (Nautilus Environmental 2005), was found in the backwater areas, side-channels, and sloughs of the lower Santa Clara River (Figure 6-1). By 1927, with the exception of McGrath Lake, the vegetation in these areas had changed; wetland vegetation and a small amount of salt-marsh now occur as a ring around the perimeter of the SCRE and McGrath Lake (Figure 6-1). It is likely that changes in hydrology due to the reductions in instream flow and lower groundwater elevations and grading to prepare the land for agriculture all contributed to the elimination of these features. The ramifications of this change in freshwater wetland distribution on aquatic and terrestrial wildlife are discussed in Sections 7 and 8.

6.2.3 Invasion by non-native plant species

Giant reed and iceplant, both aggressive non-native invasive plants, have become widespread in and around the SCRE. Over 29 ha (50 ac) of iceplant are mapped in the foredunes both north and south of the SCRE, and it is a frequent co-occurring species in other mapped dune vegetation types (Stillwater Sciences 2009). Iceplant, native to South Africa, is a mat-forming or trailing low-growing perennial, with long succulent leaves and bright yellow or pink flowers. Iceplant around the SCRE was likely intentionally planted to stabilize dunes near the harbor or escaped cultivation (it is extensively planted along highways as an ornamental and has escaped cultivation in many coastal areas). Once established, iceplant spreads vegetatively and thrives in dune communities. Beach and/or harbor maintenance and recreational activities have likely contributed to the spread of iceplant in the dunes around the SCRE, by more broadly dispersing viable fragments of the plant. Iceplant displaces native dune species, and large infestations change the ecology of the community by increasing soil organic matter; in time, increased amount of organic matter in the sandy soil can promote invasion by other weed species that otherwise would not be able to inhabit dune soils (DiTomaso and Healy 2007). Iceplant traps more sand than native dune species and generally stabilizes dune communities unnaturally (DiTomaso and Healy 2007).

Over 6.5 ha (16 ac) of giant reed occurs in the SCRE, primarily as freshwater wetland vegetation along the northern perimeter (Table 6-1), and it is a dominant or co-occurring species in much of the riparian shrublands and woodlands in the area (Stillwater Sciences 2009). As early as the late 1700s, giant reed was often planted in California for erosion control and as windbreaks (DiTomaso and Healy 2007), although it was not established in the SCRE until after 1929 (Nautilus Environmental 2005). In the Santa Clara River watershed, giant reed was likely cultivated to produce reeds for woodwind instruments and used as an ornamental landscaping plant. It propagates vegetatively, most often from rhizomes that wash downstream from eroded banks (DiTomoso and Healy 2007). As such, even if giant reed was not intentionally planted around the SCRE, it would have easily spread from upstream infestations. Until effectively treated, these infestations, of which there are many throughout the Santa Clara River watershed, will continue to supply viable vegetative fragments of giant reed to the SCRE. Giant reed is a highly aggressive member of the grass family that invades riparian zones by establishing dense, monospecific clonal stands. Although it is shade-tolerant, giant reed most often colonizes disturbed areas where native vegetation has been cleared; on the Santa Clara River, this is frequently revetted banks and recently scoured floodplains. Giant reed is a strong competitor in systems with increased nutrient supply, and heavy fertilizer use may be an important factor aiding its dominance in many California riparian systems (Coffman 2007). Dense stands typically develop, which often displace native vegetation, diminish wildlife habitat, and increase flooding and siltation in natural areas (DiTomoso and Healy 2007). Giant reed is also readily flammable, and has been shown on the Santa Clara River to increase the susceptibility of the riparian corridor to fire and to re-sprout vigorously after fire by quickly exploiting released nutrients (Coffman et al. 2010). Large stands of giant reed can significantly increase surface and groundwater loss in semiarid regions due to a high evapotranspiration rate, which is estimated to be approximately three times higher than that of native riparian species (DiTomoso and Healy 2007). While recognizing the continued supply of giant reed propagules from upstream infestations, USFWS determined that giant reed treatment in the SCRE would be an ecological benefit to native vegetation (Greenwald et al. 1999). Recently, giant reed and iceplant, have been the subject of volunteer removal efforts around the SCRE.

7 AQUATIC HABITATS AND FOCAL SPECIES

This section describes current and historical aquatic habitat conditions. Species' status, distribution, life history information, and habitat requirements in the SCRE are discussed for the two focal fish species, steelhead and tidewater goby.

While little is know regarding steelhead use of SCRE habitats, it is known that the SCRE provides a migratory corridor for upstream adult steelhead spawners and outmigrant smolts as well as valuable rearing habitat. For this reason, water quality conditions and habitat availability may have an important influence on anadromous fish survival. Adult migratory steelhead generally need relatively cool water temperatures and minimum depth and maximum velocity requirements for successful upstream migration. Smolts—which may migrate downstream to estuaries in late spring or rear in streams for up to four years as juveniles before outmigrating to the estuary and ocean—generally require cool temperatures, DO concentrations near saturation, low turbidity, and ample cover for growth and maturation.

Tidewater goby are the most common and widespread fish species in the SCRE, and appear to use the backwater habitat found in the VWRF outfall channel (Nautilus Environmental 2005). Tidewater goby live near the bottom of stable, shallow, lagoon or off-channel habitats with brackish water. Coarse sand is the species' preferred substrate for burrow-building and egg-laying. Tidewater gobies can tolerate a fairly broad range of water temperatures, levels of DO, and salinities (though generally prefer water with relatively low salinities).

7.1 Contemporary Aquatic Habitat Conditions

Aquatic habitat types present in the SCRE include open water, freshwater wetlands and a minimal amount of salt marsh (Section 6.1). Depending upon river flows and mouth closure status, the largest aquatic habitat type is open water and backwaters characterized by shallow sand-bedded flats with scattered emergent vegetation (Nautilus Environmental 2005, Stillwater Sciences 2010). Larger patches of emergent vegetation are found around the margins of the estuary, especially along the left (southern) bank and in the vicinity of the outfall channel on the north side of the lagoon. Very little submerged aquatic vegetation or other instream cover was found within the SCRE during recent surveys (Stillwater Sciences 2010), although low levels were found along the channel margins, increasing in the lowermost reaches of the Santa Clara River upstream of Harbor Blvd. Substrate found in the SCRE is predominantly sand and silts although some patches of coarser substrate are found in bar complexes in the lower reaches of the Santa Clara River (Section 3.1.2).

Current and historical assessments of water quality conditions important to aquatic species are summarized in Section 5. The strongest influences on water quality conditions during open-mouth conditions are open-mouth flows (particularly in winter and spring, and corresponding to high river flows and tidal inflow), and the strongest influence on water quality during closed-mouth conditions is the VWRF (particularly in late summer and/or fall, and corresponding to low river flows). On an annual basis, the SCRE has varying salinities as result of: (1) continued freshwater inflow from the VWRF into a backwater area, (2) varying inflow levels from Santa Clara River with natural inflow from less than 0.03 cfs approximately 60% of most years (Entrix, Inc. 2004), and (3) periodic marine influence from breaches of the sand berm.

7.1.1 Benthic Macroinvertebrates

Reviews of the BMI communities in other similarly-sized estuaries within coastal southern California by Entrix (2002b) as well as direct sampling in the SCRE (ABC Laboratories 2004–2009) shows large variations in community structure, with the SCRE BMI assemblage varying substantially within and among years, largely in response to changes in water levels, salinity, substrate, and disturbance associated annual seasonal changes in river flows and the opening and closing of the SCRE. Monitoring since 1997 indicated that the BMI community is composed of organisms common to freshwater or estuarine environments, and tends to be dominated by three or four species that are tolerant of the naturally dynamic salinity and sediment conditions in the SCRE in most years and seasons (USFWS 1999, Entrix 2002b, ABC Laboratories 2004–2009; see Appendix B for more detail). In particular, species of midge larvae (Diptera: *Chironomidae*), oligochaete worms, copepods, ostracods, and amphipods have tended to dominate during the 1997–2008 monitoring period reviewed for this report, although the taxonomic composition of the dominant species varies within years (i.e. seasonally) and among years. Weather patterns and associated river flows (dry, normal or wet years), combined with timing and duration of open versus closed estuarine conditions affect key habitat conditions (e.g., salinity, substrate, water depth and velocity), which in turn appear to be the primary drivers of invertebrate composition and distribution in estuaries (Kennish 1986, Chapman and Want 2001 [both as cited in ABC Laboratories 2009]).

The most unique BMI assemblage documented by ABC Laboratories during monitoring from 2003–2008 occurred in May 2005 when a variety of opportunistic aquatic insect taxa dominated that differed from the more typical suite of tolerant taxa documented in every other year and in Fall 2005. This shift in BMI composition likely occurred in response to the high flows and scour that occurred in the main channel and lagoon habitats following the large winter floods the preceding winter. Aerial photographs from before and after the flood indicate that main channel and lagoon habitats in the estuary were highly disturbed or reset, while the vegetated wetlands associated with the VWRF outflow channel were noticeably less disturbed and remained largely intact. By fall of 2005, benthic conditions in the SCRE had begun recovering and the BMI community was similar to that observed in previous years, although abundance still appeared depressed (ABC Laboratories 2006). BMI communities from 2006 on were similar to the pre-2005 communities (ABC Laboratories 2007–2009). This provides further evidence that the SCRE BMI community is adapted to the harsh conditions found in this dynamic environment.

Performance of commonly accepted BMI multimetric indices have not been demonstrated as reliable indicators of estuarine community condition (Ranasinghe et al 2009). Within the SCRE BMI monitoring data, comparisons among sites and years are further complicated due to changes in sampling methods and variability in environmental conditions. However, the weight of evidence to date indicates that the VWRF effluent is not adversely affecting BMI populations in the SCRE (ABC Laboratories 2009). As evidenced following the large floods in January and February 2005, the location of the outflow channel, combined with its relatively consistent flows and associated inundated vegetated wetland habitat, likely creates an important refuge area for at least some BMI and fish populations during large episodic floods that occur approximately every 10–15 years during the wettest El Niño years (Section 4.1.1).

7.1.2 Native and Introduced Fish Species

Using data collected as part of the City's *Resident Species Study* (Entrix 2002b) and other sources, recent monitoring in the SCRE has documented seventeen fish species, including 12 native and 5 non-native species (Table 7-1). This variety of fishes includes freshwater species,

euryhaline species (species adapted to variable salinity conditions found in estuarine habitats), stenohaline species (species that move into the estuary opportunistically as part of their life cycle), and saltwater fishes (Entrix, Inc. 2004, Nautilus Environmental 2009, USFWS 1999, Stillwater Sciences, unpubl. data) (Table 7-1). The most abundant fish species present include native tidewater goby, arroyo chub (*Gila orcutti*), and staghorn sculpin (*Leptocottus armatus*) as well as numerous non-native species including mosquitofish (*Gambusia affinis*), fathead minnow (*Pimephales promelas*), Green sunfish (*Lepomis cyanellus*), and common carp (*Cyprinus carpio*). Four species found in the estuary for at least a portion of their life history are federally listed, including tidewater goby, steelhead, unarmored threespine stickleback (*Gasterosteus aculaeatus williamsoni),* and Santa Ana sucker (*Catostomus santaanae*).

Of the fish species found in the SCRE, most prefer lower salinities (i.e., freshwater), but can tolerate higher salinities for shorter periods of time (Table 7-1). Because the salinity levels within the SCRE vary seasonally and spatially, many of these fishes are expected to find refuge in the estuary. Four native species also use the estuary for migratory or opportunistic purposes, including two anadromous fishes (steelhead and Pacific lamprey), and two marine species, (topsmelt and striped mullet). Topsmelt and striped mullet may enter the estuary as larva, which may explain why there have been several years with no observations of these fish, yet others where they were abundant. Several stranded *O. mykiss* were identified following a mechanical breaching event in October 2010 (Cardno/Entrix 2010). Although test results regarding anadromy were not available at the time of this report, these results suggest that the SCRE lagoon currently supports suitable habitat for steelhead rearing.

Table 7-1. Fish species qualitative abundance within the SCRE, 1997–2010.[1]

Species	Origin	Occurrence	Goby predator (yes / no)	Preferred salinity range (tolerance)	Listed species status
tidewater goby (*Eucyclogobius newberryi)*	native	abundant	—	0–12 ppt (<28 ppt)	Federally Endangered
arroyo chub (*Gila orcutti*)	introduced[2]	abundant	no	freshwater	CA Species of Special Concern
staghorn sculpin (*Leptocottus armatus*)	native	common	yes	0–67 ppt	—
mosquitofish (*Gambusia affinis*)	introduced	common	yes (larval)	0–25 ppt (<58 ppt)	—
fathead minnow (*Pimephales promelas*)	introduced	common	yes (larval)	freshwater	—
green sunfish (*Lepomis cyanellus*)	introduced	common	yes	freshwater (<2 ppt)	—
common carp (*Cyprinus carpio*)	introduced	common	disruptive	0–12 ppt (<16 ppt)	—
topsmelt (*Atherinops affinis*) [3]	native	common (marine visitor)	no	0–34 ppt (<80 ppt)	—
striped mullet (*Mugil cephalus*) [3]	native	common (marine visitor)	no	0–75 ppt	—
southern steelhead DPS (*Oncorhynchus mykiss*)	native	migratory (rearing unknown)	yes	anadromous	Federally Endangered
Pacific lamprey (*Lampetra tridentata*)	native	migratory (rearing unknown)	no	anadromous	—
prickly sculpin (*Cottus asper*)	native	uncommon	yes	fresh–seawater	—
arrow goby (*Clevelandia ios)*	native	uncommon	not expected	marine / tidal	—
unarmored threespine stickleback (*Gasterosteus aculeatus williamsoni)*	native	uncommon	no	fresh–marine	Federally Endangered
Santa Ana sucker (*Catostomus santaanae*)	introduced[2]	uncommon	no	freshwater	Federally Threatened
yellowfin goby (*Acanthogobius flavimanus*) [4]	introduced	uncommon	yes	5–40 ppt (0–40 ppt)	—
California killifish (*Fundulus parvipinnis*)	native	uncommon	no	0–33.8 ppt (<128 ppt)	—

[1] Sources: Entrix, Inc. 2004, Stillwater 2010, Moyle 2002, USFWS 1999.
[2] Otherwise native, but considered introduced in the Santa Clara River drainage (Swift et al. 1993).
[3] Topsmelt and striped mullet may enter the estuary as larva, which may explain why there have been several years of no observations, yet others where they were abundant.
[4] Last observed in 1997.

7.1.3 Inter-Specific Interactions

Inter- and intraspecific competition in southern California estuaries has not been well studied. Emmett (1997) indicates that research on the feeding habits of juvenile salmonids in Northwest estuaries indicates a high diet overlap between different species, but little conclusive data indicating that competition is occurring (McCabe et al. 1983). Although increased competition may result from the introduction of exotic species which can alter estuarine food webs, the frequent disturbance of the SCRE prevents the establishment of invasive clams (e.g., *Potamocorbula amurensis*) found to have altered food webs of some west coast estuaries (Kimmerer et al 1994). The presence of some freshwater adapted species in the SCRE (Table 7-1) that either prey upon larval or adult Tidewater goby (e.g. fathead minnow, green sunfish, prickly sculpin) or are disruptive (e.g., common carp) suggests that some degree of competition or direct predation by introduced species may be occurring.

7.2 Focal Species

Steelhead and tidewater goby are the two fish species selected to represent aquatic focal species for the Estuary Subwatershed Study. Information regarding their life history and habitat requirements is described below with an emphasis on estuarine residency. For tidewater goby, this includes all life stages; for steelhead, this includes estuarine rearing and adult and juvenile migration.

7.2.1 Steelhead

7.2.1.1 Status and distribution

Steelhead found in the Santa Clara River belong to the southern California steelhead Distinct Population Segment (DPS), which extends from the Santa Maria River in San Luis Obispo County to the U.S-Mexico border (NMFS 2006). This DPS is listed as threatened under the federal Endangered Species Act (NMFS 2006). Steelhead is the term used to denote the anadromous life-history form of rainbow trout (*O. mykiss*); because both anadromous and resident *O. mykiss* are found within the watershed, the term *O. mykiss* is used in situations where distinguishing juvenile steelhead from resident rainbow trout would be problematic. Preservation of both life-history forms is considered a high priority in the *Southern Steelhead Recovery Plan* (NMFS 2009). Steelhead occur throughout the North Pacific Ocean and spawn in freshwater streams along the west coast of North America. Historically, steelhead 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, only an estimated 500 adult steelhead remain in the Southern California DPS north of Malibu Creek, California, although 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).

Under historical conditions (before 1946) within the Santa Clara River watershed, steelhead likely spawned and reared in the major tributaries west of the Piru Creek confluence (Kelley 2004, Harrison et al. 2006). These tributaries included primarily Sespe and Piru creeks, although Santa Paula and Hopper creeks also likely provided significant steelhead habitat. The present-day distribution of steelhead in the Santa Clara River watershed is limited by a number of migration barriers that restrict upstream passage of adults, both in the lower mainstem river and most major tributaries (Titus et al. in press).

7.2.1.2 Life history

The relationship between anadromous and resident forms of *O. mykiss* is the subject of ongoing research. The two forms are capable of interbreeding and current evidence suggests that, under some conditions, either type can produce offspring that exhibit the alternate form (i.e., resident rainbow trout can produce anadromous progeny and vice-versa) (Burgner et al. 1992, Hallock 1989, Donohoe et al. 2008, Zimmerman et al. 2009). However, in some watersheds, the two types are distinct (e.g., Pearse et al. 2009).

Based on variability in life history timing, steelhead are broadly categorized into winter and summer reproductive ecotypes. Steelhead return to spawn in their natal stream, usually in their third or fourth year of life (Shapovalov and Taft 1954, Behnke 1992). Only the winter ecotype (winter-run) occurs in Santa Clara River watershed. Winter-run steelhead generally enter spawning streams from late fall through spring as sexually mature adults following the first storm flows (a.k.a. "freshets") of the winter, with males typically returning to fresh water earlier than females (Shapovalov and Taft 1954, Behnke 1992, Busby et al. 1996). Spawning can occur from late fall into spring, with peak spawning activity between January and March, based upon other steelhead populations in the southern California steelhead DPS (Busby et al. 1996). Adult steelhead are known to stray from their natal streams to spawn in nearby streams and, in more hydrologically variable streams of the central and southern California coast, straying is often more prevalent than in more northern streams (Clemento et al. 2009, Pearse et al. 2009).

Within Scott Creek (Santa Cruz County, CA), competition for space within upstream freshwater habitats has been attributed to three possible life history strategies based upon usage of lagoon and stream rearing habitat: stream rearing, lagoon rearing, and combination stream and lagoon rearing (Hayes et al. 2008). Stream-reared steelhead spend up to four years in the stream, and then outmigrate to the ocean with minimal lagoon residence (Shapovalov and Taft 1954). Lagoon-reared steelhead spend only a few months in the stream before migrating to the lagoon where they will rear for typically 1 year. Steelhead exhibiting the combination strategy will rear for 1 to 2 years in the stream and 1 to 10 months in the lagoon before emigrating to the ocean. Conditions for growth can be very good in lagoons relative to stream habitat, and thus fish in lagoons tend to achieve a larger size-at-age then their stream-reared counterparts (Smith 1990, Hayes et al. 2008). Since larger smolts tend to have higher ocean survival, growth during lagoon rearing may increase ocean survival of steelhead smolts. Lagoon systems, therefore, can provide a potential demographic boost in two ways. First, lagoons may relax to some degree the density-dependent bottleneck occurring in stream habitat. Second, lagoons provide an adjustment to a saline environment, which may increase smolt sizes and consequently improve ocean survival for both stream-reared and lagoon-reared fish. Smith (1990) concluded that even tiny lagoons unsuitable for summer rearing can contribute to the maintenance of steelhead populations by providing feeding areas during winter or spring smolt outmigration.

Although algae, aquatic vegetation, and detritus support an abundance of aquatic invertebrates consumed by steelhead and other juvenile salmonids, the diet of steelhead rearing in southern California lagoons and estuaries has not been well studied. A variety of vegetation, substrates, and salinities may be associated with different invertebrate assemblages used as food (Simenstad and Cordell 2000). Shallow intertidal and inner portions of estuaries typically support rich populations of benthic invertebrates that juvenile salmonids feed upon (McCabe et al. 1983, Healey 1982), while pelagic organisms are more important as prey in deeper estuaries (Healey 1982). In many estuaries and lagoons, gammarid amphipods (e.g., *Gammarus*, *Corophium*, *Eogammarus, Anisogammarus* spp.) and chironomid midge larvae are the most abundant invertebrates and they have been found to make up a large proportion of the diet of steelhead

(Needham 1939, Bond 2006) and are likely to be important food resources in ephemerally closed lagoons in California estuaries similar to the SCRE. These prey items have been well documented in the SCRE, although relative abundance appears to vary significantly over space and time (ABC Laboratories 2004–2009). Further study would be required to document whether food resources might be a limiting factor for a lagoon-rearing life-history strategy of juvenile steelhead in the SCRE.

Timing of smolt outmigration may depend on when adequate flow conditions are present to open the streams and lagoons to the ocean, since sandbars build up and seal off many confluences, including the SCRE, in low flow conditions (Stoecker 2002). In Santa Rosa Creek to the north, outmigrant trapping suggests that some individuals rear in upstream reaches before outmigrating as smolt, and some outmigrate to the lower reaches and lagoon at smaller sizes/younger ages (Nelson et al. 2009). 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 lack of high quality nearshore habitats in this portion of their range (NMFS 1996).

7.2.1.3 Habitat requirements

Although there is limited specific 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). Relatively cool water temperatures (between 10° and 15°C [50 and 59°F]) are preferred by adults, although they are relatively tolerant of elevated water temperatures (NMFS 1997) and may survive temperatures as high as 27°C (81°F) for short periods (Moyle et al. 1989). Optimal temperatures for growth of steelhead juveniles has been reported at a range of 15–19°C (59–66.2°F) (Myrick and Cech 2005), though the highest growth rates were observed at temperatures between 15–24°C (59–75.2°F) in the Scott Creek estuary along the central California coast (Hayes et al. 2008). Although potential temperature-related threats were characterized using lethal temperature threshold considerations, sublethal bioenergetic effects of increased water temperatures may limit growth of steelhead (Stillwater Sciences and URS Corporation 2007), and consequently survival in the ocean environment. Lastly, DO concentrations near saturation are generally required for steelhead growth, they can survive with oxygen concentrations as low as 1.5–2.0 mg/L (Moyle 2002).

Adult migratory steelhead need water with a minimum depth of 18 cm (0.6 ft) for successful upstream migration with velocities below 240 cm/s (8 ft/s) (Thompson 1972). In riverine settings, Waite and Barnhart (1992) suggested that steelhead fry occupy a restricted range of depths and low velocities near stream banks, with smolts showing a greater preference for open water habitats and deeper water (Cavallo et al. 2003). Woody debris, overhanging vegetation, man-made objects, and overhanging vegetation provide some degree of cover for juvenile life stages regardless of water level. Although woody debris, man-made objects, and overhanging vegetation provide some degree of cover for juvenile life stages, Kelley (2008) found very limited cover within the SCRE and suggested other potential threats to steelhead juveniles in the SCRE, including high water temperatures and high turbidity.

7.2.1.4 Habitat use within the Santa Clara River Estuary

Both anadromous and resident *O. mykiss* occur within the Santa Clara River watershed (Stoecker 2002, Stillwater Sciences and URS Corporation 2007), although detailed information on the relative proportion of each type is not available. Little data on steelhead spawning timing exists for the Santa Clara River, although both spawning timing and distribution within the basin is

related to timing, frequency, and duration of sandbar opening and winter flow conditions. Adult steelhead occurrence in the SCRE will be limited to periods when the estuary is open, and are expected to use it as a migration corridor to the upper watershed as soon as water depth in the river allows. Timing of smolt outmigration also depends on when adequate flow conditions are present to connect the estuary to the ocean.

Juvenile southern steelhead may rear for extended periods within upstream freshwater habitats of the Santa Clara River depending upon seasonal variations in rainfall that control the extent of wetted channel and connectivity from the SCRE to upstream locations (Stoecker and Kelley 2005). Steelhead in the San Gregorio and Scott Creek lagoon's to the north use a rearing strategy that includes 1–2 yrs of residency in upstream habitats followed by lagoon-rearing on the order of weeks to months before emigrating to the ocean. Although closed-mouth conditions may force extended periods of lagoon rearing, little is known regarding habitat use within the SCRE. Although no steelhead have been documented in routine seining surveys conducted by the City (Nautilus Environmental 2009), several stranded *O. mykiss* in the size range of 10–12 inches were identified following a mechanical breaching event in October 2010 (Cardno/Entrix 2010), suggesting the SCRE lagoon currently supports suitable habitat for steelhead rearing.

Under open-mouthed conditions in the SCRE, steelhead likely use the lagoon habitat primarily as a migratory corridor with only short periods of residency. Surveys of Santa Clara River steelhead smolts captured at the Vern Freeman Diversion dam further support this hypothesis. As part of a larger study of southern California steelhead outmigration survival, movement, and straying, Kelley (2008) captured 133 steelhead smolts upstream of the Freeman Diversion Dam and tagged 81 of them with acoustic and PIT tags prior to release in the SCRE. Acoustic receivers were placed in the nearshore ocean outside of the SCRE in order to detect fish that reached the ocean. Under open-mouth conditions, Kelley (2008) found that the majority of these steelhead smolts spent less than 3 days in the SCRE before exiting to the ocean and half of the smolts migrated to the ocean within 2 days of release.

From the studies above, it has been shown that the SCRE may potentially provide valuable rearing habitat for juvenile steelhead, for at least short periods of time, particularly when water quality conditions are favorable. Since habitat availability under open-mouth conditions is determined by tidal stage, we developed a set of habitat availability curves based on depth, cover, and water quality criteria (Figure 7-1). We then used these curves to map "potential" steelhead habitat in the SCRE under closed-mouth conditions when the SCRE is full (Figure 7-2). We assumed that steelhead will utilize open water areas (primarily used for adult migration) if depths are greater than 1 m (3.0 ft), and assumed juveniles will utilize shallow water (defined as 0.15–1 m [0.5–3.0 ft]) for rearing provided there is sufficient vegetative cover.

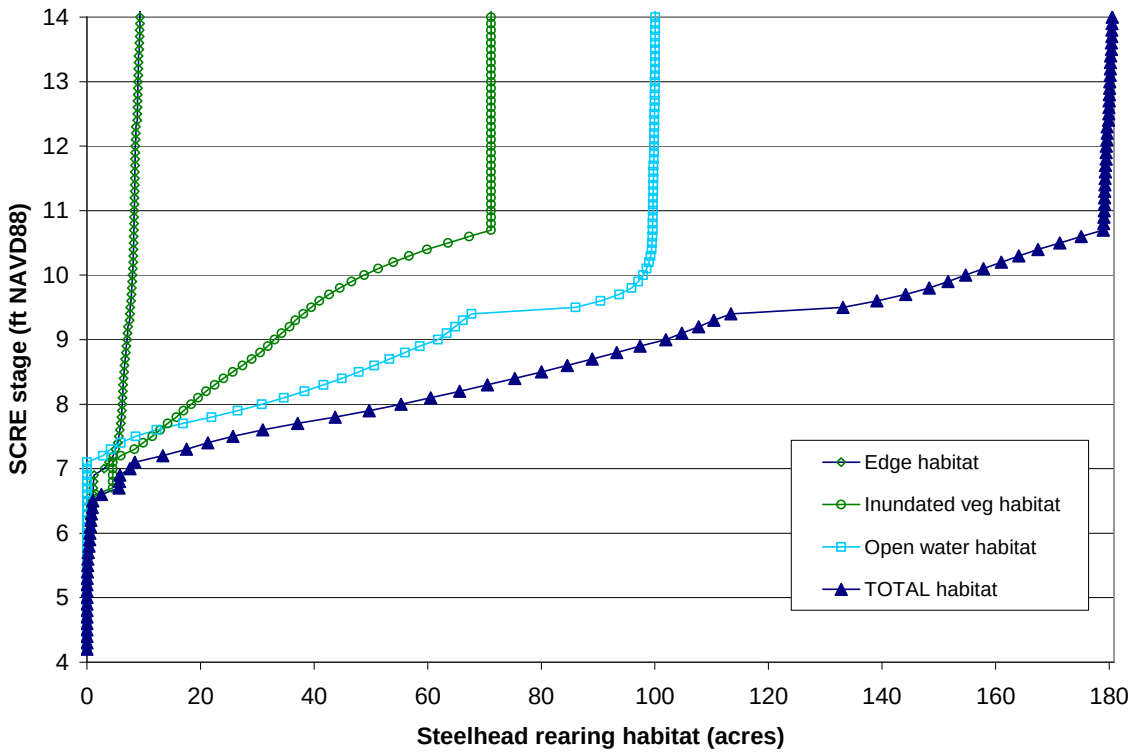


Figure 7-1. Estimate of suitable steelhead habitat as a function of SCRE stage.

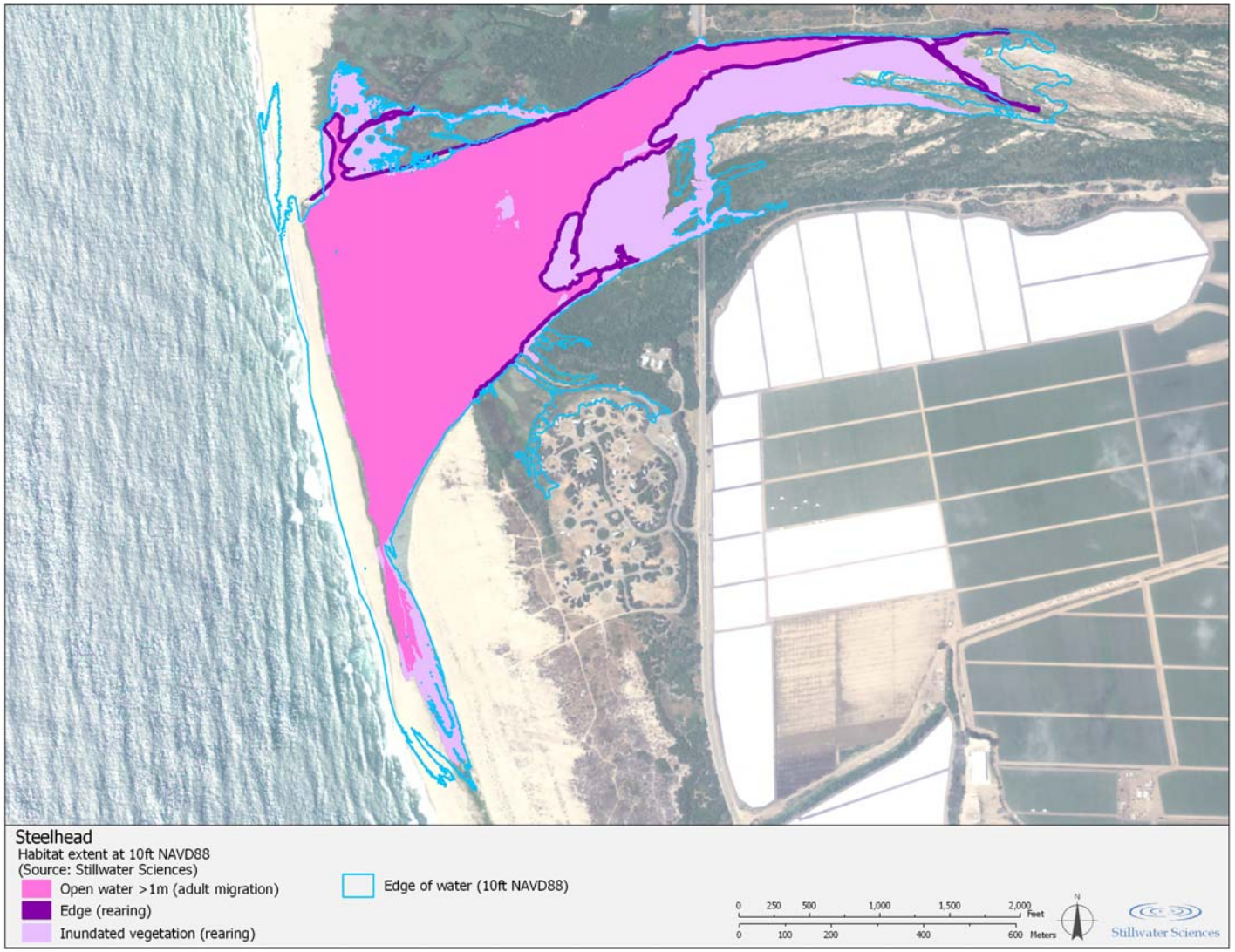


Figure 7-2. Spatial extent of suitable steelhead habitat for the SCRE under full conditions (approximately 10 ft NAVD88).

7.2.2 Tidewater goby

7.2.2.1 Status and distribution

Tidewater goby are an estuarine/lagoon adapted species that are endemic to California coast, mainly in small lagoons and near stream mouths in the uppermost brackish portion of larger bays (Moyle 2002, USFWS 2005). Tidewater goby historically occurred in at least 134 localities along the California Coast, in coastal lagoons, marshes, and estuaries from Tillas Slough in the Smith River of Del Norte County to Agua Hedionda Lagoon in San Diego County (Moyle 2002, USFWS 2005). The fish still occur within this range, but over half of the populations at these localities are extirpated or extremely small with uncertain long-term persistence (USFWS 2005). The population in the SCRE is federally listed as endangered under the Endangered Species Act (62 FR 43937).

Tidewater gobies often migrate upstream into tributaries, as far as 1.0 km (0.5 mi) from the estuary (SCR Project Steering Committee 1996). However, in the Santa Ynez River, Santa Barbara County, tidewater gobies have been found as far as 5–8 km (3–5 mi) upstream of tidal lagoon areas, sometimes in sections of stream impounded by beavers (*Castor canadensis*) (SCR Project Steering Committee 1996, USFWS 2005).

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 1999). According to the USFWS, 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).

7.2.2.2 Life history

Tidewater goby are a short-lived (generally 1 year) and highly fecund species (females produce 300–500 eggs per batch and spawn multiple times per year) that disperse infrequently via marine habitat but have no dependency on marine habitat for its life cycle (Swift et al. 1989, Lafferty et al. 1999). Tidewater gobies inhabit 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).

Tidewater gobies feed mainly on small animals, usually mysid shrimp (*Mysidopsis bahia*), gamarid amphipods (*Gammarus roeseli*), and aquatic insects, particularly chironomid midge (Diptera: *Chironomidae*) 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 use a variety of prey items in the SCRE and the relative abundance of midges found in BMI samples from the lagoon sediments (ABC Laboratories 2004–2009) suggests that food resources do not currently limit gobies in the SCRE.

Reproduction begins in spring, usually late April or May, and continues into the fall, although usually the greatest numbers of fish are produced in the first half of this time period. The reproduction period is generally associated with the closure and filling of the estuary (late spring–

fall). Breeding occurs in slack shallow waters of seasonally disconnected or tidally muted lagoons, estuaries, and sloughs. Males dig burrows vertically into sand, 100–200 mm [4–8 in] deep, and defend the burrows until hatching (SCR Project Steering Committee 1996). The eggs take approximately 6–10 days to hatch at about 15–25 °C, with tidewater gobies reaching a standard length of approximately 4–5 mm (0.17–0.25 in) (Moyle 2002). Larvae are planktonic (unable to swim freely) for 1–3 days before they become benthic (USFWS 2005) and the larvae apparently spend a day to a few weeks in open water until they reach approximately 16–18 mm (0.6–0.7 in) length, and sink down to the lagoon floor to enter the benthic juvenile lifestage until reaching sexual maturity at 24–27 mm (0.9–1.1 in) (Moyle 2002).

The average size of adult tidewater gobies tends to be significantly larger in marshes (43–45 mm [1.7–1.8 in] 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).

7.2.2.3 Habitat requirements

Gobies require stable lagoon or off-channel habitats, particularly during their relatively short larval stage (Lafferty et al. 1999, Chamberlain 2006). The dominant substrate of the SCRE, coarse sand (Stillwater Sciences 2010), was found to be the preferred substrate of gobies; Swenson (1999) showed experimentally that coarse sand (average diameter of about 0.5 mm) was necessary for the construction of breeding burrows (Entrix, Inc. 2004).

Generally, tidewater gobies are found in areas with water less than 1 m (3.3 ft) deep. Dewatering of burrows is a common threat to gobies under natural conditions. Even if not dewatered, if water depth in the estuary at a burrow stays at 30 cm (12 in) or shallower for any amount of time, clutches of eggs in burrows may not survive since the males guarding them may abandon the nest leaving the burrows to collapse (Swift et al. 1989).

Tidewater gobies have been documented in water with temperatures ranging from 8–25°C [46–77°F]) and salinities that range from 0–41 ppt (Swift et al. 1989, Moyle 2002, Chamberlain 2006). Tidewater gobies have been observed spawning regularly in water temperatures of 17–22°C (62–71°F) and salinities of 8–15 ppt (USFWS 2005), similar to ranges in the SCRE (Section 5, Tables 5-2 through 5-5). Small changes in salinity probably do not prevent successful hatching of the eggs (Entrix, Inc. 2004). Tidewater gobies have also been found over a broad range of DO levels (4–19 mg/l) (Irwin and Soltz 1984 as cited in Chamberlain 2006) similar to ranges found in the SCRE (Section 5, Tables 5-2 through 5-5).

Flood and breaching events can result in dispersal of tidewater gobies between estuarine/lagoon habitats, although survival is likely low and dispersal is limited. Gobies can persist in habitats that wash out under high flows as long as a velocity refuge is present (Moyle 2002, Lafferty et al. 1999). 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). 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).

7.2.2.4 Habitat use within the Santa Clara River Estuary

Tidewater goby represent the most common and widespread fish species in the estuary and have been observed using all parts of the estuary as well as the lower Santa Clara River as far as 3 mi

upstream from the estuary/lagoon, between Ventura and Oxnard (SCR Project Steering Committee 1996, Entrix, Inc. 2004). Ongoing tidewater goby surveys during the spring and fall have found tidewater gobies to be common and the population is stable, although the populations are higher in the fall following the spawning period (Nautilus Environmental 2009).

Tidewater gobies have been observed in the main channel areas of the SCRE and backwater areas, including the VWRF outfall channel, over the past several years of monitoring (Entrix, Inc. 2004). In addition to supplying freshwater marsh habitat, the VWRF discharge channel appears to provide the majority of side channel/ backwater refugia during river peak flow periods as well as stable flows and minimum depths during periods when the lagoon is draining (Nautilus Environmental 2005). Higher densities within the VWRF outfall channel have been attributed to more rapid population increases following high-flow scour events and gobies have been found at all locations within the SCRE (Entrix, Inc. 2004). Although tidewater goby are usually collected in water less than 1 m (3.3 ft) deep, because they have been found at depths greater than this (Worcester 1992, Swift et al. 1997), we have mapped "potential" tidewater goby habitat in the SCRE as occurring within a depth range of 0.3–1.5 m (1.0–4.9 ft). Figure 7-3 shows the tidewater goby habitat availability curve for the SCRE and Figure 7-4 provides an estimate of the spatial extent of suitable habitat under full conditions.

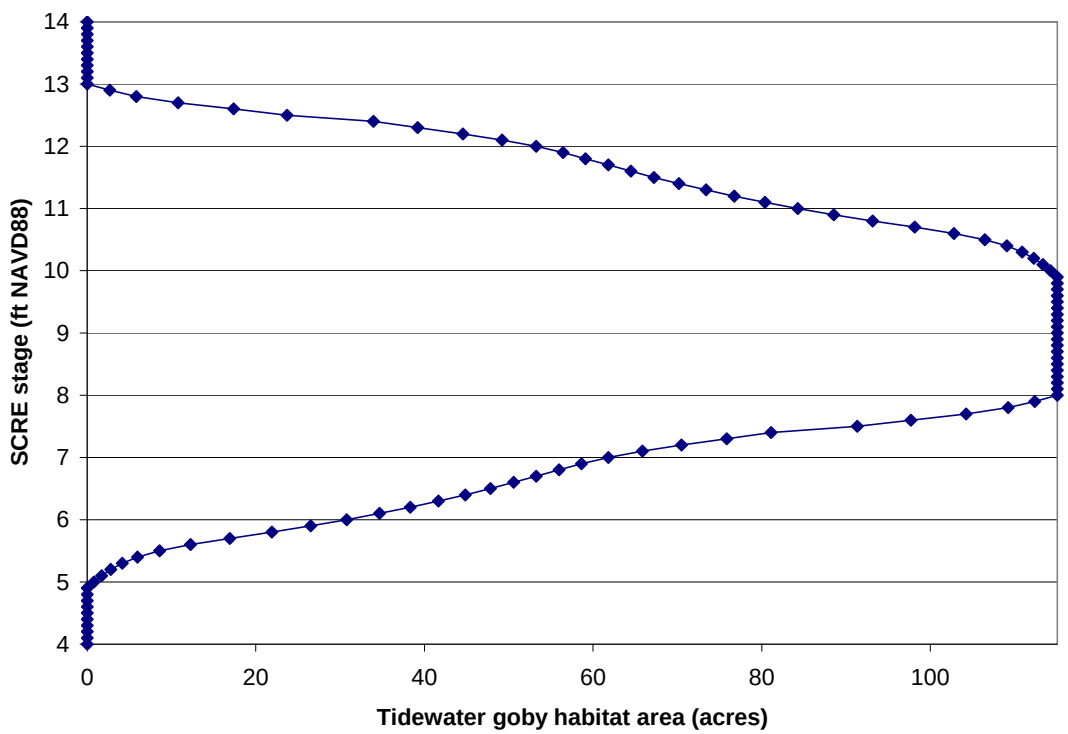


Figure 7-3. Estimate of suitable tidewater goby habitat as of function of SCRE stage.

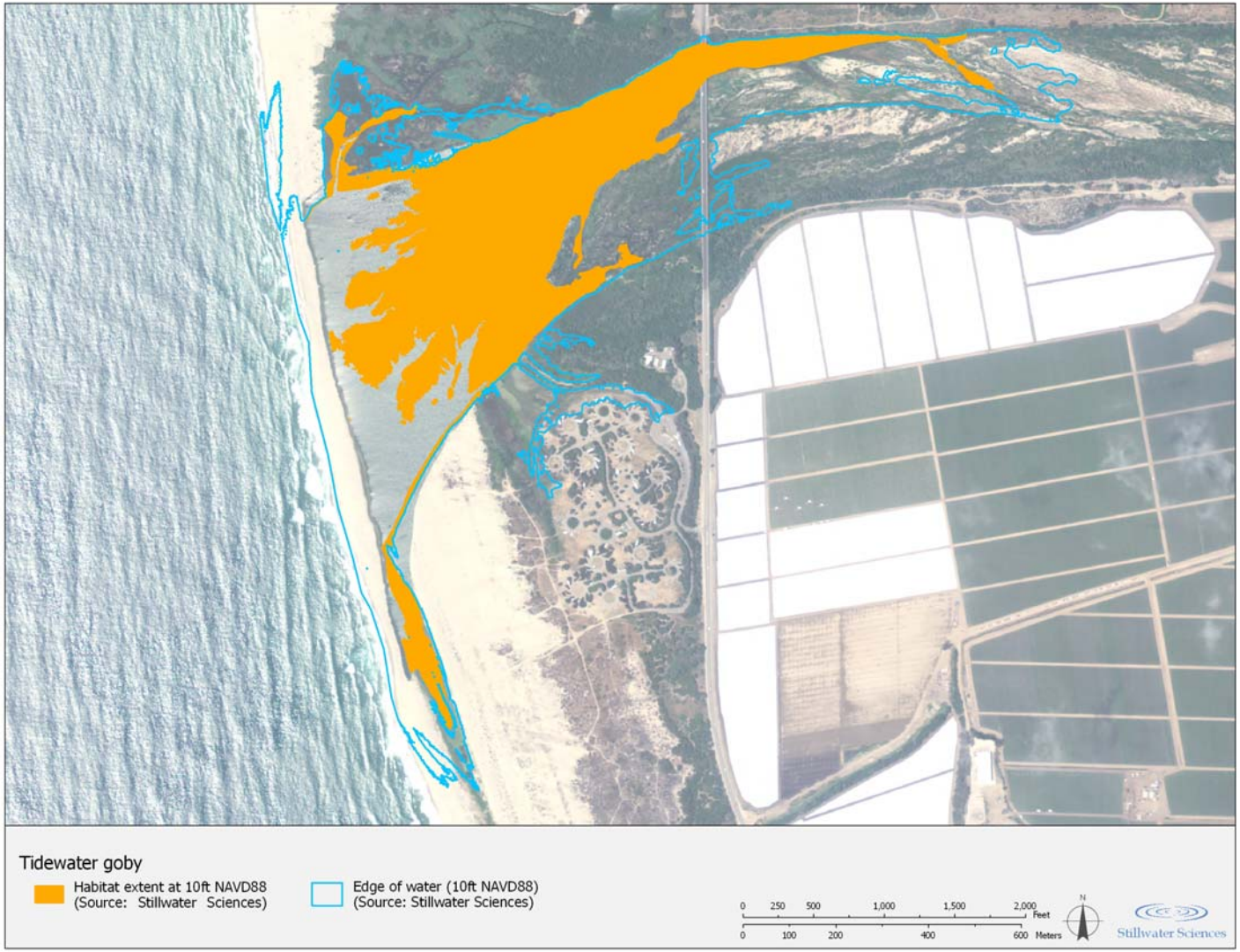


Figure 7-4. Spatial extent of suitable tidewater goby habitat for the SCRE under full conditions (approximately 10 ft NAVD88).

7.3 Historical Aquatic Habitat Conditions

In the recent past along the California coast, many lagoons and waterways such as the SCRE were channelized with levees to accommodate infrastructure, prevent flooding, and to reclaim farmland (USFWS 2005). As described in Section 3.2, channelization of the SCRE resulted in the disconnection of marsh habitat from the main channel, eliminating rearing habitats and also eliminating low-velocity marsh habitat that serves as refugia during high flows. Historically, it is likely that the salinity of the lagoon was more variable, with larger increases in salinity during summer and fall. This has resulted in an increase in freshwater species relative to other estuaries along the California coast. In historical fisheries survey results, it was noted that sampling techniques may have favored smaller fish (Entrix, Inc. 2004, USFWS 1999). Therefore, many of the larger predatory, or disruptive fishes to steelhead or tidewater goby, that are known to exist upstream of the estuary may have been underestimated or missed completely within the estuary.

8 WILDLIFE HABITATS AND FOCAL SPECIES

This section describes contemporary and historical wildlife habitat conditions. Species' status, distribution, life history information, and habitat requirements in the SCRE are discussed for the two avian focal species, western snowy plover and California least tern.

Western snowy plovers are found near the SCRE during both the summer (nesting) and winter seasons, while California least terns are found near the SCRE only during the summer (nesting) season. Open beach and foredune habitats are used by both species for nesting, which takes place on the ground above the high tide line on barren to sparsely-vegetated beaches. Western snowy plover uses the same open beach and foredune habitats foraging, by gleaning for invertebrates such and insects and crustaceans found on the sand, in stranded seaweed on the beach, or from low-growing plants. California least tern forage in aquatic habitats where small bait fish are abundant, including shallow estuaries, lagoons, coastal ponds, or nearshore waters.

8.1 Contemporary Wildlife Habitat Conditions

The SCRE habitats that are of considerable size and of generally high value to wildlife include open beach and foredune, freshwater wetland, riparian, and open water habitats. Situated along the Pacific flyway—an important north-south migration route for many bird species—the SCRE is an especially rich area for bird species. Special-status species mentioned in the sections below include species that are listed as endangered, threatened, or candidate under the federal or California Endangered Species Acts, and species designated as species of Special Concern or Fully Protected by CDFG.

8.1.1 Open beach and foredune

Open beach refers to the strip of sandy substrate that begins at the mean tide line and extends to the top of the foredune (Barbour et al. 2007). The foredune is a vegetated ridge of sand parallel to the beach, just beyond the ordinary high tide line. The surf zone, the part of open beach exposed to wave action, provides habitat for fish, birds, and aquatic invertebrates. Such invertebrates, including bivalves, snails, crabs, amphipods, and marine worms, attract a variety of shorebirds including gulls, sandpipers, dowitchers, and plovers. Insects, reptiles, birds, and small mammals may be observed in the upper beach, the area from the drift line to the base of the foredune, including western fence lizard (*Sceloporus occidentalis*), side-blotched lizard (*Uta stansburiana*), gulls (*Larus* spp.), killdeer (*Charadrius vociferous*), and California ground squirrels (*Spermophilus beecheyi*). Invertebrates found in sand, in organic matter, or from low-growing plants may provide foraging opportunities for species utilizing the upper beach and foredune.

Gulls and shorebirds identified using the open beach and foredune habitats during 2010 bird surveys include willet (*Tringa semipalmata*), whimbrel (*Numenius phaeopus*), western gull (*Larus occidentalis*), and California gull (*Larus californicus*) (Entrix, Inc. 2010 a,b). Other species that may be commonly found using open beach and foredune habitats in the SCRE include black-bellied plover (*Pluvialis squatarola*), semi-palmated plover (*Charadrius semipalmatus*), black-necked stilt (*Himantopus mexicanus*), American avocet (*Recurvirostra americana*), long-billed curlew (*Numenius americanus*), marbled godwit (*Limosa fedoa*), sanderling (*Calidris alba*), western sandpiper (*Calidris mauri*), least sandpiper (*Calidris minutilla*), and dowitchers (*Limnodromus* spp.).

Two special-status bird species are known to nest yearly in the open beach and foredunes near the SCRE: the federally threatened western snowy plover and the federally and state-listed California least tern. These two birds are also the focal species for this study and are discussed in more detail in Section 8.2. California legless lizard (*Anniella pulchra*) and coast horned lizard (*Phrynosoma coronatum*), both state Species of Special Concern, may utilize foredune habitats around the SCRE. California legless lizards prefer loose soil for burrowing, sparse vegetative cover, and require some moisture in the substrate (Jennings and Hayes 1994). They often use surface cover objects such as logs, rocks, or debris (Jennings and Hayes 1994). Coast horned lizards typically prefer open areas with areas of loose soil and an abundance of native ants (Jennings and Hayes 1994).

8.1.2 Freshwater wetland

Emergent freshwater wetland, one of the most productive wildlife habitats in California, offers high-quality wildlife habitat that provides nesting, foraging, roosting, and cover for a variety of species. Emergent freshwater wetland typically contains numerous invertebrates that in turn provide an important food source for other species. Dabbling ducks eat invertebrates, plant material, and seeds in the water. Freshwater wetland vegetation provides nesting substrate for birds; some bird species construct nests directly suspended in tules or cattails (e.g., marsh wren [*Cistothorus palustris*], common yellowthroat [*Geothlypis trichas*], red-winged blackbird [*Agelaius phoeniceus*]) while some construct nests on matted vegetation or mud while concealed behind emergent vegetation (e.g., ducks, rails, and grebes).

Common herpetofaunal species found in the SCRE that use emergent freshwater marsh and edges include California toad (*Bufo boreas halophilus*), Pacific chorus frog (*Pseudacris regilla*), and garter snake (*Thamnophis* spp.). These species also provide a food source for other wildlife including birds and mammals. The non-native African clawed frog (*Xenopus laevis*), ubiquitous in the SCRE (USFWS 1999), has been documented to feed on native amphibians and fish including tidewater goby. Freshwater wetlands provide habitat for four California species of special concern that have potential to occur within the emergent freshwater marsh, though have not been documented in the SCRE: western pond turtle (*Actinemys marmorata*), California red-legged frog (*Rana draytonii*), south coast garter snake (*Thamnophis sirtalis* ssp.) and two-striped garter snake (*Thamnophis hammondii*). Western pond turtles, which have been documented in the upper Santa Clara River watershed near Santa Clarita and in the vicinity of Piru Creek (Stillwater Sciences 2007), prefer permanent, slow-moving fresh or brackish water with basking sites such as logs or mats of vegetation in marsh and open water habitats. California red-legged frog has been recorded upstream along the Santa Clara River but not in the SCRE (Stillwater Sciences 2007). California red-legged frogs prefer still or slow-moving water with emergent and overhanging vegetation, including wetlands, wet meadows, ponds, lakes, and low-gradient, slow moving stream reaches with permanent pools.

Common and uncommon bird species typically associated with emergent freshwater marsh that may be found in the area of the SCRE include rails (e.g., Virginia rail [*Rallus limicola*], common moorhen [*Gallinula chloropus*]), herons, egrets, shorebirds, marsh wren, and common yellowthroat. California species of special concern bird species that have been identified in the SCRE—though uncommon or rarely sighted—include least bittern (*Ixobrychus exilis*), white-tailed kite (*Elanus leucurus*), and northern harrier (*Circus cyaneus*) (Nautilus Environmental 2005, Reed Smith et al. 2007). Tricolored blackbird (*Agelaius tricolor*), a state Species of Special Concern that has not been documented in the SCRE but has the potential to occur, typically nests in large colonies (at least about 50 pairs) within protected substrate such as cattail, tule, blackberry, or willow near open water.

Mammal species using habitats typically along the edges of freshwater wetlands include western harvest mouse (*Reithrodontomys megalotis*), deer mouse (*Peromyscus* spp.), California meadow vole (*Microtus californicus*), and river otter (*Lutra canadensis*); muskrats (*Ondatra zibethicus*) and beaver (*Castor canadensis*) may use vegetation for cover, food, and/or hut construction. Bat species may forage for insect prey over wetlands.

8.1.3 Riparian shrublands and woodlands

Riparian shrublands and woodlands are valuable for wildlife since they provide water, favorable microclimates, cover, foraging habitat, and important movement corridors. Bird species typically associated with riparian shrublands and woodlands that are common to this area include Pacific-slope flycatcher (*Empidonax difficilis*), black phoebe (*Sayornis nigricans*), warbling vireo (*Vireo gilvus*), bushtit (*Psaltriparus minimus*), ruby-crowned kinglet (*Regulus calendula*), orange-crowned warbler (*Vermivora celata*), Wilson's warbler (*Wilsonia pusilla*), song sparrow (*Melospiza melodia*), and black-headed grosbeak (*Pheucticus melanocephalus*). Special-status bird species with the potential to occur within the riparian forest include the southwestern willow flycatcher (*Empidonax traillii extimus*) and least Bell's vireo (*Vireo bellii pusillus*) (both federally and state-endangered species), yellow warbler (*Dendroica petechia brewsteri*) and yellow-breasted chat (*Icteria virens*) (both state Species of Special Concern); all of which are rare or uncommon in the SCRE with the exception of the yellow warbler, which is common. A variety of bat species may roost in trees of riparian habitats including the western red bat (*Lasiurus blossevillii*), a state Species of Special Concern. Other mammals that can be found using riparian habitats in the SCRE include shrews (*Sorex* spp.), coyote (*Canis latrans*), bobcat (*Lynx rufus*), raccoon (*Procyon lotor*), and brush rabbit (*Silvilagus bachmani*).

8.1.4 Open water

Open water habitats support a diversity of bird species including dabbling ducks, diving ducks, gulls, herons, and grebes. Swallows and flycatchers feed on insects that they catch in flight, often over open water. Bird species that may be observed using open water habitats in the SCRE or Wildlife Ponds include gadwall (*Anas strepera*), mallard (*Anas platyrhynchos*), blue-winged teal (*Anas discors*), bufflehead (*Bucephala albeola*), ruddy duck (*Oxyura jamaicensis*), pied-billed grebe (*Podilymbus podiceps*), great blue heron (*Ardea herodias*), American coot (*Fulica americana*), western gull (*Larus occidentalis*), tree swallow (*Tachycineta bicolor*), and brown pelican (*Pelecanus occidentalis*) (Entrix, Inc. 2010 a,b).

8.2 Avian Focal Species

The two avian focal species for the Estuary Subwatershed Study are western snowy plover (*Charadrius alexandrinus nivosus*) and California least tern (*Sternula antillarum browni*). Western snowy plover and California least tern populations have been monitored at McGrath State Beach for over 25 years (R. Smith, Ventura Audubon Society, pers. comm., 2010). Ventura Audubon Society has been conducting annual monitoring for these two species in recent years. Information regarding their habitat requirements and life history is described below.

8.2.1 Western snowy plover

8.2.1.1 Status and distribution

The western snowy plover is a small shorebird closely associated with sandy beaches. Pacific coastal populations of western snowy plover are federally listed as threatened, while interior populations are a state species of special concern. The Pacific Coast breeding population of western snowy plover ranges from Washington to Baja California, Mexico. Nesting takes place on the ground above the high tide line on barren to sparsely-vegetated beaches. In California, breeding habitat primarily includes coastal dune-backed beaches, barrier beaches, and salt-evaporation ponds, and less commonly bluff-backed beaches, dredged material disposal areas, salt ponds, and river bars (Page et al. 2009, USFWS 2007). Wintering habitat includes the same beaches used for nesting, as well as non-nesting beaches and flats. Many inland nesting western snowy plovers migrate west and spend winters along the coast. This species forages by gleaning for invertebrates such and insects and crustaceans found on the sand, in stranded seaweed on the beach, or from low-growing plants in foredune habitats.

8.2.1.2 Life history

In California, western snowy plover breeding and nesting begins in March, peaks from mid-April to mid-June, and ends by late September (USFWS 2007). Hatching occurs from early April through mid-August, and chicks take approximately one month to fledge (USFWS 2007). A nest site consists of a shallow scrape or depression created by the male during courtship; the female chooses the scrape in which to lay her eggs (Page et al. 2009). Often the nest scrape is lined with bits of debris such as small pebbles, shell fragments, or plant debris. In southern California, western snowy plovers have been found to nest in areas with 6 to 18% vegetative cover and 1 to 14% inorganic cover; height of vegetation is usually less than 6 cm (2.3 in) (USFWS 2007). After either successful hatching or loss of a nest, both sexes may double-brood and females may even triple-brood (Page *et al*. 2009). There may be some correlation between plover hatching success and nesting within active least tern colonies; least terns may provide protection from certain predators such as American crows (*Corvus brachyrhynchos*) (Powell 2001). In addition to disturbance from humans or predators, nesting success may depend on tides or weather.

Primary threats to the western snowy plovers include human disturbance (e.g., off-road vehicles, pets, or direct harassment of eggs or chicks), loss of breeding and wintering habitat from development, expanding predator populations, and introduced beachgrass (*Ammophila* spp.) (USFWS 2007).

8.2.1.3 Recent survey protocol and results

The Channel Coast District (CCD) began a Western Snowy Plover Protection Program in 2001–2002, whose goal is to increase the amount of suitable habitat on CCD beaches, reduce disturbance to nesting and wintering plovers, prevent the take of nests and chicks, and simultaneously provide recreational and educational opportunities for visitors (More 2008). The CCD supports comprehensive nesting surveys, which involves collection of breeding data as well as winter foraging and roosting data. A permanent fence was installed around McGrath Lake by CCD staff in March 2002 (More 2008). In addition, anti-predator exclosures have been used on a subset of nests at McGrath Beach to protect the nests from predation.

Recent western snowy plover visual nest surveys have been conducted once or twice weekly by a USFWS 10(a)(1)(A) recovery permit holder, who walked meandering transects throughout the

study area in search of nests or sign (More 2008, Smith 2009). Nesting surveys were typically initiated in March and finished after nesting was complete, typically around August. Efforts were made to minimize any disturbance to nests. The survey area for McGrath State Beach included McGrath Lake, "McGrath North" near the campground, and the SCRE bar. General population surveys were conducted the rest of the year.

The number of nests at McGrath State Beach between 2003 and 2009 has ranged between 10 and 23 (Smith 2009). Number and locations of western snowy plover nests observed between 2007 and 2009 are illustrated in Figure 8-1. Documented nesting areas for western snowy plovers have included the strip of beach west of the south arm of the SCRE, near the surf zone; west of the campground and inland of the south arm of the SCRE; just north of the SCRE; and near McGrath Lake (Figure 8-1). Over the last 4 years, nest locations have shifted from next to McGrath Lake, to along the outer beach from the lake to opposite the campground (Smith 2009).

Western snowy plover nesting activity in this area usually peaks in May, though in 2009 the peak was in June followed closely by July (Smith 2009). In 2009, the primary cause of nest failure was human disturbance; people taking, moving, or stepping on eggs, or people vandalizing nest exclosures placed by biologists for protection from predators (Smith 2009). Other documented causes of failed nest sites at McGrath State Beach include: dogs, predators (coyotes, crows, gulls, ravens [*Corvus corax*], red-tailed hawks [*Buteo jamaicensis*], northern harriers, great blue herons, American kestrels [*Falco sparverius*], merlins [*Falco columbarius*], coyotes, ground squirrels, and loggerhead shrikes [*Lanius ludovicianus*]), and fluctuating water (More 2008, Smith 2009).

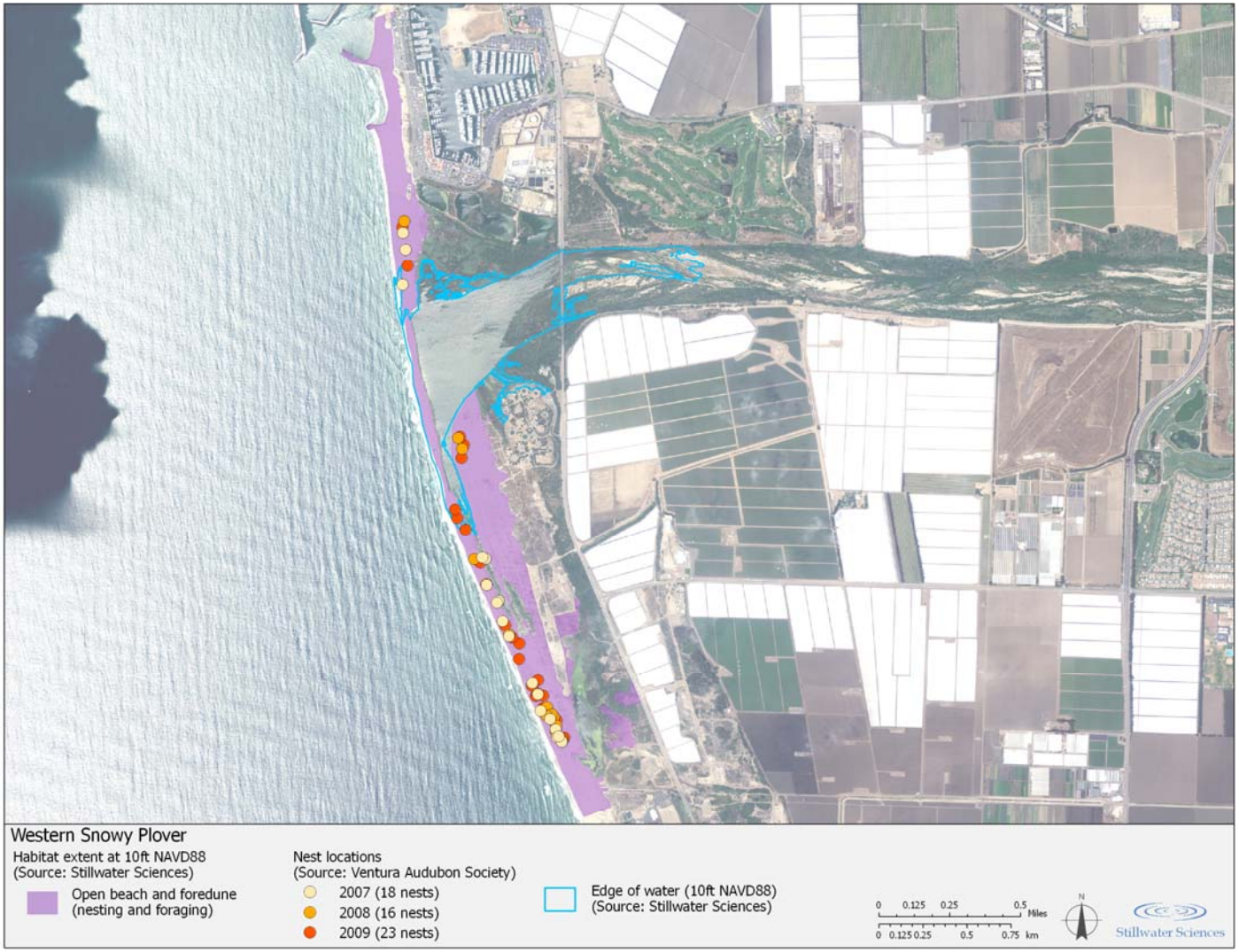


Figure 8-1. Habitats and recent western snowy plover nest locations near the SCRE, 2007–2009.

8.2.1.4 Habitat use within and around the Santa Clara River Estuary

The area surrounding the SCRE—including McGrath State Beach and the beach northwest of the SCRE—provides both nesting and wintering habitat for western snowy plovers. The western snowy plover breeding, nesting, and rearing season typically occurs here between March and August, while the non-breeding (winter) season usually takes place between September and February. During winter, numbers of western snowy plover individuals increase because inland nesting populations migrate to the coast to spend the winter.

In the context of the upland habitat mapping described above in Section 6.1, both western snowy plover foraging and nesting habitats are associated with open beach and foredune habitats. Suitable vegetated foredune habitat is specifically the *Abronia* spp. - *Ambrosia chamissonis* Alliance, which is characterized by sparse to moderate cover of low-stature perennial forbs. After a breach event, western snowy plovers have been observed foraging on the newly-formed mud flats and the dry sand in the middle of the SCRE (R. Smith, Ventura Audubon, e-mail message to N. Hume, Stillwater Sciences). Individuals have also been observed nesting on deposited sediment the southeast corner of the SCRE, prior to the area later being vegetated by willows and giant reed (R. Smith, Ventura Audubon, e-mail message to N. Hume, Stillwater Sciences).

We have used an understanding of western snowy plover habitat requirements and usage to map "potential" available western snowy plover foraging and nesting habitat in the SCRE under full conditions (Figure 8-1). Figure 8-2 shows suitable western snowy plover foraging and nesting habitat area (defined by the vegetation mapping as "open beach" or "*Abronia* spp. - *Ambrosia chamissonis* Alliance") as a function of SCRE stage. As stage increases, the area of western snowy plover habitat decreases with the inundation of open sandy beach and/or low stature foredune vegetation. The stage-habitat curve shows an inverse of that modeled for the two focal fish species, since the higher water levels equates to less habitat. It is important to note that barren sandy areas with little or no vegetation are created and maintained by scour events that occur due to fluctuating water levels and flooding. Such events suppress the growth of terrestrial vegetation by inundation with water and/or scour action when river flows are high. Consequently, while simply stabilizing the lagoon at a lower level would temporarily increase the area of dry sandy nesting and foraging habitat, vegetation—such as willows—would eventually adapt to the new water levels and inundation periods and replace such habitat. In addition, the area of additional surface water under higher stage conditions may act as a physical barrier, protecting western snowy plovers from certain predators and humans.

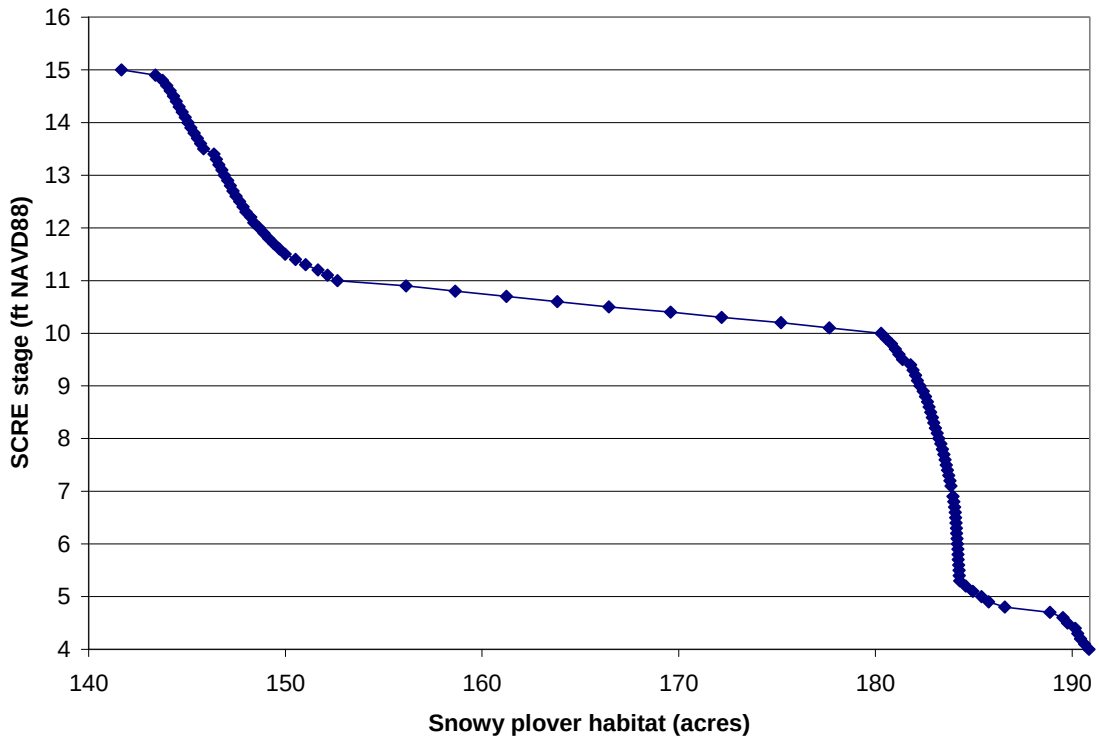


Figure 8-2. Estimate of suitable western snowy plover nesting and foraging habitat as a function of SCRE stage.

8.2.2 California least tern

8.2.2.1 Status and distribution

The California least tern, a small seabird generally associated with lagoons, estuaries, rivers, and the coast, is the smallest of North American terns. It is federally and state listed as endangered, and the only subspecies of least tern found in California. During the summer breeding season California least terns range from the San Francisco Bay area along the Pacific coast to Baja California, Mexico. Nesting habitat is typically sand or gravel beaches above high tide that are relatively free of vegetation as a result of scour from periodic high storm tides (Thompson et al. 1997, USFWS 2006). Little is known about winter distribution and habitats, though the species is generally thought to winter along marine coasts in Central America (Thelander 1994).

8.2.2.2 Life history

California least terns nest in loose colonies of around 30–50 pairs, and breeding begins as early as late April or May. Nests are bowl-shaped depressions in the sand, sometimes lined with shell fragments. Eggs hatch after 20–25 days (Thelander 1994), and are fed by both parents. After the first young fledge, there is generally a second wave of nesting. They locate their breeding colonies near an abundance of very small fish, foraging in shallow estuaries, lagoons, coastal ponds, or nearshore waters where small bait fish, such as anchovies (*Engraulis sp.*), smelts or silversides (*Atherinops sp.*), are abundant. They may take an assortment of fish and aquatic invertebrates that occur in the upper 15 cm of water (Thompson et al. 1997). They hover above

water and then plunge to feed after spotting fish. Fall migration begins in late July or early August (USFWS 2006).

Primary threats to the species include loss of breeding and wintering habitat from development, human-related disturbance (e.g., direct harassment of eggs or chicks, off-road vehicles, pets, or noise), and expanding predator populations (USFWS 2007). In addition to disturbance from humans or predators, nesting success may depend on tides or weather.

8.2.2.3 Recent survey protocol and results

Recent California least tern nest surveys have been conducted once weekly by a USFWS 10(a)(1)(A) recovery permit holder, who walked meandering transects throughout the study area in search of nests or sign (Smith 2008, 2009). California least tern nesting surveys were initiated in spring as early as late April, and completed after nesting was complete, typically around August or September (Smith 2008, 2009). Efforts were made to minimize any disturbance to nests. The survey area for McGrath State Beach included McGrath Lake, "McGrath North" near the campground, and the SCRE bar.

The number of nests at McGrath State Beach between 2000 and 2009 has ranged between 3 and 97, with a general upward trend in the number of nests, especially in recent years (Smith 2008, 2009). Number and locations of California least tern nests observed between 2007 and 2009 are illustrated in Figure 8-3. Documented primary summer nesting areas for California least terns have included: (1) just north of the SCRE, (2) west of the campground and inland of the south arm of the SCRE, (3) the strip of beach west of the south arm of the SCRE, near the surf zone, and (4) near McGrath Lake; though not all areas have been populated each year (Smith 2008, Smith 2009) (Figure 8-3). Many nests occurred on a wide, level, sandy area in between the small (0.3–0.9 m [1.0–3.0 ft] high) dunes and the surf zone (Smith 2008).

In this area, California least tern nesting activity usually peaks between June and July (Smith 2008, 2009). Documented causes of failed tern nest sites at McGrath State Beach and near the SCRE include human disturbance, abandonment, and predation by coyotes, opossums (*Didelphis virginiana*), American crows, and possibly ground squirrels (Smith 2008). Efforts to protect least tern nesting have been ongoing for over 20 years by the Ventura Audubon Society (Smith 2008). Various types of protective fencing have been placed in the spring and removed in the fall particularly to discourage disturbance by humans or their dogs.

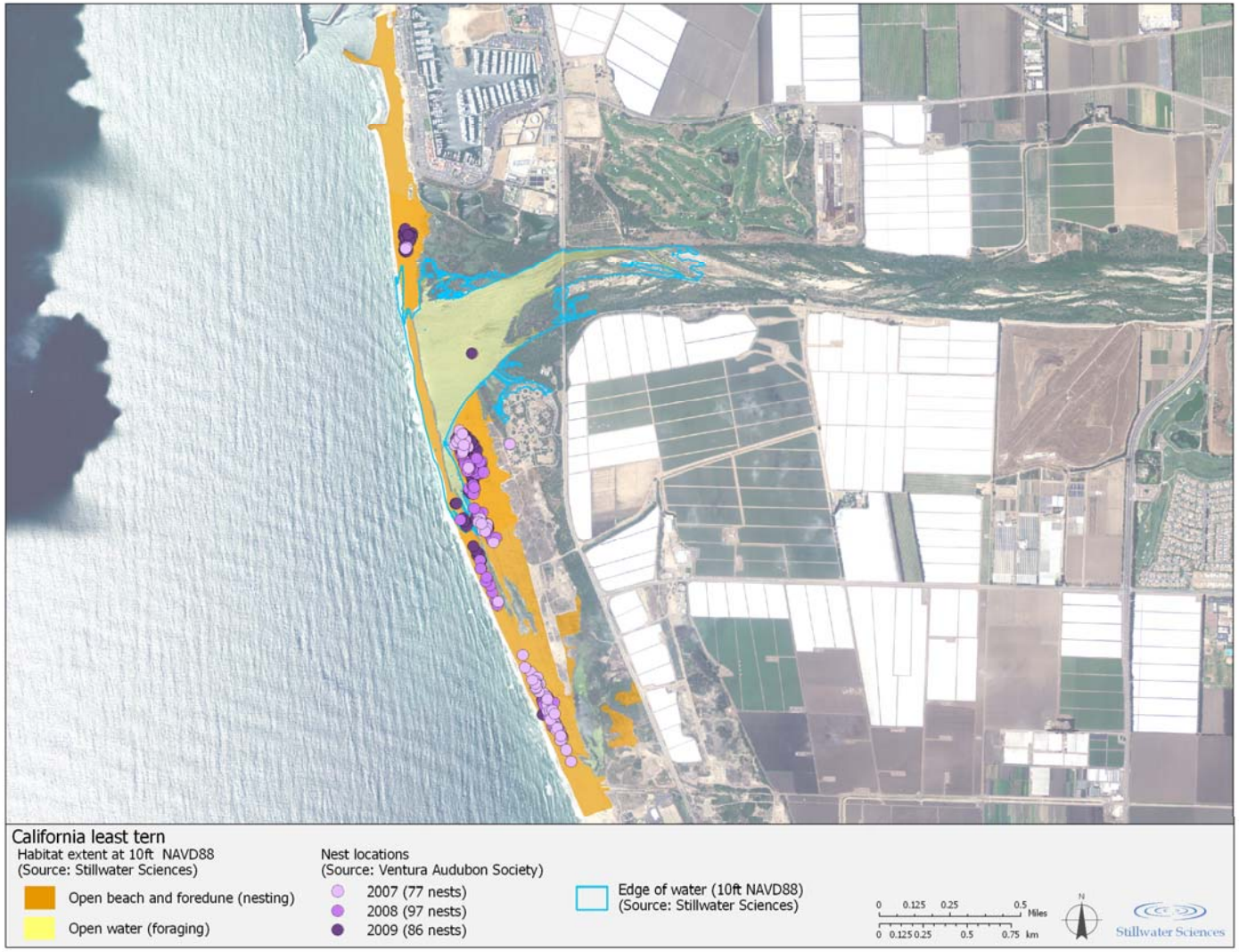


Figure 8-3. Habitats and recent California least tern nest locations near the SCRE, 2007–2009.

8.2.2.4 Habitat use within and around the Santa Clara River Estuary

The area surrounding the SCRE provides only summer nesting and rearing habitat for California least terns. This summer nesting and rearing season for California least tern typically occurs between April and September. During the non-breeding (winter) season, generally between October and March, California least terns migrate south, likely to Central America (Thelander 1994).

In the context of the upland habitat mapping described above in Section 6.1, California least tern nesting habitat is associated with open beach and foredune habitats. Suitable vegetated foredune habitat is specifically the *Abronia* spp. - *Ambrosia chamissonis* Alliance, which is characterized by sparse to moderate cover of low-stature perennial forbs. Sparse vegetation provides least tern chicks shelter from the sun and predators (Smith 2008). Primary foraging areas for California least terns using this area include the main SCRE lagoon, the south arm of the SCRE, inland freshwater ponds, McGrath Lake, the VWRF ponds, Ventura Harbor, nearshore ocean waters, and golf course ponds (Entrix, Inc. 2010a,b; Smith 2007, 2009).

As with the western snowy plover, we used an understanding of habitat requirements and usage to map "potential" available California least tern nesting and foraging habitat in the SCRE under full conditions (Figure 8-3). Figure 8-4 shows suitable California least tern nesting and foraging habitat as a function of SCRE state. The relationship between nesting habitat area (defined by the vegetation mapping as "open beach" or "*Abronia* spp. - *Ambrosia chamissonis* Alliance") and stage is as the relationship between nesting and foraging habitat and stage for western snowy plover, described above. The relationship between California least tern foraging habitat (defined here as open water) and SCRE stage shows that as stage increases, the area of nesting habitat decreases with relation to the loss of either open sandy beach or low stature vegetation types associated with foredune; therefore the higher water levels equate to less habitat. As described above for western snowy plover, barren sandy areas with little or no vegetation are created and maintained by scour events that occur due to fluctuating water levels and flooding, and while simply stabilizing the lagoon at a lower level would temporarily increase the area of dry sandy nesting habitat, vegetation would eventually replace such areas.

The relationship between California least tern foraging are and SCRE stage also illustrates that a dry estuary would dramatically decrease foraging area (Figure 8-4). However, it is important to note that the lagoon is not the sole source of least tern foraging habitat; near the SCRE, least terns also feed in McGrath Lake, the VWRF ponds, Ventura Harbor, and nearshore ocean waters. Compared with foraging habitat, availability of nesting habitat is much more limited for California least terns; therefore, nesting habitat should be a conservation priority for the species.

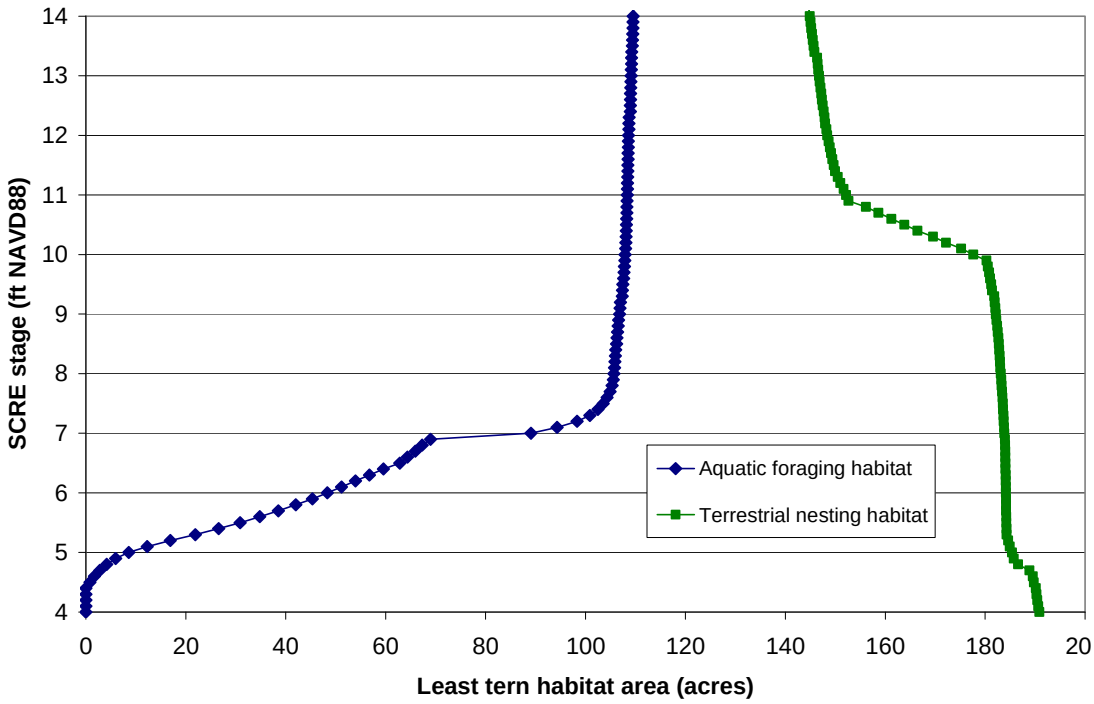


Figure 8-4. Estimate of suitable California least tern nesting and foraging habitat as a function of SCRE stage.

8.3 Historical Wildlife Habitat Conditions

Major physical changes in the SCRE since 1855, as discussed previously in Section 3.2, include the reduction in estuary size and loss of backwater-channels. In addition, as explained above in Section 6.2, several key changes in vegetation since historical times point to a loss of riparian habitats, changes in freshwater wetland distribution, and invasion by non-native plant species, primarily iceplant and giant reed.

Though there is a lack of available information regarding historical wildlife sightings, the change in size and location of riparian and freshwater wetland areas would likely correspond to a subsequent change in the abundance and distribution of the wildlife species associated with those habitats. In addition, the increase in giant reed has presumably had a negative effect on riparian bird species around the SCRE; as giant reed cover increases in area, bird species abundance and richness has been documented to decrease (Kisner 2004).

The current, 2009 vegetation mapping effort shows 69.6 ha (172 ac) of foredune habitat and 35 ha (87 ac) of open beach, for a total of 105 ha (259 ac) of western snowy plover and California least tern nesting habitat. Vegetation mapping using the aerial images from 1855 shows 91 ha (225 ac) of "dune" habitat and 17 ha (43 ac) of "beach" for a total of 108 ha (268 ac) of nesting habitat for the two focal species. This indicates that there has been little change in the overall total available open beach and foredune habitats that provide nesting substrate for the two focal species. However, invasions of non-native iceplant may have implications for reduced area of available nesting habitat for western snowy plover and California least tern. Also, the reduction in the size

of the SCRE may correlate with changes in California least tern foraging opportunities; since the SCRE has been converted from a wide, open delta to a river channel, there may be correspondingly less overall foraging habitat in the waters of the SCRE.

California least tern breeding surveys been conducted throughout the state since 1984. The area around the SCRE and McGrath State Beach has been included in these comprehensive yearly surveys. Between 1984 and 2009, the number of nests observed in and around the SCRE has fluctuated, though the trend has been an overall increase in number of nests (Figure 8-5). This trend of an increase in breeding California least terns is consistent with other populations throughout the southern California coast (Marschalek 2007); fluctuations may particularly be associated with ENSO years (Massey et al. 1992). The years with the fewest recorded least tern nests were 1988, 2003, and 2005 (years with two, three, and nine documented nests, respectively). The years with the highest number of recorded nests were 2001, 2004, and 2008 (years with 59, 83, and 97 observed nests, respectively).

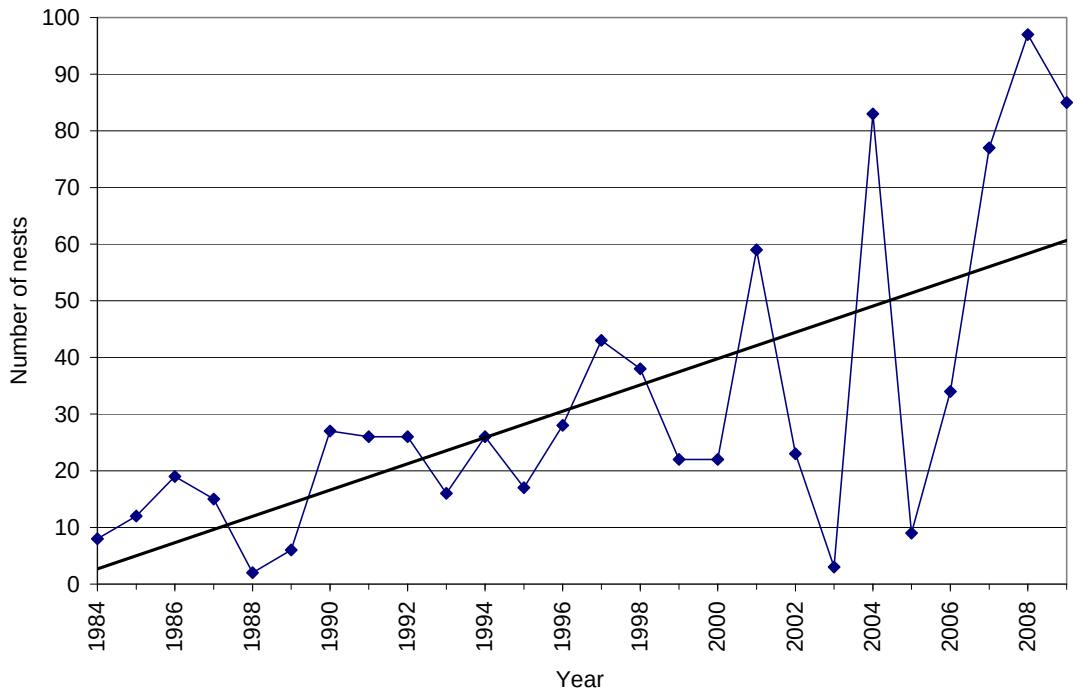


Figure 8-5. Number of California least tern nests observed in and around the SCRE, 1984–2009.

As discussed in Section 6.2, there is little evidence for salt marsh historically existing in the SCRE during the early part of the 20th century. Presently, there are several patches of salt marsh vegetation distributed around the south edge of the SCRE, totaling 1.9 ha (4.7 ac). Associated with salt marsh habitat is the Belding's savannah sparrow (*Passerculus sandwichensis beldingi*), a California species of special concern that is strongly linked with pickleweed (*Salicornia virginica*). Belding's savannah sparrow was historically observed in salt marsh habitats near McGrath State beach; the most recent documented observations being twelve breeding pairs in 1977 and 1 pair in 1991 (Swanson et al. 1990, Zembal and Hoffman 2002). Though Belding's savannah sparrows had occasionally been reported near the mouth of the SCRE, the habitat there was considered too "marginal, scant, and/or disturbed to support true subpopulations of the sparrows" (Zembal and Hoffman 2002).

9 FUNCTIONALITY OF THE SANTA CLARA RIVER ESTUARY AND SUBWATERSHED

This section includes an overall conceptual model of the SCRE, which examines the functional linkages among SCRE ecosystem processes, resultant habitats, and biotic needs using a focal species approach to evaluate existing beneficial uses. Ecosystem functions under current conditions are described to help to evaluate the effects of the VWRF discharge on the SCRE, focusing upon habitat conditions and recreational opportunities that are either directly affected by VWRF flows or that may be affected by management outcomes related to potential changes in these flows.

9.1 Overall Conceptual Model of Santa Clara River Estuary

Developing an understanding of the influences of the VWRF upon the physical processes, water quality, habitats, and recreational uses is complicated by the dynamic nature of the SCRE. As described in the preceding sections, the SCRE experiences large pulses of water and sediment from the Santa Clara River, which affect water quality, channel form, bed substrate, vegetation assemblages, and estuary extent on short-term and long-term timescales. The largest pulses occur primarily during ENSO years, which currently have a 3–8 year recurrence interval. These changes are superimposed upon more regular influences of VWRF effluent discharge, baseflow river discharge, and tidal fluctuations on mouth breaching and ocean water exchange. Here, we present an overall conceptual model of the SCRE in terms of its primary inputs, dominant physical processes, the structure of the aquatic, tidal, and riparian habitats, and biotic responses to historical changes in these habitats (Figure 9-1).

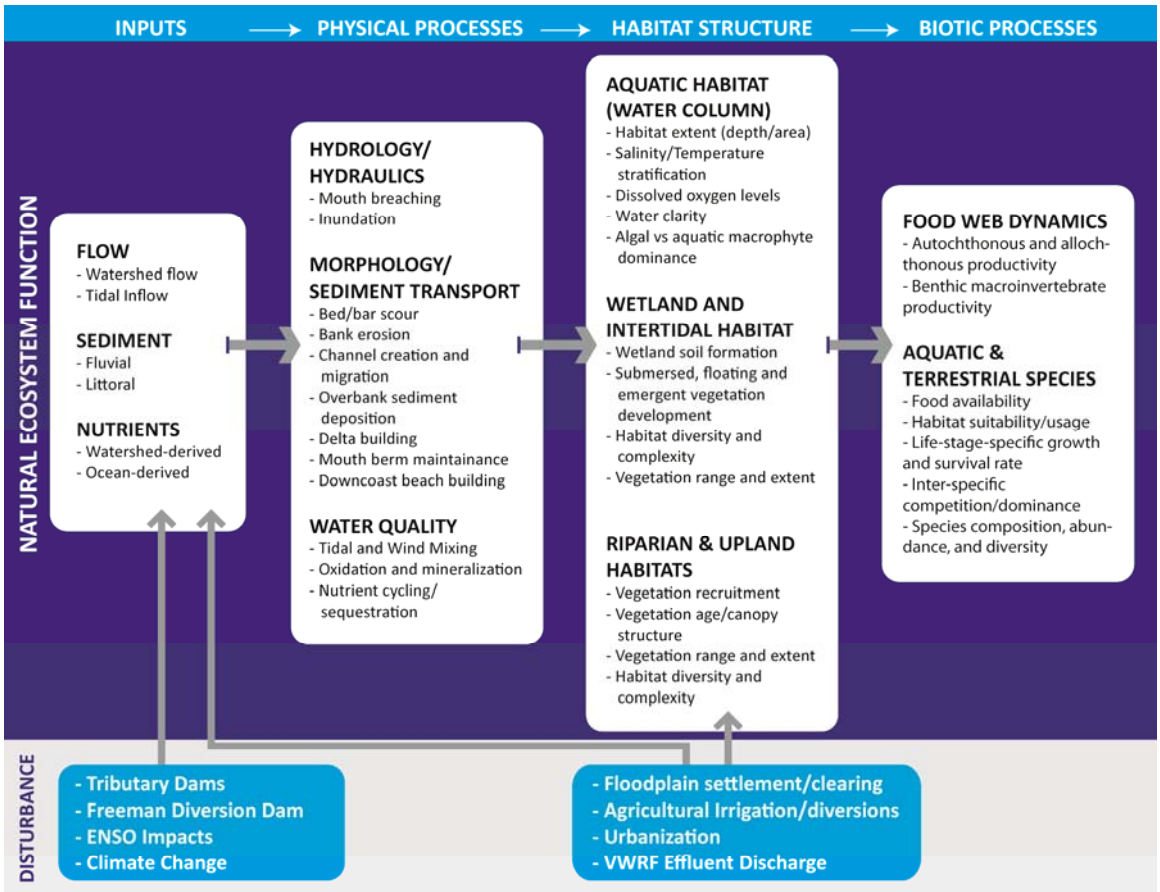


Figure 9-1. Conceptual model of the SCRE ecosystem functioning.

As an idealized conceptual model of the Santa Clara River, natural watershed inputs (such as water, sediment, and nutrients) drive physical processes (such as sediment transport and channel migration) that, in turn, determine geomorphic attributes and physical habitat structure of the Santa Clara River, its floodplain, as well as the formation and stability of the SCRE lagoon and mouth berm. These geomorphic attributes and habitat structure in turn control biological processes and are important determinants of plant and animal species abundance, distribution, and composition. Modification of any of these key inputs, including VWRF inflows, may affect physical processes and, subsequently, affect plant communities, fish and wildlife populations.

9.1.1 Primary inputs and physical processes

Habitat conditions within the SCRE are controlled primarily by the movement of water, sediment, and nutrients arriving from the upper watershed and water and nutrients delivered by the VWRF. Like most rivers within southern California, natural flow conditions of the Santa Clara River are characterized by low flows in summer and early fall, with large but infrequently occurring flow peaks in winter caused by storm events. Each of these components of the natural hydrograph is important in shaping and sustaining the river-floodplain system. Changes to any of the hydrographs components can impact natural ecosystem function.

Sediment is a fundamental building block of alluvial river systems, providing material for the river to construct in-channel bed forms, banks, and maintain the adjacent floodplain. In an undisturbed state, sediment is supplied from the upper watershed, transported through and temporarily stored in the alluvial reaches of the river, and deposited at the mouth in an estuary/lagoon or in an adjacent river mouth delta. In turn, wave action and on-shore winds affect the sediment redistribution and the formation and maintenance of beach dunes, sand bars, and estuary/lagoon morphology. As flow in the river increases, the threshold for mobilizing sediment on the channel bed surface is eventually surpassed causing bed scour. The channel bed may not be mobilized at all in low flow years but may be mobilized several times during flood years. Banks may also frequently erode, meander bends may migrate and be cutoff as "oxbows." Such rivers are in a state of dynamic equilibrium between sediment supply and transport such that erosional and depositional processes are continuously modifying the system, yet the amount of sediment entering and exiting the system is very similar and the overall channel width, depth, and slope fluctuate only narrowly over the long term (Schumm 1977). A decrease in downstream sediment supply by activities such as flow regulation in the upper watershed can lead to chronic channel incision downstream, thereby resulting in subsequent bank failure and long-term channel instability.

Separate from these in-channel processes, floodplain inundation serves to reduce flood flow magnitude and promote exchange of nutrients, organisms, sediment, and energy between terrestrial and aquatic systems. Alterations in floodplain inundation timing, magnitude, frequency, and duration due to flood-control levees and upstream flow regulation by dams can result in changes in riparian plant species composition and age-class structure, which can alter habitat suitability for native birds and cause a shift in bird community species composition. In turn, riparian vegetation can feed back to hydraulic and geomorphic processes. For instance, increased roughness provided by newly established riparian vegetation can increase fine sediment deposition during flood events, thereby promoting floodplain building processes.

9.1.2 Habitat structure and biotic responses

The physical processes discussed above, such as flooding, scour, and sediment deposition strongly influence riparian plant species composition, distribution, and physical structure and are major drivers of riparian community succession. Riparian vegetation dynamics are tightly coupled with river processes and typically exhibit two distinct patterns of vegetation development. Newly deposited sediment within the active channel and on floodplains provides surfaces for plants to establish. Willows (*Salix* spp.) are typically among the first species to colonize bare stream banks and high bar surfaces. These species have physiological traits, such as high seed output and rapid growth rates, that are well suited for quickly colonizing these areas. As vegetation from one cohort matures, it traps sediment and can extend the bar surface, creating new seedbeds for successive recruitment events (Strahan 1984, Scott et al. 1996).

In addition to the establishment of pioneer plant species on newly deposited floodplain and bar surfaces, successional processes alter vegetation composition in established riparian stands. "Later successional" species, such as box elder (*Acer negundo*), typically produce larger seeds and are more shade tolerant than the early pioneer plants, which allows them to persist in the seedbank and germinate under the forest canopy when soil temperature and moisture conditions are appropriate. Over time, these species alter soil composition, soil moisture, light penetration, and nutrient conditions within the riparian zone and outlive or outcompete the early pioneer species. At any one site, the spatial and temporal patterns of physical processes (such as flooding and sediment deposition dynamics) and biological processes (such as plant establishment and

competition) can be quite complex, and vegetation composition is often more patchy than the generalized patterns described above.

The aquatic habitat structure within the SCRE is also strongly linked to the inputs and physical processes discussed above. Historically, the floodplain of the lower Santa Clara River contained a dense riparian zone with marshy areas that were regularly re-connected with in-channel habitats at higher flows. Prior to the establishment of levees in the lower river in the 1900s, meander and migration processes regularly eroded channel banks causing a retreat of floodplain habitat on the outside of meander bends, depositing fresh sediment and forming new floodplain habitat on the inside of meander bends. Under idealized conditions, this process of erosion on the outside of meander bends maintains scour pools and causes tree recruitment into the river, which provides cover and other habitat functions for fish and other aquatic organisms. Interruption of these processes has resulted in greatly simplified habitat structure with limited available cover resident fish species.

Much of the discussion above is an idealized simplification of habitat evolution within the SCRE. The overall conceptual model developed for the SCRE presents a functional view of the ecosystem using a geomorphic process-based approach that considers the impact of several external disturbances (Figure 9-1). All disturbances included in the conceptual model impact water, sediment, and nutrient inputs, with several disturbances having the potential to directly affect habitat structure as well.

9.2 Ecosystem Functions under Current Conditions

Ecosystem functions and services are frequently used to evaluate the benefits of natural resource management decisions. Ecosystem functions may be defined as the internal functioning of the ecosystem (e.g., nutrient and energy fluxes, food-web interactions) or in terms of related ecosystem services (e.g., food production, water supply) (Costanza et al. 1997, De Groot et al. 2002). Here, we use these concepts to evaluate the effects of the VWRF discharge on the SCRE, focusing upon several functions that are either directly affected by VWRF flows or that may be affected by management outcomes related to potential changes in these flows. These functions are:

- Flow regulation
- Sediment storage and beach building
- Water quality regulation
- Aquatic habitat maintenance
- Wildlife habitat maintenance
- Recreational opportunities

In the sections below, we focus on the effects of the VWRF discharge, a primary disturbance, on ecosystem functioning with emphasis on effects related to focal species and human recreational uses of the SCRE and McGrath State Beach.

9.2.1 Flow regulation

An important function of estuaries and adjacent floodplain areas is to store floodwater delivered from the contributing watershed during large storm events. Floodwater storage within and adjacent to an estuary affects sediment deposition and associated off-channel and upland habitat

maintenance, and in-channel bed scour and associated aquatic habitat maintenance. In addition, flood storage affects groundwater recharge rates, which has more subtle effects upon river baseflows and groundwater salinity.

The amount of floodwater stored in a seasonally-closed estuary like the SCRE is primarily a function of available volume (the difference between total estuary volume and the volume of water already in the estuary), while floodwater storage within the adjacent floodplain is a function of both available volume and the connectivity between the main river channel and floodplain. Available estuary storage volume can vary substantially among storm events as a function of differences in bed elevation (scour- or deposition-induced variation), adjacent tide elevation during open-mouth conditions, and pre-storm inflow volume to the estuary during closed-mouth conditions. Floodplain storage also varies due to changes in available floodplain area and/or floodplain elevation. The degree of connectivity between floodwaters and the adjacent floodplain is controlled by the interaction between flood magnitude (and associated stage height) and floodplain elevation.

From the analysis presented in Sections 3.2 and 4.1 pertaining to changes in SCRE and lower Santa Clara River geomorphology and hydrology over the past 150 years, it is apparent that there has been a fundamental change in the way the SCRE and adjacent floodplain regulate flow. Under historical conditions, the Santa Clara River had an extensive floodplain once it entered the lower watershed and floodwaters were readily able to access this floodplain during high flow events. This resulted in the maintenance of upland riparian vegetation communities along the lower Santa Clara River channel and adjacent to the SCRE, and wetland and scrub vegetation communities within the SCRE footprint adjacent to side channels. Historically, floodwater storage also likely helped maintain an elevated water table and baseflows to the SCRE during winter months, which may have affected the proportion of time the SCRE mouth remained open. Since the onset of settlement in the lower Santa Clara River watershed in the early 20th century, the flow regulation potential of both the SCRE and adjacent floodplain has decreased dramatically. Extensive land reclamation and levee building since the 1930s has reduced the SCRE and floodplain area by 75% to 90%, and largely disconnected the main channel from the adjacent floodplain. This disconnection has resulted in the rapid movement of flood waters through the SCRE due to limited floodwater storage, increased flow constriction through the lower Santa Clara River due to flood control levees, and increased bed scour (Section 3.2).

Although the current VWRF flows may be seen as replacing historical baseflow that are diverted from the Santa Clara River upstream of the SCRE, changes in flow regulation have been largely altered by watershed development within and adjacent to the SCRE since the onset of European-American settlement. The establishment of the VRWF in 1958 decreased the SCRE and adjacent floodplain footprint by approximately 33 ha (80 ac), however this developed area represents a relatively small portion of the total area developed over the past 150 years and it occurred after an extensive levee system disconnecting the SCRE from the adjacent floodplain had already been established. The constant VRWF discharge is currently a dominant source of water to the SCRE during dry months (August–September) and between storms (as shown in Section 4.2), which often results in minimal river flow storage capacity for early fall storms and between major storm events in the winter/spring. It is unclear, however, if this degree of river flow storage is drastically different from the conditions that existed prior to VRWF establishment. Upstream flow regulation and groundwater pumping has likely contributed to a substantially decreased average annual baseflow compared to pre-development conditions, and it is likely that the SCRE historically received more intra-storm baseflow than it does currently (Section 4.1). Therefore, the volume of water currently within the SCRE at the onset of storm events may be similar to historical conditions and the VWRF discharge as well as local groundwater sources may be

currently providing surface and groundwater flows that historically were provided from upstream sources.

9.2.2 Sediment storage and beach building

Related to the flow attenuation and floodwater storage functions, estuaries and adjacent floodplain areas can serve as storage zones for sediment transported during storm events. Sediment deposited over-bank on adjacent floodplains acts to maintain existing vegetation communities and habitats, create new floodplain habitats, and maintain floodplain elevation by counteracting soil compaction and ground subsidence. Sediment deposited within the estuary and within nearshore deltas can substantially affect short-term and long-term beach building and coastline position. Following storm events, this sediment can be redistributed by fluvial, tidal, and/or wind action and can go towards maintaining geomorphic features within and around the estuary (e.g., dunes, mouth berm, beach and coastline position), or can go into offshore littoral cells and be deposited on down-coast beaches. The amount of sediment deposited and stored is a function of several factors including the volume of sediment delivered from the upper watershed, the connectivity of the river with storage zones (both in the estuary and the adjacent floodplain), the degree of flow confinement and associated flow velocity, and the overall volume available for deposition during the storm event (as impacted by adjacent tidal elevation and estuary bathymetry). Variations in estuary bathymetry and tidal elevations can contribute to inherent differences in transport and deposition dynamics for storms with similar peak flows and sediment yields.

The sediment storage dynamics of the SCRE and adjacent floodplain and the role that these features have in maintaining local and down-coast beaches has been drastically altered by development both adjacent to the SCRE and within the contributing watershed. Flow regulation in tributaries to the Santa Clara River and along the mainstem channel has reduced average annual sediment yield to the SCRE by over 20%, and flow diversion along the mainstem Santa Clara River is thought to have prevented hundreds of thousands of cubic meters of sediment from reaching the SCRE. This upstream sediment storage, combined with sediment deposition caused by other in-channel development features, has likely decreased the amount of sediment deposited within and adjacent to the SCRE on an annual average basis and has thereby impacted local beach building, mouth berm formation, and down-coast beach building processes compared to pre-development conditions.

In addition to the impact of upstream dams, levee construction has greatly affected sediment deposition. Prior to levee-building along the length of the Santa Clara River, flood flows had a greater capability to spread out onto adjacent floodplains, thereby dissipating flow energy (and scour potential) and depositing a considerable portion of the suspended sediment load. The building of levees to contain flood flows from the Los Angeles County Line down to the SCRE has affected sediment deposition and storage in the SCRE and adjacent areas in three primary ways. First, levee-building has contributed to the overall decrease in the area of the SCRE and adjacent floodplain and has therefore contributed to an overall decreased sediment storage potential. Second, levee-induced flow constraint has caused flows to have a relatively higher velocity and scour potential when reaching the SCRE compared to pre-developed conditions, which has likely caused a decrease in short-term sediment storage within and adjacent to the SCRE. The prevention of over-bank fine sediment deposition caused by levees may be contributing to an increased frequency of dense hyperpycnal flow events that scour the main channel areas. As hyperpycnal flows have the potential to bypass the SCRE before being deposited offshore or to be transported directly to the Santa Barbara littoral cell, the frequency of

hyperpycnal flows can affect both the timing and volume of sediment delivered to beaches adjacent to and down-coast from the SCRE.

Similar to flow regulation, the VWRF impact on sediment storage and beach building processes is a relatively small part of the entirety of development-related impacts since the onset of European-American settlement. As discussed in Section 3.2, the establishment of the VWRF decreased the available floodplain within the current SCRE footprint, thereby decreasing over-bank flow and sediment retention potential along the north bank. However, as with the flow regulation effects, this decrease SCRE area is represents a relatively small portion of the total area decrease over the past 150 years. Furthermore, the extensive levee system upstream of the VWRF that prevents overbank flow and floodplain sediment storage is a primary cause of sediment-laden floodwaters often bypassing the SCRE before reaching the ocean. The relatively high VWRF discharge to the SCRE during dry months and between storm events can cause the SCRE mouth berm to breach, resulting in the evacuation of sediment into the Santa Barbara littoral cell and down-coast. Although this translates to a loss of sediment to the offshore delta that did not exist historically, the impact of upstream sediment retention and increased hyperpycnal flow frequency on the river delta volume is likely considerably more substantial than any impact associated with the VWRF operations. In this regard, the VWRF has minimal effects upon sediment storage and beach building functions.

9.2.3 Water quality regulation

Water quality regulation in estuaries is a function of geologic setting, water balance (proportion of inflows, outflow, and storage), the quality of incoming waters, types of soils and vegetation, and the effects of human activity in the immediate vicinity of the contributing watershed. Because of its geographic setting at the transition between riverine and marine systems, estuaries regulate riverine water quality arriving from the upper watershed (e.g., minerals, nutrients, natural organic matter, and potential contaminants) though the structuring influences of bathymetry (depth and volume), orientation and extent of open water areas (because of wind fetch as it relates to wave heights and mixing depths), and tidal exchanges with the ocean (e.g., mixing and density stratification due to salinity).

Variations in inflows to the SCRE from riverine sources including seasonal changes in baseflow, episodic stormwater runoff, and tidal exchanges are the strongest driver of water quality within the SCRE on an annual basis, primarily due to their influences on mouth closure status. The SCRE historically functioned largely as a lagoon-type system, and continues to function in a similar manner under current conditions. A lagoon-type system is characterized by a narrow tidal inlet and a shallow (i.e., less than 2 m [6 ft] deep) open-water area separated from the ocean by a bar or barrier beach (USEPA 2001). In contrast to other estuary types that are characterized by high organic matter content in bottom sediments, the shallow depth of the lagoon within the SCRE enhances sediment-water exchanges of minerals, nutrients, and oxygen due to the influence of vertical mixing of wind-driven waves.

The VWRF exerts the strongest influences upon water quality during closed-mouth conditions, which normally occurs from late spring through fall in most years (Section 4.2). However, because of the stochastic nature of annual rainfall patterns related to ENSO and other factors, periods of stable lagoon formation may extend through dryer winter/spring periods when outflow of the Santa Clara River is insufficient to initiate a breaching event or maintain open-mouth conditions. Although the majority of open-water sites sampled as part of this study exhibited uniform DO with depth (Stillwater Sciences 2010), like other shallow lagoonal systems in many locations, the SCRE is characterized by local areas of low DO (hypoxic) water with strong

vertical salinity stratification, subject to the influences of freshwater flows, ocean exchanges, and wind mixing. This is particularly evident within the stands of tule and giant reed along the north side of the lagoon (Stillwater Sciences 2010), which serve to inhibit wind-mixing and allow salinity stratification to isolate bottom waters from atmospheric exchanges of oxygen. Although DO levels in the SCRE were found to be uniform throughout the water column during *in situ* monitoring for this study, significant excursions of DO above and below saturation were observed during continuous monitoring of *in situ* water quality conducted by the City (Section 5.1).

Under current conditions, water quality regulation in the SCRE is directly impacted by excess nutrients arriving from the VWRF. The elevated nutrient levels in the VWRF outfall channel relative to other locations in the SCRE combined with the elevated trophic state index values suggest the SCRE is currently eutrophic. Although recent scour events may influence the relative amounts of rooted submerged aquatic vegetation (SAV), the lack of SAV has also been linked with nutrient enrichment in estuaries (e.g., Orth and Moore 1983).

9.2.4 Aquatic habitat maintenance

One of the primary ecological functions of estuaries is the maintenance of aquatic habitat. As described in Section 3.2, historical changes in physical processes in the Santa Clara River watershed and SCRE have resulted in the loss of floodplain habitats, altered stream flows, and a likely increase in the frequency of mouth berm breaching. The formation and stability of the SCRE lagoon and suitability of this habitat with respect to depth, hiding cover, and water quality needs of steelhead and tidewater goby is the primary focus of this section.

As described in detail in Section 3.2, coastal lagoons such as the SCRE form in response to seasonal rainfall patterns, with sandbar closure during dry periods (spring and summer) and breaching during wet periods (fall and winter). During wet months, high energy waves erode and breach sandbars, while high stream flows widen and deepen the estuary mouth; whereas in dry months, low-energy waves deposit sand and build up sandbars. After sandbar formation, water surface elevation rises as the impounded estuary (lagoon) fills with freshwater streamflow. Under natural conditions, this annual cycle serves to provide relatively stable water levels, salinity, temperature, and DO from late spring to early winter, with some variation due to occasional storms. Sandbar breaching during the onset of fall or winter storms supplies an exchange of salt and freshwater, provides temporary habitat access for upstream and downstream steelhead migrants, and offers opportunity for dispersal of passively transported species such as tidewater goby. Below, we discuss recent changes to the lagoon with respect to these two species to represent habitat needs of other estuarine species within the SCRE.

9.2.4.1 Steelhead

Estuary conditions may have an important influence on anadromous fish survival since southern steelhead must pass through these areas during upstream adult migration and downstream smolt outmigration, 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). Perhaps the most important aspect for understanding steelhead population dynamics is that the average fecundity is high relative to the amount of suitable juvenile rearing habitat available within upstream stream reaches where spawning takes place. As discussed in Section 7.2.1, this means that rather than being controlled by reproductive success, steelhead population size within a particular river system may often be limited by physical habitat constraints during the juvenile freshwater rearing stage.

Steelhead can smolt at a variety of ages[10], but most frequently smolt at age 2+. Because juvenile steelhead must spend at least one summer and winter in freshwater prior to outmigrating to the sea, they tend to establish localized territories within areas of suitable rearing habitat soon after emergence from the gravel. The relatively extended freshwater rearing of steelhead has important consequences for the species' population dynamics. The maximum number of steelhead that a stream can support is limited by food and space through this territorial behavior, and is necessary to produce steelhead smolts that are large enough to have a high probability of ocean survival relative to all other sources of mortality throughout the steelhead life cycle.

Although the physical habitat requirements for different age classes of steelhead are relatively similar, as fish age and grow they require more space for foraging. The presence of lagoon habitat in coastal watersheds can serve to relieve density-dependent pressures from competition between individuals. That is, competition for food and space results in downstream emigration or mortality of juvenile steelhead that are smaller or less aggressive. In watersheds without lagoon habitat, emigrants are presumed to be lost from the population. However, in coastal watersheds with lagoons, a portion of the density-dependent emigration from stream habitat may be accommodated by lagoons, depending on seasonal habitat suitability. Because predation mortality and fish size are likely to be inversely related, estuarine rearing and increased growth relative to stream rearing 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). As stated in Section 7.2.1.4, the identification of stranded *O. mykiss* following a mechanical breach in October 2010 suggests that the lagoon currently provides suitable rearing habitat for steelhead.

Overall, water levels in the SCRE may currently provide steelhead rearing habitat, water quality conditions may limit the suitability of this habitat during warmer months (Section 5). Kelley (2008) suggested that bar and lagoon formation prior to late April potentially affected one third of emigrating smolts from either increased predation or from adverse water quality conditions in the SCRE. Kelley (2008) included high water temperatures are suspected as a major factor limiting suitability of over-summer rearing habitat for juvenile steelhead in the SCRE. In addition, DO is frequently at or below required levels for steelhead life history needs (Table 5-6, Figure 5-6) and provides for only marginally suitable conditions for rearing of juvenile steelhead in the SCRE, particularly under closed-mouth conditions. Under current conditions, the VWRF discharge may be directly linked to periods of low DO levels due to algal growth (Section 9.2.3), although the main SCRE lagoon currently provides relatively cool water temperatures as compared to upstream tributary habitats.

Recognizing the water quality limitations above, we have modeled "potential" steelhead habitat in the SCRE as the superimposition of depth and hiding cover requirements (Figure 7-1), which shows suitable habitat increasing with estuary stage, reaching a maximum above 3 m (10 ft) NAVD88. As stated above, a number of factors in addition to depth and cover may influence habitat use in the SCRE and that current water quality conditions may limit over-summering in the SCRE lagoon. Further, it should be noted that stream and lagoon habitat should not be viewed as disconnected habitat features. Just as upstream conditions such as seasonal changes in freshwater inflow and sediment delivery affect lagoon characteristics, demographic processes such as immigration and emigration link steelhead population dynamics within stream and lagoon

[10] In this discussion, we follow conventional methods for assigning fish ages to year classes. Age 0+ refers to fish in their first year of life, sometimes called young-of-the-year; age 1+ to fish in their second year of life, and so on. A fish changes from age 0+ to age 1+ based on the time of hatching, which in the case of steelhead occurs in the spring.

habitat. Steelhead populations are typically limited by a combination of density-dependent processes occurring within stream reaches, and by the degree to which seasonal rearing opportunities in lagoon habitat may relax those density-dependent bottlenecks. Because SCRE conditions are apparently not favorable for steelhead over-summering, quick emigration to the ocean may be most advantageous for smolts arriving on the SCRE during winter and spring. Whether the SCRE lagoon historically provided suitable water quality conditions for a life history strategy that includes over-summering is not known. Nevertheless, we assume that the ability of lagoon habitat to support steelhead in excess of stream carrying capacity is dependent on the annual variability in timing of sandbar formation and the maintenance of suitable water quality conditions following sandbar closure during spring.

9.2.4.2 Tidewater goby

As with steelhead, the tidewater goby life cycle is closely tied to the dynamics of estuary closure and lagoon formation. On a population scale, presence of tidewater goby along the California coast follows the distribution of littoral cells that distribute sand and allow the development of closed estuaries (Capelli 1997). Geographic gaps in distribution correspond to steep coastlines with no embayments that are likely to form lagoons. Sandbars that separate lagoon habitat from the ocean can prevent tidal influence, reduce water velocity, and subsequently diminish the probability of tidewater goby displacement.

Although water quality conditions may limit the habitat suitability for over-summering steelhead, there is no evidence that water quality is currently limiting tidewater goby populations. However, completely freshwater environments are generally not preferred by tidewater goby. For example, a study in the Big Sur River watershed, where the mouth never closes and a brackish lagoon does not form, resulted in no observations of tidewater goby suggesting that closed lagoons may provide a habitat feature necessary for stable populations of tidewater goby (Swenson 1996). As noted above, tidewater gobies have been observed in the main channel areas of the SCRE and backwater areas such as the outfall channel (Entrix 2004), suggesting the formation and stability of this lagoon habitat is important for the continued recovery of this species. Although Entrix (2004) indicated that mid-summer breaching events were associated with stranding and mortality of tidewater gobies in the SCRE, Nautilus Environmental (2009) found relatively similar seine haul densities of gobies under both open- and closed-mouth conditions.

Swenson and McCray (1996) reported that field studies of tidewater gobies in the estuaries of two central California creeks (San Gregorio and Pescadero) revealed different patterns in population ecology among different habitats. Feeding ecology differed for gobies in lagoon, creek and marsh habitats. Tidewater gobies in the marsh were significantly larger, more fecund and potentially longer-lived than tidewater gobies in the lagoon or creek. However, males in lab studies preferred to dig spawning burrows in sand rather than mud, which suggests that sandy lagoons may be more important than muddy marshes as spawning habitat. Although lagoons are considered the typical habitat of tidewater gobies, Swenson and McCray's research suggests marshes can also be important, perhaps due to better food resources or more moderate disturbance regimes. Marshes may serve as refugia during winter high-flow events, providing a potential source population for recolonization of adjacent creek and lagoon habitats following major resetting flood events (USFWS 2005).

BMI monitoring in the SCRE suggest that the lagoon and emergent wetland habitat near the VWRF effluent channels both provide an array of food resources for tidewater gobies (Section 7.1.1). In addition, the lower reach of the Santa Clara River upstream from the estuary likely provides additional suitable habitat with a different mix of prey items. During most years, food

availability for tidewater goby likely increases subsequent to closing of the estuary and from spring to summer following disturbance from winter high flows, and decreases after sand bar breaching. Larger, infrequent flood events, such as the January and February 2005 flood events, reset the main channel and lagoon, depleting invertebrate abundance and allowing a short-term shift in species composition from tolerant estuarine species to a broader mix of opportunistic riverine and estuarine species. This shift was evident in sampling conducted in 2005, which showed a shift in taxonomic compositions in May, but a return to the long-term composition observed in other years by October. The emergent wetland vegetation associated with the VWRF outfall channel may be particularly important as a refuge during high flow years, such as 2005, since it appears to be some of the only suitable tidewater goby habitat in the estuary and lower river that, under current conditions, is not likely to be highly disturbed or reset during big flood events. The mosaic of primary habitat types (lagoon, emergent wetland, and main channel), variety of substrates (including sandy substrates for spawning), and range of suitable BMI prey found in the Santa Clara River estuary appear to provide favorable conditions for persistence of the local tidewater goby population,

As presented in Section 7.2.2, we have modeled "potential" goby habitat primarily as a function of depth in the SCRE. Goby habitat generally increases with increasing estuary stage until the preferred depth range is exceeded at stages above 2–3 m (8–10 ft) NAVD88 (Figure 7-3). Because food resources for Tidewater goby do not appear to be currently limiting the population (Section 7.2.2.2) and gobies are found throughout the SCRE at all times of year, the current population levels are likely limited by a combination of flood scour events, predation, and suitable spawning locations. Swift et al. (1999) describes tidewater goby spawning beginning after estuary closure in late spring from individuals that had survived during the several months of open water exchange with the ocean. Swift et al. (1989) found that mortality among early-season spawners from the prior year and the absence of large individuals later in the season suggest that the late-season spawning population largely consists of individuals derived from more recent (within-season) spawning. This suggests that early-season spawning tidewater goby surviving from the prior year are vulnerable to the effects of storms that occur after estuary closure. These individuals may retreat upstream to freshwater marshes, channel margins, or other backwater habitats such as the outfall channel (Swift et al. 1989, Swenson 1997). High-flow events may displace pelagic and benthic larvae, juveniles, and adults, and scour burrows, which are laid 2.5 cm (1 in) below the surface (Swenson 1997). Although instances of intentional breaching of the SCRE lagoon observed in 2009–2010 as well as some prior years may have the same effects, elevated flows (including summer flows from the VWRF) may breach newly formed sandbars causing changes in estuary depth, salinity, and temperature that preclude embryo survival (Capelli 1997).

9.2.4.3 Summary

Maintenance and restoration of the natural hydrologic regime and breaching frequency of the SCRE is an important factor in preserving lagoon function, and the physical and chemical environment required for both steelhead and tidewater goby (Hayes et al. 2008, Smith 1990, Mistch and Gosselink 1993, Capelli 1997). The amount of time that the estuary mouth has been open has varied by decade (Figure 4-7), with approximately mouth conditions occurring approximately 60–80% of the time during winter and pring, and only 40–50 percent of the time during summer and fall (Figure 4-8). Rearing habitat in the SCRE under closed-mouth conditions found during summer and periods of low rainfall has been shown in this study to increase with estuary stage (Figure 7-3), reaching a maximum at levels above 3 m (10 ft) NAVD88. Improvements to over-summering habitat conditions for rearing steelhead may be accomplished

through planned improvements in the VWRF treatment process and reductions in nutrients arriving in the SCRE.

For tidewater goby, complete sandbar closure, sufficient freshwater inflow, availability of sand substrate, and availability of hydraulic refuge are all important factors in providing stable conditions necessary for tidewater goby spawning and early rearing**.** The discharge from the VWRF is consistent with general water chemistry parameters preferred by the goby and reduces salinity to a preferred range during dry seasons (Nautilus Environmental 2005). Also, suitable habitat has been shown to increase with increasing stage, reaching a maximum between 2.4–3 (8–10 ft) NAVD88. However, sand bar breaching associated with VWRF flows in excess of the infiltration capacity of the sandbar during summer or intentional breaching events may cause rapid dewatering of spawning burrows or alter chemical conditions (salinity, temperature, and DO) within the estuary. Too many breaches in one season may impact the goby population given their short lifespan. However, their high fecundity, frequent spawning, and territorial behavior allow goby populations to rebound, but alterations to the natural dynamics of lagoon formation can prevent this natural resiliency. It should be noted that frequent salinity exchanges with the ocean may also limit the number of predators in the lagoon.

Incremental mixing of saltwater with freshwater within the estuary may benefit both steelhead and tidewater goby, due to maintenance of water quality conditions and control of non-native predatory fish, for example. In addition, increased breaching frequency during spring may promote steelhead emigration. However, periods when the estuary is open provide fewer habitats for tidewater goby, and breaching during the late spring or early summer may dewater tidewater goby nests. In summary, stable lagoon habitat from early summer through fall is consistent with the needs for early-season steelhead emigration and tidewater goby. Nevertheless, current water quality conditions in the SCRE may not allow a steelhead life history strategy that includes estuarine rearing.

9.2.5 Wildlife habitat maintenance

The SCRE provides yearly nesting habitat for the two focal terrestrial avian species—western snowy plover (Pacific coastal populations) and California least tern. This is a unique function of the SCRE as habitat is generally uncommon along the California coastline due in large part to habitat loss from urbanization, which has contributed to the decline of and subsequent listing of these two species. For nesting, both the California least tern and western snowy plover require barren or sparsely-vegetated sand or gravel beaches above the high tide line. Both species build a nest by creating a shallow depression in the open sand, which exposes their eggs to humans, predators, and elements including weather and inundation.

For these two avian focal species, habitat management and conservation considerations are especially important during breeding and rearing seasons, since nests and eggs are immobile and subject to damage and predation. California least terns are only present near the SCRE during the spring and summer breeding and nesting season, arriving to breed as early as late April and typically finishing nesting and migrating south by mid-September. Western snowy plovers breed and nest near the SCRE between March and August. Contribution of effluent from the VWRF is more evident during the drier months, which coincide with the nesting season for both species.

During September through February, numbers of western snowy plover increase around the SCRE because inland nesting populations migrate to the coast during winter to forage and roost. While California least terns migrate south and are absent from the SCRE in winter, western

snowy plovers also spend winter around the SCRE; therefore habitat management and conservation considerations for western snowy plovers using the SCRE are year-round.

The elevated estuary stage resulting from high relative contribution of VWRF effluent to total inflow volume during low-flow conditions may be partly responsible for creation of the backwater area that forms along McGrath State Beach south of the SCRE. This additional inundated area may decrease the amount of usable nesting habitat for California least tern and nesting and foraging habitat for western snowy plover. Conversely, the area of additional surface water may provide added foraging habitat for California least tern, and may act as a physical barrier protecting nests of both species from certain predators. Increased summer base flow from the VWRF and the gradual increase in water levels following SCRE mouth closure could potentially impact either the available area for nesting avian focal species, or submerge active nests established below the full lagoon level. For example, during the 2007 nesting season, one western snowy plover nest was documented as lost due to inundation from increasing water levels in the lagoon (Smith 2007).

California least terns are primarily piscivorous—foraging for small fish in the estuary, ocean, and other nearby waters—while western snowy plovers are insectivorous. Consequently, California least terns are more directly linked with aquatic organisms using the SCRE. There is a potential threat of contaminants in the SCRE that could adversely affect water quality and resident fish populations, which could then affect California least tern through bioaccumulation. Elevated levels of contaminants could be detrimental to least terns as a result of accumulated substances, including Hg, Se, and PCBs, that can then affect reproduction and chick survival (Ohlendorf et al. 1986, Hothem and Powell 2000). As described in Section 5.2, although a number of these substances are present in the SCRE that could lead to potential bioaccumulation, no tissue data for fish or other organisms was available to evaluate such contaminants. Nonetheless, there has been no evidence of least tern or western snowy plover reproductive failure in the SCRE due to possible water quality contamination.

Generally, the trend has been an increase in the number of nesting California least terns in the SCRE, which is consistent with other populations throughout the southern California coast (Marschalek 2007). The data are limited regarding nesting success of western snowy plovers in the study area; however, based on 2007–2009 study results, western snowy plovers have been nesting successfully in relatively stable numbers for at least the past several years. Generally, the primary issues facing nesting success of the western snowy plover and California least tern in the SCRE are direct and indirect human disturbance and predation, both factors not influenced by the VWRF.

As discussed in Section 8.3, there has been little change in the overall total available open beach and foredune habitats that provide nesting substrate for the two focal species over the last century. However, invasions of non-native iceplant may have implications for reduced area of available nesting habitat for western snowy plover and California least tern if the invasive species continues to proliferate. Presence of iceplant is not linked to the VWRF.

Outside the SCRE, the VWRF Wildlife Ponds provide habitat for many other avian species, particularly those associated with open water habitat including dabbling ducks, diving ducks, gulls, herons, grebes, swallows, and flycatchers. The VWRF provides year-round open water habitat in the Wildlife Ponds and provides additional riparian habitat for birds and other wildlife associated with the SCRE.

9.2.6 Recreational opportunities

Recreation at McGrath State Beach is very diverse and includes camping, surfing, hiking, bird watching, nature observation, swimming, and sunbathing (California Department of Parks and Recreation 1979). The State Beach is comprised of a campground, recreational beach, and the Santa Clara River Natural Preserve. In addition to access to these areas along the south side of the SCRE, there is also access from the north side of the SCRE, allowing visitors many of these uses, with the exception of camping.

The McGrath State Beach camping facilities along the south side of the SCRE include 174 campsites, two group campgrounds, and one limited day use parking area. Paved roads provide access to individual campsites and the day use area. Other facilities include three centralized toilet and shower buildings, a small visitor center, a visitor area for campfire talks presented by the park staff, and an onsite park staff residence. Access to the beach, the Natural Preserve, and other areas of the park are mostly on dirt or sand paths and a boardwalk extends north through the riparian section of the nature preserve.

Use of the recreational facilities is highly seasonal with warm spring, summer, and fall weather causing a significant increase in visitors. One major impact to recreational use of the McGrath State Beach during these times is ground saturation and flooding, which frequently results in partial campground closure. During low-flow conditions when the mouth is closed, surface flow coming into the SCRE causes subsurface flow to be directed towards the south bank floodplain and campground. When the mouth remains closed and the filling persist, the south bank water table elevation continues to rise and the campground ground surface becomes saturated. In addition, surface water flow up the tidal slough on the northwest end of campground currently floods the access road around the northern campground perimeter when the SCRE stage rises above 3 m (10 ft) NAVD88. This can occur throughout the year, but consistently occurs by mid-summer when high river flows have subsided and the mouth has been closed for an extended time period (A. Hamilton, CA State Parks, pers. comm., 2010). The flooding persists for weeks to months until the mouth berm breaches, the SCRE drains, and the groundwater flows towards the SCRE and the water table elevation decreases. Data collected from 2009–2010 suggest that the water table responds somewhat rapidly, decreasing by up to 1 m (4 ft) just 1 to 2 weeks after mouth berm breaching.

The impact of the VWRF discharge on current ground saturation and flooding within McGrath State Beach is apparent from the data compiled and analyses conducted for this study. The data presented in Section 4.2 suggest that the high frequency of mouth closure combined with the relatively high VWRF discharge rate consistently cause the SCRE to fill and subsequently increase adjacent floodplain water table elevations during the drier summer months (August and September). The water balance presented in Section 4.2 suggests that the degree of ground saturation during summer months and between storm events would likely decrease on average if the VWRF discharge was removed because there appears to be no other water source that would increase enough to compensate for the loss. Therefore, there appears to be a direct link between current VWRF discharge operations and decreased recreational opportunities at McGrath State Beach.

It is not clear, however, from existing information or analyses conducted for this study, how present-day floodplain flooding during closed-mouth, low-flow conditions differs from historical, pre-development flooding dynamics. In fact, it is possible that the VWRF discharge acts to maintain a degree of saturation that is somewhat similar to what existed historically. Existing information suggests that the SCRE mouth berm has always closed and impounded incoming

surface and subsurface flow during drier months. It has been hypothesized that there was once likely more year-round baseflow coming into the SCRE than currently exists and that the longer mouth berm that existed historically allowed more subsurface seepage out to the ocean than at present. Therefore, the SCRE likely remained full historically to some quasi-equilibrium stage during dry months and there was a balance between inflows and outflows. As McGrath State Beach is located in an area that was once within the SCRE active boundary and contained salt flat, vegetated wetland, and woody vegetation (Figure 6-1), the area was probably regularly flooded historically during low-flow conditions when the SCRE mouth was closed. Both surface and subsurface flow to the SCRE has decreased on average due to upstream development impacts (e.g., flow regulation, groundwater pumping, and floodplain development). It is not obvious then if the VWRF discharge is increasing floodplain ground saturation on average compared to historical conditions, or if the discharge is compensating for the decrease in flow to the SCRE and thereby causing the floodplain to saturate as it did prior to the establishment of McGrath State Beach.

9.2.7 Summary of Santa Clara River Estuary functionality and resulting effects

Table 9-1 below provides a summary of the current conditions of ecosystem functions in the SCRE, describes if and how the current conditions are affected by the VWRF, and relates any effects with focal species and recreational opportunities. Overall, the VWRF provides freshwater flows that generally increase habitat area for focal species up to an elevation of approximately 3 m (10 ft) NAVD88 under current conditions, with higher water levels bringing losses in rearing habitat for tidewater goby (Figure 7-3), losses in nesting and foraging habitat for western snowy plover (Figure 8-2) and a decrease in nesting habitat for California least tern (Figure 8-4). Although current water levels above 3 m (10 ft) NAVD88 also coincide with periods of campground flooding, limiting recreational opportunities, it is unclear whether this condition occurred under historical conditions (Section 9.2.6). Current water quality conditions related to nutrient loading and algal growth may limit the suitability of the SCRE for over summering of steelhead (Section 9.2.3). In summary, the current VWRF discharge to the SCRE provides an enhancement of aquatic and wildlife beneficial uses with respect to habitat area up to SCRE water stages near 3 m (10 ft) NAVD88 under current conditions, but may limit habitat at higher SCRE stage. Current water quality conditions in the SCRE are not enhanced by the VWRF discharge. Current recreational opportunities may be limited by the current VWRF discharge regime due to increased SCRE stage associated with extended periods of berm closure. After a summary of climate change impacts on future conditions (Section 10), we develop a series of discharge and treatment alternatives to determine whether the VWRF discharge can provide an enhancement to existing beneficial uses under a different discharge or treatment scenario (Section 1.2, Study Question 9).

Table 9-1. Current condition of ecosystem functions in the SCRE with association to VWRF and effects on focal species and recreation.

Ecosystem function	Current conditions in the SCRE	Conditions affected by VWRF?	Effects on focal species	Effects on Recreation
Flow regulation	Decreased flood storage potential as compared to historical conditions.	**Moderate**: • Primarily result of land reclamation and levee-building. • VWFR discharge reduces available SCRE volume, though effects are similar to historical conditions.	None.	Flooding results in occasional partial closure of McGrath campground.
	Main river channel constricted/disconnected from floodplain compared to historical conditions.	**None**: • Primarily result of land reclamation and levee-building.	None.	None.
	Increased baseflow during dry periods.	**High**: • Although estimation of historical baseflows is not possible, it is likely that summer flows under historical conditions were lower than those under the current VWRF discharge regime.	• Benefit from increased amounts of freshwater habitat for focal fish species. • Potential for stranding as well as dewatering of goby spawning burrows following intentional breach events. • Potential increased population of warm water predator fish species.	Flooding results in occasional partial closure of McGrath campground.
Sediment storage and beach building	Decreased amount of sediment deposition and storage compared to historical conditions; reduced beach-building and mouth berm-formation.	**Minimal:** • Primarily result of upstream sediment storage, in-channel development, and levee-building. • VWRF footprint decreases available floodplain, though small portion of total area reduced historically.	Small reduction in beach-building does not substantially reduce focal avian species nesting habitat.	None.
	VWRF discharge to SCRE can cause mouth to breach during dry months and between storm events.	**Minimal:** • VWRF discharge causes occasional mouth-breaching/ sediment loss during dry months and between storm events, though upstream sediment retention is more substantial of a cause.	Mouth-breaching and mixing of saltwater/ freshwater benefits aquatic focal species; though summer breaching may cause rapid dewatering of tidewater goby spawning burrows.	None.

Ecosystem function	Current conditions in the SCRE	Conditions affected by VWRF?	Effects on focal species	Effects on Recreation
Water quality regulation	SCRE can have local areas of low DO water with vertical salinity stratification, particularly within vegetated areas along the north bank due to lack of wind mixing.	**High**: • Although direct BOD loading from the VWRF is low, DO is lower within the outfall channel than at open water sites.	Low DO may cause physiological stress of focal fish species.	None.
	The SCRE is currently eutrophic.	**High**: • VWRF discharge is contributing to high nutrient levels in SCRE, though strongest influences on WQ during are closed-mouth conditions.	High nutrient levels may be associated with algal growth and periods of anoxia, which may limit over-summering steelhead habitat.	None.

Ecosystem function	Current conditions in the SCRE	Conditions affected by VWRF?	Effects on focal species	Effects on Recreation
Aquatic habitat maintenance	Steelhead temporarily use SCRE for immigration and emigration to and from the ocean; steelhead may use the SCRE for rearing	**Moderate**: • VWRF provides increased flows and ponded habitat area • VWRF discharge contributes to high nutrient levels; strongest influences on WQ during are closed-mouth conditions.	• Increased area allows greater forage area and reduced predation risk. • Stable freshwater conditions may increase predation risk from enhanced suitability for non-native predator species. • Low DO and high nutrient levels may contribute to limiting over-summering steelhead habitat.	None.
	Tidewater gobies are found throughout the SCRE year-round.	**Moderate**: • VWRF discharge consistent with water chemistry preferred by tidewater gobies, and reduces salinity to preferred range during dry seasons. • VWRF flows during summer may cause breaching and rapid dewatering of tidewater goby spawning burrows or alter chemical habitat conditions.	• Increased area allows greater forage area and reduced predation risk. • Stable freshwater conditions may increase predation risk from enhanced suitability for non-native predator species. • Breaching associated with VWRF flows may dewater spawning burrows/ alter salinity and other water quality conditions.	None.
Wildlife habitat maintenance	Western snowy plovers and California least terns use nearby beaches for nesting; California least terns forage in estuary.	**Minimal:** • High relative contribution of VWRF discharge to total inflow volume during low-flow conditions, in part, creates backwater area that forms along McGrath State Beach. • VWRF provides increased flows and ponded habitat area.	Lagoon provides California least tern foraging habitat and physical barrier protecting both species' nests from predators, which mitigates effects of small decrease in nesting habitat.	Maintenance of wildlife habitat improves recreational opportunities for birding and wildlife viewing.

Ecosystem function	Current conditions in the SCRE	Conditions affected by VWRF?	Effects on focal species	Effects on Recreation
Recreational opportunities	Recreation at McGrath State Beach includes camping, surfing, hiking, bird watching, nature observation, swimming, and others.	**High:** • VWRF discharge during mouth closure causes the SCRE to fill, saturate the floodplain, and partially flood McGrath State Beach campground.	None.	Campground is often partially flooded during drier summer months.

10 ESTIMATED FUTURE CONDITIONS DUE TO CLIMATE CHANGE

Carollo Engineers (2011) developed a technical memorandum to present results of a scientific literature review of potential effects of global climate change relevant to the region of the SCRE (see Appendix E). The results were used to ultimately provide reasonable projections of changes in local atmospheric temperature, mean sea level (including the extreme high and low tide levels), and precipitation to incorporate into the analysis of alternative management scenarios to existing VWRF discharge conditions and assessment of the effects on physical, water quality, and biological dynamics of the SCRE and the subwatershed ecosystem. The following subsections summarize information used in the technical memorandum to develop the projections for temperature, sea level/tides, and precipitation patterns/events.

For the most part, recent estimates of projected ranges of changes in atmospheric temperature, tide levels, and precipitation patterns/events due to global climate change were extracted from published scientific literature identified as being the most relevant or, in the case of the IPCC, internationally recognized analyses with implications for California.[11] [12] Estimates were then checked against a recent comprehensive report by the Point Reyes Bird Observatory (PRBO 2011) that includes an assessment of climate change specific to the southern California coast.

10.1 Atmospheric Temperature

There is inherent uncertainty regarding the future of the earth's climate since: (1) no one can definitively predict what humans will do in the future concerning intensity of energy demand, efficiency of energy utilization, and consumption of fossil fuels; and (2) there is uncertainty about future emissions of GHGs and how and when the earth's climate will respond. However, the IPCC (2007) AR4 concluded that improved modeling studies were converging on a global mean surface temperature increase of 2.8°C (5.1°F) by 2100. Consequences of such a temperature increase include increased intensity, duration, and frequency of storm events, changes to mountainous snowfall and snowmelt dynamics (Cayan et al. 2006), melting of land ice, thermal expansion of the marine mixed layer of the ocean, subsequent rise in sea level, and changes in soil moisture.

The SCRE's climate is moderated by the proximity to the Pacific Ocean, resulting in an average annual atmospheric temperature of approximately 15.5°C (60°F) with an average annual maximum of 21°C (70°F) and an average annual minimum of 9.5°C (49°F) based on observations collected at Ventura's National Weather Station (No. 049285). Projections from the IPCC and the United Kingdom Hadley Centre's climate model (HadCM2) show temperatures in California increasing by an average of 2.8°C (5.1°F) with a full range of 1.1 to 5.0°C (2.0 to 9.0°F) by 2100 (Hayhoe, et al. 2004).

Based on this information, the recommended projected ranges of increase in the atmospheric temperature in the region of the SCRE are:

- 3.0˚C (1.8 to 5.4˚F), average of 2.0˚C (3.6˚F) by year 2050; and

[11] Air temperature and precipitation for Ventura were retrieved from the National Climatic Data Center (NCDC) (http://www.ncdc.noaa.gov/oa/mpp/freedata.html).

[12] Precipitation and water level data, as measured at the Santa Barbara and Santa Monica precipitation and tide gages, were retrieved from NOAA's National Ocean Service (NOS) Center for Operational Oceanographic Products and Services (COOPS) (http://tidesandcurrents.noaa.gov/index.shtml).

- 5.0˚C (2.0 to 9.0˚F), average of 3.0˚C (5.5˚F) by year 2100.

10.2 Sea level and tides

In the context of global climate change, mean sea level (MSL) is usually presented as monthly or annual averages. The sea level and tide data, analysis, and other related information presented here are applicable to the ocean at the entrance of the Estuary and along the coast in much of southern California, but do not consider if or to what extent sea level or tide conditions may be different inside the Estuary. Since there is no tide gage at Ventura, tidal description and analysis was dependant on tide gages at Santa Barbara and Santa Monica.

There has been little, if any, MSL rise observed along the California coast (and along most of the west coast of the Americas) since at least 1992 when global satellite data first became available (Aviso 2010). Local tide gage data shows that sea level in southern California has been essentially static since about 1980. These extended episodes of relatively static sea levels are likely due to regional sea level suppression as a result of effects connected to the Pacific Decadal Oscillation[13] and the subsequent response of the Pacific Ocean to a combination of surface warming or cooling and changes in wind stress patterns. Sea level rise will likely increase when the suppression of MSL rise along the Pacific coast of the Americas relaxes or reverses. When this happens in the coming decades, the southern California region could see rates of sea level rise above the ever-increasing global average rates, which now average about 0.3 m (1 ft) per century (Aviso 2010).

The average daily tidal fluctuation for tide gages at Santa Barbara and Santa Monica is characterized by two ranges, the diurnal tide range (defined as the difference between mean higher-high water [MHHW] and mean lower-low water [MLLW]) and the mean tide range (defined as the difference between mean high water [MHW] and mean low water [MLW]). For the 19-year tidal epoch between 1983 and 2001, the diurnal tide range was 1.64 m (5.39 ft) at Santa Barbara and 1.66 m (5.43 ft) at Santa Monica. The mean tide range was 1.11 m (3.65 ft) at Santa Barbara and 1.15 m (3.76 ft) at Santa Monica. Santa Barbara's monthly maximum tide is observed as increasing at a slightly higher rate than MSL. This means that the high tide, on average, has been getting higher faster than MSL. Concurrently, the minimum tide level is decreasing. Similarly, Santa Monica's monthly maximum tide level is increasing at a slightly higher rate than MSL, and MSL is increasing at a slightly higher rate than the minimum sea level. In other words, the tide ranges for each gage have been increasing over time. The cause of the observed changes in tide range is not understood. Therefore, it is not known whether past trends will or will not continue into the future. Assuming moderately high GHG output in the future, projected ENSO events and related storm surges and wave action would be expected to produce larger fluctuations and much more severe erosion and coastal damage and flooding than has been observed in the past.

Based on this information, the recommended projected ranges of sea level rise for the SCRE are:

- 0.2 to 0.6 m (0.7 to 2.0 ft) by year 2050;
- 0.5 to 2.0 m (1.6 to 6.6 ft) by year 2100; and
- consider the likelihood of extreme tides.

[13] Pacific Decadal Oscillation is regarded as the predominant source of long-term climate variability in the Pacific Northwest.

10.3 Precipitation patterns and events

Between 1900 and 2010, the average annual precipitation amount at the SCRE was approximately 380 mm (15 in) (as recorded at Ventura's National Weather Station No. 049285). Upstream the SCRE (as recorded at the Santa Paula-UWCD Station No. 049245a), the average annual precipitation is approximately 17 in between 1820 and 2010. To examine the potential future impacts of global climate change on annual average precipitation, scientists have developed general circulation and regional climate computer models. While projected temperature changes are broadly consistent across most modeling efforts, projected total average annual precipitation changes have varied widely across models and emissions scenarios (Kiparsky and Gleick 2003, Madsen and Figdor 2007). In addition, as models are run at finer levels of geographic resolution (e.g., regional or metropolitan level) the accuracy decreases. Most regional studies performed in California have focused on Northern California, and there is no scientific consensus on future projections for total average annual precipitation for the Ventura area. The regional circulation models show little change, maybe slightly more or slightly less in total annual precipitation. There is no scientific consensus on the projected change in total average annual precipitation.

Although projections for total average annual precipitation vary significantly, most regional climate model results in the U.S. suggest that the extreme daily (24-hour) precipitation rate will increase relative to changes in the average annual precipitation rate. Both general circulation and regional climate models project the intensity of precipitation is likely to increase around the world, with the most significant increases occurring in the middle to high latitudes (Meehl, 2005). Kharin and Zwiers (2005) project the probability of 24-hour precipitation events considered to be extreme will increase by a factor of about two by the period of 2046 to 2065 and by a factor of three by the end of the 21st century relative to those that occurred during the period of 1981 to 2000. This means 24-hour precipitation events having return periods of 10, 20, 50, and 100 years will occur two or more times as often by the year 2100 due to global climate change (Kharin and Zwiers 2005, Kharin et al. 2007). By the year 2100, northern California is projected to experience an increase in both low and high intensity events (Dettinger 2005).

Based on this information, the recommended projected changes in precipitation patterns and events for the SCRE are:

- Little change in total annual precipitation amount;
- an increase in the frequency of extreme daily (24-hour) events[14] by:
    - a factor of 2 by year 2050; and
    - a factor of 3 by year 2100.

10.4 Summary of Climate Change Effects

The climate change projections described above provide a basis for assessing the future rate and magnitude of natural surface and subsurface flows into and out of the SCRE. Changes in future sea levels are considered to be of greatest importance in determining changes to future SCRE outflows, with less consideration given to increases in air temperatures as well as peak runoff events associated with berm breaching and channel bed scour. In the next section, we combine estimates of future sea levels with several VWRF discharge alternatives to predict future SCRE stage, volume, inundated area, and water quality parameters in relation to focal species habitat and recreational opportunities.

[14] The frequency of these extreme daily events has already increased by 58% to 69% since 1948, which exceeds the projected increase in frequency.

11 ALTERNATIVES DEVELOPMENT AND ANALYSIS

As outlined in Section 1.2, the primary purpose of the Study is to confirm that the VWRF effluent discharge to the SCRE provides enhancement of existing beneficial uses as compared to the absence of the discharge. In addition, the City has agreed to further evaluate whether different discharge conditions or discharge management measures, such as elimination of discharge, a decreased discharge volume, or increased levels of effluent treatment, might further improve beneficial uses of the SCRE.

Based upon a review of existing conditions, and the current ecosystem functioning of the SCRE detailed in the preceding sections, the VWRF discharge currently provides usable habitat for the four focal species selected for this study, with periods of unsuitable DO levels attributable to excess nutrient loading to the SCRE during summer months (Section 9.2). Habitat area for aquatic and avian focal species was estimated to be either unaffected or to generally increase with increasing SCRE water levels up to approximately 3 m (10 ft) NAVD88 (Figures 7-1, 7-3, 8-2, and 8-4). However, water levels monitored during 2009–2010 were frequently higher and incremental habitat benefits to some species at these levels were minor. Further, higher water levels above the 3 m (10 ft) NAVD88 may contribute to the incidence of breach events during dry conditions, which did not normally occur under historical conditions. Although continued periods of uniformly higher SCRE stage benefit some focal species, it is apparent that management measures applied to the high SCRE stage should be considered to evaluate additional improvement to beneficial uses that might be attained. Since higher water levels above 3 m (10 ft) NAVD 88 do not appear to be necessary to increase wildlife habitat and water levels at and above 3 m (10 ft) increase flooding risks and reduce recreational area at McGrath State Beach, it may be possible to also improve both wildlife and, secondarily, recreational beneficial uses by considering different discharge management measures corresponding to lower SCRE stage.

In order to provide a rational basis of comparison to examine whether the VWRF provides "enhancement of water quality," which is defined as "further realization of existing beneficial uses" under current conditions as compared conditions without discharge, this section examines ecosystem functioning under current conditions as compared to ecosystem functioning in the absence of discharge. In addition, several discharge volume reduction and treatment alternatives are described and evaluated below to determine if additional or different discharge management measures can further improve beneficial uses. Alternatives to existing VWRF discharge conditions were developed with the City and the Project Stakeholder Group during the September 28, 2010 workshop. In all, six discharge alternatives were developed to represent the full range of possible actions, from no change, to discharge elimination, to volume reductions, to additional treatment, including planned VWRF upgrades and potential implementation of constructed wetlands for additional removal of nitrates (NO_3). Also incorporated within each alternative are future changes anticipated due to climate change as discussed in Section 10. The following is a description of each alternative:

Alternative 1: No Action

Alternative 1 represents future conditions in the SCRE under current VWRF effluent flows and effluent water quality. This alternative provides a basis of comparison of ecosystem functioning of the remaining alternatives below.

Alternative 2: Planned VWRF upgrades
Alternative 2 represents an estimate of future conditions assuming current flows but also including upgrades intended to reduce nutrient concentrations arriving in the SCRE. Upgrades to the VWRF include denitrification to reduce nitrate (NO_3) levels in the effluent discharge below the 10 mg-N/L water quality objective established under the Basin Plan.

Alternative 3: Planned VWRF upgrades with enhanced denitrification
Alternative 3 represents estimated future conditions assuming current flows, including planned VWRF upgrades discussed for Alternative 2 above, but including enhanced denitrification to 5 mg-N/L through the implementation of either onsite or offsite treatment wetlands or other means such as implementation of further VWRF improvements. For the purposes of this assessment, this alternative assumes that the wildlife Ponds will continue to receive 10 mg-N/L from the VWRF, with flows associated with the enhanced denitrification step discharged to the SCRE at 5 mg-N/L.

Alternative 4: VWRF effluent flow reduction
Alternative 4 was developed to assess the effects of planned VWRF effluent water quality improvements to 10 mg-N/L discussed under Alternative 2, combined with decreases in VWRF effluent discharges to the SCRE. The degree of effluent flow reduction was developed iteratively to meet an estimated SCRE Stage of 2.7–3 m (9–10 ft) NAVD88 to maximize both focal species habitat area and to reduce flooding related impacts to recreational area at McGrath State Beach.

Alternative 5: VWRF effluent flow reduction with enhanced denitrification
Alternative 5 was developed to determine the combined benefits of improved VWRF effluent water quality reflected in Alterative 3 along with reduced loading and SCRE stage reduction benefits associated with the VWRF effluent flow reduction reflected in Alternative 4. As with Alternative 3, the wildlife ponds are assumed to receive 10 mg-N/L from the VWRF, but flows associated with the enhanced denitrification step would be discharged to the SCRE at 5 mg-N/L.

Alternative 6: Complete VWRF effluent flow removal
Alternative 6 illustrates conditions under complete removal of the VWRF effluent flow and thus nutrient inputs to the SCRE. Although implementation of this alternative could potentially include elimination of flows to the VWRF wildlife ponds, we have assumed continued make-up water supply for habitat maintenance in the ponds and TIN levels of 10 mg-N/L corresponding to planned VWRF upgrades.

Below, we provide an overall alternative assessment for purposes of maximizing beneficial uses using an approach that focuses on the ecosystem functions most strongly linked to changes in VWRF effluent discharge. We focus on the impacts of VWRF discharge on habitat maintenance and necessary water quality conditions for focal species, while considering, secondarily, impacts of VWRF effluent discharge on recreational opportunities.

11.1 Assessment Approach

The approach developed for assessing ecosystem functions affected by each alternative during a late spring to summer time period included applying the water balance (Section 4.3) and nutrient balance (Section 5.5) to future conditions that accounted for climate change and modifications to VWRF effluent discharge and nutrient concentration. The estimated SCRE stage for each alternative was combined with developed relationships between stage and focal species habitat area (see Sections 7.2 and 8.2) to provide estimates of available habitat area for the focal species selected for this Study.

The impact of climate change within this analysis was reflected in values of future sea level elevation (approximate 0.40 m [1.3 ft] increase in MSL) and air temperatures (approximately 2°C [3.6°F]) increase in average temperatures) projected to 2050. The data presented in Section 10 indicates that the projected increase in average annual precipitation is minimal; suggesting that the average annual groundwater elevation adjacent to the SCRE and the average annual river flow into the SCRE will likely be similar to current conditions. Although increased evaporation of the SCRE has been included in this assessment, increased air temperatures could also result in increased evapotranspiration and a decrease in local groundwater elevation and thus base flows to the river and SCRE during the drier months. We have not attempted to model these impacts and have also not included temperature related effects upon bacterial and algal respiration in the water column.

The critical period for assessing discharge affects on water quality and habitat extent and quality in the SCRE area is during closed-mouth, low-flow conditions in the spring and summer months, when the high relative contribution of VWRF effluent to total inflow volume results in a relatively rapid rate of estuary filling and sustained periods of elevated estuary stage. This condition is also the critical period for assessing impacts to recreation, because the elevated SCRE stage is the stage that has the potential to increase inundation extent and saturation of adjacent floodplain areas, thereby impacting habitat quality and availability for several species and decreasing visitor usage of McGrath State Beach. Although maintenance of the side-channel high flow refuge by the VWRF outfall channel may be shown to provide an enhancement of existing aquatic habitat uses for endangered tidewater goby and steelhead (RARE) during the winter and spring, as discussed in Section 4, conditions in the SCRE are dominated by river flows and the VWRF effluent has little effect upon breaching, flooding, or other potential impacts during winter. For this reason, we have not considered a reduction of effluent discharges during the winter for the alternatives, as it would have little to no effect on water levels or water quality due to the increased occurrences of open mouth conditions. Therefore, our approach to evaluating the alternatives was to focus on the critical summer period by assessing habitat, water quality, and, secondarily, recreation impacts for each alternative under a hypothetical extended closed-mouth period extending from late spring through summer (June through September).

11.2 Caveats and Uncertainty

There are several important caveats to this alternatives assessment that need to be acknowledged in order to put the following results in the proper context. The analytical tools used to assess the alternatives were developed during a relatively "wet" year with higher-than-average rainfall (Sections 4.1 and 4.2), resulting in a considerable amount of river discharge passing through the SCRE and extended periods of tidal exchange through an open mouth. This likely resulted in developed relationships between SCRE stage and groundwater inflow rates and total SCRE inflow rates that do not adequately represent drier, more average conditions. In addition, these alternatives were assessed under idealized conditions assuming future normal flows and a hypothetical SCRE state where the future stage-area relationship will be the same as the current relationship. The analyses presented in Sections 3.2 and 4.2 illustrate the dynamic nature of the SCRE morphology, and how morphologic change is directly linked to ENSO year frequency. It is possible, then, that the future relationship between SCRE stage and area may be different from current conditions if and when the ENSO year frequency changes. Also, there are several positive feedbacks that would, in reality, accompany each alternative that were not considered in this analysis. For example, the effect of sustained flow reduction on wetland vegetation encroachment and altered stage-habitat relationships compared to current conditions was not explicitly included.

Similarly, water quality in the SCRE has been shown to be highly variable (Sections 5.1 and 5.5) and modeling approaches may not capture variations in future conditions on seasonal and shorter time-scales due to variations in lagoon morphology as well as rapid changes in inflows and water levels. These caveats underpin the notion that the results from this analysis are intended to illustrate average conditions that should therefore be viewed relative to each other and not as an absolute prediction of a future SCRE state.

11.3 Estimates of Future Flow Contributions to the Santa Clara River Estuary

The compiled water balance (Section 4.3) was used to provide a hypothesized time series of SCRE stage as a function of inflows and outflows assuming average values based on measured data. As stated above, each alternative was assessed for a single, idealized SCRE state, assuming similar lagoon morphology and baseline flows as those encountered during the 2009–2010 monitoring period. The SCRE inflows and outflows used in the alternatives assessment were for an idealized future dry season (June through September) conditions when the mouth berm would be closed. The SCRE mouth was presumed to remain closed during this period as long as the modeled stage remained below 3.4 m (11 ft) NAVD88, or approximately 0.2–0.3 m (0.5–1 ft) higher than the current equilibrium SCRE stage under low-flow conditions (see Section 4.3).

Assumptions used to develop the flow rates and the average flow rate for each inflow and outflow component used in the alternatives assessment are given below. The inflow and outflow rates for normal conditions were derived from data collected over the past several decades and were adjusted for future conditions based on the climate change predictions developed in Section 10.

Santa Clara River flow
The rate of Santa Clara River flow into the SCRE for all alternatives was derived from the monthly mean flow rates from WY 1981 (the year that had the median total flow volume for all water years between 1927 and 2009 and was therefore a normal water year). These flows were presumed to remain the same under future conditions primarily due to an anticipated minimal increase in future mean annual precipitation over the next several decades (see Section 10).

VWRF effluent flow
The VWRF effluent inflow rate for Alternatives 1 through 3 was developed from first determining the average monthly ETS flow since 1984, and then using the ratio of ETS flow to VWRF effluent flow for current conditions (i.e., October 2009 through September 2010) to reduce the ETS flow value. The future VWRF effluent flow was set at the current average flow rate for each month. For Alternatives 4 and 5, the VWRF effluent flow rate was reduced to values that would decrease the threshold SCRE stage associated with the onset of ground saturation at McGrath State Beach during dry season, closed-mouth conditions, but at the same time would not have a negative impact on the habitat available to focal species. Anecdotal information and observations made during this study suggest that the threshold stage is approximately 2.9 m (9.5 ft) NAVD88, or approximately 0.3 m (1 ft) below the "full" stage (see Section 4.3). Using an iterative process, we determined that reducing the estimated future VWRF effluent flow by 30% results in a steady-state stage that remains below that threshold stage.

Evaporation
Evaporation was determined from combining the median monthly SCRE evaporation rates over the period of record (1972–2010) with an estimate of the average increase in evaporation due to anticipated increases in future air temperatures. In order to ascertain the increase in evaporation,

the Linacre (1992) equation was used to get a relative change in evaporation. The Linacre (1992) equation calculates mean monthly evaporation and can be expressed as:

$$E_o = (0.015 + 0.000042T + 10^{-6z})(0.8R_s - 40) + 2.5Fu(T - T_d)$$

where E_o is mean monthly evaporation, T is the daily mean air temperature, z is elevation, R_s is solar irradiance, F is a function of z, u is wind speed, and T_d is dew point temperature. Assuming a mean monthly air temperature increase on the order of 2°C (3.6°F) (i.e., the anticipate increase in annual average air temperature by 2050 as described in Section 10) and holding all other variables constant results in a 4% increase in monthly evaporation compared to existing conditions. The median monthly evaporation rates were therefore increased by 4% to arrive at the future evaporation rates. Lastly, evaporative losses associated with implementation of on-site or off-site treatment wetlands were not assessed as part of this analysis.

Subsurface flow through the mouth berm

The flow through the closed-mouth berm was derived from the hydraulic variables determined as part of the water balance analysis and the gradient between the SCRE stage and the adjacent tidal elevation. A future tidal elevation time series was compiled using current normal tidal elevations combined with the anticipated increase in mean sea level. The current normal tidal elevations for each month were determined using the tidal time series for June 2010 through September 2010 and adjusting the elevations according to how much the mean monthly elevation differed from the long-term value. The current normal tidal elevations were then increased by 0.41 m (1.35 ft), or the average of the range of values given in Section 10 for anticipated sea level rise.

Groundwater flows

Similar to the subsurface flow through the mouth berm, the groundwater inflows and outflows were derived from the calculated hydraulic variables combined with local hydraulic gradients. For groundwater flow across the south bank at McGrath State Beach, the hydraulic gradient was derived from the calculated SCRE stage and an assumption that the water table elevation remained fixed at an elevation of 2.0 m (6.5 ft) NAVD88 and that flow would be directed out of the SCRE when the stage was above this elevation. This assumption regarding the change in gradient direction was based on the data collected during the water balance period. For the groundwater flow from the VWRF ponds, the hydraulic gradient was derived from the calculated SCRE stage and an assumed constant VWRF pond surface elevation that is the same as current conditions (5.8 m [19 ft] NAVD88). For the unmeasured groundwater flow, the amount of inflow was estimated from the calculated stage and the relationship between flow rate and SCRE stage developed in the water balance for spring-summer conditions (see Section 4.3). As the relationship between unmeasured groundwater flow and SCRE stage was developed during a wet year, it likely over-estimates the flow rate somewhat during a "normal" year when the water table is presumably lower. However, the developed relationship is the only tool currently available to assess unmeasured groundwater flow and was deemed appropriate for use in this analysis.

Average SCRE source flow estimates by discharge Alternative

Table 11-1 summarizes the calculated average flow rates used to assess each alternative, with SCRE stage during a hypothetical 4-month filling period beginning June 1 shown in Figure 11-1. Under the idealized future normal flow conditions associated with Alternatives 1 through 3, the maximum modeled stage remained below the presumed breaching elevation of 3.4 m (11 ft) NAVD88. The modeled SCRE filling rate closely matched observations during 2009–2010, in which the SCRE exhibited a relatively rapid stage increase during the first month following mouth closure. For Alternatives 4 and 5, reducing the VWRF flows by 30% resulted in near

equilibrium stage of approximately 2.9 m (9.5 ft) NAVD88 whereas complete VWRF flow removal resulted in an equilibrium SCRE stage of approximately 2.4 m (8 ft) NAVD88 under Alternative 6.

Although not shown in Figure 11-1, the impact of removal of all VWRF flow (VWRF effluent discharge and VWRF pond percolation) on SCRE stage and volume was also assessed during the Alternative 6 modeling runs. Removal of the pond flow resulted in essentially the same rate of filling as shown in Figure 11-1 for Alternative 6, but the maximum equilibrium stage reached was slightly less (2.3 m [7.7 ft] NAVD88 vs. 2.4 m [8.0 ft] NAVD88) without the pond percolation flow included.

Table 11-1. Estimated average future summertime flows by VWRF discharge alternative in millions of gallons per day (MGD).

Discharge alternative	Surface water flow contributions					Groundwater flow contributions		
	Santa Clara River	VWRF Discharge	Precipitation	Evaporation	Berm Outflow	South Bank	VWRF ponds	North Bank and other sources
Alternative No.s 1–3	0.4	5.0	0	-0.7	-4.0	-0.2	0.9	0.4
Alternative No.s 4–5	0.4	3.5	0	-0.6	-3.1	-0.2	1.0	0.6
Alternative No. 6	0.4	0	0	-0.6	-2.1	-0.4	1.1	2.3

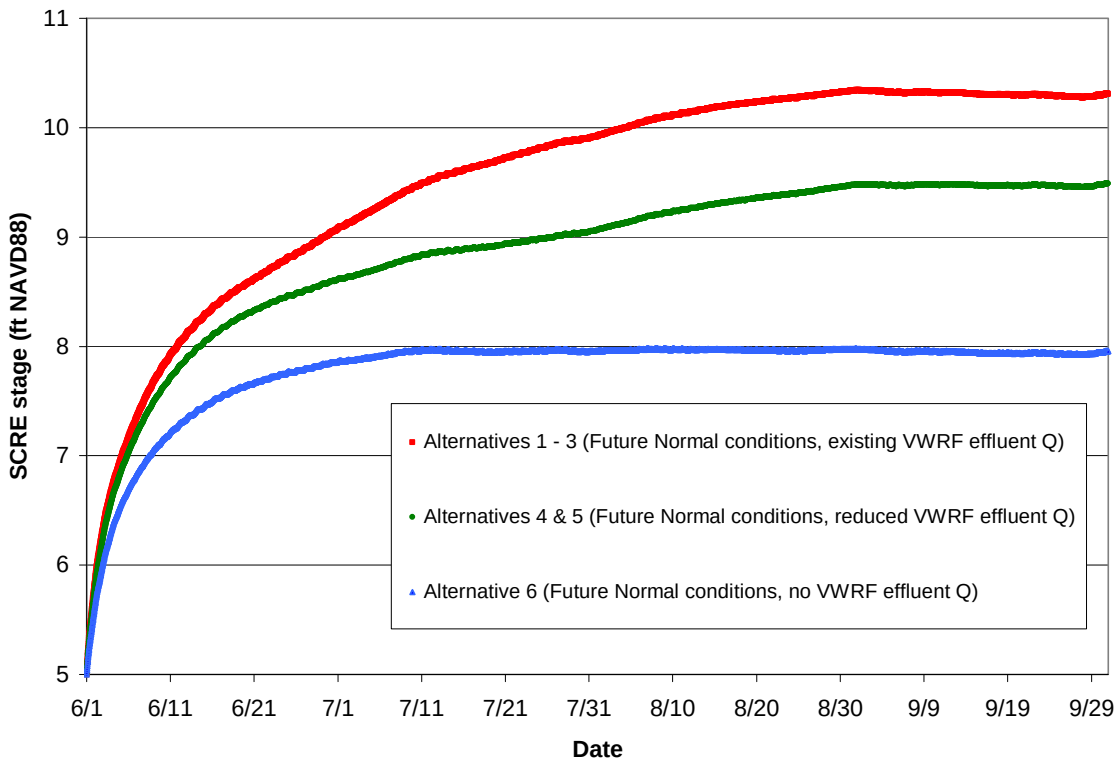


Figure 11-1. Modeled SCRE stage times series for each alternative.

11.4 Estimates of Future Water Quality Conditions

Estimates of future water quality conditions under the alternatives described above were based upon both quantitative and qualitative approaches. Using the nutrient balance developed in Section 5.5 and seasonal average flows of the primary water balance components (Section 11.3), future nutrient loads to the SCRE were estimated. The analysis below centers upon variations in the amount of NH_4^+ and NO_3^- (i.e., TIN) arriving in the SCRE due to flow reductions and various onsite and offsite treatment alternatives such as wetlands denitrification (Carollo Engineers 2010b), or the implementation of additional upgrades to the VWRF unit processes.

Santa Clara River Inflows

As detailed in Section 5, TIN levels arriving in the SCRE from the Santa Clara River were primarily comprised of NO_3 and have historically ranged from 5.5–6.4 mg-N/L (Tables 5-8 and 5-9). For the purposes of developing future TIN levels in the SCRE, we have assumed that TIN arriving from riverine sources will average 6 mg-N/L during the spring/summer months for all alternatives (Table 11-2).

VWRF Inflows

Some improvements in water quality conditions within the SCRE are expected to result from completion of planned implementation of denitrification wastewater treatment processes at the VWRF under Alternatives 2 and 4, which are expected to reduce NO_3^- and TIN levels in the discharge to the SCRE below the 10 mg-N/L water quality objective established under the Basin Plan. Under Alternatives 3 and 5, we have assumed that summertime NO_3^- levels from either an

onsite or offsite treatment wetland or improvements to the treatment processes could reduce TIN levels in the effluent to 5 mg-N/L. Because the flow configuration of this step has not been established, we have assumed the VWRF wildlife ponds would receive 10 mg-N/L as described for Alternatives 2 and 4, with the ponds being used as source water for enhanced denitrification. That is, all underflow from the wildlife ponds would reach the SCRE at 10 mg-N/L, with all surface water discharges treated to 5 mg-N/L. Lastly, as discussed above, Alternative 6 assumes no flows arrive in the SCRE from the VWRF or the ponds.

Ground water sources

Overall, nutrient levels in groundwater were found to be low in the SCRE floodplain, based upon monitoring well sampling conducted in 2009–2010 (Appendix B). For all alternatives, we have assumed TIN levels in groundwater are on the order of 1 mg-N/L.

TIN uptake and removal within the SCRE

Based upon summertime observations of lower NO_3^- and TIN levels than at other times of year under closed-mouth conditions (Section 5.1.3) and levels below the major sources to the SCRE (e.g., VWRF and Santa Clara River inflows), it is apparent that some combination of algal uptake and denitrification effectively reduces TIN levels during summer months. We have assumed conservative rates of TIN removal of 50–200 mg-N/m^2-d on an areal basis based upon removal estimates of 79–359 mg-N/m^2-d due to denitrification only in other estuaries (Seitzinger 1988, Horne 1995).

Table 11-2 shows estimates of future TIN levels in the SCRE using the assumptions above along with future flow estimates (Section 11.3) and the nutrient balance approach developed in Section 5.5. For each alternative evaluated here, future TIN levels were estimated by summation of the total of all loads arriving from each source and removed by algal uptake and denitrification (Table 11-2). Next, the total SCRE TIN loading was divided by the total water volume represented by the sum of SCRE outflows and storage terms from the water balance (Table 11-1) to arrive at an estimate of average TIN levels expected under the alternative (Table 11-2). In circumstances when the estimated TIN levels were low, we have assumed a practical lower limit of 2 mg-N/L due to the potential for additional nitrification to NO_3^- from mineralization of algal and plant detritus in the SCRE.

Table 11-2. Estimated average future summertime TIN loading and SCRE concentration by VWRF discharge alternative.

Source	Parameter	Alt. No. 1	Alt. No. 2	Alt. No. 3	Alt. No. 4	Alt. No.5	Alt. No. 6
SCR inflow	(MGD)	0.3	0.3	0.3	0.3	0.3	0.3
	(mg-N/L)	6	6	6	6	6	6
	(lb-N/day)	15	15	15	15	15	15
VWRF inflow	(MGD)	5.0	5.0	5.0	3.5	3.5	-
	(mg-N/L)	15	10	5	10	5	-
	(lb-N/day)	620	420	210	290	150	-
VWRF Pond Groundwater inflow	(MGD)	0.9	0.9	0.9	1.0	1.0	1.1
	(mg-N/L)	15	10	10	10	10	10
	(lb-N/day)	110	73	73	79	79	88
Unmeasured Groundwater inflow	(MGD)	-	-	-	-	-	1.7
	(mg-N/L)	1	1	1	1	1	1
	(lb-N/day)	-	-	-	-	-	14.3
Total Mass Loading	lb-N/day	750	500	300	390	240	120
Estimated SCRE Area	acres	174	174	174	151	151	131
Potential N-removal by algal uptake or denitrification (50-100 mg/m^2-day)	Low (lb-N/day)	78	78	78	68	68	63
	High (lb-N/day)	160	160	160	140	140	130
Estimated SCRE TIN Levels	**Low (mg-N/L)**	***10***	***4***	***2***	***3***	***2***	***2***
	High (mg-N/L)	***15***	***9***	***6***	***9***	***6***	***6***

As stated in the development of the nutrient balance (Section 5.5) and illustrated in Table 11-2, future TIN levels will rapidly approach the largest flow and thus load contribution to the SCRE under the future discharge conditions. That is, TIN levels of the SCRE will approach levels in the VWRF effluent discharge for Alternatives 1 through 5 and will approach the levels found in the Santa Clara River for Alternative 6. Assuming implementation of enhanced denitrification under Alternatives 3 and 5, or complete flow removal (Alternatives 6), TIN levels in the SCRE will approach those found in the Santa Clara River.

It should be noted that reductions in NO_3^- alone may not result in reduced algal levels or improve DO conditions within the SCRE due to the apparent phosphorus limitations upon algal growth (Section 5.4). That is, algal levels may not decrease at all times of year unless nutrient levels in the SCRE fall below typical saturation levels for algal growth (200 ug/L for inorganic nitrogen and from 20–80 ug/L for orthophosphate) (Horne and Goldman 1994). Given nutrient levels in

the Santa Clara River are well above these saturating conditions (Tables 5-7 through 5-9), continued algal bloom episodes are likely to occur under all alternatives. Nevertheless, it should be expected that the frequency and duration of algal blooms should decrease with reduced TIN levels approaching the 1.5–4.5 mg-N/L range identified for algal growth limitation (Section 5.4). Although PO_4 levels are elevated in the SCRE, TIN load reductions have been associated with reductions in algal blooms other southern California estuaries (Sutula et al 2006).

11.5 Estimates of Future Habitat Conditions

Changes in habitat area under various VWRF effluent discharge alternatives were based upon the stage vs. habitat relationships developed in Section 7 for focal fish species (Figures 7-1 and 7-3) and in Section 8 for focal bird species (Figures 8-2, and 8-4).

Southern California steelhead
Figure 7-1 shows steelhead habitat availability to increase with rising stage. As noted in Section 7.2, the SCRE may provide valuable rearing habitat for juvenile steelhead, for at least short periods of time, particularly when water quality conditions are favorable. For this reason, steelhead habitat conditions were assessed by the provision of inundated area and depth (as it corresponds to habitat availability with rising stage), as well as by the potential for discharge management to reduce the frequency of low DO conditions that have occurred in the SCRE.

Tidewater goby
Figure 7-3 shows that tidewater goby area generally increases with rising SCRE stage, but is relatively unchanged at depths of 2.4–3 m (8–10 ft) NAVD88 and actually decreases at depths above 3 m (10 ft) NAVD88. Since tidewater goby are relatively insensitive to changes in salinity and DO, habitat conditions were primarily assessed by changes in habitat area available, as well as potential reductions in habitat suitability for predator fish species at higher salinity levels.

Western snowy plover
Figure 8-2 shows that nesting habitat area for western snowy plover decreases with increasing SCRE stage above 3 m (10 ft) NAVD88 due to inundation of open areas with bare sand or low-stature vegetation (Section 8.1.1.4), though some consideration must be given to the fact that permanently decreased stage results in encroachment of upland vegetation and reduced habitat suitability for nesting. Since plover are unaffected by changes in water quality conditions, habitat conditions were assessed primarily using this stage vs. nesting habitat area relationship.

California least tern
Figure 8-4 shows that nesting habitat area for California least tern decreases with increasing SCRE stage above 3 m (10 ft) NAVD88 due to inundation of open sand areas along the SCRE perimeter (Section 8.2.2). However, foraging habitat within the SCRE is associated with inundated area, which generally increases with SCRE stage. Because algal blooms may affect sight feeding success of least tern foraging within the SCRE lagoon, habitat conditions were assessed using this stage vs. habitat area relationship, as well as by the potential for discharge and water quality management to reduce nutrient loads and the frequency of algal bloom conditions. However, because algal blooms are ephemeral and patterns subject to a range of factors, no attempt was made to directly predict changes in suitable foraging area due to increases or decreases in nutrient loading to the SCRE.

11.6 Estimates of Future Recreational Opportunities

Although there are a number of recreational opportunities associated with visitors to McGrath State Beach that are largely unaffected by SCRE stage (Section 9.2.6), we have limited our evaluation to changes in the frequency and duration of flooding conditions affecting the campground areas at a SCRE stage of approximately 3 m (10 ft) NAVD88, and above rather than attempting to identify and assess specific impacts of discharge to other recreational uses of the SCRE. The amount of total time the modeled stage was over the threshold stage associated with the onset of ground saturation within the campground (2.9 m [9.5 ft] NAVD88) was used as an indicator of the alternative's impact on recreation area, and therefore on human recreation.

11.7 Alternatives Assessment Results and Discussion

11.7.1 Alternative 1: No action

11.7.1.1 Focal species habitat and water quality requirements

Combining the modeled stage time series for the VWRF effluent discharge alternatives (Figure 11-1) with the developed relationships between stage and focal species habitat area from Section 7.2 and 8.2 clearly shows how habitat availability changes as stage increases while the SCRE fills following the hypothetical mouth closure on June 1st. However, nutrient loads in continued VWRF effluent discharges through the closed-mouth period may be contributing to periods of anoxia/hypoxia in some locations of the SCRE due to algal growth, with these conditions exacerbated in locations where wind mixing is minimal. Below, we detail habitat conditions for each of the focal species.

Southern California steelhead
Estimated steelhead habitat as a function of SCRE stage increased as the SCRE filled to the highest stages modeled (approximately 3.1 m [10.3 ft] NAVD88), which was reached approximately one month before the end of the model simulation period (Figure 11-2). However, water quality conditions for steelhead limit its value as rearing habitat in the SCRE at certain times during summer due to the high potential for algae blooms to occur during closed mouth conditions. In addition, despite compliance with discharge effluent limitations, nutrient loading from continued VWRF effluent discharges may be contributing to periods of anoxia/hypoxia in some locations of the SCRE. Extended discharge related freshwater flows may also be associated with increased predator populations that are adapted to more warm and fresh water conditions. Further, higher water levels in the SCRE may be contributing to episodic man-made breaches during summer and fall months. These events may cause fish stranding, thereby limiting the suitability of the SCRE for rearing steelhead. Comparison to the no discharge alternative (Alternative 6) is provided in Section 11.7.6.

Tidewater goby
Like steelhead, estimated tidewater goby habitat as a function of SCRE stage increased as the SCRE filled, but reaches its maximum value only two weeks after the start of the model simulation period when the SCRE was still filling rapidly (Figure 11-3). Estimated tidewater goby habitat area remained close to this maximum for the remainder of the simulation period. Although tidewater goby are not considered to be particularly sensitive to existing water quality conditions, current discharges may provide increased habitat for warm or fresh water predator species, and higher water levels in the SCRE may contribute to episodic man-made breaches during summer and fall months. While discharges facilitate the creation of side channel refuge habitat protecting fish in breach events, dry weather breach events may cause rapid dewatering of

burrows or fish stranding. Comparison to the no discharge alternative (Alternative 6) is provided in Section 11.7.6.

Western snowy plover

Estimated western snowy plover habitat area decreased with increasing SCRE stage (Figure 11-4), but the total amount of habitat decrease over the entire simulation period was relatively small. Overall, habitat area decreased by approximately 3% over the first 2 months of the model simulation period and decreased by approximately 8% over the entire period. The snowy plover shows no sensitivity or adverse reaction to current water quality conditions, which would not change under the No Action alternative. Comparison to the no discharge alternative (Alternative 6) is provided in Section 11.7.6.

California least tern

Although estimated California least tern nesting habitat decreased slightly over the entire period with increasing SCRE stage, foraging habitat increased with increasing stage. Foraging habitat reached its maximum value less than two weeks after the start of the model simulation period (Figure 11-5). Estimated least tern foraging habitat area remained close to this maximum throughout the remainder of the simulation period. Although least tern shows no sensitivity or adverse reaction to current water quality conditions, sight feeding success may be impaired by periodic development of algal blooms or floating mats associated with excess nutrients in the SCRE. Comparison to the no discharge alternative (Alternative 6) is provided in Section 11.7.6.

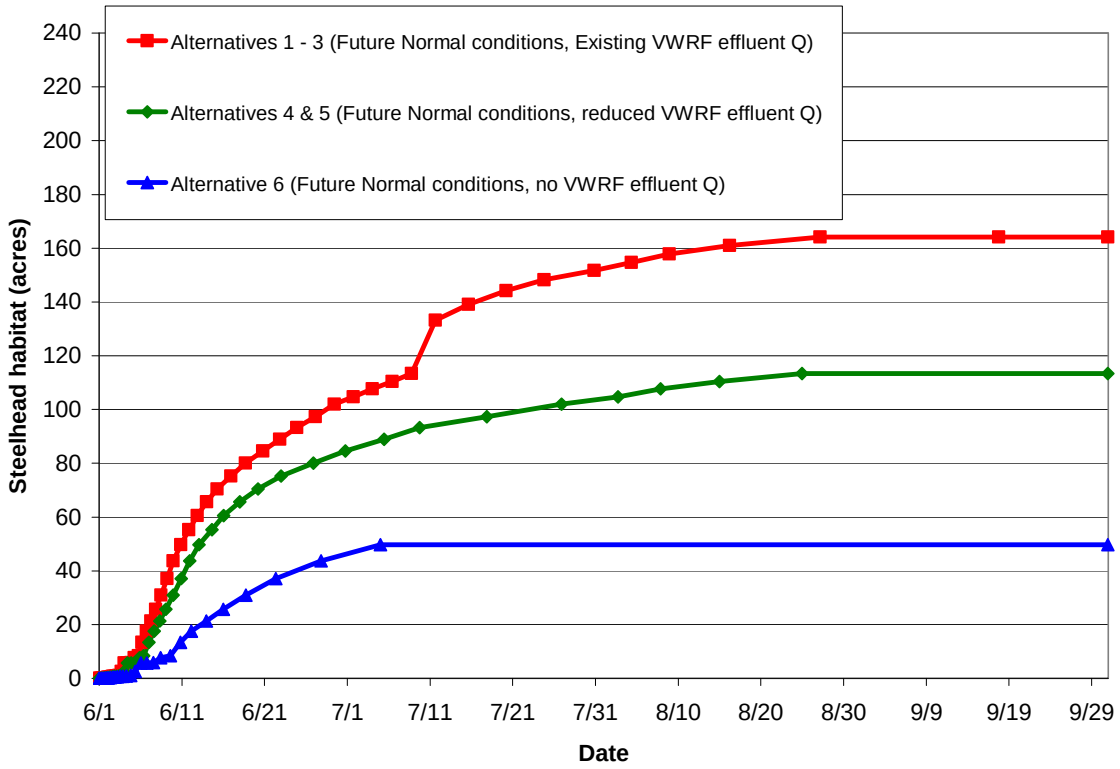


Figure 11-2. Modeled steelhead habitat area for each alternative.

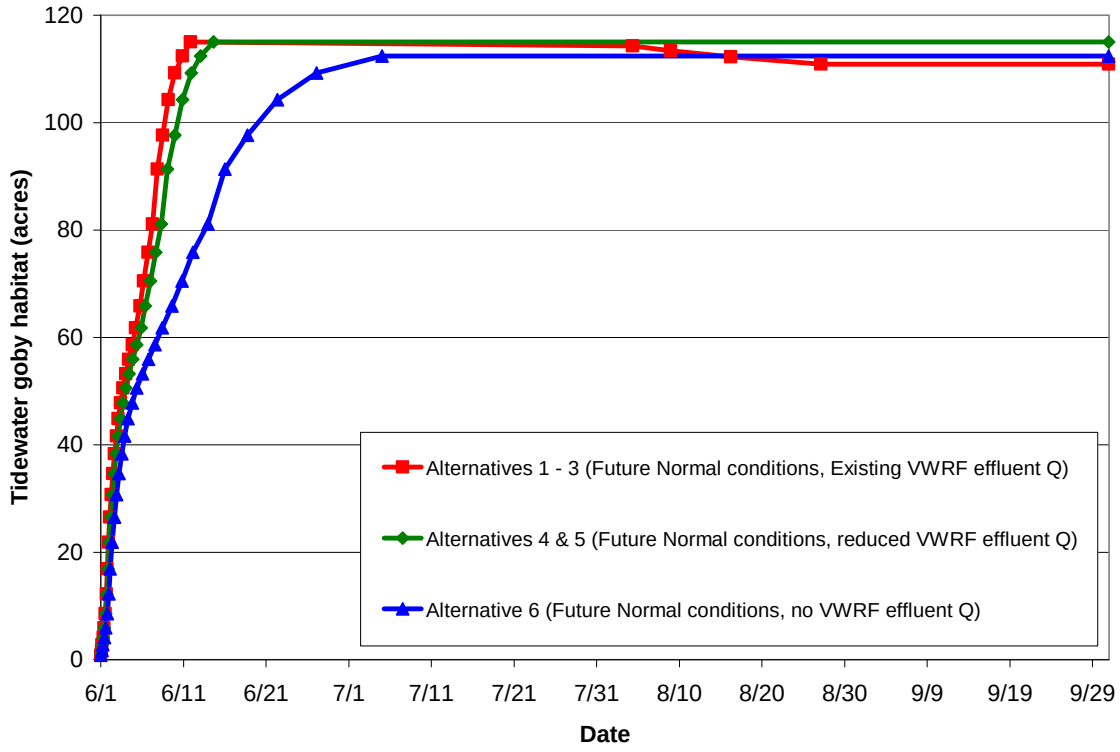


Figure 11-3. Modeled tidewater goby habitat area for each alternative.

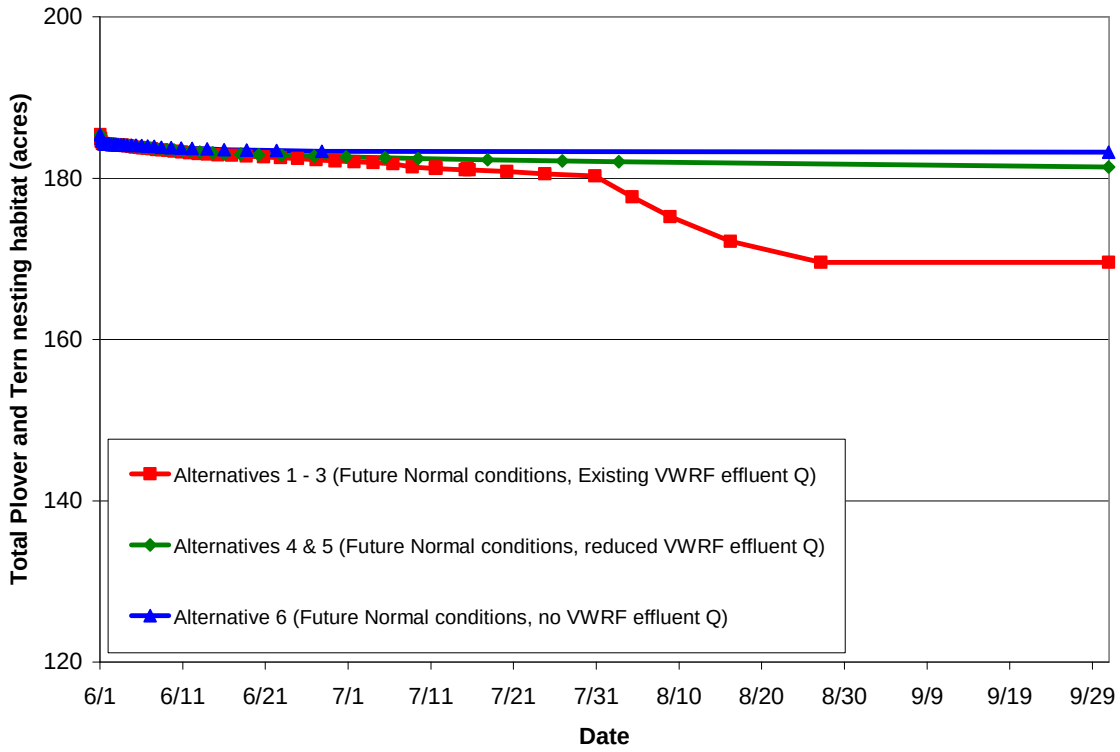


Figure 11-4. Modeled western snowy plover and California least tern nesting habitat area for each alternative.

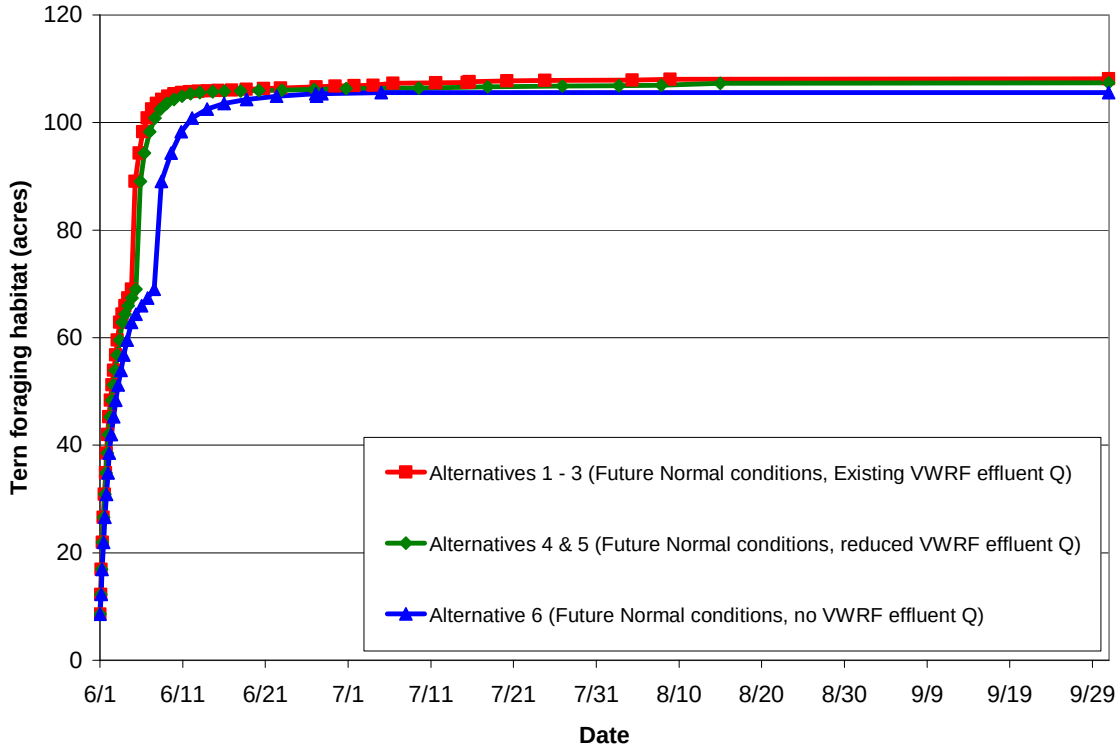


Figure 11-5. Modeled California least tern foraging habitat area for each alternative.

11.7.1.2 Human recreation

For Alternative 1, the stage was above 2.9 m (9.5 ft) NAVD88 for approximately 70% of the model simulation period, suggesting that flooding conditions in the McGrath State Beach campground that reduce recreational area may occur by mid-July under the No Action alternative, or less than two months after the beginning of the model simulation period (Figure 11-1). Comparison to the no discharge alternative (Alternative 6) is provided in Section 11.7.6.

11.7.2 Alternative 2: Planned VWRF upgrades

11.7.2.1 Focal species habitat and water quality requirements

The modeled SCRE stage for Alternative 2 is shown in Figure 11-1. The estimated focal species habitat area time series for Alternative 2 was the same as for Alternative 1 (Figures 11-2 through 11-5). Following implementation of planned improvements to the VWRF, additional reductions in the discharge contribution to SCRE nutrient loads have the potential to decrease algal growth and to provide a well-oxygenated and food-rich environment. This in turn, has the potential to create higher quality habitat for juvenile steelhead rearing, which may relax to some degree the density-dependent bottleneck occurring in stream habitat discussed in Section 7.2.1, potentially improving steelhead production within the upper watershed. However, continued discharges at current levels may also provide increased habitat for fresh water predator species. While existing discharges facilitate the creation of side channel refuge habitat protecting tidewater goby during breach events, higher water levels in the SCRE may contribute to episodic man-made breaches

during summer and fall months that may cause fish stranding and disturbance of goby spawning burrows.

Although improved water quality conditions under Alternative 2 associated with treatment upgrades would not directly improve habitat conditions for Western snowy plover due to their relative insensitivity to water quality conditions, anticipated nutrient load reductions may reduce the frequency, duration and extent of algal blooms that occur under current conditions. These water quality improvements may directly benefit foraging success of least terns using the SCRE lagoon.

11.7.2.2 Human recreation

For Alternative 2, the amount of time the modeled SCRE state was above 2.9 m (9.5 ft) NAVD88 was the same as for Alternative 1 (Figure 11-1) and it is apparent that periodic of flooding at McGrath State Beach may continue to occur.

11.7.3 Alternative 3: Planned VWRF upgrades with enhanced denitrification

11.7.3.1 Focal species habitat and water quality requirements

The modeled SCRE stage for Alternative 3 is shown in Figure 11-1. The estimated focal species habitat area time series for Alternative 3 was the same as for Alternative 1 (Figures 11-2 through 11-5). Further reductions in SCRE nutrient levels through the implementation of on-site or off-site treatment wetlands may further reduce contribution of nutrient loads and the potential for algal blooms; thereby further improving growth conditions for rearing steelhead. However, denitrification that can be achieved by treatment wetlands is quite variable since treatment efficiency is dependent on ambient water temperatures, wetland planting density and other variations associated with a natural treatment system (Carollo Engineers 2010b). For this reason, enhanced denitrification may not reliably and substantially reduce SCRE TIN levels beyond that achieved by planned treatment upgrades. Also, while discharges facilitate the creation of side channel refuge habitat protecting tidewater goby in breach events, elevated SCRE water levels associated with man-made breach events would continue to occur as discussed under Alternatives 1 and 2.

Although improved water quality conditions under Alternative 3 associated with treatment upgrades would not directly improve habitat conditions for Western snowy plover due to their relative insensitivity to water quality conditions, anticipated nutrient load reductions may reduce the frequency, duration and extent of algal blooms that occur under current conditions. These water quality improvements may directly benefit foraging success of least terns using the SCRE lagoon.

11.7.3.2 Human recreation

For Alternative 3, the amount of time the modeled SCRE stage was above 2.9 m (9.5 ft) NAVD88 was the same as for Alternative 1 and 2 (Figure 11-1) and it is apparent that periods of flooding at McGrath State Beach may continue to occur.

11.7.4 Alternative 4: VWRF effluent flow reduction

11.7.4.1 Focal species habitat

Decreasing the VWRF effluent discharge by 30% caused the SCRE stage to increase somewhat less rapidly compared to the modeled normal effluent discharge rate from Alternatives 1 through 3, and resulted in a maximum SCRE stage of approximately 2.9 m (9.5 ft) NAVD88 (Figure 11-1). As discussed below, when compared to the results from Alternatives 1 through 3, the decreased VWRF effluent discharge did not have the same effect on habitat area for all focal species.

Southern California steelhead
As steelhead habitat area is driven primarily by stage, it follows that the decreased modeled stage associated with Alternative 4 resulted in a slower rate of habitat area increase and also a reduced overall habitat area, approximately 30% less than that for Alternatives 1 through 3 (Figure 11-2). As with Alternative 2, however, improved water quality conditions in the SCRE following planned VWRF upgrades may improve growth conditions for rearing steelhead. It is anticipated that VWRF flow reductions may result in reduced spawning areas for fresh water predator species, and increased lagoon salinity may further reduce habitat suitability for fresh water predators to rearing steelhead. Reduced SCRE water levels will also reduce man-made breach events and potential stranding impacts to rearing steelhead.

Tidewater goby
Because tidewater goby habitat area is relatively insensitive to changes in SCRE Stage between 2.4 and 3 m (8 and 10 ft) NAVD88, the rate of increase and the maximum modeled tidewater goby habitat was essentially the same as that for Alternatives 1 through 3 (Figure 11-3). As with steelhead discussed above, reduced SCRE water levels will result in reduced occurrence of man-made breach events and potential dewatering of goby burrows or stranding, while still providing for creation of side channel habitat that may serve as high flow refuge. Although tidewater goby are relatively insensitive to improved water quality conditions associated with upgraded nutrient treatment, it is anticipated that VWRF flow reductions may result in reduced spawning areas for warm-water predator species, and increased lagoon salinity may further reduce habitat suitability for predators to tidewater goby.

Western snowy plover
Estimated western snowy plover habitat area increases for Alternative 4 were essentially the same as Alternatives 1 through 3 for the first 2 months of the model simulation period, but the maximum habitat area reached by the end of this period was approximately 6% greater than that for Alternatives 1 through 3 (Figure 11-4). It is not anticipated that the improved water quality conditions under Alternative 4 associated with treatment upgrades would greatly improve habitat conditions for the snowy plover substantially due to their relative insensitivity to water quality conditions.

California least tern
The estimated California least tern nesting habitat area for Alternative 4 increases were the same as the tern nesting habitat area increases under Alternative 4 for the entire model simulation period (Figure 11-4). California least tern foraging habitat increases were also essentially the same as Alternatives 1 through 3, reaching the maximum value less than 2 weeks after the start model simulation (Figure 11-5). Improved water quality conditions under Alternative 4 associated with treatment upgrades would potentially improve habitat conditions for tern by reducing the

frequency, duration and extent of algal blooms that occur under current conditions. These water quality improvements may directly benefit foraging success of least terns using the SCRE lagoon.

11.7.4.2 Human recreation

As previously discussed, Alternative 4 was designed so that the reduced VWRF effluent discharge would result in a modeled SCRE stage that did not rise above the 2.9 m (9.5 ft) NAVD88 saturation threshold stage (Figure 11-1). Therefore, the stage was above 2.9 m (9.5 ft) NAVD88 for 0% of the model run and it is assumed that the potential for flooding at McGrath State Beach campground would be low under this alternative.

11.7.5 Alternative 5: VWRF flow reduction and enhanced denitrification

11.7.5.1 Focal species habitat

The modeled SCRE stage for Alternative 5 is shown in Figure 11-1. The estimated focal species habitat area increases over the time series were the same as for Alternative 4 (Figures 11-2 through 11-5). As indicated in Alternatives 2 and 3, further reductions in SCRE nutrient levels may improve growth conditions for rearing steelhead. Reductions in freshwater inputs and breach events will be associated with improved conditions for steelhead and tidewater goby, as discussed above for Alternative 4, and continued discharge would likely maintain the existing provision of side channel refuge for gobies in the VWRF outfall channel.

Although improved water quality conditions under Alternative 5 associated with treatment upgrades would not directly improve habitat conditions for Western snowy plover due to their relative insensitivity to water quality conditions, anticipated nutrient load reductions may reduce the frequency, duration and extent of algal blooms that occur under current conditions. These water quality improvements may directly benefit foraging success of least terns using the SCRE lagoon.

11.7.5.2 Human recreation

The amount of time the modeled SCRE stage was above 2.9 m (9.5 ft) NAVD88 under Alterative 5 was the same as for Alternative 4 (Figure 11-1) and it is assumed that the potential for flooding at McGrath State Beach campground would be low under this alternative.

11.7.6 Alternative 6: Complete VWRF effluent flow removal

11.7.6.1 Focal species habitat

Complete removal of the VWRF effluent discharge resulted in a modeled SCRE stage time series that increased less rapidly than what was modeled for Alternatives 1 through 5, and resulted in a maximum equilibrium SCRE stage of approximately 2.4 m (8 ft NAVD88), or a maximum stage approximately 0.8 m (2.5 ft) less than that for Alternatives 1 through 3 and approximately 0.5 m (1.5 ft) less than that for Alternatives 4 and 5 (Figure 11-1). When compared to the results from Alternatives 1 through 5, the elimination of VWRF effluent discharge did not have the same impact on habitat conditions for all focal species.

Southern California steelhead
Under a complete flow removal scenario, potential steelhead rearing habitat area would be substantially reduced. Maximum steelhead habitat area for Alternative 6 is lower than

Alternatives 1 through 5; approximately 70% lower than the value for Alternatives 1 through 3 and approximately 56% lower than the value for Alternatives 4 and 5 (Figure 11-2). Water quality conditions for steelhead would improve under a complete flow removal scenario as compared to existing conditions (Alternative 1) and SCRE water quality would be entirely controlled by Santa Clara River water quality during closed mouth conditions. However, Alternative 6 is only expected to improve nutrient levels in the SCRE to the same degree as anticipated under Alternatives 3 or 5 (per Table 11-2). Further, improvements in water quality under Alternative 6, as with those anticipated under Alternative 5, may not offset anticipated reductions in steelhead rearing habitat area. Nevertheless, reductions in freshwater inputs from the VWRF as well as reductions in the occurrence in man-made breach events will be associated with improved conditions for steelhead as discussed above.

Tidewater goby

Under Alternative 6, the maximum tidewater goby habitat area was the same as for Alternatives 1 through 5, but the rate of habitat area increase at the beginning of the model simulation period was much less (Figure 11-3). Reductions in freshwater inputs from the VWRF may also reduce the occurrence of man-made breach events, which may result in reduced spawning areas for warm-water predator species, and increased lagoon salinity may reduce habitat suitability for predators to tidewater goby. However, the elimination of the VWRF discharge will likely result in vegetation encroachment and the elimination of the outfall channel that may provide valuable side channel habitat used by tidewater goby during high flow events. Although it is unknown whether tidewater goby, which are found in low numbers throughout the SCRE (Entrix 2002b, 2004), will find suitable low velocity refuge during high flow events in the absence of the VWRF outfall channel, the extent of emergent vegetation along the SCRE shoreline suggests that suitable low velocity refuge during high flows may be available under a complete flow removal alternative.

Western snowy plover

The rate of decrease and the maximum modeled western snowy plover habitat area for Alternative 6 was essentially the same as for Alternatives 4 and 5 (Figure 11-4). It is not anticipated that the improved water quality conditions under Alternative 6 as compared to Alternatives 1, 2 and 4 would improve habitat conditions for the snowy plover due to their relative insensitivity to water quality conditions.

California least tern

The rate of decrease and the maximum modeled California least tern nesting habitat area for Alternative 6 was essentially the same as for Alternatives 4 and 5 (Figure 11-5). However, the rate of increase and the maximum modeled California least tern foraging area for Alternative 6 was essentially the same as that for Alternatives 1 through 5. Improvements in water quality conditions under Alternative 6 would improve habitat for least tern in comparison to Alternatives 1, 2 and 4 by reducing the frequency, duration and extent of algal blooms. These water quality improvements may directly benefit foraging success of least terns using the SCRE lagoon.

11.7.6.2 Human recreation

As the maximum modeled stage for Alternative 6 was approximately 2.4 m (8 ft) NAVD88, the saturation threshold stage was not reached and it is assumed that the impact to human recreation would be minimal under this alternative in comparison to current conditions (Alternative 1).

11.8 Discussion

Based upon the results above, maintenance of current discharge under the No Action alternative (Alternative 1) will continue to be associated with periods of unsuitable DO levels attributable to excess nutrient loading to the SCRE, as well as continued episodes of flooding and man-made breach events that may affect the focal species selected for this study. Alternatives 2 through 5 provide some benefit to focal species habitat and human recreational uses within and adjacent to the SCRE. However, decreases in SCRE stage have been shown to limit potential rearing habitat for steelhead, suggesting that the VWRF flows provide some degree of enhancement relative to the expected lower SCRE stage under a complete flow removal alternative. Table 11-3 shows a simplified ranking of VWRF effluent discharge alternatives based upon the ecosystem functions used for this evaluation. We have identified the degree of expected changes in each ecosystem function relative to the same function under the No Action alternative. In addition, we provide a brief summary discussion of each alternative in relation to both the No Action alternative, including assessment of a complete flow removal under Alternative 6.

Table 11-3. Qualitative comparison of focal species' habitat conditions and recreational opportunities to the 'No Action' alternative.

Alternative	Focal species habitat					Recreational opportunities
	Habitat area				Water quality conditions	
	Tidewater goby	Steelhead	Plover & tern nesting	Tern foraging		
Alternative 1 Existing Conditions	=	=	=	=	=	=
Alternative 2 VWRF upgrades	=	=	=	=	↑	=
Alternative 3 VWRF upgrades and enhanced denitrification	=	=	=	=	↑↑	=
Alternative 4 VWRF flow reduction	=	↓	↑	=	↑	↑
Alternative 5 VWRF flow reduction and enhanced denitrification	=	↓	↑	=	↑↑↑	↑
Alternative 6 Complete VWRF effluent flow removal	=	↓↓	↑	=	↑↑↑	↑↑

Note: = indicates no change, ↑ indicates improved conditions, ↓ indicates reduced habitat availability or quality

Alternative 1: No Action

As stated above, although current SCRE water levels benefit steelhead under the No Action alternative due to increased habitat area availability, continued VWRF effluent discharges may be associated with periods of anoxia/hypoxia in some locations of the SCRE due to excess nutrient loading. Algal bloom conditions may at times impair sight feeding success of least terns using the

SCRE lagoon for foraging. That is, suitable open water habitat for rare, threatened, or endangered species (RARE) represented by steelhead rearing and least tern foraging may not be available in the SCRE during summer due to the high potential for algae blooms, with the potential for bloom conditions to occur during spring and fall periods corresponding to berm closure. In addition to the high potential for flooding at McGrath State Beach, artificial breaches, generally occurring at high SCRE stages during summer and fall months, may cause rapid dewatering of burrows or stranding, thereby limiting the suitability of the SCRE for spawning, reproduction, and/or early development (SPWN) of tidewater goby.

When compared to complete flow removal (Alternative 6), the No Action alternative generally results in poorer water quality conditions for focal fish species as well as for least tern foraging. In addition to elevated TIN levels (Table 11-2) and the nutrient related impacts discussed above, extended periods of stable freshwater flows associated with the VWRF effluent discharge under the No Action alternative in comparison to complete flow removal may provide increased spawning opportunities for non-native predator species adapted to these conditions. The No Action Alternative corresponds to slightly lower nesting habitat area for California least tern and this area would likely be larger immediately following implementation of a complete flow removal alternative. However, vegetation succession under a complete flow removal would likely reduce the suitability of these areas within a few years (See Section 8.2.1.4). Lastly, excess VWRF effluent flows have been shown to limit non-contact recreational opportunities of the SCRE (REC-2) due to flooding of the McGrath State Beach campgrounds that would not regularly occur under a complete flow removal alternative.

Alternative 2: Planned VWRF upgrades

Alternative 2 includes improvements to the VWRF treatment processes, including denitrification to reduce nitrate (NO_3) levels in the VWRF effluent discharge below the 10 mg-N/L level established under the Basin Plan. Although TIN levels would be in excess of those arriving from the Santa Clara River (Table 11-2), nutrient load reductions to the SCRE in comparison to the No Action Alternative during closed-mouth conditions may provide new opportunities for rare, threatened, or endangered species (RARE) because of potential use of the SCRE by steelhead for rearing as well as foraging use by least tern. Nutrient load reductions may reduce the frequency, duration and extent of algal blooms for the benefit of steelhead and least tern. As discussed under Alternative 1, maintenance of current discharge levels may limit non-contact recreational opportunities of the SCRE (REC-2) due to flooding of the McGrath State Beach campgrounds during extended closed mouth conditions, potentially resulting in man-made breaches associated with disturbance of tidewater goby spawning and rearing (SPWN) and also potentially increasing populations of non-native predator species adapted to freshwater conditions.

Alternative 3: Planned VWRF upgrades with enhanced denitrification

Alternative 3 would provide further reductions of TIN levels below those in Alternative 2 and approaching background levels in the Santa Clara River (Table 11-2). In comparison to the No Action alternative, it is anticipated that these nutrient load reductions to the SCRE during closed-mouth conditions may provide new opportunities for rare, threatened, or endangered species' (RARE) because of potential use of the SCRE by steelhead for rearing, as well as foraging use by least tern. Nutrient load reductions may reduce the frequency, duration and extent of algal blooms for the benefit of steelhead and least tern. Despite these potential improvements, however, maintenance of current discharge levels may limit non-contact recreational opportunities of the SCRE (REC-2) due to flooding of the McGrath State Beach campgrounds during extended closed mouth conditions, potentially resulting in man-made breaches associated with disturbance of tidewater goby spawning and rearing (SPWN) and also potentially increasing populations of non-native predator species adapted to freshwater conditions.

Alternative 4: VWRF effluent flow reduction
In comparison to the No Action alternative, VWRF effluent flow reductions may slightly reduce available habitat for rare, threatened, or endangered species (RARE) in the SCRE, as indicated by the reduction in the area of potential steelhead rearing habitat. However, this decrease may be offset by improvements in water quality. TIN levels would be slightly in excess of those arriving from the Santa Clara River (Table 11-2), but would likely result in reduced frequency, duration and extent of algal blooms for the benefit of steelhead and least tern. In addition to nutrient load reduction benefits, reduced freshwater flows would potentially improve conditions for aquatic uses (RARE, SPWN) discussed under Alternative 1–3 by limiting habitat suitability for warm water predator fish species that are adapted to the stable freshwater flows delivered from the VWRF. VWRF effluent reductions will reduce flooding in the McGrath State Beach campground, thereby providing increased recreational (REC-2) opportunities. Lastly, reduced SCRE stage will provide slight increases in nesting area for wildlife (WILD) as represented by the focal bird species in comparison to the No Action Alternative.

Alternative 5: VWRF effluent flow reduction with enhanced denitrification
Alternative 5 would provide further reductions of TIN levels below those in Alternative 2 and approaching background levels in the Santa Clara River (Table 11-2). In comparison to the No Action alternative, increased nutrient removal under Alternative 5 will improve water quality conditions and increase opportunities for rare, threatened, or endangered species (RARE) uses represented by the increased suitability of the SCRE for steelhead rearing as well as foraging use by least tern. Nutrient load reductions may reduce the frequency, duration and extent of algal blooms for the benefit of steelhead and least tern. Flow reductions may also provide similar benefits to habitat area as well as reduced frequency of flooding, as described under Alternative 4. Lastly, reduced SCRE stage under Alternative 5 will provide slight increases in nesting area for wildlife (WILD) as represented by the focal bird species in comparison to the No Action Alternative.

Alternative 6: Complete VWRF effluent flow removal
In comparison to the No Action alternative, complete VWRF flow removal from the SCRE will improve water quality conditions for rare, threatened, or endangered species (RARE) uses by reducing duration and extent of algal blooms for the benefit of steelhead and least tern. Increases in SCRE salinity due to removal of freshwater flows from the VWRF during closed mouth conditions may reduce habitat suitability for warm water predator fish species. However, a decrease in available habitat area for steelhead rearing with complete flow removal may reduce the beneficial use of rare, threatened, or endangered species (RARE). Complete VWRF effluent flow removal from the SCRE will reduce or eliminate flooding in the McGrath State Beach campground, thereby enhancing recreational (REC-2) opportunities.

Summary of alternatives
It is apparent from the preceding discussion that the No Action Alternative, while potentially providing enhancement of aquatic beneficial uses (RARE) with respect to steelhead habitat area, is associated with periods of anoxia/hypoxia due to the regular occurrence of algal blooms that may also affect foraging success of least tern. Elevated inflows from the VWRF during the critical summer period may also limit recreational use opportunities due to flooding at McGrath State Beach. However, complete flow removal from the SCRE under Alternative 6 will also reduce available habitat area for steelhead rearing, and potentially eliminate Tidewater goby access to high flow refuge provided by the VWRF outfall channel, although it is likely that other low velocity areas in the SCRE are available. Balancing affected beneficial uses through intermediate flow reductions without the implementation of water quality improvement measures

will likely not support aquatic beneficial uses at some times of year, and implementation of water quality improvement measures in the absence of flow reductions will likely not support recreational or aquatic uses under conditions of flooding. As has been the case with numerous applications of the ESA in the past 30 years, the difficulty of balancing species needs with each other and other resource uses is apparent here and some degree of "trade-off" between the ecological functions discussed in this report, and thus between one alternative's support of one beneficial use or another may inevitably occur.

Depending on whether one of the alternatives or a future alternative is developed, it is anticipated that management actions involving effluent flow reductions and water quality improvements would only be necessary during summer months when the SCRE lagoon is closed. During times of year when the SCRE mouth berm is likely to be breached due to storm runoff events (November through May in most years), it is anticipated that the water quality improvements identified here would be unnecessary due to lower water temperatures, reduced solar irradiance, and higher occurrence of breaching events due to Santa Clara River inflows. We have not attempted to identify the amount of VWRF effluent flows that could be diverted from the SCRE as part of an expanded water recycling program (Carollo Engineers 2010a) and we have also not attempted to rank onsite and offsite opportunities for wetlands denitrification (Carollo Engineers 2010b) as part of this report. A pairing of the reuse and wetland options with the alternatives discussed here is part of a future project following input from stakeholders and the Regional Board.

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Appendices

Appendix A

The Santa Clara River Watershed
Physical and Geomorphic Setting

Geology

The Santa Clara River watershed is located within the San Andreas Fault system, a geologically active region that forms the dynamic boundary between the Pacific and North America tectonic plates (Figure A-1). Tectonic uplift rates within and around the watershed are estimated to range between 1 mm a-1 to 5 mm a-1 (Orme 1988), with possibly reduced rates over the last several thousand to tens of thousands of years. Geologic structure and local active faults associated with the larger San Andreas Fault system control the position of unchanneled valleys and major and minor tributaries, and result in a high rate of watershed sediment production. In the lower portion of the Santa Clara River downstream of the Sespe Creek confluence, the channel roughly follows the axis of a west-trending synclinal valley along an active strand of the Oak Ridge fault and is crossed by the McGrath fault approximately 2.5 km (1.5 mi) upstream of the river mouth (Clahan 2003) (Figure A-1). The Oak Ridge fault, a thrust fault that dips to the south at a shallow angle, poses a local earthquake risk (Fisher et al. 2005) and has been estimated to have a maximum vertical displacement of between 3.5 to 6 mm/yr (Peterson and Wesnousky 1994) to 5.9 to 12.5 mm/yr (Yates 1981) over the past 200,000–400,000 years. Persistent regional geologic instability throughout the watershed over the last 28 million years has exposed a wide variety of highly deformed, fractured, faulted, and highly erodible rock types (Yeats 1981, Rockwell et al. 1984, Rockwell 1988). The frequent and intense seismic activity in the watershed then acts to accelerate sediment production from these exposed rock types and subsequent sediment delivery to adjacent channels. Seismic shaking during the magnitude 6.7 Northridge event in 1994, for example, triggered nearly 7,400 landslides (Harp and Jibson 1996). A large amount of sediment mobilized from such seismically induced landslides is transported directly to channels while the remaining sediment is stored on the hillsides and is available for transport during subsequent storm events.

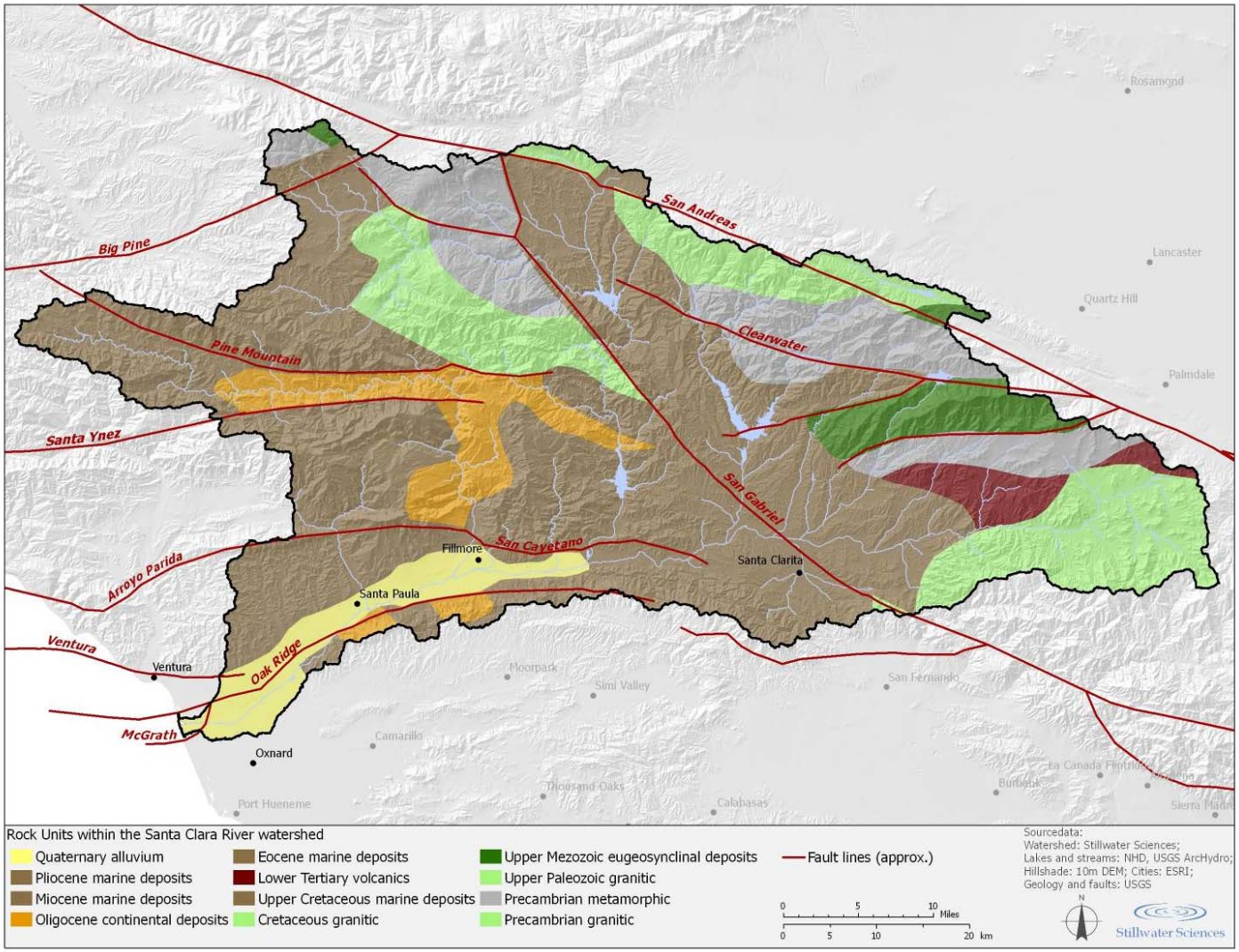

Figure A-1. Major rock units and fault traces in the Santa Clara River watershed.

Deep sediment deposits underlying the Santa Clara River contain an extensive aquifer system that is part of the larger Santa Clara-Calleguas groundwater basin, a primary water supply for the region (Figure A-2). The basin covers over 480 km^2 (300 mi^2) within the Santa Clara River and Calleguas Creek watersheds and contains 12 groundwater sub-basins across the two watersheds. Within the Santa Clara River watershed, the sub-basins are (from east to west) Piru, Fillmore, Santa Paula, Mound, and Oxnard Plain (Hanson et al. 2003). The general vertical structure of each of the basins has recent and relatively thin alluvial deposits overlying older, much thicker alluvial deposits. The recent alluvial deposits are generally younger than 10,000 years and are locally derived from the mountains draining to the Santa Clara River; the underlying older deposits are as old as 1.8 million years and up to approximately 3,000 m (10,000 ft) thick (UWCD and CLWA 1996, as cited in Nautilus Environmental 2005). Since the alluvial deposits are relatively porous, nearly all of the groundwater basins along the lower Santa Clara River are unconfined aquifers (Oxnard Plain being the exception): they are able to receive water from the surface and their water table surface fluctuates freely in response to recharge and discharge rates.

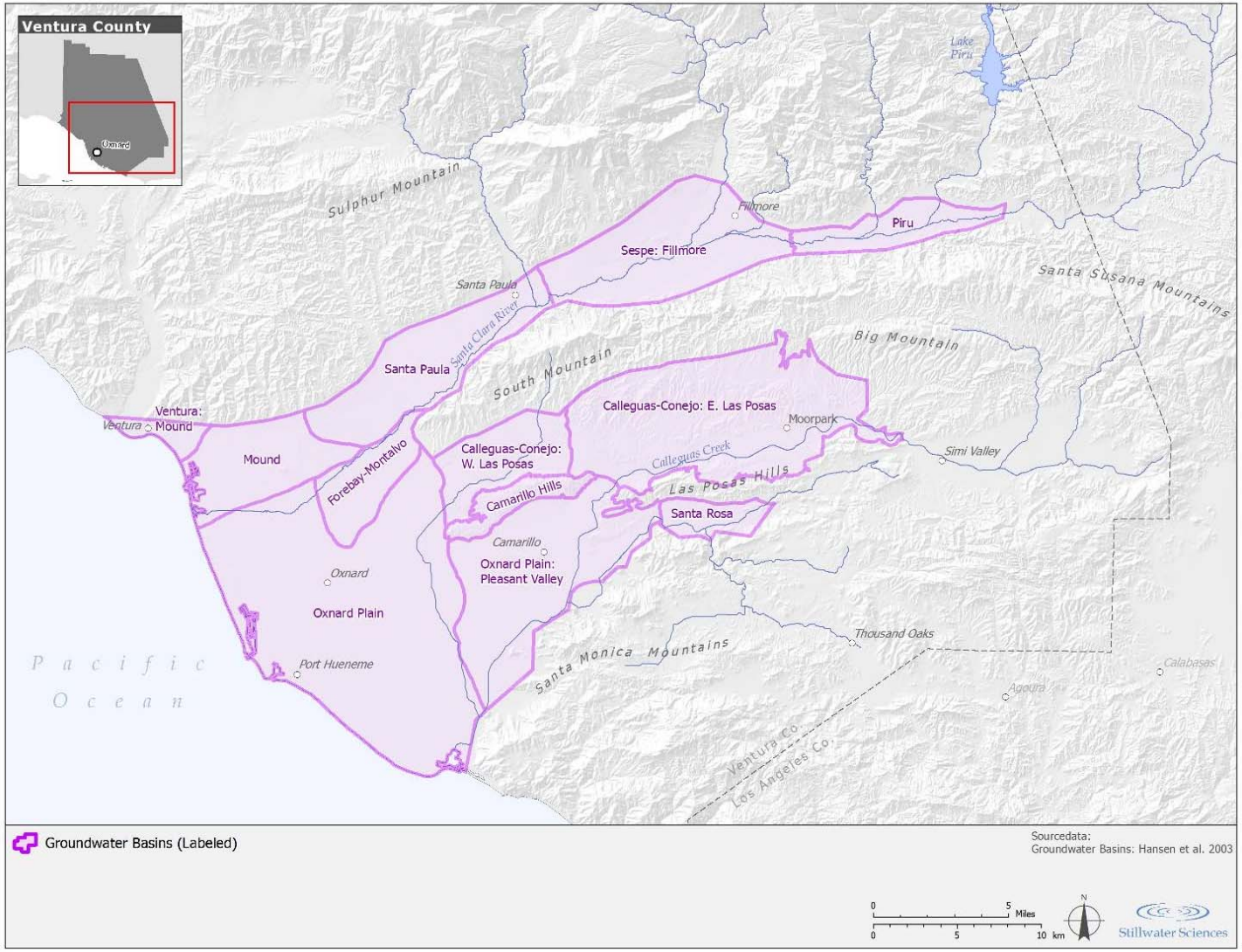

Figure A-2. Santa Clara-Calleguas Hydrologic Unit and groundwater basin.

Land Use/Land Cover

When compared with many southern California coastal watersheds, the Santa Clara River watershed is relatively undeveloped; over 50% of the watershed is National Forest land (Angeles National Forest and Los Padres National Forest). Land cover in upland areas of the Santa Clara River watershed is dominated by scrub/shrub (chaparral) vegetation; grasslands and mixed, deciduous, and evergreen woodlands compose the remainder of upland land cover. Along floodplain and valley bottom areas of the Santa Clara River Valley, orchard and row crop agriculture is the dominant land use, with significant urban areas in the upper (Santa Clarita, Newhall) and lower (Santa Paula, Fillmore, Oxnard) valley areas. In the lower Santa Clara River below the confluence with Sespe Creek, agricultural and urban use account for 22% and 9% of land cover, respectively (Warrick 2002).

Land use and climatic conditions within the Santa Clara River watershed make the area prone to frequent wildfire. Historical records indicate that over two-thirds of the Santa Clara River watershed has burned at least once since the late 19th century, with many areas of the lower watershed, including South Mountain and the lower Sespe, Hopper, and Piru creek watersheds, burning up to nine times since 1878 (CAL FIRE 2007). Fires in 2003 burned 119,000 ac (48,200 ha) or approximately 11% of the watershed slopes, and the recent 2006 Day Fire burned 162,800 ac (65,900 ha) in total, most of which was in the Santa Clara River watershed. As in most southern California watersheds, the "flood-fire" sequence, or phenomenon where post-fire winter rains result in increased runoff and accelerated erosion, strongly impacts the frequency of floods, landslides, and debris flows within the Santa Clara River watershed.

Since the onset of European settlement over 150 years ago, there has been considerable change in the land use and water-management practices throughout the Santa Clara River watershed. These changes can be synthesized into five distinct historical periods that span the time period from pre-European colonization, through periods of increased agricultural impact and river flow modification, to increased development and urbanization impacts under present-day conditions (Table A-1). Management practices within each period have had a profound influence on channel geomorphic process and water and sediment discharge dynamics from the watershed to the SCRE (see Stillwater Sciences 2007 for a more detailed discussion).

Table A-1. Historical periods characterizing land use and impacts to the Santa Clara River watershed.

Time period	Description
Pre-European Colonization Pre-1820	Human activities have a relatively minor impact on ecosystem processes, with the exception of Native American grassland management.
Ranching and Colonization 1820–1890	Significant European arrival in the watershed, extensive riparian vegetation removal, the introduction of livestock.
Irrigation and Diversions 1890–1955	Floodplain agriculture requires tributary diversions and the construction of small dams.
Dams and River Modifications 1955–1990	Completion of major dams, construction of an extensive network of levees and bank protection, and intensive river and floodplain aggregate mining.
Urbanization 1990 to present	Continued rapid population increases in the watershed cause large-scale conversion of floodplain agricultural land to residential developments.

Climate and Hydrology

The Santa Clara River watershed experiences a semi-arid Mediterranean-type climate, with cool wet winters and warm dry summers. Proximity to the Pacific Ocean moderates both seasonal and diurnal temperatures, air moisture, and precipitation. Air moisture is greatest at the coast and decreases to near-desert conditions towards the eastern watershed boundary. Most precipitation occurs between November and March and varies significantly throughout the watershed due to topographic features. For example, average annual rainfall is more than 114 cm (45 in) in the mountainous headwaters of Sespe Creek but only about 22 cm (9 in) in the drier eastern portions of the watershed near the Mojave Desert (Figure A-3).

The El Niño Southern Oscillation (ENSO), a climatic phenomenon characterized by warming and cooling cycles (oscillations) in the waters of the eastern equatorial Pacific Ocean, also impacts local climactic and subsequent hydrologic conditions within the watershed. ENSO events typically have 1–1.5 year durations, a 3–8 year recurrence interval, and are related to changes in atmospheric circulation, rainfall, and upper ocean heat content (see Deser et al. 2004 and references contained therein). In southern California, ENSO years are characterized by relatively high rainfall intensities, with rivers and streams exhibiting higher annual peak flow magnitudes than they do in non-ENSO years (Cayan et al. 1999, Andrews et al. 2004). The recent wet-period ENSO cycle (1967 to the present) has been marked by strong ENSO years every 3–7 years, a relative increase in the number of large storms, and associated mean sediment fluxes for southern California rivers (from the Pajaro River south to the Tijuana River) that have been approximately five times greater than during the preceding dry period (1944–1968) (Inman and Jenkins 1999).

Regional climatic conditions result in inherently variable inter- and intra-annual flow conditions throughout the watershed. Overall, the flow in the lower Santa Clara River is relatively low: 75% of the time discharge is less than 150 cfs near Montalvo, CA (approximately 7.2 km (4.5 mi) upstream of the SCRE) and 50% of the time discharge is less than 10 cfs (URS Corporation 2005). During the dry summer season, discharge in the mainstem and tributaries is derived largely from rising groundwater, inflows from dam releases, or other anthropogenic sources (e.g., irrigation runoff or treated wastewater effluent). Rising groundwater occurs in areas where bedrock constriction forces groundwater to the surface (e.g., the Piru and Fillmore narrows). During winter months, however, peak flows can be quite large; flows in the lower Santa Clara River at Montalvo, CA exceed 10,000 approximately once every 2 years and exceed 90,000 cfs approximately once every 10 years (URS Corporation 2005). Over the past 60 years, every one of the 12 largest annual peak floods have occurred during ENSO years, including the January 1969 flood event (~165,000 cfs) and the more recent January 2005 flood event (~136,000 cfs) (Figure A-4). During the rainy season, flows can increase, peak, and subside rapidly in response to high intensity rainfall, with the potential for severe flooding under saturated or near-saturated watershed conditions. Between winter rainfall events in wet years, the river may exhibit continuous baseflow to the ocean from residual watershed discharge; in dry years, flow may be intermittent or non-existent in the lower watershed.

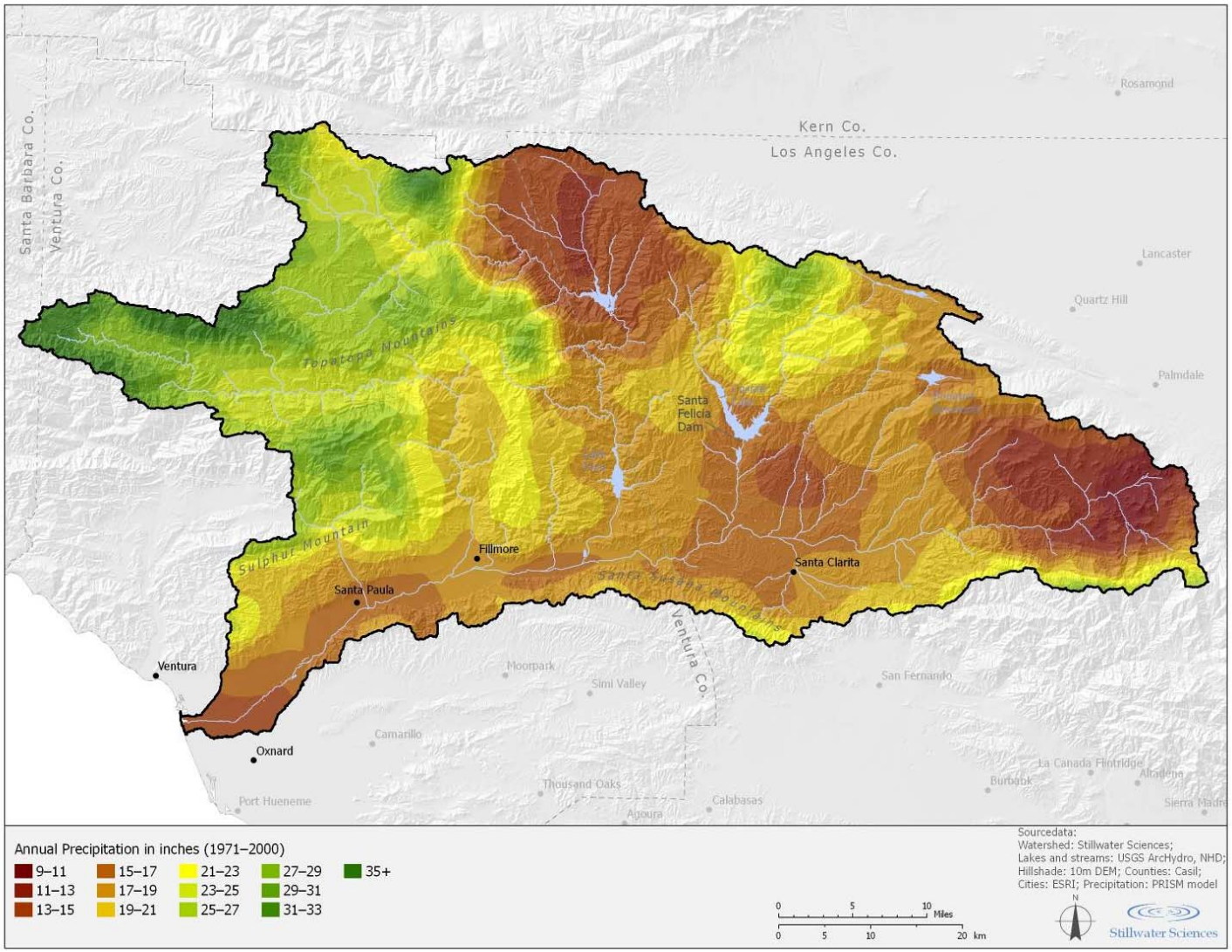

Figure A-3. Distribution of mean annual precipitation for the Santa Clara River watershed.

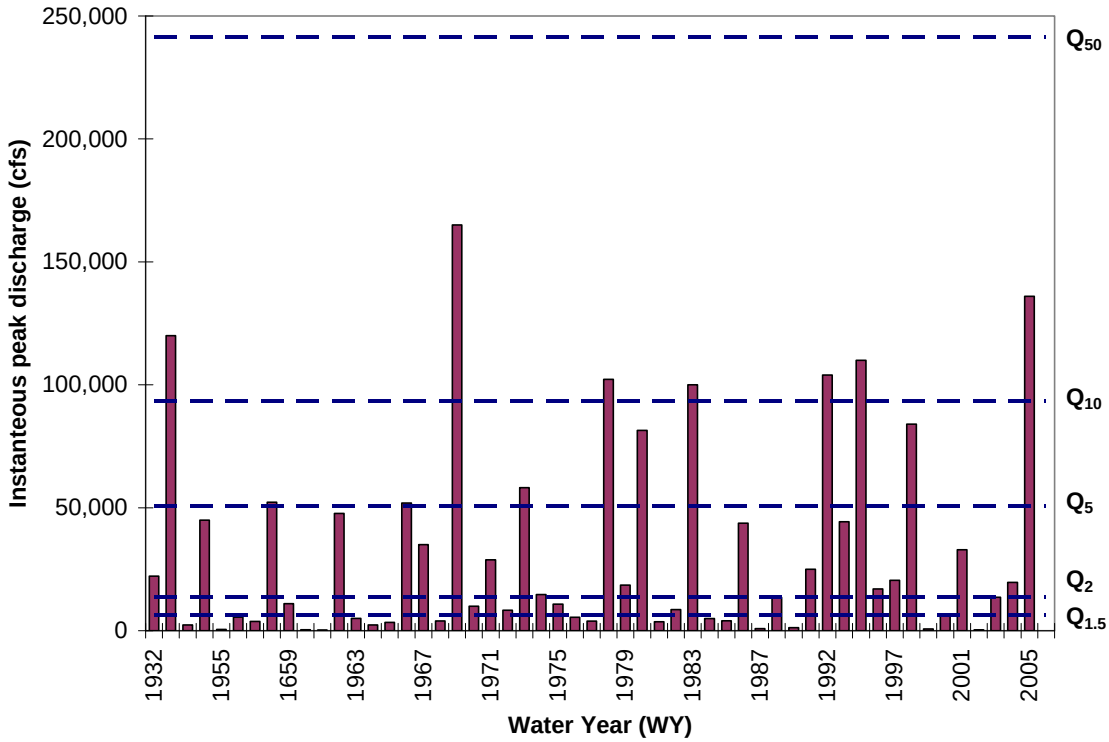

Figure A-4. Annual maximum instantaneous discharge for the lower Santa Clara River at Montalvo, CA (USGS 11114000).

Water supply dams impact both the low-flow and high-flow discharge to the lower Santa Clara River. More than one-third of the watershed lies upstream of tributary dams, which has caused run-off to the mainstem Santa Clara River to be reduced by approximately 25% (Warrick 2002). The major dams in tributaries to the mainstem Santa Clara River include Santa Felicia Dam on Piru Creek and Castaic Dam on Castaic Creek (Table A-2). Throughout the year, controlled releases of water from Piru Reservoir supplement surface flows in the river between the confluence with Piru Creek and the Freeman Diversion Dam. Freeman Diversion Dam, located approximately 15 km (9 mi) upstream from the SCRE, can divert up to 375 cfs continuously from the Santa Clara River at any given time with a maximum allowable annual diversion volume of 144,000 acre-feet.

Table A-2. Major tributary dams and regulated areas within the Santa Clara River watershed.

Tributary name	Drainage area (km^2)[a]		Drainage area (mi^2)[a]		Dams
	Total	Regulated[b]	Total	Regulated[b]	
Bouquet Canyon Creek	187	34	72	13	Bouquet Canyon
Castaic Creek	526	397	203	153	Castaic Dam
Piru Creek	1,134	1,095	438	423	Santa Felicia
Total	**1,847**	**1,526**	**713**	**589**	

[a] Areas calculated from 10 m (32 ft) USGS DEM using ArcHydro.
[b] Regulated areas are areas upstream of dams, which impound sediment and water.
[c] Debris basins do not impound water and only impound the coarse fraction of the sediment load to the limit of their capacity (e.g., Warrick 2002).

Sediment Production and Transport

The combination of rapid tectonic uplift rates, highly erodible bedrock, frequent high-intensity storm events, and the "flood-fire" sequence lead to extremely high rates of sediment production and yield from hillslopes to tributaries to the mainstem Santa Clara River and out to the SCRE. Over the past several decades, the average annual sediment production and yield are on the order of 1.5 x 10^7 tonnes/yr of hillslope sediment production (3,580 tonnes/km^2/yr; Stillwater Sciences 2007) and 3.5 x 10^6 tonnes/yr of sediment yield in the lower river at Montalvo, CA (1,390 tonnes/km^2/yr; Warrick 2002). Average annual estimates of sediment discharge from the Santa Clara River over the past 80 years are shown in Table A-3. Sediment yield from tributaries to the mainstem channel is impacted by the three major waters supply dams (listed above), a number of small-head water supply dams (e.g., Harvey Diversion Dam on Santa Paula Creek), other in-channel structures (e.g., bridges and culverts), and natural geologic controls. The water supply dams, which regulate over one-third of the watershed, act as effective sediment traps (e.g., Santa Felicia Dam traps 100% of incoming sediment [Williams 1979]), reducing suspended and bedload sediment yield to the mainstem Santa Clara River by an estimated 21% compared to pre-dam conditions (Brownlie and Taylor 1981, Warrick 2005). Freeman Diversion dam also acts as an effective sediment trap, resulting in the short-term deposition of 200,000–300,000 yd^3 of sediment following completion (Swanson et al. 1990) and upstream bed aggradation on the order of 1.6–3 ft (0.5–1m) between 1993 and 2005 (Stillwater Sciences 2007).

Table A-3. Summary of sediment discharge estimates for the Santa Clara River.

Sediment discharge class	Sediment discharge (tonnes yr^{-1})	Sediment discharge (m^3 yr^{-1})
Suspended Sediment	$3.5x10^6$ [for 1950–1999] (Warrick and Milliman 2003)	
Sand		$1.35x10^5$ (Noble Consultants 1989)
		$5.71x10^5$ (PRC Toups Corp. 1980)
Sand & Gravel	$9.6x10^5$ [for 1928–1975] (Brownlie and Taylor 1981)	$9.12x10^5$ [for 1971–1999] (Willis and Griggs 2003)
Total Sediment	$3.3x10^6$ [for 1928–1975] (Williams 1979)	
	$3.5x10^6$ [for 1928–1999] (Warrick 2002)	

Over the long term, the supply of sediment from hillslopes to channels is thought to be fairly continuous (Scott and Williams 1978), with wet-season contributions from overland flow, landslides, and soil slumps, and dry-season contributions from dry ravel. Over the short-term, sediment is delivered from hillslopes to tributaries to the mainstem more episodically during flows associated with big storms (particularly during ENSO years) and during moderate storms following wildfires (Florsheim et al. 1991, Wells 1981). These flows effectively transport a substantial sediment load during very short time periods. Within the lower mainstem Santa Clara River at Montalvo, CA, recent studies have shown that an estimated 25% of the total sediment transported between 1928 and 2000 occurred over just 4 days (Warrick 2002), approximately 55% of the total sediment transported between 1968 and 1975 occurred over just two days

(Williams 1979), and approximately 94% of the suspended sediment transported between 1968 and 1995 occurred during less than 100 days (1% of the time) (Stillwater Sciences 2007). These data help illustrate the concept that the Santa Clara River's channel-forming "dominant" discharge, or flow that transports the greatest volume sediment over the long-term and does the most geomorphic work (Wolman and Miller 1960, Emmett and Wolman 2001), is in fact the largest flow on record. A consequence is that the river does not have equilibrium tendencies (i.e., small annual fluctuations around a long-term average condition) and experiences dramatic geomorphic changes during these episodic high flow events that can persist for years of decades (see Stillwater Sciences 2007 for a more complete discussion).

Relatively high watershed rates of fine-sediment production and yield contribute to frequently occurring hyperpycnal flows. Hyperpycnal flows are characterized by a sediment-laden flood discharge that is denser than ocean water and can induce off-shore turbidity currents (Normark and Piper 1991, as cited in Warrick and Milliman 2003; Normark et al. 1993, as cited in Lamb et al. in press). Buoyancy theory suggests a threshold of suspended sediment concentration for hyperpycnal flows of approximately 40 g L^{-1} (resulting in an approximate total flow density of 1,040 kg m^{-3}) for southern California rivers (ocean density = 1,025 kg m^{-3}) (Warrick 2002). During the 1969 flood events, for example, Santa Clara River suspended sediment concentrations exceeded this 40 g L^{-1} threshold for periods of hours to days. The hyperpycnal threshold is thought to be surpassed in the Santa Clara River during flows no more than 1 to 3 times the value of the mean annual flow (recurrence interval of approximately 1–4 yrs), resulting in hyperpycnal flows being responsible for transporting approximately 75% of the estimated 170 million tonnes of sediment discharged from the Santa Clara River between 1950 and 1999 (Warrick and Milliman 2003).

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Appendix B

Water Quality and Benthis Macroinvertebrates Data Compilation

(see accompanying data CD)

CD Contents

Compiled VWRF Annual Reports

Folder: City_Annual_Reports
Files: Annual reports in PDF format for 1997 to 2009

Compiled Annual Benthic Macroinvertebrate data for the Santa Clara Rive Estuary

Folder: ABC_BMI_Reports
Files: Annual benthic macroinvertebrate reports in PDF format for 2003 to 2008
Letter to the Board in Word format describing changes to sampling techniques as of 2004

Compiled grab sample data for Santa Clara River Estuary

File; Appendix_B_WQ_data.xls

- Historical analytical water quality data collected from 2001-2008
- Historical *in situ* water quality data (Temp., DO, pH, Sp.C, Turb.) collected from 1997-2008
- Analytical water quality data collected by Stillwater Sciences in 2009–2010
- *In situ* water quality (Temp., DO, pH, Sp.C., Turb) data collected by Stillwater Sciences in 2009–2010
- Historical trace metals data collected from 2001-2008
- Historical total coliform data collected from 2000-2008
- Historical fecal coliform data collected from 2001-2008

2009–2010 Continuous *in situ* water quality data for the Santa Clara River Estuary

File: Appendix_B_City_Sonde_WQ_Data.xls

- North Sonde (SCR1) Raw Data (2/24/2009-7/7/2010)
- South Sonde (SCR2) Raw Data (2/19/2009-7/7/2010)
- North Sonde (SCR1) Final Data (3/12/2009-7/7/2010)
- South Sonde (SCR2) Final Data (2/20/2009-7/7/2010)

2009-2010 Laboratory Reports

Folder: SEM_Lab_Reports
Files: Appendix_B_SEM_WQ_data.xls

- Electronic data delivery (EDD) of 2009–2010 analyses

Laboratory reports in PDF format detailing nutrient concentrations from estuary and groundwater samples collected during 2009-2010

Folder: ASL_Lab_Reports
Files: Laboratory reports in PDF format detailing priority pollutant concentrations within groundwater samples collected during 2009-2010

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Appendix C

Water Quality Data (2009–2010)

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Laboratory Reports

See CD attached to Appendix B for Sierra Environmental Monitoring analytical water quality reports.

Table C-1. Water quality sampling locations within the Santa Clara River Estuary (2009-2010)

Site Code	Site Name	GPS Coordinates						
		9/1/2009	11/17/2009	1/27/2010	3/25/2010	5/27/2010	7/27/2010	10/12/2010[1]
R1	Riverine	34° 14.121'N 119° 14.976'W	34° 14.110'N 119° 14.911'W	34° 14.068'N 119° 15.581'W	34° 14.108'N 119° 14.889'W	34° 14.112'N 119° 14.887'W	34° 14.114'N 119° 14.840'W	34° 14.114'N 119° 14.840'W
EU1	Upper Estuary	34° 14.092'N 119° 15.561'W	34° 14.097'N 119° 15.547'W	34° 13.581'N 119° 15.789'W	34° 14.001'N 119° 15.541'W	34° 14.119'N 119° 15.520'W	34° 14.126'N 119° 15.518'W	34° 14.126'N 119° 15.518'W
EU2	Upper Estuary	34° 14.055'N 119° 15.496'W	34° 14.054'N 119° 15.466'W	34° 13.638'N 119° 15.782'W	34° 14.126'N 119° 15.511'W	34° 14.019'N 119° 15.541'W	34° 14.042'N 119° 15.510'W	34° 14.042'N 119° 15.510'W
EU3	Upper Outfall	34° 14.132'N 119° 15.760'W	34° 14.132'N 119° 15.760'W	34° 14.042'N 119° 15.887'W	34° 14.016'N 119° 15.789'W	34° 14.157'N 119° 15.740'W	34° 14.000'N 119° 15.832'W	34° 14.000'N 119° 15.832'W
EM1	Middle Estuary	34° 13.984'N 119° 15.761'W	34° 14.010'N 119° 15.754'W	34° 13.872'N 119° 15.851'W	34° 14.154'N 119° 15.746'W	34° 14.018'N 119° 15.790'W	34° 13.996'N 119° 15.832'W	34° 13.996'N 119° 15.832'W
EM2	Lower Outfall	34° 14.041'N 119° 15.899'W	34° 14.050'N 119° 15.882'W	34° 13.941'N 119° 15.855'W	34° 14.070'N 119° 15.858'W	34° 14.046'N 119° 15.882'W	34° 14.053'N 119° 15.880'W	34° 14.053'N 119° 15.880'W
EL1	Near Mouth	34° 13.869'N 119° 15.909'W	34° 13.883'N 119° 15.899'W	34° 13.867'N 119° 15.891'W	34° 13.853'N 119° 15.900'W	34° 13.876'N 119° 15.884'W	34° 13.816'N 119° 15.872'W	34° 13.816'N 119° 15.872'W
EL2	Lower Estuary	34° 13.771'N 119° 15.829'W	34° 13.774'N 119° 15.840'W	34° 13.826'N 119° 15.868'W	34° 13.741'N 119° 15.789'W	34° 13.612'N 119° 15.792'W	34° 13.643'N 119° 15.809'W	34° 13.643'N 119° 15.809'W
GW1	Well #1	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W	34° 13.622'N 119° 15.619'W
GW2	Well #2	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W	34° 13.473'N 119° 15.545'W
GW3	Well #3	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W	34° 13.204'N 119° 15.485'W

[1] GPS coordinates were not taken but the sample locations were nearly identical to the July event.

Table C-2. *In situ* water quality data for the Santa Clara River Estuary (2009-2010)

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
9/1/2009	R1	1345	1.1	0.5	20.3	3214	127.0	11.4	8.29	0.9
9/1/2009	EU1	1300	3.2	0.5	25.0	2548	258.8	21.2	9.70	21.4
9/1/2009	EU1	1300	3.2	2	24.0	2582	268.1	22.3	9.60	NM
9/1/2009	EU1	1300	3.2	2.5	23.3	2647	240.6	20.4	9.30	NM
9/1/2009	EU2	1315	1.5	0.5	25.4	2538	257.0	20.9	9.70	22.3
9/1/2009	EU2	1315	1.5	1	25.1	2542	258.0	21.0	9.70	NM
9/1/2009	EU3	1210	5.8	0.5	24.5	2284	61.4	5.1	7.80	5.0
9/1/2009	EU3	1210	5.8	2	24.2	2281	59.1	4.9	7.80	NM
9/1/2009	EU3	1210	5.8	3	22.2	2389	52.1	4.5	7.70	NM
9/1/2009	EU3	1210	5.8	4	21.2	2431	50.9	4.5	7.60	NM
9/1/2009	EM1	1230	3.1	0.5	24.7	2536	257.0	21.1	9.57	22.1
9/1/2009	EM1	1230	3.1	2	24.3	2531	259.8	21.6	9.55	NM
9/1/2009	EM1	1230	3.1	2.5	24.0	2541	245.1	20.4	9.55	NM
9/1/2009	EM2	1155	5.4	0.5	23.2	2401	80.6	6.8	8.20	3.8
9/1/2009	EM2	1155	5.4	2	22.6	2387	69.9	6.0	8.10	NM
9/1/2009	EM2	1155	5.4	3	22.0	2426	61.0	5.3	8.00	NM
9/1/2009	EM2	1155	5.4	4	21.5	2477	65.1	5.7	7.90	NM
9/1/2009	EM2	1155	5.4	5	21.5	2478	67.0	5.3	7.90	NM
9/1/2009	EL1	1125	5.6	0.5	24.0	2536	216.1	18.1	9.50	18.3
9/1/2009	EL1	1125	5.6	2	23.2	2538	179.0	15.3	9.41	NM
9/1/2009	EL1	1125	5.6	3	23.1	2537	146.8	12.6	9.34	NM
9/1/2009	EL1	1125	5.6	4	23.1	2536	140.9	12.0	9.34	NM
9/1/2009	EL2	1105	6	0.5	23.9	2540	184.1	15.4	9.45	17.5
9/1/2009	EL2	1105	6	2	23.7	2544	188.3	15.8	9.46	NM
9/1/2009	EL2	1105	6	3	23.8	2544	191.5	15.9	9.48	NM
9/1/2009	EL2	1105	6	4	23.7	2544	196.2	16.5	9.47	NM
9/1/2009	EL2	1105	6	5	23.1	2553	165.8	14.4	9.37	NM
9/1/2009	EL2	1105	6	5.5	23.1	2553	141.9	11.9	9.33	NM

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
11/17/2009	R1	1125	0.7	0.5	16.27	3139	137.5	13.33	7.53	3.9
11/17/2009	EU1	1220	2.8	0.5	16.3	9435	81.1	7.77	8.47	14.8
11/17/2009	EU1	1220	2.8	1	16.01	9581	80	7.5	8.48	NM
11/17/2009	EU1	1220	2.8	1.5	15.89	9691	76.2	7.28	8.5	NM
11/17/2009	EU1	1220	2.8	2	15.76	9737	74.5	7.09	8.51	NM
11/17/2009	EU2	1030	2.1	0.5	15.34	9437	95	9.23	8.55	15.8
11/17/2009	EU2	1030	2.1	1	15.17	9437	91.6	8.95	8.55	NM
11/17/2009	EU2	1030	2.1	1.5	14.61	9535	91.2	8.68	8.58	NM
11/17/2009	EU2	1030	2.1	2	14.61	9601	94.2	9.05	8.56	NM
11/17/2009	EU3	1305	4.7	0.5	18.44	2987	84.3	7.91	7.61	6.6
11/17/2009	EU3	1305	4.7	1	18.35	3397	79.5	7.42	7.59	NM
11/17/2009	EU3	1305	4.7	1.5	17.71	5121	70.2	6.6	7.84	NM
11/17/2009	EU3	1305	4.7	2	17.16	9851	60.4	5.74	8.31	NM
11/17/2009	EU3	1305	4.7	3	16.46	11360	45.1	4.32	8.44	NM
11/17/2009	EU3	1305	4.7	4	16.61	12490	42.5	3.53	8.45	NM
11/17/2009	EU3	1305	4.7	5	17.14	17580	18.2	2.09	8.38	NM
11/17/2009	EM1	1236	2.3	0.5	16.27	9713	73.3	7.04	8.54	20.5
11/17/2009	EM1	1236	2.3	1	16.278	9718	75.9	7.17	8.54	NM
11/17/2009	EM1	1236	2.3	1.5	16.29	9722	74.1	6.93	8.54	NM
11/17/2009	EM2	1330	4.8	0.5	18.1	5995	68.6	6.39	7.85	5.3
11/17/2009	EM2	1330	4.8	1	17	8826	43.4	4.26	8.2	NM
11/17/2009	EM2	1330	4.8	1.5	16.24	10280	41.2	4.23	8.4	NM
11/17/2009	EM2	1330	4.8	2	16.07	10760	43.3	4.3	8.49	NM
11/17/2009	EM2	1330	4.8	2.5	16	11320	47.8	4.53	8.51	NM
11/17/2009	EM2	1330	4.8	3	16.01	11300	48.1	4.56	8.52	NM
11/17/2009	EM2	1330	4.8	3.5	16.02	11280	49.7	4.7	8.51	NM
11/17/2009	EL1	1405	4.8	0.5	16.21	9968	81.3	7.66	8.56	15.1
11/17/2009	EL1	1405	4.8	1.5	16.34	10120	75.2	7.21	8.55	NM
11/17/2009	EL1	1405	4.8	2.5	16.4	10130	75.2	7.12	8.55	NM

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
11/17/2009	EL1	1405	4.8	3.5	16.35	10060	74.3	7.33	8.56	NM
11/17/2009	EL1	1405	4.8	4	16.32	10070	74.1	7.17	8.56	NM
11/17/2009	EL2	1415	3.1	0.5	16.3	9779	81.8	7.63	8.57	16.5
11/17/2009	EL2	1415	3.1	1.5	16.33	9785	83.4	7.9	8.57	NM
11/17/2009	EL2	1415	3.1	2.5	16.33	9788	83.1	7.87	8.57	NM
11/17/2009	GW1	1515	N/A	N/A	17.83	6434	NM	NM	NM	NM
11/17/2009	GW2	1545	N/A	N/A	18.67	3514	NM	NM	NM	NM
11/17/2009	GW3	1635	N/A	N/A	19.03	3255	NM	NM	NM	NM
1/27/2010	R1	1045	0.8	0.5	12.4	1682	104.2	11.2	7.92	90.5
1/27/2010	EU1	1500	1.1	0.5	16.3	10540	115.7	10.8	7.89	22.5
1/27/2010	EU2	1440	1.2	0.5	17.8	15180	107.0	9.6	7.93	106.7
1/27/2010	EU3	1215	0.7	0.5	17.7	2695	107.9	10.1	7.41	8.1
1/27/2010	EM1	1305	0.8	0.5	14.1	2275	106.1	10.9	7.93	87.1
1/27/2010	EM2	1240	0.6	0.5	18.2	3033	104.6	9.9	7.38	77.9
1/27/2010	EL1	1315	1	0.5	15.9	13810	116.4	10.7	7.85	95.5
1/27/2010	EL2	1335	0.5	0.5	17.9	25190	98.9	8.6	7.77	79.4
1/26/2010	GW1	16:10	N/A	N/A	16.3	5052	NM	NM	NM	NM
1/26/2010	GW2	16:45	N/A	N/A	17.5	3160	NM	NM	NM	NM
1/26/2010	GW3	17:25	N/A	N/A	17.5	3134	NM	NM	NM	NM
3/25/2010	EU2	10:00	3	0.5	16.43	4492	155.5	15	8.51	NM
3/25/2010	EU2	10:00	3	1.5	16.43	4553	152.1	14.68	8.54	NM
3/25/2010	EU2	10:00	3	2.5	16.34	4452	151	14.6	8.52	NM
3/25/2010	R1	10:50	0.8	0.5	16.8	2492	118.4	11.39	7.66	NM
3/25/2010	EU1	11:35	2.1	0.5	17.24	2727	113.3	10.77	7.66	NM
3/25/2010	EU1	11:35	2.1	1.5	17.18	2712	128.1	12.12	7.64	NM
3/25/2010	EM1	12:00	2.2	0.5	17.96	3905	177	16.69	8.42	NM
3/25/2010	EM1	12:00	2.2	1.5	17.99	4198	202.2	18.87	8.57	NM
3/25/2010	EU3	12:20	3.1	0.5	21.3	2365	92.9	8.23	7.59	NM
3/25/2010	EU3	12:20	3.1	1.5	21.04	2366	93	8.22	7.58	NM

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
3/25/2010	EU3	12:20	3.1	2.5	21.04	2367	92.6	8.19	7.57	NM
3/25/2010	EM2	12:40	1.5	0.5	21.4	2641	87.5	7.73	7.6	NM
3/25/2010	EL1	13:00	2.2	0.5	19.39	6787	245.5	22.1	8.98	NM
3/25/2010	EL1	13:00	2.2	1.5	18.94	7277	243.3	22.1	8.95	NM
3/25/2010	EL2	13:20	3.4	0.5	19.06	5530	255	23.01	9.01	NM
3/25/2010	EL2	13:20	3.4	1.5	19.02	5500	257.2	23.35	9	NM
3/25/2010	EL2	13:20	3.4	2.5	19.04	5530	255.6	23.27	9	NM
3/25/2010	GW1	15:35	N/A	N/A	NM	7003	NM	NM	NM	NM
3/25/2010	GW2	15:45	N/A	N/A	NM	4712	NM	NM	NM	NM
3/25/2010	GW3	15:55	N/A	N/A	NM	1555	NM	NM	NM	NM
5/27/2010	R1	1105	0.5	0.5	17.75	2833	124.2	11.74	7.78	2.5
5/27/2010	EU1	1140	4.1	0.5	20.47	4011	205.8	18.41	9.33	17.8
5/27/2010	EU1	1140	4.1	1.5	20.42	4012	210.3	18.71	9.33	NM
5/27/2010	EU1	1140	4.1	2.5	20.33	4011	205.3	18.22	9.31	NM
5/27/2010	EU1	1140	4.1	3.5	20.01	4009	192	17.09	9.27	NM
5/27/2010	EU2	1020	2.6	0.5	20.33	4140	203.1	18.14	9.43	17.8
5/27/2010	EU2	1020	2.6	2	20.29	4170	205.6	18.34	9.45	NM
5/27/2010	EU3	1230	4.2	0.5	21.9	1939	115.9	10.09	8.15	1.0
5/27/2010	EU3	1230	4.2	1.5	21.9	1940	115.9	10.04	8.07	NM
5/27/2010	EU3	1230	4.2	2.5	21.87	1939	114.3	9.96	8.01	NM
5/27/2010	EU3	1230	4.2	3.5	21.83	1939	131.8	11.78	7.95	NM
5/27/2010	EM1	1210	3.3	0.5	21.06	4145	203.2	18.46	9.48	16.8
5/27/2010	EM1	1210	3.3	1.5	21.08	4152	214.3	18.81	9.5	NM
5/27/2010	EM1	1210	3.3	2.5	20.77	4199	197.7	17.3	9.45	NM
5/27/2010	EM2	1250	2.9	0.5	21.22	2774	89.8	7.87	8.16	4.1
5/27/2010	EM2	1250	2.9	2	20.98	3006	85.3	7.55	8.38	NM
5/27/2010	EL1	1310	2.5	0.5	21.09	4456	239.8	21.01	9.78	18.9
5/27/2010	EL1	1310	2.5	1.5	21.07	4442	234.9	20.58	9.71	NM
5/27/2010	EL2	1320	2.1	0.5	21.42	4561	142	12.38	9.62	19.1

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
5/27/2010	EL2	1320	2.1	1.5	21.27	4569	141.5	12.36	9.64	NM
5/27/2010	GW1	1525	N/A	N/A	16.74	8894	NM	NM	NM	NM
5/27/2010	GW2	1545	N/A	N/A	18.04	5764	NM	NM	NM	NM
5/27/2010	GW3	1610	N/A	N/A	18.65	3398	NM	NM	NM	NM
7/27/2010	R1	1050	1.2	0.5	18.24	3109	93.1	8.56	7.64	2.5
7/27/2010	EU1	1010	4.4	0.5	20.07	2472	148.8	13.4	9.29	10.4
7/27/2010	EU1	1010	4.4	1.5	20.08	2474	148.6	13.37	9.34	NM
7/27/2010	EU2	1130	2	0.5	20.84	2520	158	14.02	9.08	7.4
7/27/2010	EU2	1130	2	1.5	20.84	2521	156.9	13.91	9.11	NM
7/27/2010	EU3	1130	2	2.5	20.8	2524	155.5	13.82	9.12	NM
7/27/2010	EU3	1130	2	3.5	20.72	2527	153.3	13.63	9.12	NM
7/27/2010	EU3	1215	4.8	0.5	21.99	2273	82	7.11	7.8	0.9
7/27/2010	EU3	1215	4.8	1.5	21.54	2275	74.4	6.5	7.63	NM
7/27/2010	EU3	1215	4.8	2.5	21.46	2273	71	3.22	7.55	NM
7/27/2010	EU3	1215	4.8	3.5	21.43	2272	69.4	3.09	7.5	NM
7/27/2010	EU3	1215	4.8	4.5	21.41	2271	69.3	6.08	7.46	NM
7/27/2010	EM1	1200	2.8	0.5	10.76	2464	142.6	12.66	9.12	9.9
7/27/2010	EM1	1200	2.8	1.5	20.73	2470	143.3	12.76	9.16	NM
7/27/2010	EM1	1200	2.8	2.5	20.7	2473	143.9	12.84	9.18	NM
7/27/2010	EM2	1230	5.2	0.5	21.04	2307	61.8	5.42	7.49	2.5
7/27/2010	EM2	1230	5.2	1.5	20.8	2316	57.7	5.1	7.47	NM
7/27/2010	EM2	1230	5.2	2.5	20.41	2344	49.4	4.38	7.46	NM
7/27/2010	EM2	1230	5.2	3.5	20.01	2392	50.9	4.61	7.48	NM
7/27/2010	EL1	1250	1.5	0.5	21.01	2472	156	13.93	9.32	19.3
7/27/2010	EL2	1310	2.2	0.5	21.21	2462	150.7	13.28	9.64	24.2
7/27/2010	EL2	1310	2.2	1.5	21.19	2462	150.9	13.31	9.65	NM
7/27/2010	GW1	1440	N/A	N/A	16.53	8124	NM	NM	NM	NM
7/27/2010	GW2	1500	N/A	N/A	18.49	4782	NM	NM	NM	NM
7/27/2010	GW3	1525	N/A	N/A	19.04	3445	NM	NM	NM	NM

Date	Site Code	Sample Time	Bottom Depth	Sample Depth	Temp	Specific Conductivity	Dissolved Oxygen	Dissolved Oxygen	pH	Turbidity
			ft	ft	°C	µS/cm	%	mg/L	s.u.	ntu
10/12/2010	EU2	1025	1	0.5	21.65	5988	92.8	7.97	8.8	28.5
10/12/2010	R1	1050	1.4	0.5	18.38	3154	73.6	6.77	7.71	2.6
10/12/2010	EU1	1110	4.1	0.5	21.54	6032	98.6	8.54	8.91	15.7
10/12/2010	EU1	1110	4.1	1.5	21.54	6047	97.6	8.43	8.95	NM
10/12/2010	EU1	1110	4.1	2.5	21.52	6059	96.4	8.31	8.96	NM
10/12/2010	EU1	1110	4.1	3.5	21.52	6063	93.7	8.11	8.98	NM
10/12/2010	EM1	1130	1.4	0.5	21.55	6000	135.5	11.78	9.08	9.0
10/12/2010	EU3	1150	4.7	0.5	23.5	2275	108.1	9.3	8.27	8.6
10/12/2010	EU3	1150	4.7	1.5	22.48	2716	105.2	8.98	8.24	NM
10/12/2010	EU3	1150	4.7	2.5	21.87	6218	98.7	8.44	8.81	NM
10/12/2010	EU3	1150	4.7	3.5	21.77	6281	96.9	8.32	8.95	NM
10/12/2010	EM2	1200	3.1	0.5	22.25	3372	126.1	10.62	8.6	8.4
10/12/2010	EM2	1200	3.1	1.5	21.99	4632	107.1	9.17	8.7	NM
10/12/2010	EM2	1200	3.1	2.5	21.89	5554	104.5	9.01	8.75	NM
10/12/2010	EL1	1215	3.5	0.5	21.54	6175	134.3	11.62	9.1	8.4
10/12/2010	EL1	1215	3.5	1.5	21.54	6166	133	11.52	9.13	NM
10/12/2010	EL1	1215	3.5	2.5	21.49	6225	132.8	11.49	9.15	NM
10/12/2010	EL2	1225	1.3	0.5	21.39	6304	104.8	9.07	9.1	12.2
10/12/2010	GW1	1415	N/A	N/A	18.89	5973	NM	NM	NM	NM
10/12/2010	GW2	1440	N/A	N/A	18.96	5293	NM	NM	NM	NM
10/12/2010	GW3	1505	N/A	N/A	20.08	3403	NM	NM	NM	NM

N/A = not applicable
NM = not measured

Table C-3. Analytical water quality data for the Santa Clara River Estuary (2009-2010)

Date	Site Code	Sample Time	Ammonia	Kjeldahl Nitrogen	Nitrate	Nitrite	Phosphorus-Ortho	Phosphorus-Total	Chlorophyll a
			mg/L N	mg/L N	mg/L N	mg/L N	mg/L	mg/L	ug/L
9/1/2009	R1	1345	<0.1	1.1	3.3	<0.05	<0.02	0.05	13
9/1/2009	EU1	1300	0.1	3.5	2.7	<0.05	0.2	0.67	170
9/1/2009	EU2	1315	<0.1	3.1	2.9	<0.05	0.21	0.51	200
9/1/2009	EU3	1210	0.9	2.6	16	<0.05	3	3.2	8
9/1/2009	EM1	1230	0.3	3.3	3.4	<0.05	0.35	0.6	150
9/1/2009	EM2	1155	0.7	1.6	11	<0.05	2.2	2.7	<2
9/1/2009	EL1	1125	0.6	2.8	3	<0.05	0.28	0.67	260
9/1/2009	EL2	1105	0.4	3.5	2.9	<0.05	0.25	0.75	170
11/17/2009	R1	1125	<0.1	<0.1	5	<0.5	<0.02	<0.02	5
11/17/2009	EU1	1220	0.3	1.7	9.3	<0.5	0.8	1	3
11/17/2009	EU2	1030	0.4	1.1	9.7	<0.5	0.8	0.84	2
11/17/2009	EU3	1305	0.2	1.5	19	<0.5	2.1	2.8	9
11/17/2009	EM1	1236	0.3	1.9	11	<0.5	1.1	1.4	3
11/17/2009	EM2	1330	0.4	1.7	16	<0.5	2.5	2.2	9
11/17/2009	EL1	1405	0.3	2.2	11	<0.5	1.1	1.3	2
11/17/2009	EL2	1415	0.3	2	12	<0.5	1.2	1.1	<2
11/16/2009	GW1	1515	1.1	1.1	<0.5	<0.5	NM	NM	NM
11/16/2009	GW2	1545	1	1	<0.5	<0.5	NM	NM	NM
11/16/2009	GW3	1635	<0.1	0.2	2	<0.5	NM	NM	NM
1/27/2010	R1	1045	<0.1	1	1.4	<0.05	0.05	0.15	<2
1/27/2010	EU1	1215	<0.1	0.6	7.4	<0.5	0.18	0.36	81
1/27/2010	EU2	1305	<0.1	1.1	4.5	<2.5	0.25	0.48	36
1/27/2010	EU3	1240	0.7	1.1	19	<0.05	2.4	2.5	4
1/27/2010	EM1	1500	0.20	0.4	1.4	<0.05	0.06	0.29	<2
1/27/2010	EM2	1440	0.9	1.5	18	<0.05	2	2.5	5
1/27/2010	EL1	1315	0.2	0.5	2	<0.5	0.08	0.53	<2
1/27/2010	EL2	1335	0.3	0.6	<2.5	<2.5	0.13	0.31	15
1/26/2010	GW1	1610	NM	NM	<0.5	<0.5	NM	NM	NM

Date	Site Code	Sample Time	Ammonia	Kjeldahl Nitrogen	Nitrate	Nitrite	Phosphorus-Ortho	Phosphorus-Total	Chlorophyll a
			mg/L N	mg/L N	mg/L N	mg/L N	mg/L	mg/L	ug/L
1/26/2010	GW2	1645	NM	NM	<0.5	<0.5	NM	NM	NM
1/26/2010	GW3	1725	NM	NM	1.3	<0.5	NM	NM	NM
3/25/2010	R1	1000	<0.1	0.4	5.5	<0.05	<0.02	0.23	4
3/25/2010	EU1	1200	<0.1	1	8.4	<0.5	0.54	0.8	38
3/25/2010	EU2	1220	<0.1	0.7	4.6	<0.05	<0.02	0.11	29
3/25/2010	EU3	1240	<0.1	0.7	18	<0.05	2.3	2.4	<2
3/25/2010	EM1	1050	<0.1	2	9.1	<0.5	0.74	1	44
3/25/2010	EM2	1135	<0.1	0.8	17	<0.05	2.2	2.4	13
3/25/2010	EL1	1300	<0.1	2.7	8.8	<0.5	0.45	0.89	130
3/25/2010	EL2	1320	<0.1	1.9	8.4	<0.5	0.46	0.92	250
3/24/2010	GW1	1535	NM	NM	<0.5	<0.5	NM	NM	NM
3/24/2010	GW2	1545	NM	NM	<0.5	<0.5	NM	NM	NM
3/24/2010	GW3	1555	NM	NM	0.8	<0.5	NM	NM	NM
5/27/2010	R1	1105	<0.1	0.4	6.1	<0.5	<0.02	<0.02	11
5/27/2010	EU1	1230	<0.1	3.6	5.6	<0.5	0.27	0.76	84
5/27/2010	EU2	1210	<0.1	2.3	5.4	<0.5	0.23	0.64	46
5/27/2010	EU3	1250	0.4	0.7	13	<0.5	2.5	2.6	<2
5/27/2010	EM1	1140	0.1	0.9	6.2	<0.5	0.43	0.93	44
5/27/2010	EM2	1020	0.2	2	11	<0.5	2.3	2.4	15
5/27/2010	EL1	1310	<0.1	2.2	5.7	<0.5	0.18	0.71	46
5/27/2010	EL2	1320	0.4	2.4	5.6	<0.5	0.22	0.8	13
5/26/2010	GW1	1525	NM	NM	<0.5	<0.5	NM	NM	NM
5/26/2010	GW2	1545	NM	NM	<0.5	<0.5	NM	NM	NM
5/26/2010	GW3	1610	NM	NM	1	<0.5	NM	NM	NM
7/27/2010	R1	1050	0.1	0.1	6.1	0.27	<0.02	0.05	3
7/27/2010	EU1	1215	<0.1	1.5	2.2	0.55	0.4	0.18	42
7/27/2010	EU2	1200	<0.1	1	2.3	0.49	0.33	0.17	21
7/27/2010	EU3	1230	0.2	0.4	13	0.5	3.8	4	<2
7/27/2010	EM1	1010	<0.1	1.5	3.2	0.54	0.41	1	42
7/27/2010	EM2	1130	0.1	0.8	8.5	0.38	3.1	3.1	11

Date	Site Code	Sample Time	Ammonia	Kjeldahl Nitrogen	Nitrate	Nitrite	Phosphorus-Ortho	Phosphorus-Total	Chlorophyll a
			mg/L N	mg/L N	mg/L N	mg/L N	mg/L	mg/L	ug/L
7/27/2010	EL1	1250	<0.1	1.8	2.5	0.57	0.39	0.22	35
7/27/2010	EL2	1310	<0.1	1.6	2.1	0.59	0.26	0.17	46
7/27/2010	GW1	1440	NM	NM	<0.5	<0.5	NM	NM	NM
7/27/2010	GW2	1500	NM	NM	<0.5	<0.5	NM	NM	NM
7/27/2010	GW3	1525	NM	NM	0.55	<0.5	NM	NM	NM
10/12/2010	R1	1050	0.1	1.4	4.5	<0.5	<0.02	<0.02	5
10/12/2010	EU1	1110	0.3	2.7	4.3	<0.5	0.63	0.97	11
10/12/2010	EU2	1025	0.2	3.6	4.1	<0.5	0.60	0.37	33
10/12/2010	EU3	1150	<0.1	1.6	11	<0.5	2.8	3.4	130
10/12/2010	EM1	1130	0.3	2.5	5	<0.5	0.3	0.48	25
10/12/2010	EM2	1200	<0.1	2.4	9.5	<0.5	2.3	2.8	160
10/12/2010	EL1	1215	<0.1	2.6	4.6	<0.5	0.67	0.2	18
10/12/2010	EL2	1225	0.2	3.2	4.8	<0.5	0.74	0.53	11
10/12/2010	GW1	1415	NM	NM	<0.05	<0.05	NM	NM	NM
10/12/2010	GW2	1440	NM	NM	<0.05	<0.05	NM	NM	NM
10/12/2010	GW3	1505	NM	NM	1.4	<0.05	NM	NM	NM

N/A = not applicable
NM = not measured

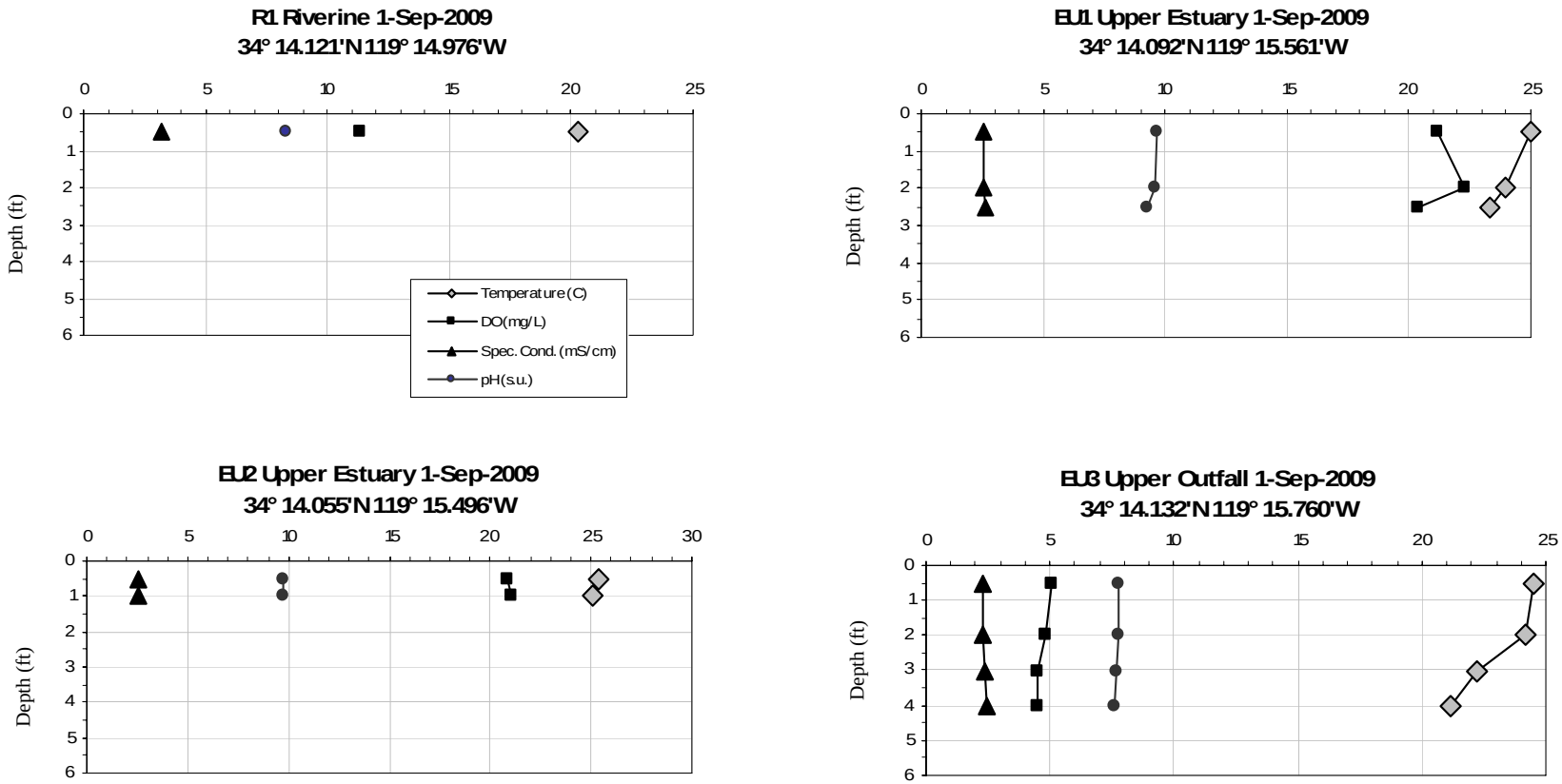

Figure C-1. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, September 2009.

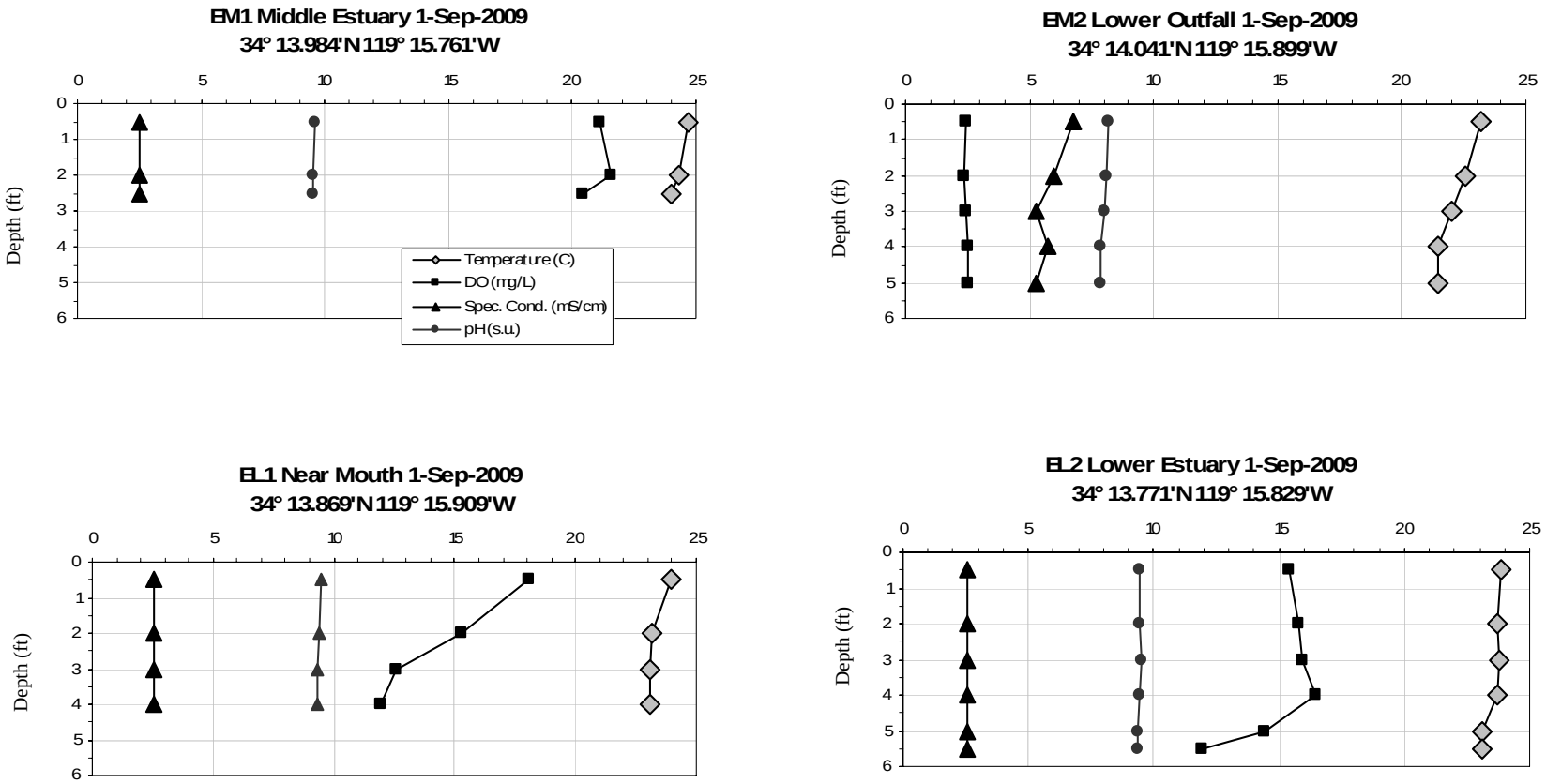

Figure C-2. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, September 2009.

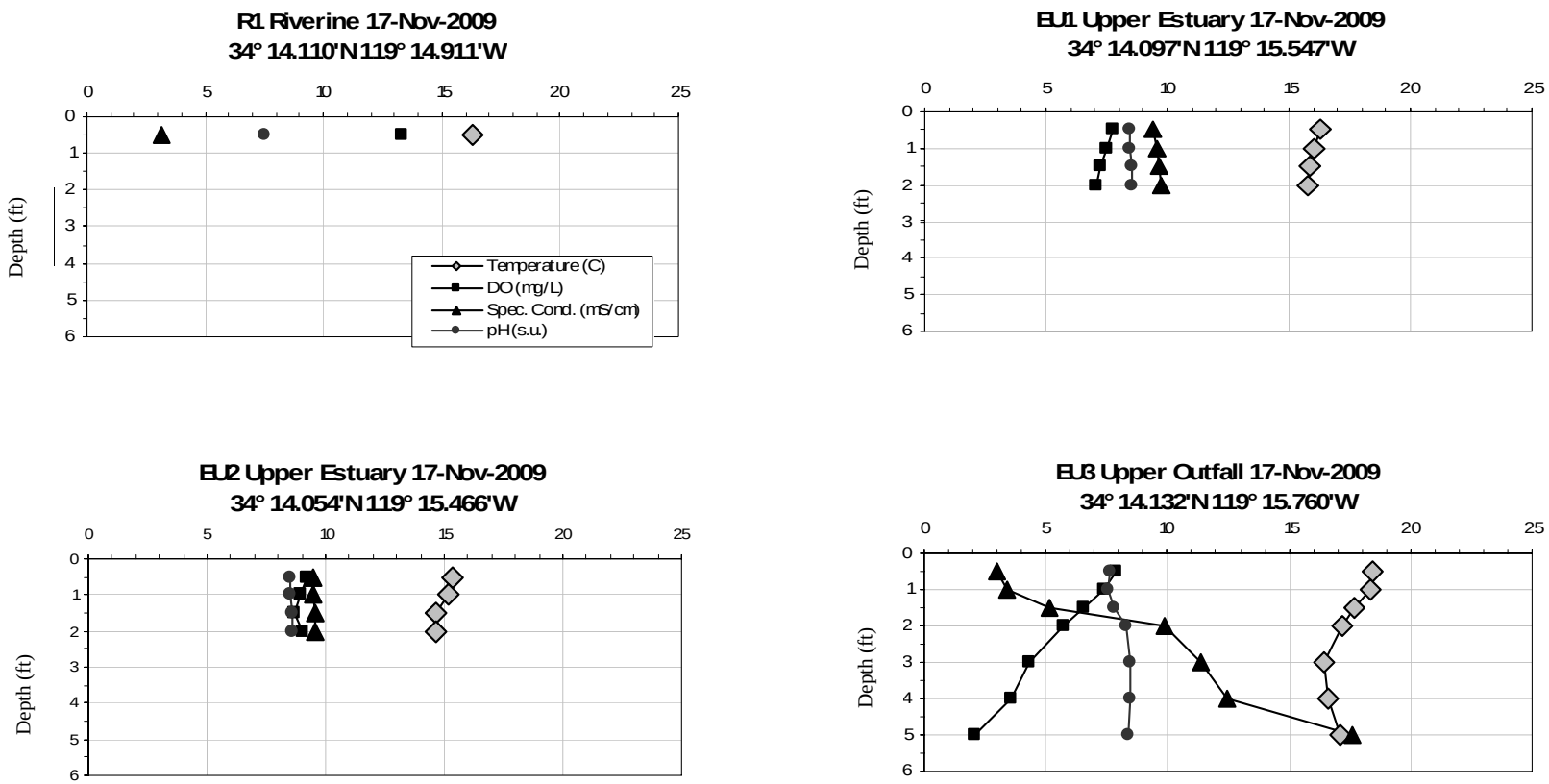

Figure C-3. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, November 2009.

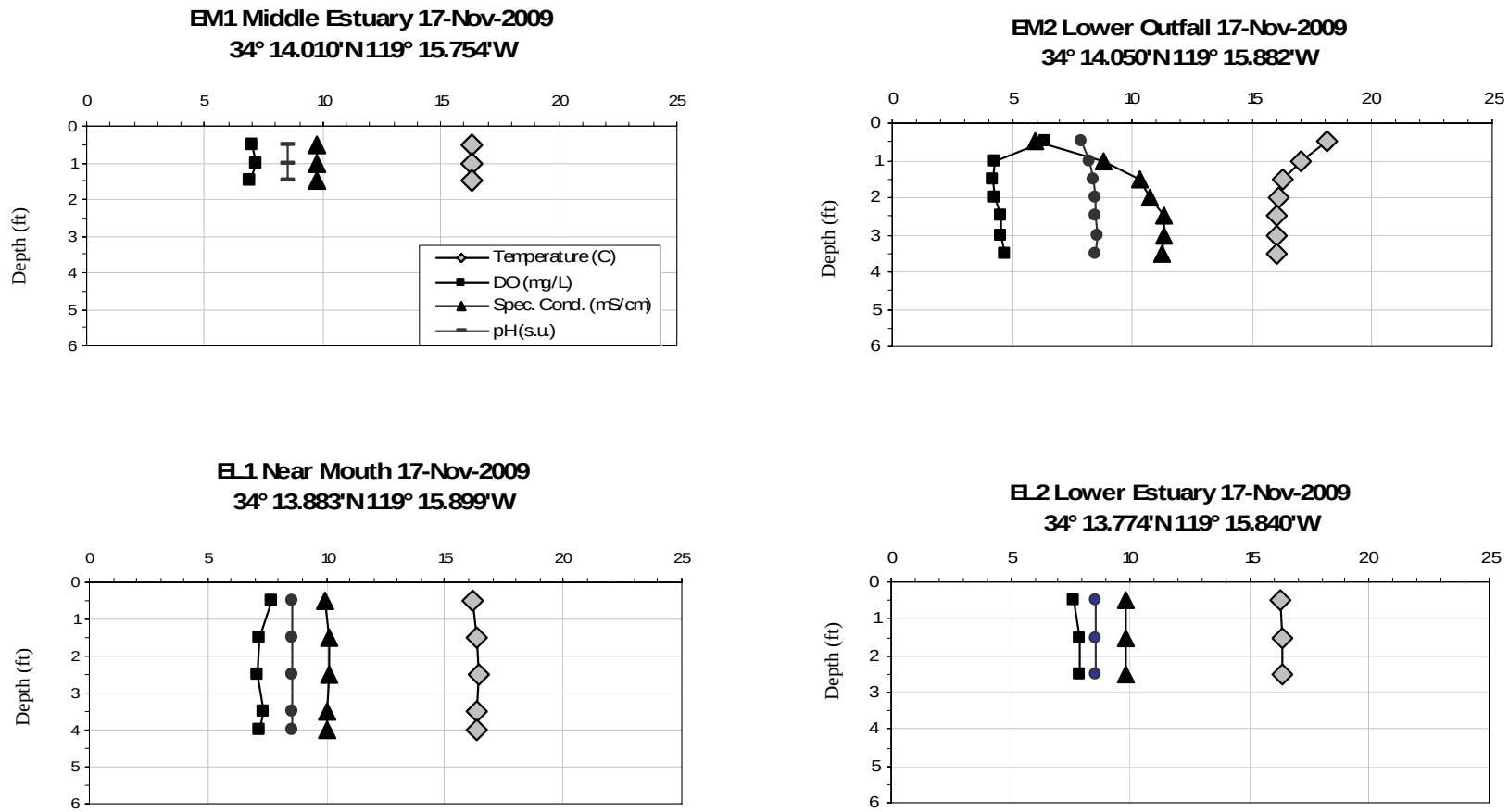

Figure C-4. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, November 2009.

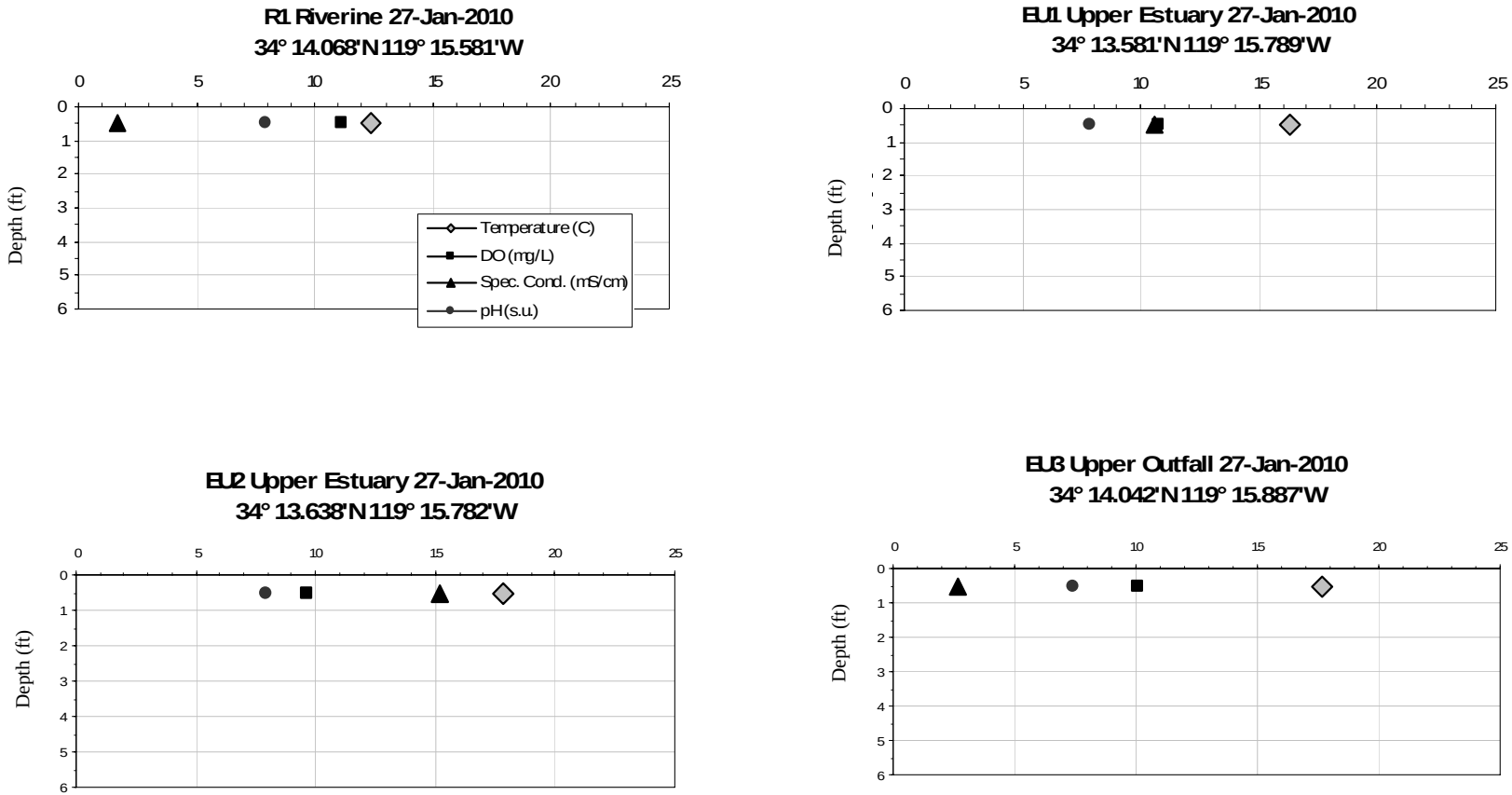

Figure C-5. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, January 2010.

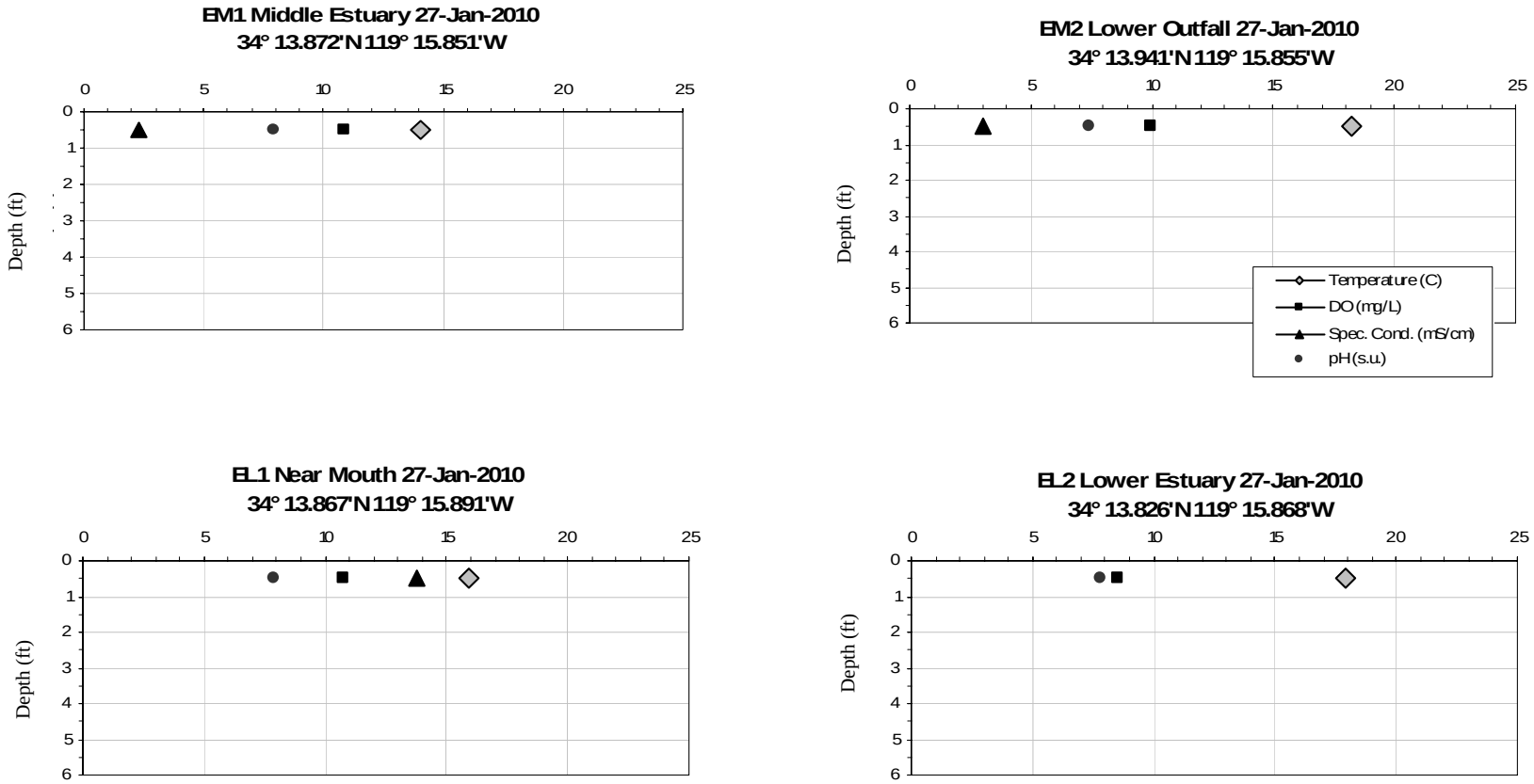

Figure C-6. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, January 2010.

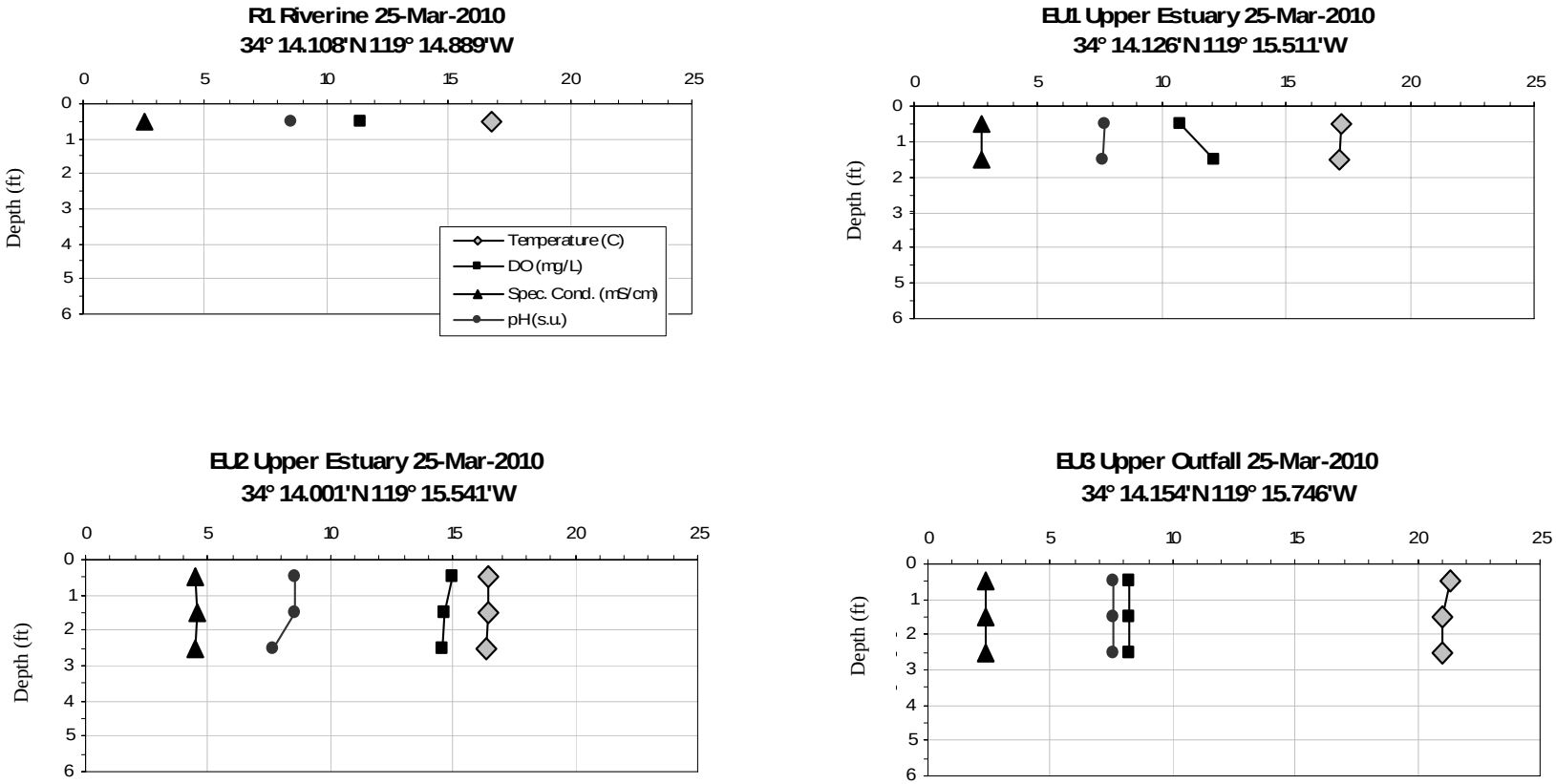

Figure C-7. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, March 2010.

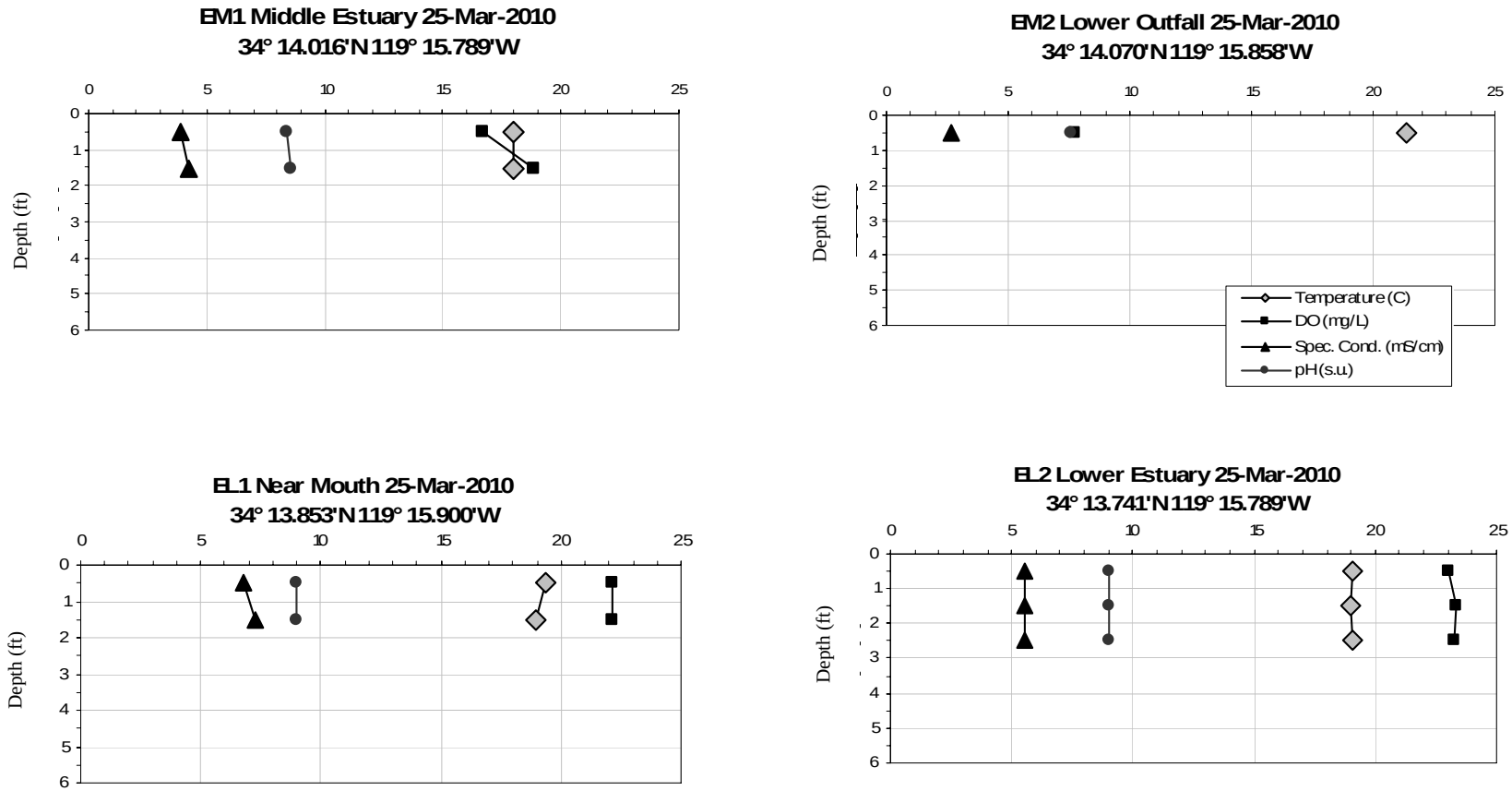

Figure C-8. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, March 2010.

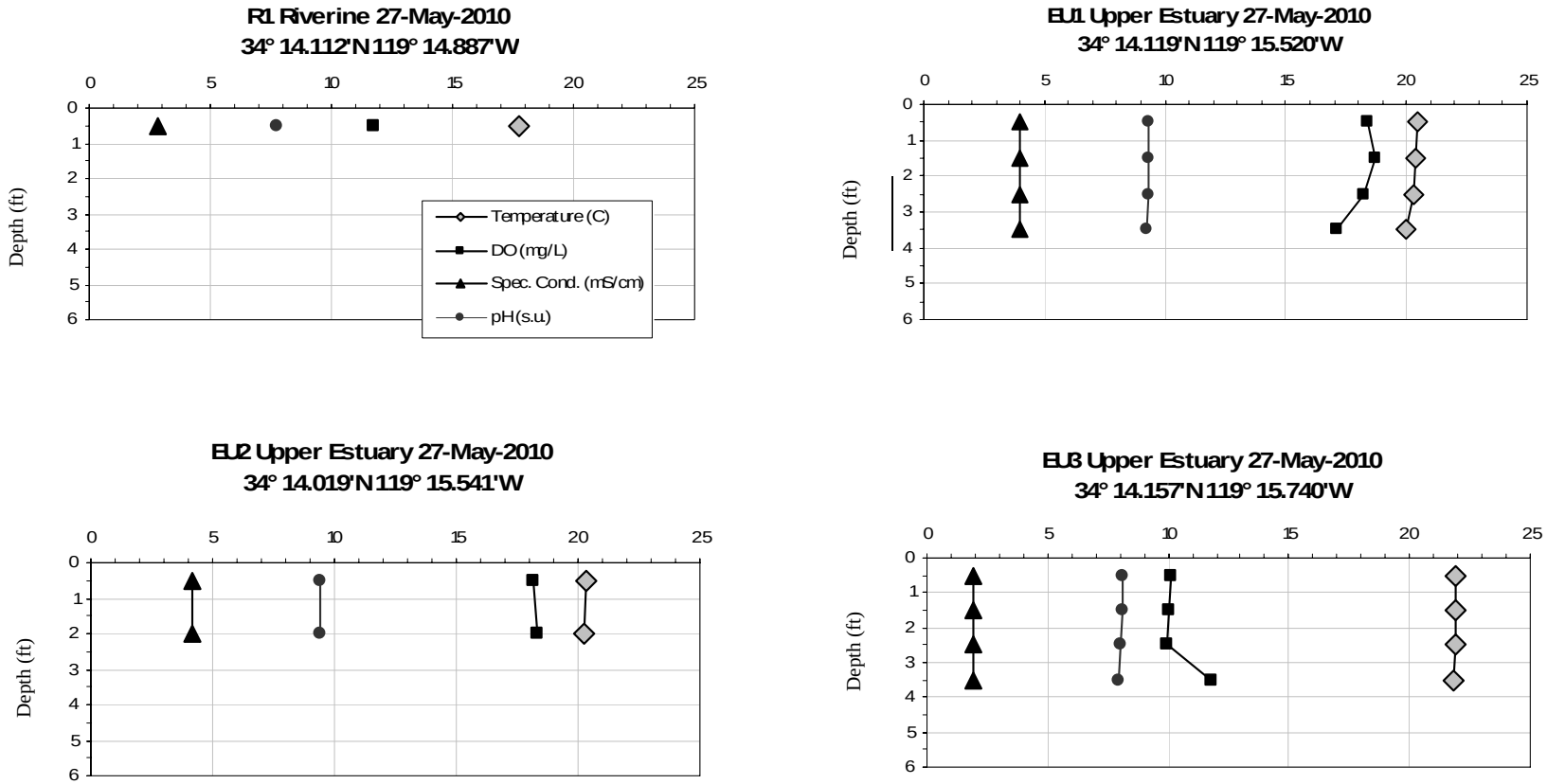

Figure C-9. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, May 2010.

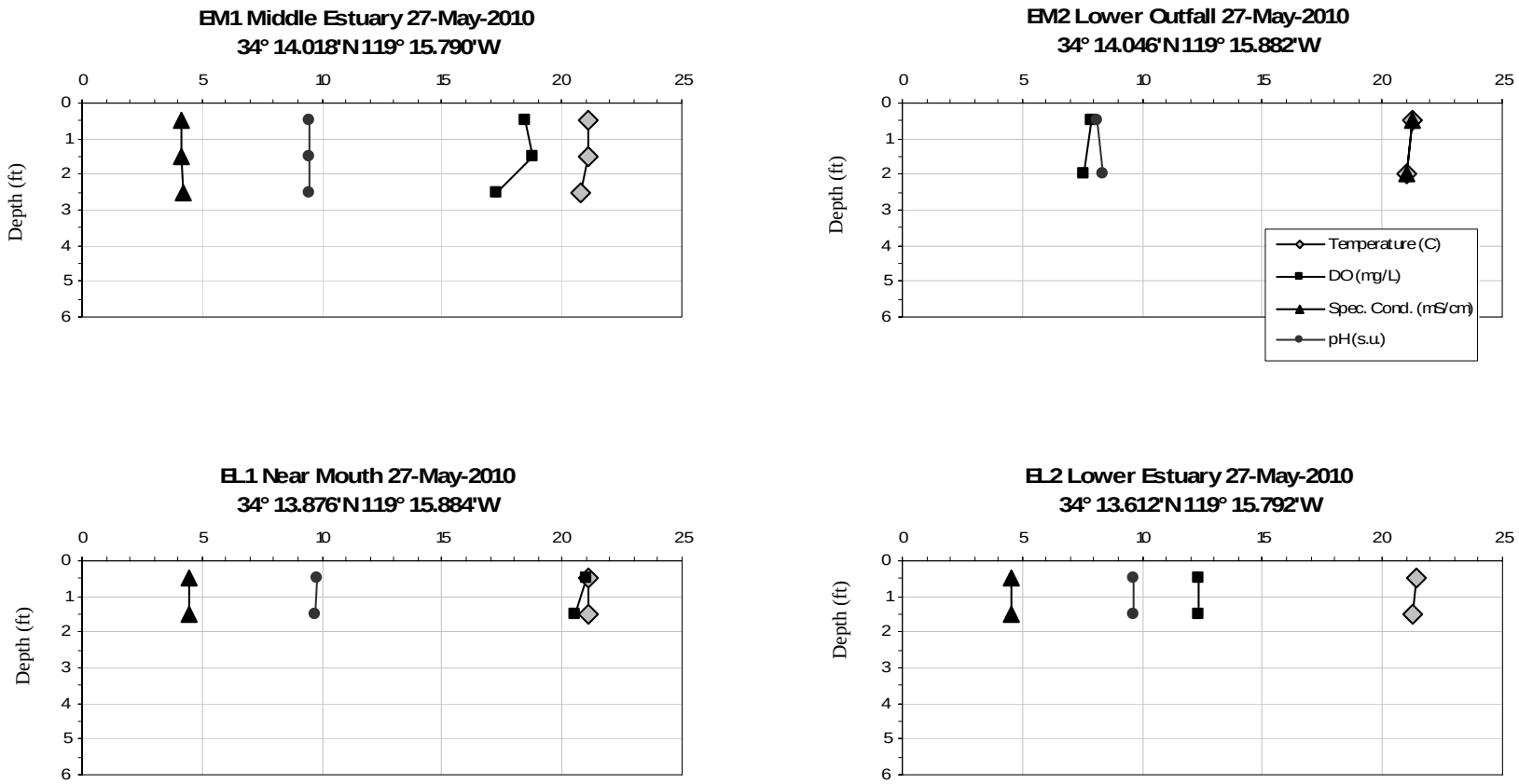

Figure C-10. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, May 2010.

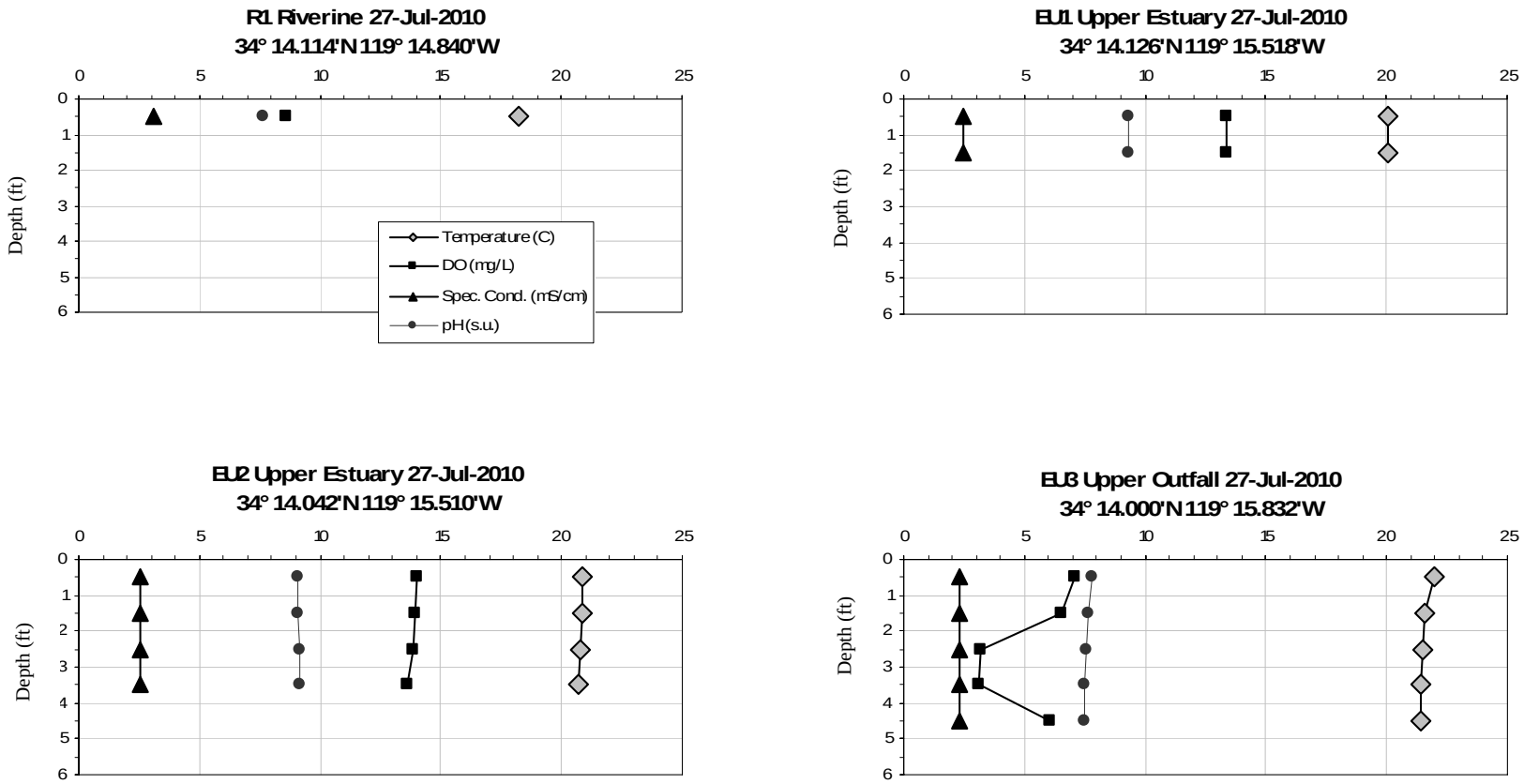

Figure C-11. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, July 2010.

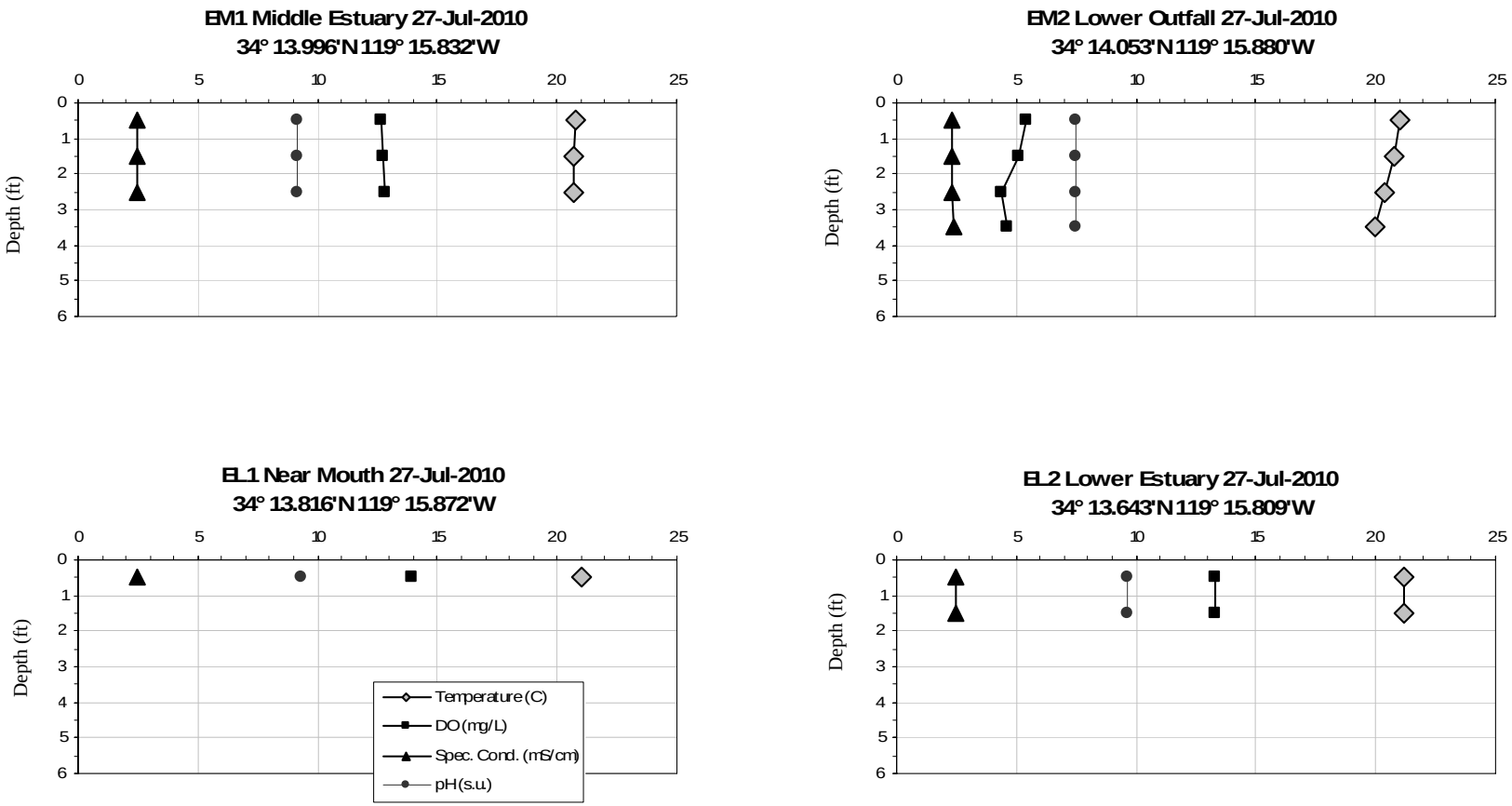

Figure C-12. *In situ* water quality at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, July 2010.

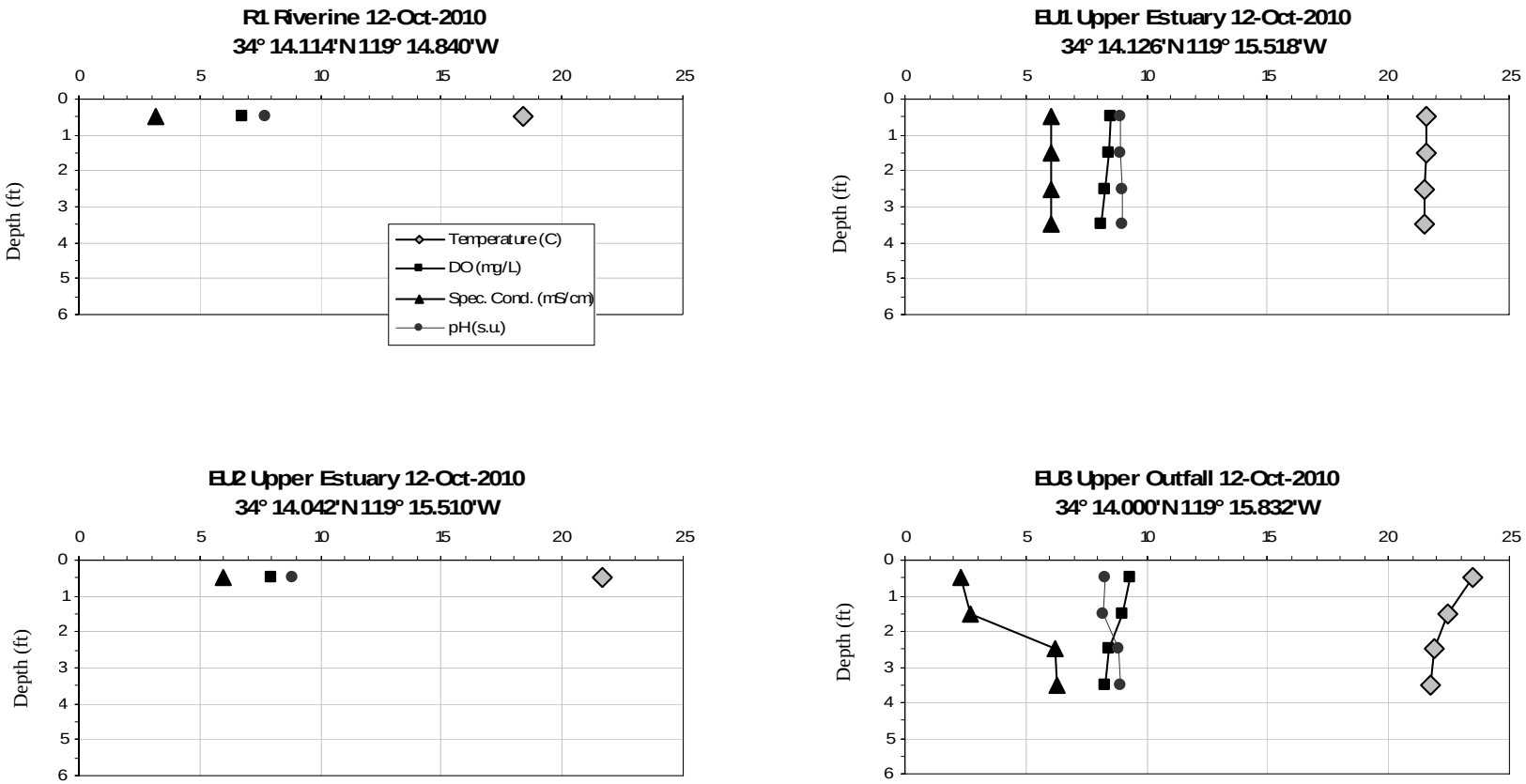

Figure C-13. *In situ* water quality profiles at the Riverine, Upper Estuary, and Upper Outfall locations in the Santa Clara River Estuary, October 2010.

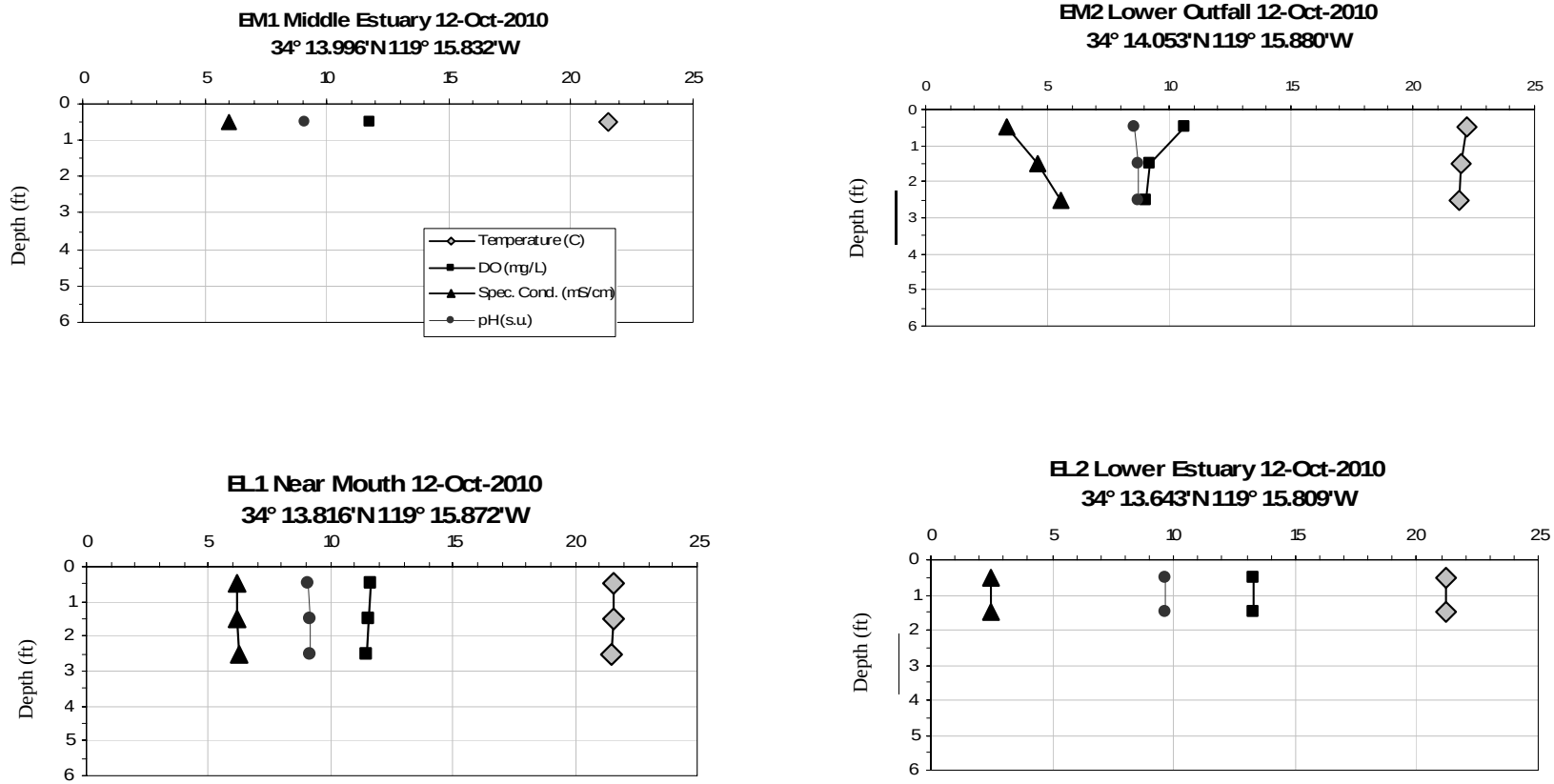

Figure C-14. *In situ* water quality profiles at the Middle Estuary, Lower Outfall, Near Mouth, and Lower Estuary locations in the Santa Clara River Estuary, October 2010.

Appendix D

Compiled Plant Species List (2009–2010)

Scientific name	Common name	Family	Native?
Abronia maritima	red sand verbena	Nyctaginaceae	Y
Abronia umbellata	pink sand verbena	Nyctaginaceae	Y
Ambrosia chamissonis	silver burr ragweed	Asteraceae	Y
Ambrosia psilostachya	Cuman ragweed	Asteraceae	Y
Anemopsis californica	yerba mansa	Saururaceae	Y
Araujia sericifera	white bladderflower	Asclepiadaceae	N
Argentina egedii	Pacific silverweed	Rosaceae	Y
Arundo donax	giant reed	Poaceae	N
Astragalus pycnostachyus var. *lanosissimus*	marsh milkvetch	Fabaceae	Y
Atriplex lentiformis	big saltbush	Chenopodiaceae	Y
Atriplex triangularis	triangle orache	Chenopodiaceae	Y
Azolla sp.	mosquitofern	Azollaceae	
Baccharis pilularis	coyotebrush	Asteraceae	Y
Baccharis salicifolia	mule's fat	Asteraceae	Y
Bolboschoenus maritimus	common threesquare	Cyperaceae	Y
Bromus diandrus	ripgut brome	Poaceae	N
Cakile maritima	European searocket	Brassicaceae	N
Calystegia soldanella	seashore false bindweed	Convolvulaceae	Y
Camissonia cheiranthifolia	beach suncup	Onagraceae	Y
Carpobrotus chilensis	sea fig	Aizoaceae	N
Carpobrotus edulis	hottentot fig	Aizoaceae	N
Chenopodium ambrosioides	Mexican tea	Chenopodiaceae	N
Clematis ligusticifolia	western white clematis	Ranunculaceae	Y
Conium maculatum	poison hemlock	Apiaceae	N
Conyza canadensis	Canadian horseweed	Asteraceae	Y
Corethrogyne filaginifolia var. *filaginifolia*	common sandaster	Asteraceae	Y
Cortaderia jubata	purple pampas grass	Poaceae	N
Cotula coronopifolia	common brassbuttons	Asteraceae	N
Croton californicus	California croton	Euphorbiaceae	Y
Cuscuta sp.	dodder	Cuscutaceae	Y
Cynodon dactylon	Bermudagrass	Poaceae	N
Cyperus esculentus	yellow nutsedge	Cyperaceae	Y
Distichlis spicata	saltgrass	Poaceae	Y
Echinochloa crus-galli	barnyardgrass	Poaceae	N
Equisetum sp.	horsetail	Equisetaceae	Y
Ericameria ericoides	California goldenbush	Asteraceae	Y
Eriogonum parvifolium	seacliff buckwheat	Polygonaceae	Y
Euthamia occidentalis	western goldentop	Asteraceae	Y
Foeniculum vulgare	sweet fennel	Apiaceae	N
Frankenia salina	alkali seaheath	Frankeniaceae	Y
Heliotropium curassavicum	salt heliotrope	Boraginaceae	Y
Heterotheca grandiflora	telegraphweed	Asteraceae	Y
Hirschfeldia incana	shortpod mustard	Brassicaceae	N

Scientific name	Common name	Family	Native?
Isocoma menziesii var. *vernonioides*	Menzies' goldenbush	Asteraceae	Y
Jaumea carnosa	marsh jaumea	Asteraceae	Y
Juncus acutus	spiny rush	Juncaceae	Y
Juncus torreyi	Torrey's rush	Juncaceae	Y
Lemna sp.	duckweed	Lemnaceae	
Leymus triticoides	beardless wildrye	Poaceae	Y
Ludwigia sp.	primrose-willow	Onagraceae	
Melilotus albus	yellow sweetclover	Fabaceae	N
Myoporum laetum	ngaio tree	Myoporaceae	N
Nasturtium officinale	watercress	Brassicaceae	Y
Nicotiana glauca	tree tobacco	Solanaceae	N
Oenothera elata	Hooker's evening-primrose	Onagraceae	Y
Opuntia littoralis	coastal pricklypear	Cactaceae	Y
Persicaria hydropiperoides	swamp smartweed	Polygonaceae	Y
Persicaria lapathifolium	curlytop knotweed	Polygonaceae	Y
Phacelia sp.	phacelia	Hydrophyllaceae	Y
Pluchea odorata	sweetscent	Asteraceae	Y
Polypogon monspeliensis	annual rabbitsfoot grass	Poaceae	N
Populus balsamifera ssp. *trichocarpa*	balsam poplar	Salicaceae	Y
Ricinus communis	castorbean	Euphorbiaceae	N
Rubus ursinus	California blackberry	Rosaceae	Y
Rumex crispus	curly dock	Polygonaceae	N
Rumex sp.	dock	Polygonaceae	
Salix exigua	narrowleaf willow	Salicaceae	Y
Salix laevigata	red willow	Salicaceae	Y
Salix lasiolepis	arroyo willow	Salicaceae	Y
Salix lucida ssp. *lasiandra*	shining willow	Salicaceae	N
Sarcocornia pacifica	pickleweed	Chenopodiaceae	Y
Schoenoplectus acutus var. *occidentalis*	common tule	Cyperaceae	Y
Schoenoplectus americanus	Olney's three-square bullrush	Cyperaceae	Y
Schoenoplectus californicus	southern bulrush	Cyperaceae	Y
Spergularia sp.	sandspurry	Caryophyllaceae	
Stephanomeria virgata ssp. *pleurocarpa*	wand wirelettuce	Asteraceae	Y
Tamarix sp.	tamarisk	Tamaricaceae	N
Toxicodendron diversilobum	Pacific poison oak	Anacardiaceae	Y
Typha domingensis	southern cattail	Typhaceae	Y
Typha latifolia	broadleaf cattail	Typhaceae	Y
Urtica dioica	stinging nettle	Urticaceae	Y
Washingtonia filifera	California fan palm	Arecaceae	Y
Xanthium strumarium	rough cockleburr	Asteraceae	Y
Yucca sp.	yucca	Agavaceae	Y

Appendix E

Climate Change Memorandum

(Carollo Engineers 2011)

City of Ventura
Santa Clara River Estuary Special Studies

TECHNICAL MEMORANDUM
ESTIMATE OF FUTURE CONDITIONS DUE TO CLIMATE CHANGE

FINAL
March 2011

2700 YGNACIO VALLEY ROAD • SUITE 300 • WALNUT CREEK, CALIFORNIA 94598 • (925) 932-1710 • FAX (925) 930-0208

City of Ventura
Santa Clara River Estuary Special Studies

TECHNICAL MEMORANDUM
ESTIMATE OF FUTURE CONDITIONS DUE TO CLIMATE CHANGE

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ESTIMATE OF FUTURE CONDITIONS DUE TO CLIMATE CHANGE

1.0 INTRODUCTION / SUMMARY

The City of Ventura owns and operates the Ventura Water Reclamation Facility (VWRF), which discharges tertiary treated municipal wastewater to the Santa Clara River Estuary (Estuary) just south of the City near the mouth of the Santa Clara River. Under the Water Quality Control Policy for the Enclosed Bays and Estuaries of California, discharges of municipal wastewater to enclosed bays and estuaries are to be phased out except in circumstances where the discharge is shown to enhance the quality of receiving waters. To address this issue regarding a finding of enhancement, the Regional Board required the City to complete the "Special Studies for the Santa Clara River Estuary" as a condition of the City's NPDES discharge permit (CA0053651).

The Special Studies include:

- Estuary Subwatershed Study (to be completed by March 2011) – to evaluate the physical and biological function of the Estuary affected by the discharge to determine whether the discharge to the Estuary provides an ecological enhancement now or under different conditions such as a decreased discharge to the Estuary.
- Treatment Wetlands Study (to be completed by March 2010) – to evaluate the feasibility of implementing a constructed treatment wetland to further improve the water quality of the VWRF tertiary discharge by reducing nutrients and other constituent concentrations to further promote receiving water quality improvements.
- Recycled Water Market Study (to be completed by March 2010) – to evaluate and quantify the feasibility of expanding the City's existing reclaimed water system through evaluation of potential users within a five-mile radius of the VWRF (study area).

As part of the Estuary Subwatershed Study, this technical memorandum (TM) provides the results of the scientific literature review of future global climate change scenarios relevant to the region of the Estuary ecosystem. Specifically, the TM describes the potential effects of global climate change in terms of subsequent changes in atmospheric temperature, sea level, and precipitation patterns/events, with the following recommended projections for use in the next subtask of this study:

- Projected range of increase in atmospheric temperature
  - 2050 – 1.0 to 3.0˚C (1.8 to 5.4˚F), average of 2.0˚C (3.6˚F); and
  - 2100 – 1.1 to 5.0˚C (2.0 to 9.0˚F), average of 3.0˚C (5.5˚F).

- Projected range of rise in mean sea levels
  - 2050 – 0.7 to 2.0 feet (0.2 to 0.6 meters); and
  - 2100 – 1.6 to 6.6 feet (0.5 to 2.0 meters).

- Projected change in precipitation patterns/events
  - There is no scientific consensus on the projected change in total average annual precipitation; however, the regional circulation models show little change, if any;
  - The frequency of extreme daily (24-hour) events has increased by 58-69% since 1948, which is exceeding the projected increase in frequency of these events. Below are the projected increases in event frequency (relative to the period of 1981-2000):
    - 2050 – by a factor of two; and
    - 2100 – by a factor of three.

The information in this TM will be used in a subsequent task to assess the impacted physical, water quality, and biological dynamics in the Estuary and Subwatershed ecosystems due to a warming climate.

2.0 PURPOSE AND SCOPE OF STUDY

The goal of this study is to conduct and present the findings of a scientific literature review and data analysis of the effects of future climate change scenarios on the Estuary ecosystem. More specifically, this TM summarizes the projected changes in local atmospheric temperature, sea levels, and precipitation patterns/events. This information will be used to examine the state of the Estuary ecosystem under projected conditions. To accomplish this, historical records of local atmospheric temperature, mean sea level (as well as the extreme high and low tide levels), and precipitation were considered.

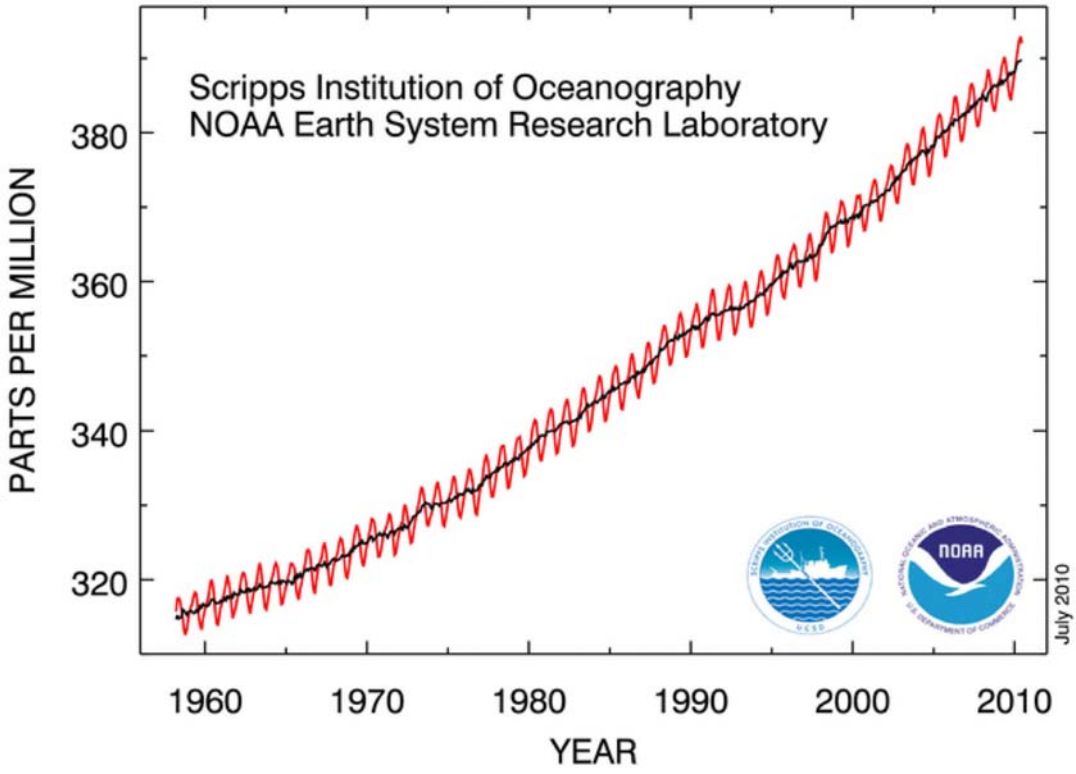
The historic records used in this study are verified data retrieved from the National Climatic Data Center (NCDC) (http://www.ncdc.noaa.gov/oa/mpp/freedata.html) for atmospheric temperature and precipitation in Ventura. Data from the National Oceanic and Atmospheric Administration's (NOAA) National Ocean Service (NOS) Center for Operational Oceanographic Products and Services (COOPS) (http://tidesandcurrents.noaa.gov/index.shtml) were used for precipitation and water levels measured at the Santa Barbara and Santa Monica precipitation and tide gages, respectively. This study then used the latest projected ranges of changes in atmospheric temperature, tide levels, and precipitation patterns/events due to global climate change, which were extracted from published scientific literature. These projected ranges will be used in subsequent tasks to estimate the potential changes in conditions that the Estuary ecosystem will face by considering impacted physical, water quality, and biological dynamics due to global climate change.

3.0 BACKGROUND

The chemistry and dynamics of atmospheric greenhouse gases (GHG) (water vapor, carbon dioxide, and other gases) act to hold heat in the atmosphere and create a natural greenhouse effect for the planet. Without these gases in the atmosphere, the earth's average temperature would be much lower and life could not sustain itself. Since the onset of the industrial revolution, however, data show that human-generated emissions of GHGs (e.g., carbon dioxide, methane, nitrous oxide, chlorofluorocarbons, and other gases) have been accumulating in the atmosphere and are intensifying the earth's natural greenhouse effect more rapidly than expected (Rahmstorf, *et al.* 2007).

Figure 1 shows the longest continuous record (1958 to present day) of monthly atmospheric carbon dioxide (CO_2) concentrations available in the world measured at the Mauna Loa Observatory in Hawaii. The Mauna Loa record shows a 22.6% increase in the mean annual concentration, from 316 parts per million by volume (ppmv) in 1959 to 387 ppmv in 2009.

Figure 1 Monthly Atmospheric Concentration of Carbon Dioxide Observed at Mauna Loa Observatory 1958 to 2010.



The Intergovernmental Panel on Climate Change's (IPCC) Fourth Assessment Report (AR4) of 2007 presented six scenarios of varying ranges in projected increases of CO_2 levels (dating from the pre-industrial level of about 260-280 ppmv) and the subsequent increase in global mean atmospheric surface temperatures. The entire range of increased temperature is 1.1 to 6.4 degrees Celsius (2.0 to 11.5 degrees Fahrenheit) by the year

2100 with an average increase of 2.8 degrees Celsius (5.1 degrees Fahrenheit). Although there is large uncertainty about future emissions of GHGs and how and when the earth's climate will respond to the enhanced concentrations of GHGs, various studies report that detectable changes are already under way.[1] The most likely outcomes are increases in atmospheric temperature and changes in precipitation, soil moisture, and sea level, which could have adverse effects on many ecological systems, as well as on human health, infrastructure, and the economy.

The scientific literature referenced in the following sections of this TM include key studies recently analyzing global climate change impacts, which generally or specifically affect the Estuary in terms of changes in atmospheric temperature, sea (tide) level, and precipitation patterns. The literature is identified as being the most relevant or, in the case of the IPCC, internationally recognized analyses with implications for California.

4.0 TEMPERATURE

Historically, much of California (including the Estuary) has warm, dry summers and cool, wet winters, close to that of a Mediterranean-type climate. The proximity of the Pacific Ocean moderates the Estuary's climate resulting in an average annual atmospheric temperature of approximately 15.5 degrees Celsius (60 degrees Fahrenheit), with an average annual maximum of 21 (70) and an average annual minimum of 9.5 (49) degrees Celsius (Fahrenheit) based on observations collected at Ventura's National Weather Station (No. 049285). Figure 2 shows the monthly average maximum, monthly mean, and monthly average minimum temperatures for the years 1900 through 2010.

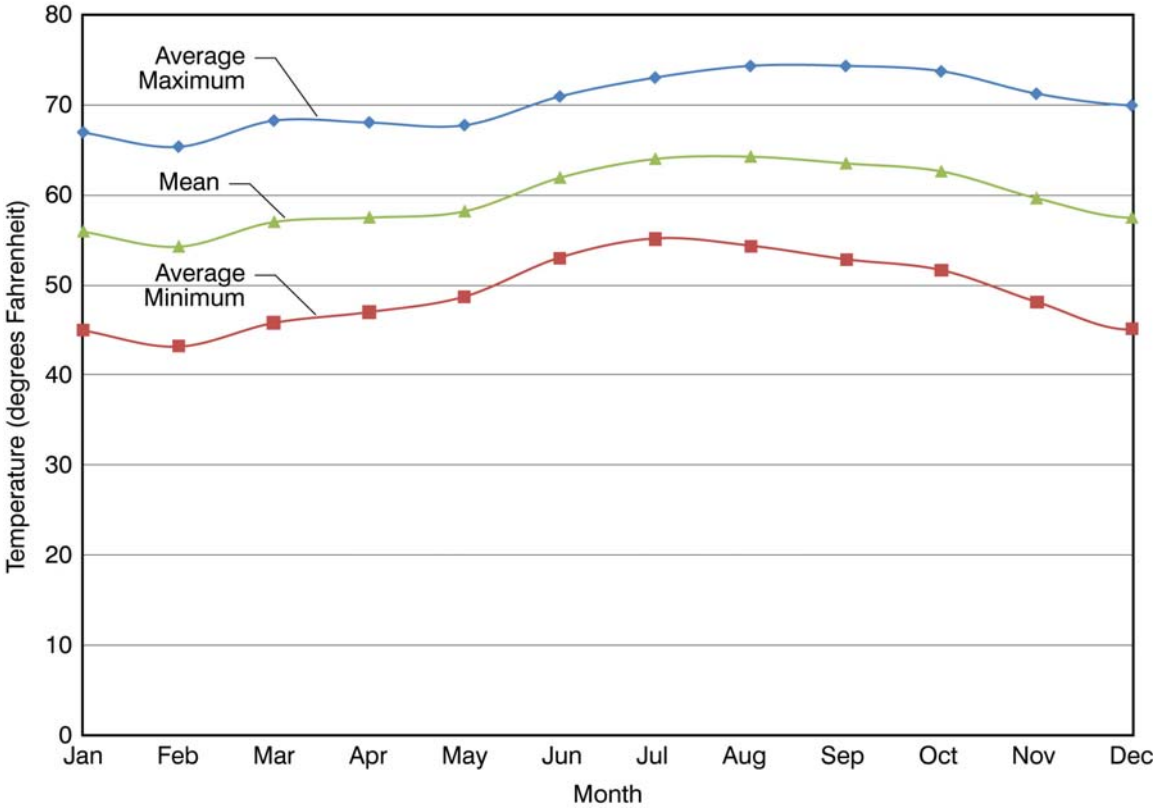
A new National Aeronautics and Space Administration Goddard Institute for Space Studies (NASA GISS) analysis of global surface temperature shows that the year 2009 was tied for the second warmest year since modern records began in 1880.[2] While 2008 was the coolest year of the decade, due to strong cooling of the tropical Pacific Ocean, 2009 saw a return to near-record global temperatures and was only a fraction of a degree cooler than 2005 (the warmest year on record) and tied with a cluster of other years — 1998, 2002,

---

[1] There are two main sources of uncertainty about future climate: First, no one knows what humans will do in the future. Will humans continue with "business as usual" and consume even greater amounts of fossil fuels for energy, or will we reduce our energy demand, increase efficiency, and turn to more sustainable and less carbon-intensive fuels? The answer is crucial to the trajectory of atmospheric GHG concentrations, and therefore to the degree of future warming. Second, we do not precisely know the "climate sensitivity," which is the atmospheric temperature response to a given change in the radiative forcing associated with a change in GHG concentrations. This is because the physics and chemistry of the atmosphere and ocean are not understood well enough to develop reliable models. Estimates range from 1.5° to 4°C warming with a "best-guess" of about 3°C for a doubling of CO_2 concentration in the atmosphere to about 540 ppmv. This is the crux of the "climate debate" over global warming: "What is the climate sensitivity?"

[2] http://www.giss.nasa.gov/research/news/20100121/

Figure 2 Monthly Average Maximum, Mean, and Average Minimum Atmospheric Temperatures in Degrees Fahrenheit Observed at Ventura for 1900-2010.



2003, 2006 and 2007. According to the GISS team, the earth has warmed 0.6 degrees Celsius (about 1 degree Fahrenheit) over the past 30 years, and 0.8 degrees Celsius (about 1.4 degrees Fahrenheit) over the past 100 years (Gutro, 2006).

The IPCC (2007) AR4 concluded that improved modeling studies were converging on a global mean surface temperature increase of 2.8 degrees Celsius (5.1 degrees Fahrenheit) by 2100. Projections resulting from the IPCC and the United Kingdom Hadley Centre's climate model (HadCM2) also show temperatures in California increasing by an average of 2.8 degrees Celsius (5.1 degrees Fahrenheit) with a full range of 1.1 to 5.0 degrees Celsius (2.0 to 9.0 degrees Fahrenheit) by 2100 (Hayhoe, *et al.*, 2004). Consequences of such a temperature increase include increased intensity, duration, and frequency of storm events, and dramatic changes to mountainous snowfall and snowmelt dynamics (Cayan, *et al.*, 2006). The temperature increase is also expected to cause increased melting of land ice (specifically in Greenland and Antarctica) and thermal expansion of the marine mixed layer of the ocean, both of which contribute to sea level rise.

The recommended projected ranges of increase in atmospheric temperature for use in the next subtask are:

- 1.0 to 3.0˚C (1.8 to 5.4˚F), average of 2.0˚C (3.6˚F) by year 2050; and
- 1.1 to 5.0˚C (2.0 to 9.0˚F), average of 3.0˚C (5.5˚F) by year 2100.

5.0 SEA LEVEL AND TIDES

Sea level is the elevation of the ocean relative to a reference elevation. Along the coast, sea level is usually measured relative to the adjacent land and is called relative sea level. For practical purposes, sea level is an average or mean elevation recorded over a time period long enough to exclude the effects of waves (usually longer than one minute). Tide gage data is often presented as 1-hour averages, but mean sea level (MSL), in the context of global climate change and MSL rise, is usually presented as monthly or annual averages. This section discusses sea level and tides near the Estuary and VWRF in Ventura, California. It describes the tide regime and past sea level fluctuations in southern California, as well as the range of possible future sea level rise to support the subsequent task of predicting impacts of climate change on the Estuary.

The sea level and tide data, analysis, and other related information presented in this TM are applicable to the ocean at the entrance of the Estuary and along the coast in much of southern California. This analysis does not consider if or to what extent sea level or tide conditions may be different inside the Estuary from those of the ocean. However, it has been shown that average water levels may be higher within California coastal lagoons, bays, and estuaries than they are at the mouth in the ocean (Walker *et al.*, 1988). These tidal differences may be associated with ebb or flood tidal bars or due to the presence of sediment sills and other normal flow constrictions at the mouth. In addition, constraints inside an estuary, such as configurations of interior channels, will modify tidal conditions.

5.1 Tidal Datums

A tidal datum is a vertical reference based on a long-term average value of a particular phase of the tide. Since there is no tide gage at Ventura, we rely on tide gages at Santa Barbara and Santa Monica to describe the general characteristics of the tide at the Estuary. Tide measurements began in 1974 and 1933, respectively. The tide data are shown on the NOAA-NOS website in real time, and tidal datum statistics are tabulated and published monthly. Table 1 summarizes the tidal datum values for the tide gages (No. 941-1340 and No. 941-0840, respectively) as calculated for the latest 19-year (1983-2001) Tidal Datum Epoch.

In Table 1, values are presented relative to the North American Vertical Datum (NAVD) of 1988. All tide and land elevation information in this TM is presented in feet relative to NAVD. If corresponding values are required relative to other vertical reference elevations, adjustments can be made using the information in Table 1 (NOAA, 2010).

Table 1	**Tidal Datum Values for Santa Barbara and Santa Monica (1983-2001 Epoch) in Feet Relative to North American Vertical Datum of 1988** **Santa Clara River Estuary Special Studies** **City of Ventura**	
Datum	**Santa Barbara**	**Santa Monica**
Highest Observed	7.26 (19 Jan 1992)	8.31 (30 Nov 1982)
MHHW	5.30	5.24
MHW	4.54	4.50
MSL	2.69	2.60
NGVD	2.72	2.72
MLW	0.89	0.74
NAVD	0	0
MLLW	-0.09	-0.19
Lowest Observed	-3.02 (17 Dec 1933)	-3.04 (17 Dec 1933)

Mean lower-low water (MLLW) is defined as the average of the lowest observed water level each day over a specific 19-year tidal epoch, currently defined as the years 1983-2001. This 19-year value is chosen as the nearest whole number of years spanning the 18.6-year lunar node cycle. It is also a sufficiently long time to average out meteorological and oceanographic variability unrelated to the tides. Mean higher-high water (MHHW) is similarly defined, except that the highest water readings for each day are averaged. Mean low water (MLW) and mean high water (MHW) are 19-year averages of the two daily low and two daily high waters, respectively. Mean sea level (MSL) is the average of all readings obtained during the tidal epoch. The highest and lowest observed values are from the entire tide gage record. The dates when the extreme values occurred are noted for reference. The tidal datums defined above are “floating” datums, in that the elevations change over time relative to the local land elevation as either mean sea level changes or tide conditions change (Flick *et al.*, 2003).

The average daily tidal fluctuation is characterized by two ranges:

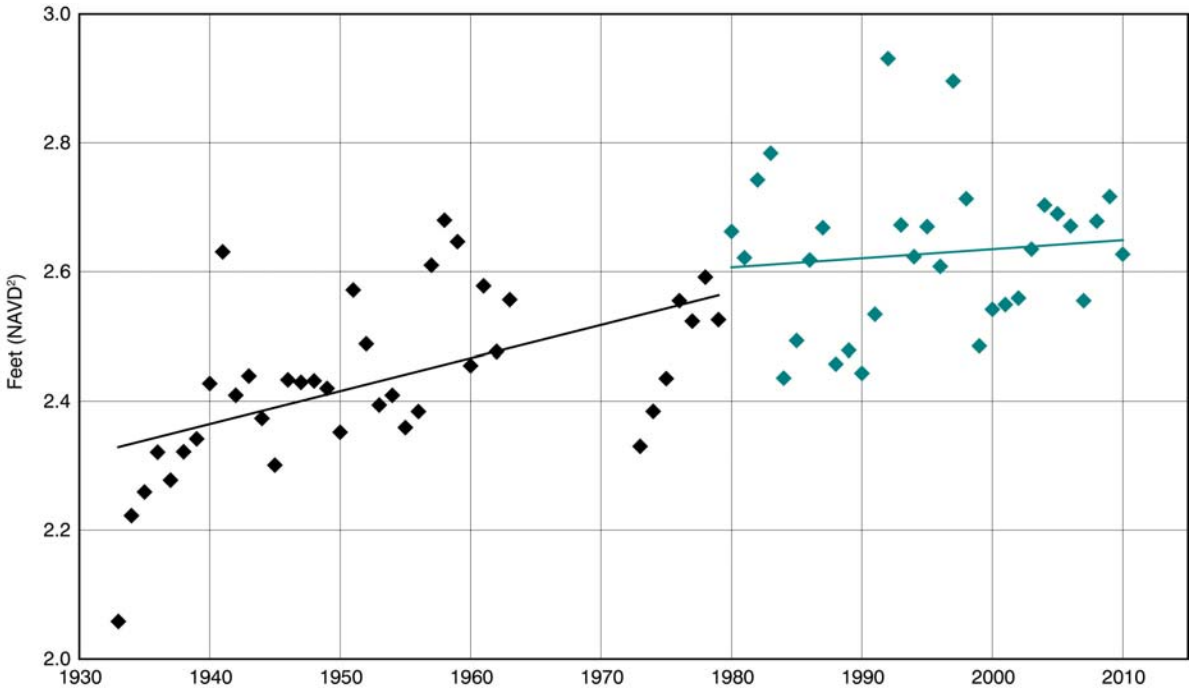
- The diurnal tide range defined as the difference between MHHW and MLLW, or 5.39 feet at Santa Barbara and 5.43 feet at Santa Monica; and
- The mean tide range, which is the difference between MHW and MLW, or 3.65 feet at Santa Barbara and 3.76 feet at Santa Monica.

Table 1 also contains entries for NAVD and the National Geodetic Vertical Datum of 1929 (NGVD), which are fixed, land-based datum reference elevations.

5.2 Observed Local Mean Sea Level and Tide Levels

There has been little if any MSL rise observed along the California coast (and along most of the west coast of the Americas) since at least 1992 when global satellite data first became available (Aviso, 2010). In fact, local tide gage data shows that sea level in southern California has been essentially static since about 1980. Figure 3 shows the annual MSL measured at the Santa Monica tide gage from 1933 through mid-2010.

Figure 3 Annual Mean Sea Levels at Santa Monica from 1933 Through Mid-2010.[1]



Notes:
1. Note the much lower rate of rise from 1980-2010 (green points and line) compared with the overall trend and the earlier trend from 1933-1979 (black points and line).
2. North American Vertical Datum of 1988 (NAVD).

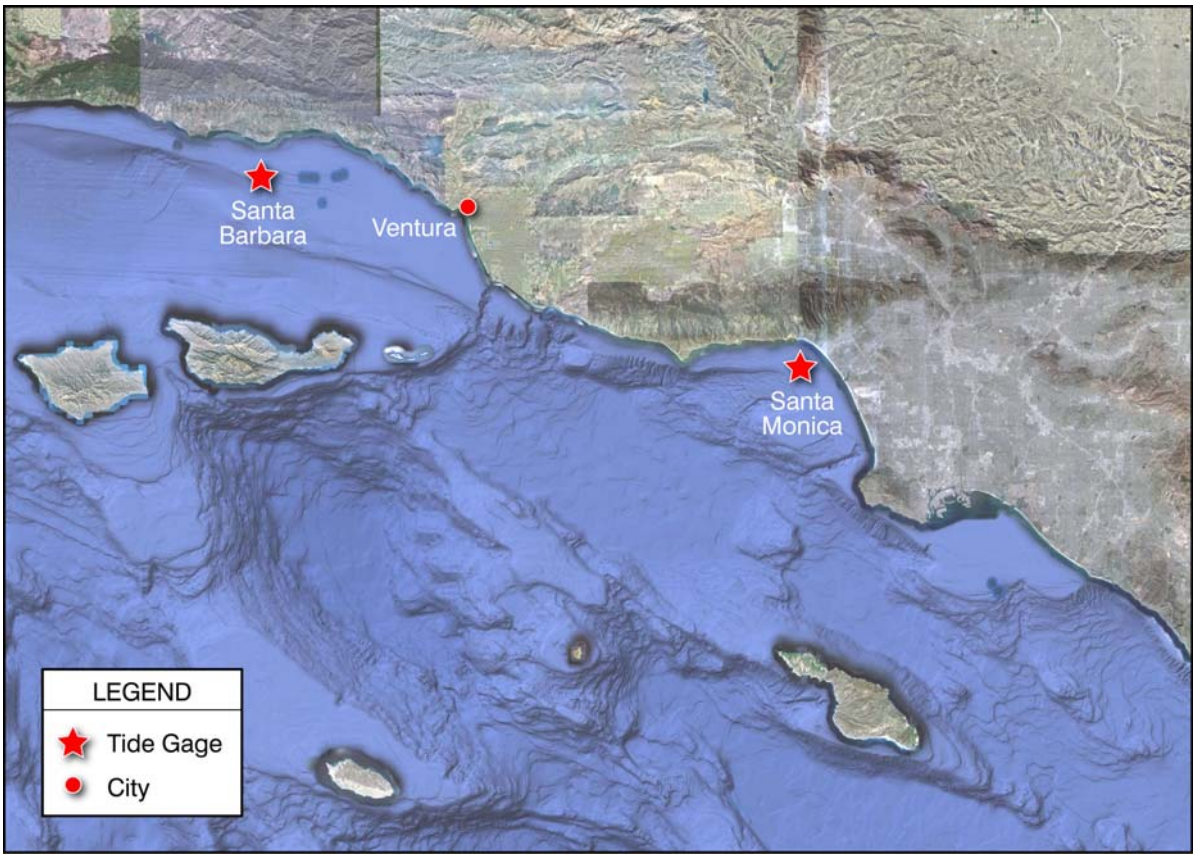
Overall, from 1933 through mid-2010 the sea level at Santa Monica increased at a rate of about 0.46 feet per century. However, the rate from 1933 to 1979 was about 0.51 feet per century, but the rate slowed by a factor of 3.6 to only 0.14 feet per century from 1980 to 2010. Similar results are seen at other locations in California, including La Jolla and San Francisco. In fact, measurements from the long-term station at San Francisco, where data is available since 1855, suggests that episodes of static or even decreasing MSL have been observed before, notably from 1855 through 1900.

The decades-long episodes of static or even falling sea levels represent periods where MSL rise is regionally suppressed. This is most likely due to the response of the north Pacific Ocean to a combination of surface warming or cooling, and changes in wind stress patterns. The ocean's response to wind forcing produces what is commonly referred to as the Pacific Decadal Oscillation (PDO) pattern in sea surface temperature and sea level

elevation across the basin (Mantua *et al.,* 1997). Risk of flooding due to sea level rise will increase when the suppression of MSL rise along the Pacific coast of the Americas relaxes or reverses. When this happens in the coming decades, the southern California region could see rates of sea level rise above the ever-increasing global average rates, which now average about 1 foot per century (Aviso, 2010).

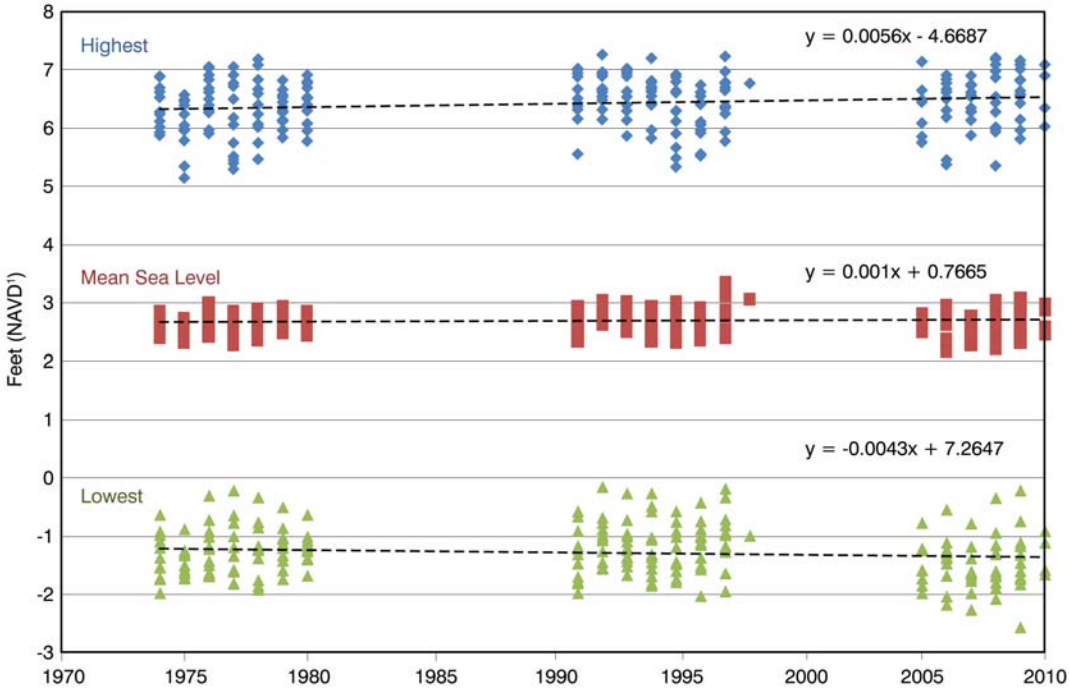
Since there is no tide gage maintained for the City of Ventura at this time, the closest (currently operating) tide gages referenced in this analysis are located in Santa Barbara and Santa Monica. NOAA maintains and reviews the data from each of the tide gages used in this analysis. Figure 4 shows the locations of the tide gages, in addition to the location of the VWRF and the Estuary.

Figure 4 Locations of the Santa Barbara and Santa Monica Tide Gages Relative to the City of Ventura.



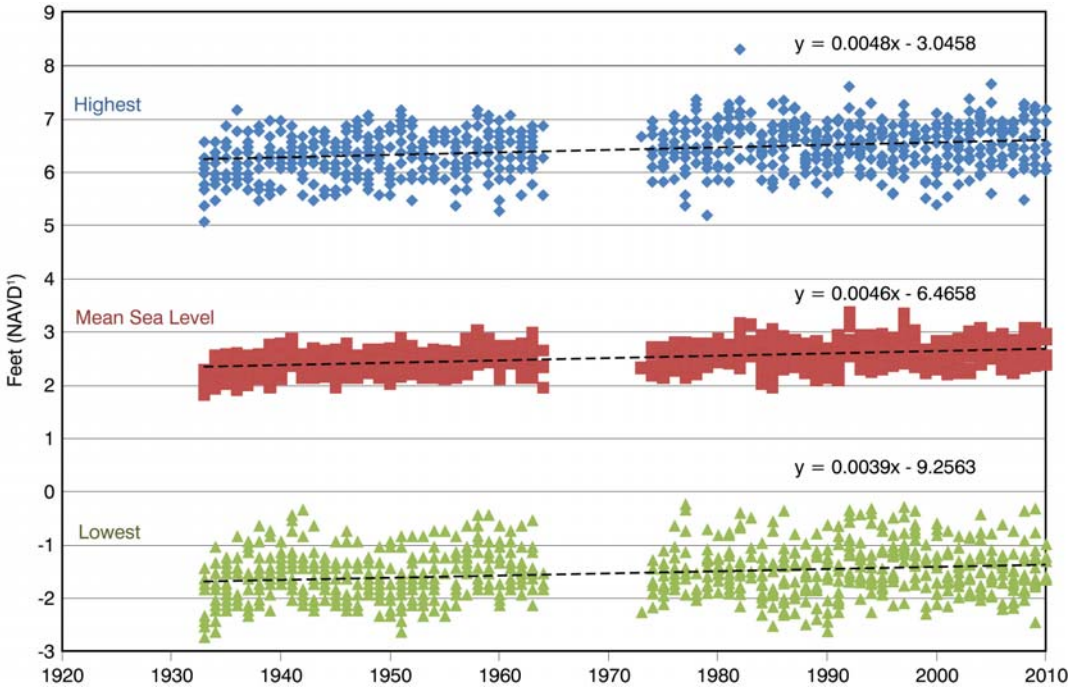
As previously mentioned, verified monthly tide data relative to NAVD from each tide gage are available and were obtained from NOAA's Tides and Currents website. Monthly maximum tide, MSL, and minimum tide levels are plotted in Figures 5 and 6 for Santa Barbara and Santa Monica, respectively.

Figure 5 Monthly Maximum, Mean, And Minimum Sea Levels Observed at Santa Barbara from January 1974 through June 2010.



Notes: 1. North American Vertical Datum of 1988 (NAVD).

Figure 6 Monthly Maximum, Mean, And Minimum Sea Levels Observed at Santa Monica from January 1933 through June 2010.



Notes: 1. North American Vertical Datum of 1988 (NAVD).

Both Figures 5 and 6 show trend lines associated with each tide level record. Figure 4 shows Santa Barbara's monthly maximum tide and MSL increasing (showing an upward trend) over time – the maximum tide increasing at a slightly higher rate than MSL. This means that the high tide, on average, has been getting higher faster than MSL. At the same time, the minimum tide level is decreasing (showing a downward trend). Similarly, Figure 5 shows Santa Monica's monthly maximum tide, MSL, and minimum tide levels are all increasing over time - where the maximum tide level is increasing at a slightly higher rate than MSL, and MSL is increasing at a slightly higher rate than the minimum sea level. In other words, the tide ranges for each gage have been increasing over time.

This result was first documented by Flick *et al.* (2003), who tabulated MSL and tide range data from 90 long-term U.S. tide stations. That study showed upward trends of tide range at several open-ocean locations along the west coast of the U.S., including La Jolla, Los Angeles, San Francisco, California; Charleston and Astoria, Oregon; and Neah Bay, Seattle, and Friday Harbor, Washington. Several other stations showed the opposite trend, with decreasing tide ranges, most notably San Diego Bay, Port San Luis, Monterey, Point Reyes, and Crescent City, California; and Cherry Point, Washington. The cause of the observed changes in tide range is not understood. Therefore, it is not known whether past trends will or will not continue into the future.

5.3 Sea Level Components

Storm surges, inter-annual fluctuations related to El Niño, as well as longer-term effects connected to the Pacific Decadal Oscillation (PDO), significantly influence sea levels on the southern California coast. In addition, global climate change is already occurring and there are likely to be significant future impacts on sea levels and coastal resources. Independent of climate change, vertical land movements also contribute to relative sea level change and astronomical tides cause changes in water level along the California coast up to about 3 meters (10 feet) (Cayan *et al*., 2006).

Based on geological data, as shown in Figure 6 from the IPCC (2001) Third Assessment Report (TAR), global MSL has risen at least 400 feet (120 m) over the past 18,000 to 20,000 years. From 15,000 to about 7,000 years ago, the average rate of MSL rise was about 4.5 feet per century (1.4 meters per century). This rise was erratic, and times of much higher rates occurred (such as during Meltwater Pulse 1 shown on Figure 7) when sea level rose at 13 feet per century (4 meters per century) for about 500 years. The rate slowed to an average of about 0.16 feet (0.05 meters) per century over the last 5,000 years, and to an average rate of 0.4-0.8 inches (0.01-0.02 meters) per century over the last 3,000 years (Church *et al*., 2001).

However, tide gage records available since the late 19th century show that MSL has risen by an average of 6.7 inches (0.17 meters) per century since 1950 (Church and White, 2006). An acceleration of the rate of MSL rise since the mid-19th century is illustrated in Figure 8.

Figure 7 Mean Sea Level Rise Over the Past 20,000 Years Associated with Ice-Cap Retreat and Global Warming of About 8 Degrees Celsius (IPCC, 2001).

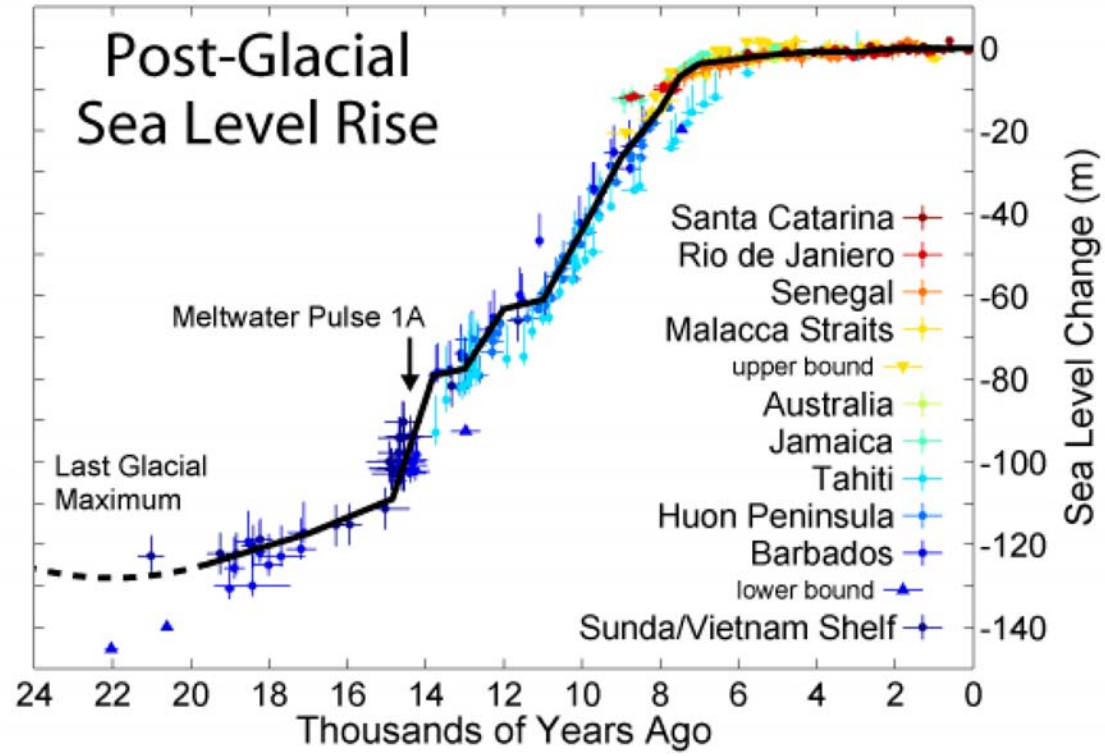
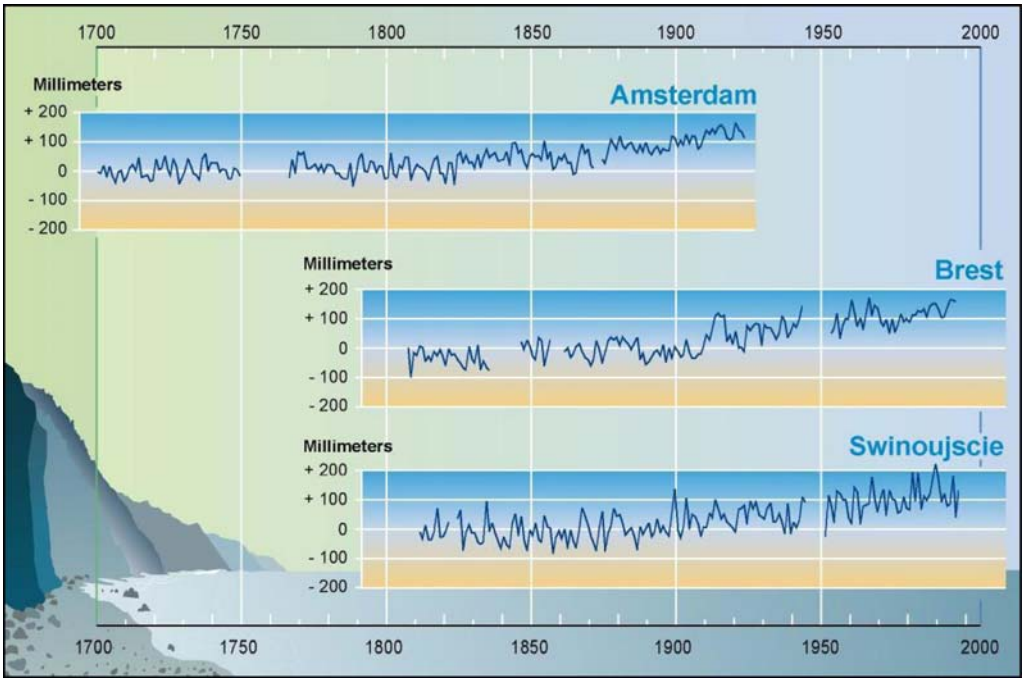


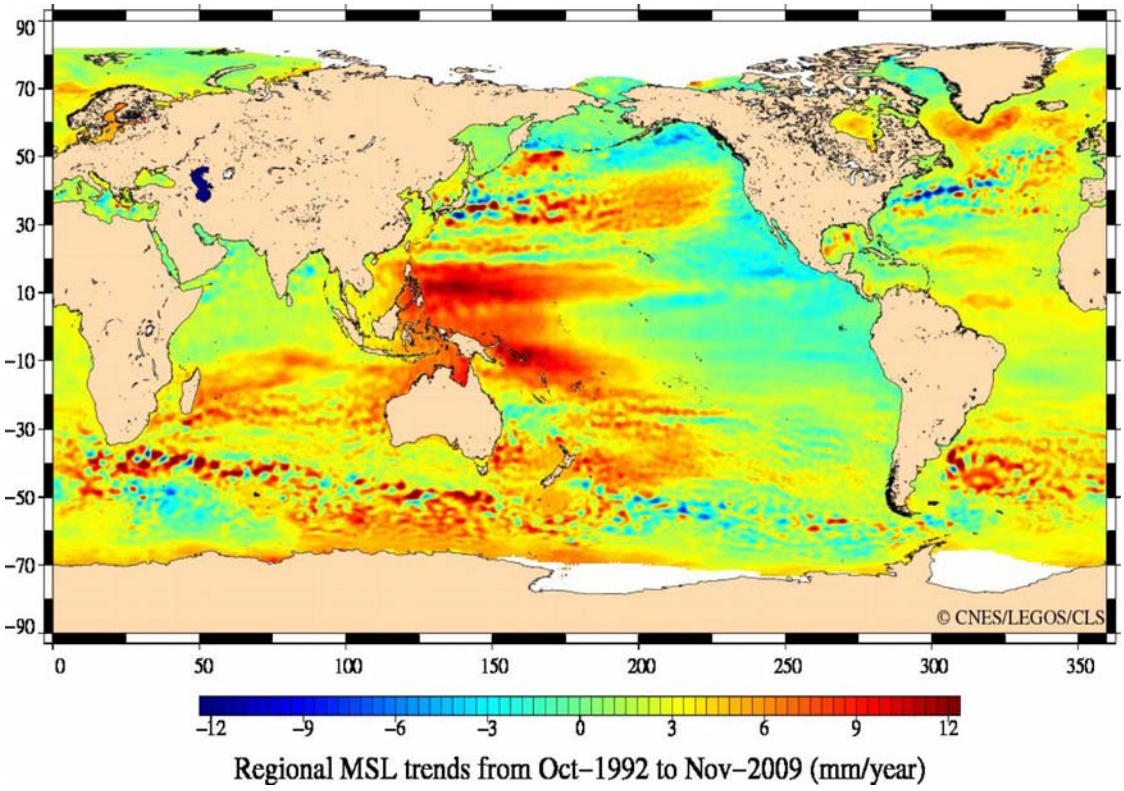
Figure 8 Long-Term Tide Gage Data From Amsterdam, Brest, And Swinoujscie (Poland) Showing Mean Sea Level Rise Over The Past 300 years (IPCC, 2001).



Since the early 1990s, MSL rise has been routinely measured by high-precision altimeter satellites. From 1993 to 2009, data show that the average rate of global MSL rise amounted to 13 inches (0.33 meters) per century (Ablain *et al.*, 2009), suggesting that global MSL rise is still accelerating. It is believed that thermal expansion and melting land ice each account for about half of the observed MSL rise to date, although there is some uncertainty in the estimates (Meehl *et al*., 2007).

Since the processes contributing to changes in sea level all have significant spatial variability, it has been suggested that there will be considerable geographic variability in changes in the rate of relative sea level rise (Walsh *et al*., 2005). In fact, satellite data and tide gage observations reveal there has been little sea level rise along the west coast of the Americas over the past several decades. Figure 9 shows much higher rates of sea level rise in the western Pacific Ocean (warm, yellow and red colors) and much lower rates, or actual decreases, in the eastern Pacific Ocean (cool, blue and green colors) since at least 1992.

Figure 9 Map Showing High Rates of Mean Sea Level Rise in the Western Pacific Ocean and Low Rates (or Actual Decreases) in the Eastern Pacific Ocean Off the Coast of the Americas (Aviso, 2010).



The following sections outline the important components of sea level and the processes that cause sea level fluctuations. The emphasis is on those processes and patterns that contribute to high water levels, since it is water level peaks that allow waves, especially simultaneously-occurring storm waves, to cause coastal flooding, longshore sand transport,

beach erosion, and lagoon entrance scour and filling. These processes include the tide, surges due to storms, inter-annual water level fluctuations that are usually related to El Niño and decadal changes correlated to the PDO mentioned above, and the longer-term MSL rise (including past and possible future levels) associated with global climate change.

5.3.1 Land Ice Melt

Future rates of melting or disintegration of the Greenland and West Antarctic Ice Sheets are the single largest uncertainty in making estimates of future rates of sea level rise (Allison *et al*., 2009). It is projected that global climate change will reduce the amount of water frozen in glaciers and ice caps, specifically in Greenland and Antarctica. Total ocean volume, and hence sea level, will increase as this meltwater moves from the land to the oceans.[3] An important uncertainty relates to whether discharge of ice from the ice sheets will continue to increase as a consequence of accelerated ice flow, as has been observed in recent years. This would add to the rate of sea level rise, but quantitative projections cannot be made with confidence, due to a limited understanding of the relevant processes (Meehl *et al.*, 2007).

Over a long period of time, more extreme sea level increases are expected. The IPCC (2001) TAR suggested that continued GHG emissions could trigger polar ice cap melting beyond 2100, resulting in a sea level rise greater than 5 meters (16 feet) within the next millennium (Church *et al*., 2001). However, the IPCC (2007) AR4 showed a decrease in the projected upper range of global MSL rise suggesting an increase in accuracy of modeling efforts and a decreased likelihood of rapid ice cap melt. Unfortunately, models currently referenced by the IPCC do not consider ice dynamics and, therefore, provide only lower limits (Rignot and Kanagaratnam, 2006). Recent (2006 and later) studies show that both the Greenland Ice Sheet and portions of the Antarctic Ice Sheet are experiencing rapid increases in near-coastal discharge over the past several decades due to climate warming and ice dynamics (Rignot and Kanagaratnam, 2006; Velicogna and Wahr, 2006; van de Wal *et al.*, 2008; Nicholls and Cazenave, 2010). Depending on some model variants, these studies show sea level rising between 12 and 71 inches (~0.30 and 1.80 meters, respectively) by 2100. The upper limit of these estimates is well above IPCC (2007) AR4 projections showing global MSL rise of less than 24 inches (~0.60 meters) for the business-as-usual A1FI GHG emissions scenario (Nicholls and Cazenave, 2010).

Since the 1960s, glaciologist John Mercer (1968) has also been warning the scientific community about the instability of the West Antarctic Ice Sheet. Though it is smaller in mass relative to the entire Antarctic Ice Sheet, Mercer argues that it is held back in a delicate balance by the floating ice shelves at its rim. A slight warming could disintegrate the ice shelves causing a sea level rise of up to 20 feet (6 meters) (Church *et al*., 2001;

[3] This contribution is often called the “eustatic” component of sea level change, since in principle it contributes to sea level changes globally. However, for dynamical reasons changes in “global” sea level are not uniformly distributed.

Weart, 2005). The probability or timing of this kind of catastrophic event is highly uncertain and not considered in this analysis.

5.3.2 Thermal Expansion

The ocean water density (and subsequent volume) changes whenever a change in the heat content or salinity takes place.[4] Global ocean water volume is increased by decreases in the density of the existing water mass through warming of the marine mixed layer.[5] Atmosphere-Ocean General Circulation Model (AOGCM) simulations project that by 2100 the increase in ocean water volume due to thermal expansion will account for more than half of the average rise, but land ice will lose mass increasingly rapidly as the century progresses (Meehl *et al.*, 2007).

5.3.3 Storm Events

Changes in wave or storm patterns may also occur under global climate change (Schubert *et al*., 1998). Meteorological forcing contributes to increases in mean water levels along the coast, independent of eustatic and steric contributions to relative sea level. The frequency, direction, intensity, and duration of winds are important variables which, when combined with water depth and coastal morphology, determine wave and storm-surge forcing at the coast (Meehl *et al*., 2007). Large-scale ocean-atmospheric circulation and climate regimes, such as the El Niño Southern Oscillation (El Niño) and the PDO (discussed earlier), also have implications for beach and barrier stability.

Southern California sea level fluctuations on time scales ranging from the synoptic storm scale of a few days to El Niño events, which occur every two to seven years, are important since they can range up to about one foot in height (Bromirski *et al.*, 2003; Flick, 1998). This is larger than the past century's total MSL rise (discussed below), which was about 0.7 foot. Extreme high coastal water levels are a result of the combined effects of one or more of the following: peak high tides, low barometric pressure, high winds, elevated sea levels due to warmer than normal ocean temperatures usually associated with El Niño, large wave-driven runup, and freshwater runoff in and around estuaries.

5.3.4 Extreme High Tides

Extreme high tides result from the coincidence of various astronomical cycles driven by the varying gravitational forces associated with the orbits of the moon around the earth, and the earth and moon around the sun. Tides provide the largest component of sea level change in southern California on every time scale except many thousands of years (ice ages).

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[4] This part of sea level change is called the "steric" height contribution, which results from changes in density as a function of temperature and salinity.

[5] The marine mixed layer extends up to only a few hundred meters in depth from the ocean surface. While it is proportionally less than the deeper ocean layers (on average ~4,000 meters in depth), deep ocean layers take centuries to millennia to warm.

Furthermore, the tide is one of the few geophysical phenomena that can be accurately predicted.

Tides along the California coast are defined as “mixed,” in that the diurnal (once-per-day) constituents that make up the tides are almost as large as the semidiurnal (twice-per-day) constituents. In southern California, there are almost always two high tides and two low tides per 24-hour 50-minute tidal day. However, the highs and lows are respectively unequal in amplitude. The diurnal components of the tides in California arise from the enhanced response of this part of the Pacific Ocean to the forcing associated with the declination of the moon and sun. The highest tide ranges occur in winter and summer when the declination of the sun is largest.

A daily tidal pattern of higher-high water followed by lower-low water, then lower-high water followed by higher-low water repeats in all seasons. This sequence means that tidal outflow (higher-high to lower-low) from lagoons, harbors, and tidal estuaries in southern California tends to be stronger, but of shorter duration – lasting only about six hours – than the average tidal inflow (lower-low to higher-high), which occurs over the approximately 18-hour remainder of the tidal day.

In California, the highest astronomical tides occur in a series of repeated cycles of increasing period starting with the daily higher-high and lower-high sequence mentioned above, moving to the twice per month spring-neap cycle, to the twice per year summer and winter fluctuation in which these peak tides are higher than those in spring and fall owing to the declination of the sun (Lamphier-Gregory *et al.*, 2006). The longer-term peak-tide cycles include a 4.42-year lunar perigee cycle and an 18.6-year lunar node cycle (Zetler and Flick, 1985). These are described and illustrated in the following section.

5.4 Future Sea and Tide Levels

Significant uncertainty exists concerning the trajectory in future sea level due mainly to uncertainty about the future emissions of GHGs, population growth rates, government policies to address emissions, and the actual dynamics of the oceans, atmosphere, and ice sheets. In IPCC (2007) AR4, the range in projected rise in MSL is 7 to 23 inches (0.18 to 0.59 meters) by the year 2100 relative to 1990 levels. However, these projections are based on physical models that do not reproduce the current rate at which the polar ice sheets are melting. During the past five years, there have been major advances in the science of sea level rise. The results of an extensive literature review conducted to determine the accepted projected ranges of rise in MSL are presented in Table 2.

Peer-reviewed studies now estimate a range of rise in MSL between 20 to 79 inches (0.50 to 2.0 meters) by 2100 and recommend that 55 inches (1.40 meters) be used for climate change adaptation planning. Therefore, the IPCC’s conservative estimates should be

Table 2	**Resources and Projections for Rise in Global Mean Sea Level due to Global Climate Change Santa Clara River Estuary Special Studies City of Ventura**				
		2050		**2100**	
Resource	**Units**	Low	High	Low	High
Meehl *et al*., 2007 (IPCC)	Inch	3.8	12.5	7	23
	Meter	0.10	0.32	0.18	0.58
Rahmstorf, 2007	Inch	10.9	30.0	20	55
	Meter	0.3	0.8	0.5	1.4
Pfeffer *et al.,* 2008	Inch			31.5	78.7
	Meter			0.8	2.0
NRC I 1987	Inch				19.7
	Meter				0.5
NRC II 1987	Inch				39.4
	Meter				1.0
NRC III 1987	Inch				59.1
	Meter				1.5
NRC SERDP 1987	Inch				78.7
	Meter				2.0

viewed as minima for planning purposes (Healy, 2007). In the near term (i.e., the next 20 or so years), the major contributors to changes in extreme high tides experienced at the Estuary remain to be the large-scale ocean-atmospheric circulation and climate regimes - El Niño and PDO.

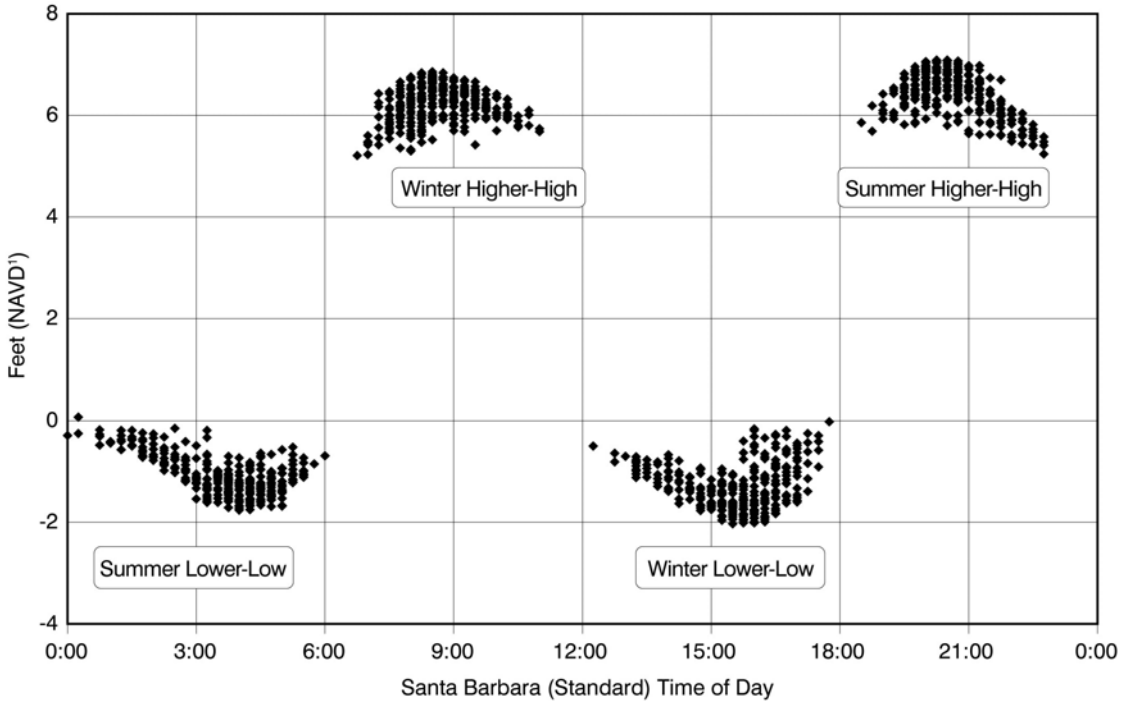
Some studies show a small probability, but extremely high consequences of more rapid and severe sea level rise, resulting from accelerated melting on Greenland and West Antarctic Ice Sheets. Glaciologist, John Mercer, argues that the ice shelves supporting the West Antarctic Ice Sheet have the potential to disintegrate within the next 40 years and result in a 20-foot (6-meter) rise in sea level (Weart, 2005). Much of the world's population lives near the coast, and a sea level rise of this magnitude could displace up to 2 billion people and result in the abandonment of many cities. We do not address this scenario here.

Future tide projections were made for the next 40 years using software developed at Scripps Institution of Oceanography. The following subsections consider how projected sea level rise scenarios will impact the Estuary in terms of daily tide patterns as well as longer lunar-related cycles through 2050 and beyond. Since there is little difference between the tidal conditions experienced at Ventura from those at Santa Barbara and Santa Monica, the projections are applied only to the Santa Barbara tide gage data.

5.4.1 Daily Tide Patterns

The "time of day" in which the highest and lowest tides occur in southern California was demonstrated by Flick (2000). The higher-high tide always occurs in the morning (loosely defined as midnight to noon) during the winter, and in the afternoon (noon to midnight) during the summer. Figure 10 shows that the Santa Barbara monthly peak higher-high and lower-low tides have a tendency to cluster. Winter peak highs tend to occur around 8 AM, with summer highs around 8 PM (20:00); summer lows tend to bunch around 4 AM, with winter lows at about 4 PM (16:00). This pattern is consistent with the previous findings of Flick (2000) who considered all high and low tides between 1983 and 2000 at Los Angeles, which has a nearly identical tide regime.

Figure 10 Shows How the "Time of Day" of Monthly Peak High and Low Tides Cluster in Summer and Winter at Santa Barbara for 2000 to 2050.



Notes: 1. North American Vertical Datum of 1988 (NAVD).

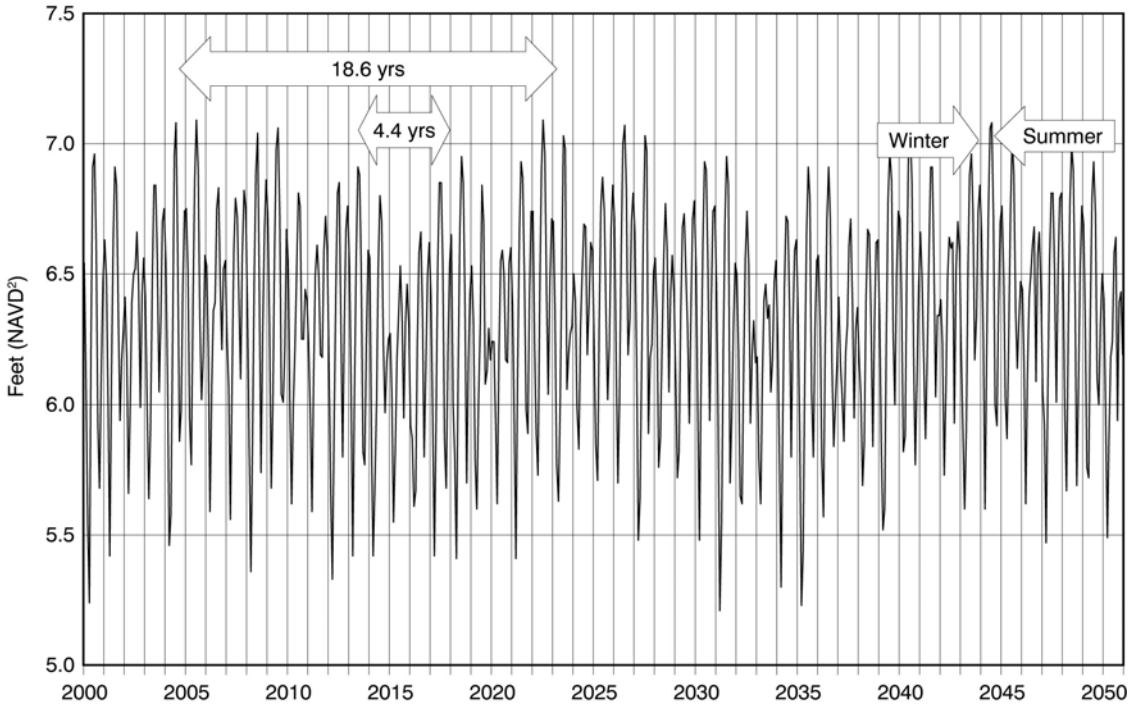
5.4.2 4.42 and 18.6-Year Lunar Cycles

Zetler and Flick (1985) showed that along the west coast certain lunar cycles produce peak high tides every 4.42 and 18.6 years. The highest winter and summer high tides are about 0.5 foot higher during the peaks of these cycles than they are during years in between. Their study revealed that the 4.42-year cycle led to unusually high maximum tides during the El Niño winter of 1982-83, specifically in late January 1983. These peak tides were an important contribution to the extensive coastal flooding and erosion in southern California that occurred at that time, and will continue to be important in the future.

The second important pattern of extreme high tides noticeable along the west coast is the 18.6-year lunar node regression cycle. Together, the 4.42 and 18.6-year cycles produce clear variations in the peak high and low tide patterns, as well as the tide range.

Figures 11 and 12, respectively, show plots of the maximum and minimum monthly predicted tides at Santa Barbara from January 2000 to December 2050. Arrows at the upper left side of Figure 11 show the length of the 4.42-year and 18.6-year periods. Arrows on the upper right of Figure 11 show successive winter and summer maximum tides. This winter-summer pattern, already discussed above, is evident in the plot.

Figure 11 Maximum monthly projected tides for Santa Barbara 2000 to 2050.[1]

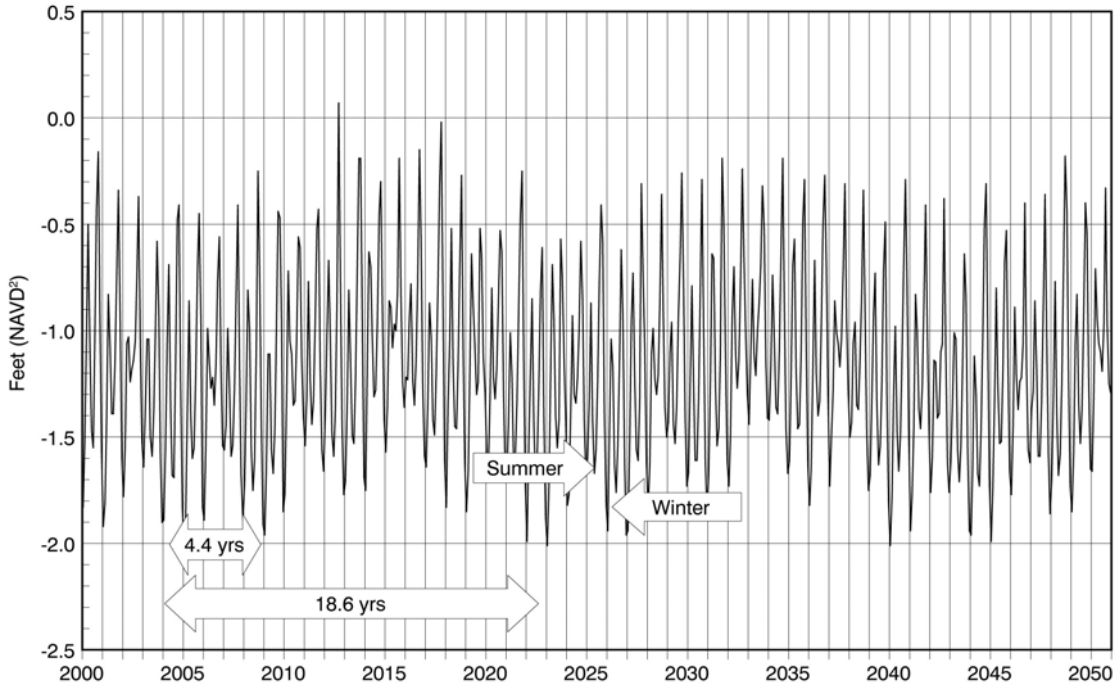


Notes:
1. Note the arrows in the upper left show length of 18.6-year and 4.42-year lunar cycles. Arrows to the right show successive winter and summer maximum tides.
2. North American Vertical Datum of 1988 (NAVD).

Figure 12 shows that the effect of the 18.6-year cycle on the maximum monthly tide values crested in about 2004-05, and will occur again in about 2023. Quantitatively, the effect is about 0.2-0.3 foot (half of the 4.42-year cycle), which causes about 0.5 foot peak tide variation over time. The 4.42-year maximum tide cycle crests in about 2005, 2009, 2013, 2018, and so on. Maximum tide heights from 2000 to 2050 are projected to reach about 7.1 feet NAVD. Peak tide heights during the crests of the 4.42-year cycle regularly exceed 7 feet.

Figure 12 shows a similar pattern of results for the minimum monthly tides at Santa Barbara for 2000-2050. The peak low tides are nearly a mirror image of the peak high tides and show similar quantitative effects for the winter-summer, and 4.42-year and 18.6-year cycles. Minimum tide heights reach about -2 feet NAVD during the 2000-2050 time interval.

Figure 12 Minimum monthly projected tides for Santa Barbara 2000 to 2050.

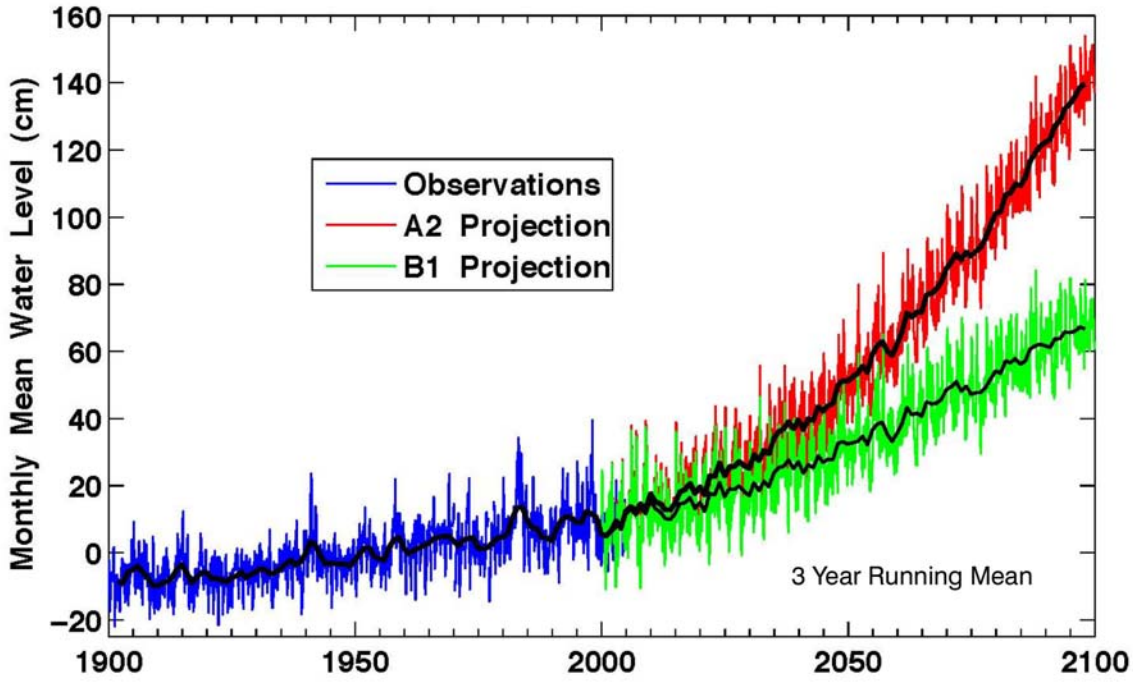


Notes:
1. Note the arrows in the lower left show length of 18.6-year and 4.42-year lunar cycles. Arrows to the right show successive summer and winter minimum tides.
2. North American Vertical Datum of 1988 (NAVD).

5.4.3 Mean Sea Level Rise Projections to Year 2100

While the projections of the 4.42- and 18.6-year high tides are important for the next 20 or more years; when looking out to year 2100, MSL rise due to global climate change will have a larger effect. This is shown in studies by Cayan *et al.* (2009) for San Francisco, where MSL rise was projected using the methods of Rahmstorf (2007). These projections include El Niño fluctuations derived from regionally-adjusted output of general circulation models. Figure 13 shows a plot of monthly MSL at San Francisco over the past century – from 1900-2000 (blue line) – and MSL rise projections for two different future GHG scenarios for 2000 through 2100 (red and green). Also included are the 3-year running mean averages of these monthly curves (black lines).

Figure 13 Observed Mean Sea Level at San Francisco For 1900-2000 (Blue Line) and Projected Sea Level for Two Different Greenhouse Gas Scenarios for 2000-2100 (Red, Green).



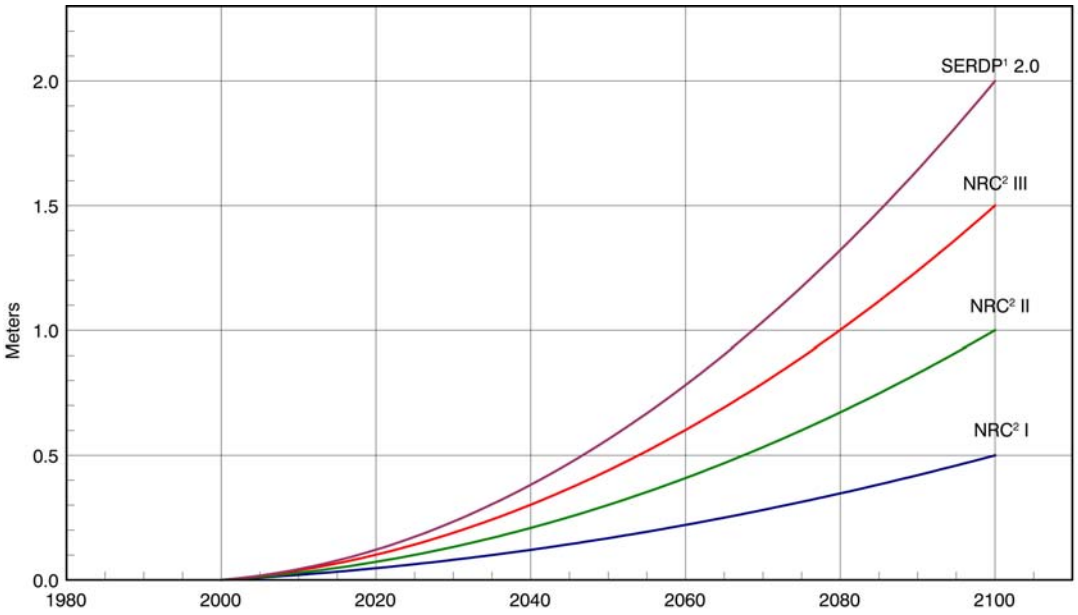
The sharp peaks in the monthly colored lines indicate past (blue) and projected (red, green) modeled El Niño-related sea level fluctuations added to the A2 (red) and B1 (green) scenario MSL projection curves (IPCC, 2001). The MSL projections shown are global and assume that over the next century regional MSL will follow the mean global trajectory. The added fluctuations attributed to El Niño are expected to also be statistically accurate and representative of the fluctuations that will be observed along the entire southern and central California coast.

By the middle of the 21st Century, the upper MSL curves for both projections begin to reach higher levels than the El Niño peaks of the past century. In other words, by the mid-21st century the background average water levels will reach heights only seen during past severe El Niño events. Projected El Niño events and related storm surges and wave action would be expected to produce much more severe erosion and coastal damage and flooding than has been observed in the past. After mid-century, the expected acceleration (the concave-upward shape of the curves) of MSL rise would be the dominant effect (instead of El Niño-related short-term elevated sea levels), if the assumption of the high A2 GHG emissions scenario turns out to be correct. These projections suggest that a total rise of up to 4.5 feet (1.4 m) by 2100 is possible, assuming moderately high global GHG output in the future.

Other approaches for assessing impacts from various future MSL scenarios have been adopted. Most recently, the U.S. Army Corps of Engineers has published a directive (USACOE, 2009) instructing all elements under its Civil Works jurisdiction to consider three specifically-shaped curves with respective end-points of 1.6, 3.3, and 4.9 feet (0.5, 1.0, and 1.5 m) of sea level rise by 2100 relative to 1986. These values are based upon work by the National Research Council of the National Academy of Sciences (NRC, 1987). Implementation by the USACOE lends engineering authority and credibility to this approach.

Flick *et al.* (2010) in ongoing work for the U.S. Navy (sponsored by the Strategic Environmental Research and Development Program, SERDP) has adopted this approach, but considers four scenarios with similarly specified MSL rise values of 1.6, 3.3, 4.9, and 6.6 feet (0.5, 1.0, 1.5. and 2.0 m) of sea level rise by 2100 relative to 2000. The practical differences between these approaches are inconsequential (except for the added fourth scenario). These sea level curves are shown in Figure 14. The curves are constructed assuming an initial sea level rise rate at year 2000 and a constant (linear) increase in that rate calibrated to produce the desired end-point sea level value.

Figure 14 The four mean sea level rise curves adopted from U.S. Army Corps of Engineers (2009) by Flick *et al.* (2010).



Notes: 1. Strategic Environmental Research and Development Program (SERDP)
2. National Research Council (NRC)

These four curves were modeled on the low, intermediate, and high range of possible future MSL rise rate increases first proposed by NRC (1987), with the fourth SERDP "2-m-by-2100" curve added. A constant rate of increase of MSL implies a quadratic-shaped, concave-upward MSL curve. Most, if not all of the published past and projected future MSL

curves are “concave-up” with rise rates that increase over time. An initial rate of rise in a given starting year together with either an end-point value of sea level at any future year, or a specified constant increase of the rate of rise (acceleration) completely defining the sea level curve in this simple quadratic form. Given all the uncertainties associated with the range of possible future MSL rise projections, the precise shape of the scenario curves is not of consequence.

Although the NRC (1987) study could now be considered “old science,” MSL rise curves based on it (and including the new SERDP curve) more than span the range of (under-estimated) MSL rise projections in the IPCC TAR (2001) and the IPCC AR4 (2007), which range from 0.3-2.9 feet (0.09-0.88 meters) and 0.6-1.9 feet (0.18-0.59 meters), respectively. In addition, the present four scenarios also span the 2.0-4.8 feet (0.60-1.46 meters) range in Cayan *et al.* (2009) shown in Figure 15, as well as the 1.6-4.6 feet (0.5-1.4 meters) range in Rahmstorf (2007), and the newer estimates of Vermeer and Rahmstorf (2009) of 2.7-5.9 feet (0.81-1.79 meters) shown in Figure 16, all by year 2100.

Figure 15 Projected Global Mean Sea Level Rise Using Rahmstorf (2007) Method for Six Models (2000-2100). Simulations for A1fi, A2, And B1 Emission Scenarios Shown for the Original Rahmstorf (Dashed) and Adjusted for Affect of Reservoirs and Dams (Solid). Historical (Black) and Projected B1 (Blue), A2 (Red), And A1fi (Gold) Simulations Shown with Observed Global Sea Level (aqua) (from Cayan *et al.,* 2009).

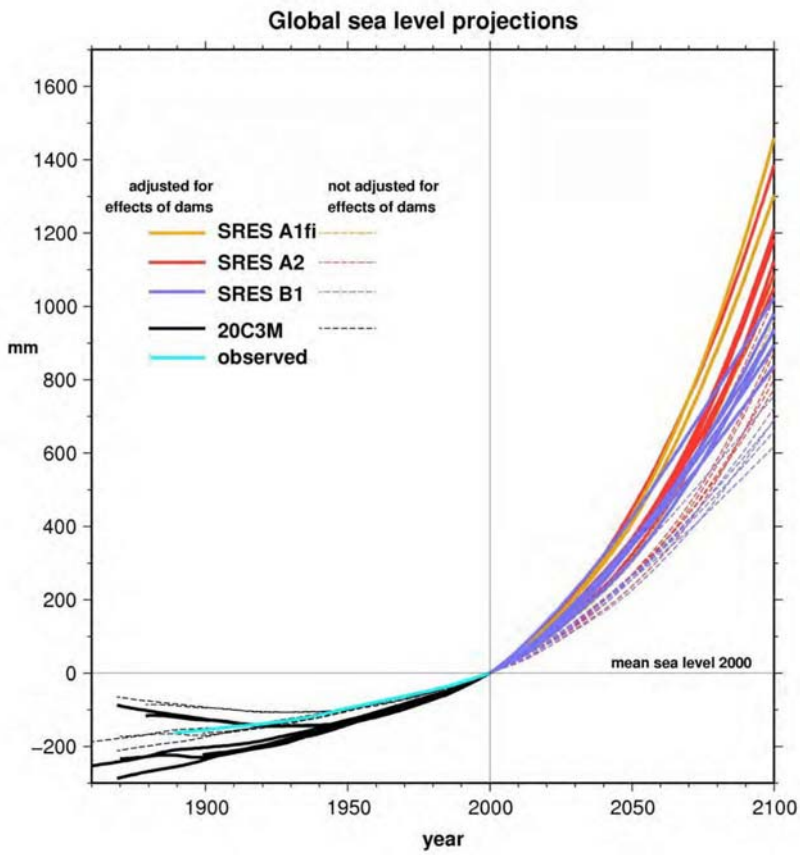
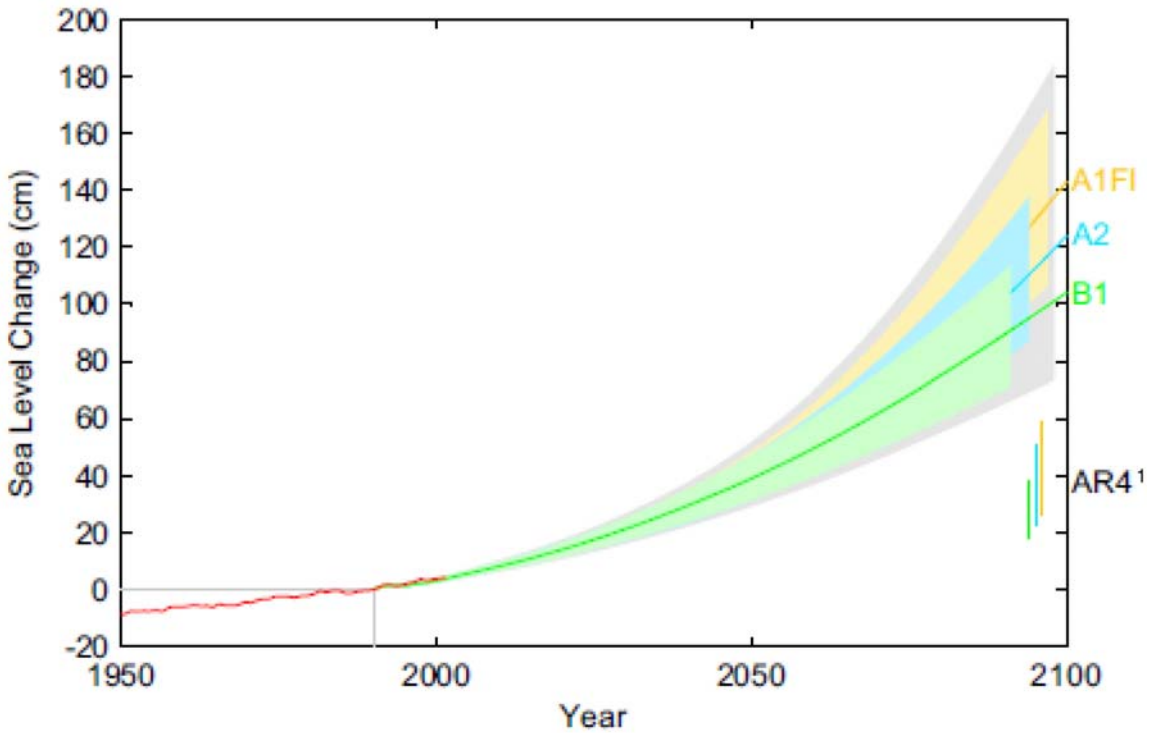


Figure 16 Projections Show Future Mean Sea Level Rise (1990-2100), Including the Uncertainty Ranges for Three Emissions Scenarios (Right).



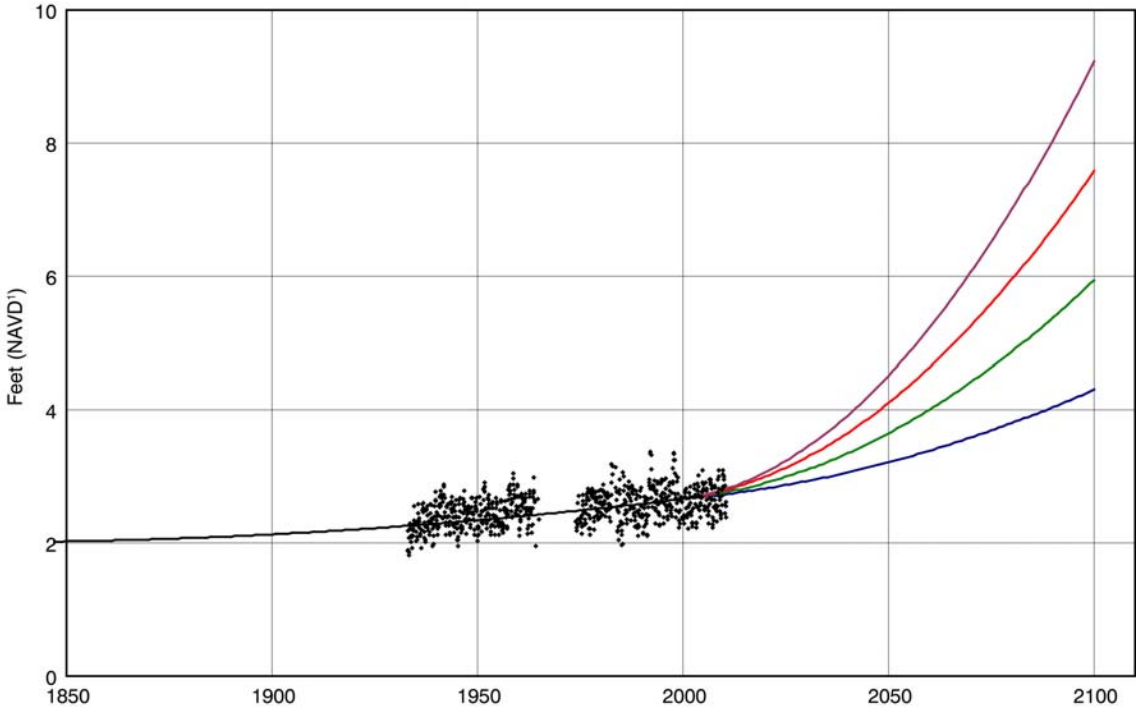
Notes: 1. IPCC AR4 (2007) mean sea level rise scenario 2100 end-points are shown for comparison (from Vermeer and Rahmstorf, 2009).

5.5 Conclusions

A thorough consideration of both sea level fluctuations on time scales of days (tides) to decades (El Niño and PDO) and possible future MSL rise rates due to global climate change is necessary to properly plan and determine the impact to the Estuary and VWRF operations. Extreme water levels are the cause of flooding and need to be considered in the design of coastal structures. Most of the potential damage will occur when high water stands due to extreme tides, weather, and climate anomalies are made more frequent by the gradual rising of MSL (Cayan *et al*., 2006).

Figure 17 shows the impact of the four MSL rise curves on Santa Monica's MSL through year 2100.

Figure 17 Santa Monica's Historical Mean Sea Level (1933-2010) and Projected Rise in Mean Sea Level by Year 2100 Using the Four Curves Adopted from U.S. Army Corps of Engineers.



Notes: 1. North American Vertical Datum of 1988 (NAVD).

Keeping in mind the need to consider the extreme tides, the recommended projected ranges of sea level rise for use in the next subtask are:

- 0.7 to 2.0 feet (0.2 to 0.6 meters) by year 2050; and
- 1.6 to 6.6 feet (0.5 to 2.0 meters) by year 2100.

6.0 PRECIPITATION PATTERNS

On average, more than 80 percent of California's annual precipitation occurs between November and April. The same is true for City of Ventura and the Estuary, which has received on average about 15 inches of precipitation annually between 1900 and 2010 (as recorded at Ventura's National Weather Station, No. 049285, for the years 1900 through 2010). Upstream the SCRE, at the Santa Paula-UWCD station No. 049245a, the average annual precipitation is about 17 inches for the years 1820 through 2010.

For this analysis a summary of current trends and projected global climate change impacts on the Estuary will be presented in terms of average annual precipitation, precipitation events, and cumulative residual precipitation.

6.1 Average Annual Precipitation

6.1.1 Current Trends

The change in trend of total average annual precipitation across the U.S. is already measurable. The results of a recent study performed using two data sets of global precipitation from 1925 through 1999 showed increased annual precipitation in temperate regions of the Northern Hemisphere (Zhang *et al.,* 2007). Since about 1970, total annual precipitation across the U.S. has tended to remain above the 20th-century mean and averaged about 5 percent more than the previous 70 years (Karl *et al.,* 1996). More importantly, in the western mountains of the U.S., approximately 74 percent of weather stations showed an increase in the fraction of average annual precipitation falling as rain rather than snow from 1949 through 2004 (Knowles *et al.,* 2006).

6.1.2 Future Projections

To examine the potential future impacts of global climate change, scientists have developed computer models (general circulation and regional climate models) simulating climate. While projected temperature changes are broadly consistent across most modeling efforts, projected changes in total average annual precipitation have varied widely across models and emissions scenarios (Kiparsky and Gleick, 2003; Madsen and Figdor, 2007). In addition, as models are run at finer levels of geographic resolution (e.g., regional or metropolitan level) the accuracy decreases.

California's precipitation patterns vary in different parts of the region. The usefulness of these models is judged based on the models' ability to reproduce recent changes; however, the general circulation models (GCMs) do not reproduce these detailed precipitation patterns accurately. Precipitation relies on meteorological conditions that occur at scales smaller than GCMs currently resolve. In recent years, they have become increasingly accurate and have been applied to more regions of the country. However, most regional studies performed in California have focused on Northern California, and there is no consensus on future projections for total average annual precipitation for the Ventura area.

6.2 Precipitation Events

Although projections for total average annual precipitation vary significantly, most regional climate model results in the U.S. suggest that the extreme daily (24-hour) precipitation rate will increase relative to changes in the average annual precipitation rate. During times of high intensity precipitation, runoff rates increase rapidly resulting in less groundwater recharge and decreased surface water quality.

6.2.1 Current Trends

The Environment California Research and Policy Center study evaluated trends in the frequency of extreme levels of precipitation (rainfall or snowfall) across the contiguous U.S.

(as well as finer levels of geography) (Madsen and Figdor, 2007). The analysis considered daily precipitation records obtained from the NCDC spanning from 1948 through 2006 at more than 3,000 weather stations in 48 states. Patterns in the timing of heavy precipitation relative to the local climate at each weather station were examined (Madsen and Figdor, 2007). The results were found to be consistent with other studies performed previously, including a study completed in 1999 by researchers at the Illinois State Water Survey and the NCDC examining the period from 1931 to 1996 (Kunkel *et al.,* 1999).

The ECRPC (2007) study focused on storms with extreme 24-hour precipitation totals that are defined relative to the local climate, selecting those with an average recurrence interval of one year or more. With 95 percent confidence, records show that the average increase in frequency of extreme precipitation events lies between 22 and 26 percent across the continental U.S. since 1948. The largest increases occurred across New England, New York, the Great Lakes area, the upper Midwest, in addition to Louisiana, New Mexico, Northern Washington, and Southern California (Madsen and Figdor, 2007).

At the state level, the trend toward increasingly frequent extreme precipitation events remains consistent. Records show a 26 percent increase in frequency of extreme precipitation events in California since 1948. Detection of statistically significant trends in the frequency of extreme precipitation events becomes more difficult at the metropolitan level. However, 55 of the 248 metropolitan areas (as defined by the U.S. Census Bureau) showed a statistically significant increase in the frequency of extreme precipitation events. A review of extreme precipitation for an area including Santa Barbara, Santa Maria, and Lompoc showed a 69 percent average increase in frequency of extreme precipitation events since 1948, and a 58 percent increase for an area including Los Angeles, Riverside and Orange County (Madsen and Figdor, 2007).

6.2.2 Future Projections

Further changes in precipitation patterns are projected to occur under global climate change. Both general circulation and regional climate models project the intensity of precipitation is likely to increase around the world, with the most significant increases occurring in the middle to high latitudes (Meehl, 2005). Global simulations show the percentage increase in extreme precipitation is greater than the percentage increase in mean annual precipitation. Kharin and Zwiers (2005) project the probability of 24-hour precipitation events considered to be extreme will increase by a factor of about two by the period of 2046 to 2065 and by a factor of three by the end of the 21st century relative to those that occurred during the period of 1981 to 2000. This means 24-hour precipitation events having return periods of 10, 20, 50, and 100 years will occur two or more times as often by the year 2100 due to global climate change (Kharin and Zwiers, 2005; Kharin *et al.,* 2007).

Across the U.S, Karl and Knight (1998) show the annual 24-hour precipitation event with the highest amount of rainfall turned out to be the one projected to return more frequently. Projected changes in regions of California are less accurate and do not cover the area of Ventura. These changes are mainly associated with changes in moisture flowing in from the Pacific Ocean and the increase in elevation of freezing levels during the winter. By the year 2100, northern California is projected to experience an increase in both low and high intensity events (Dettinger, 2005).

Although there is a lack of model simulation results relevant to projections for the Central Coast, including the area in and around the City of Ventura, it is recommended that long term planning be based on the current trends of an increase in frequency and change in distribution within the year of extreme events, as opposed to increases or decreases in average annual precipitation.

6.3 Cumulative Residual Precipitation

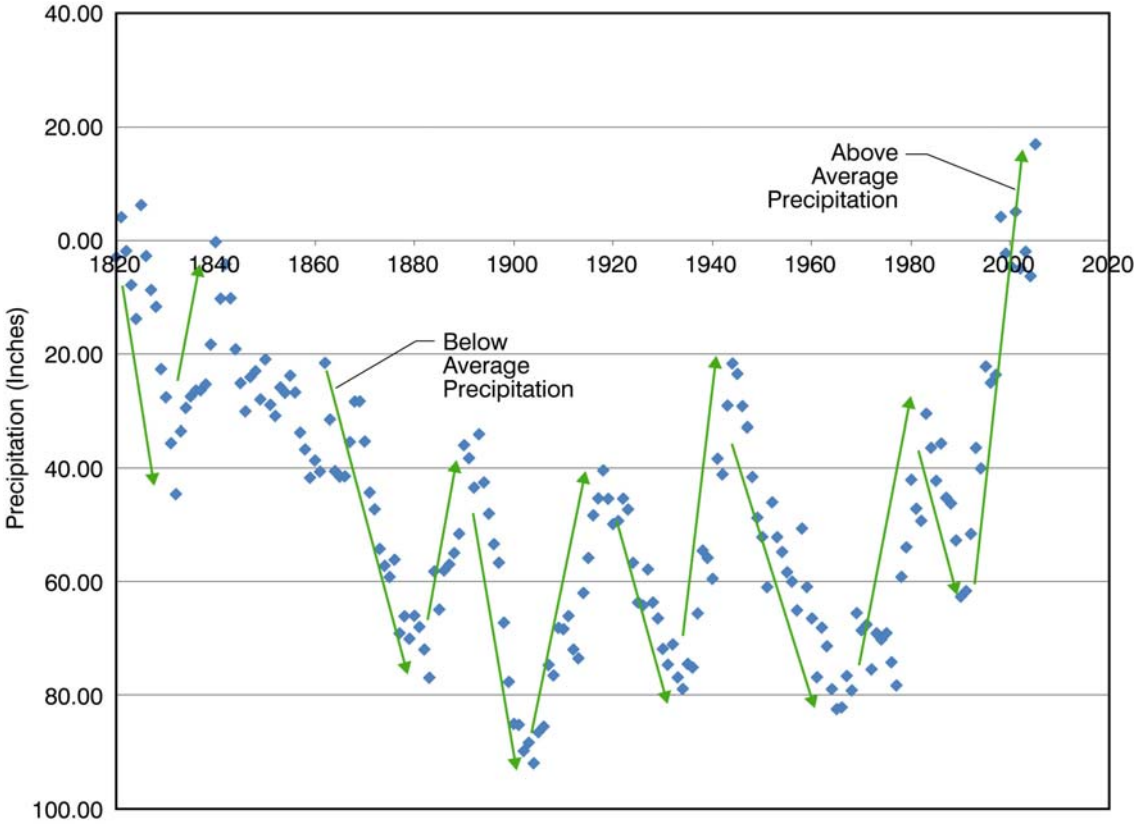
While California is classified as a “Mediterranean” and semi-arid climate, this does not describe the extreme variability of storminess that characterizes this coast. Storminess varies from year to year and also shows variation over decades-long time scales. Therefore, in addition to knowing the annual and/or monthly average precipitation and intensity, it is helpful to determine the cumulative residual precipitation. The cumulative residual precipitation (P_n) at month n after the beginning of the time series is calculated from the monthly precipitation data, p_i, by subtracting the mean, p, and then accumulating it, as shown by Equation (1) (Flick, 1993).

$$P_n = \sum_{i=1}^{n} (p_i - \bar{p}), \quad n = 1, N$$

(1) where N is the total number of months in the record.

Figure 18 shows the cumulative residual precipitation from 1820 to 2009 for Santa Paula, upstream the Estuary. The cumulative residual record has an upward trend during periods of above average precipitation, and a downward trend during times of lower than average precipitation.

Figure 18 Cumulative Residual Precipitation in Inches Observed at Santa Paula (Upstream the Santa Clara River Estuary) from 1820 through 2009.



6.4 Conclusions

The variability of precipitation in the near future (i.e., the next 20 or so years) will look like the past for the California coast since the key contributors to change in storminess are El Niño and the PDO – much more so then a slight increase in temperature due to global climate change. The recommended projected changes in precipitation patterns for considering in the next subtask include the following:

- There is no scientific consensus on the projected change in total average annual precipitation. The regional circulation models show little change, maybe slightly more or slightly less in total annual precipitation.
- The frequency of extreme daily (24-hour) events (relative to the period of 1981-2000) is projected to increase by a factor of two by year 2050 and a factor of three by year 2100. However, the frequency of these events has already increased by 58 to 69 percent since 1948, which exceeds the projected increase in frequency.

7.0 SUMMARY OF RECOMMENDATIONS

The goal of this study is to conduct and present the findings of a scientific literature review and data analysis of the effects of future global climate change scenarios on the Estuary ecosystem. More specifically, this TM summarizes the projected changes in local atmospheric temperature, sea levels, and precipitation patterns/events, with the following recommended projections for use in the next subtask of this study:

- Projected range of increase in **atmospheric temperature**
  - 2050 – 1.0 to 3.0˚C (1.8 to 5.4˚F), average of 2.0˚C (3.6˚F); and
  - 2100 – 1.1 to 5.0˚C (2.0 to 9.0˚F), average of 3.0˚C (5.5˚F).
- Projected range of rise in mean sea levels
  - 2050 – 0.7 to 2.0 feet (0.2 to 0.6 meters); and
  - 2100 – 1.6 to 6.6 feet (0.5 to 2.0 meters).
- Projected change in precipitation patterns/events
  - There is no scientific consensus on the projected change in total average annual precipitation; however, the regional circulation models show little change, if any;
  - The frequency of extreme daily (24-hour) events has increased by 58-69% since 1948, which is exceeding the projected increase in frequency of these events. Below are the projected increases in event frequency (relative to the period of 1981-2000):
    - 2050 – by a factor of two; and
    - 2100 – by a factor of three.

This information will be used to examine the state of the Estuary ecosystem under projected conditions in the next subtask (Predict Impacts of Climate Change on the Estuary) of the study.

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