FINAL REPORT ◦ FEBRUARY 2018

City of Ventura Special Studies – Phase 3: Assessment of the Physical and Biological Conditions of the Santa Clara River Estuary, Ventura County, California

PREPARED FOR

City of Ventura
P.O. Box 99
Ventura, CA 93002

PREPARED BY

Stillwater Sciences
2855 Telegraph Ave., #400
Berkeley, CA 94705

Stillwater Sciences

This Final Phase 3 Study Report contains additional materials and revisions to the November 1, 2017 Draft reflecting resource agency and Stakeholder comments received at workshops held on November 8, November 15, and December 20, 2017, as well as written comments received through January 16, 2018. Per the request of the RWQCB in their December 12, 2014 letter to the City, the results of the Phase 1 and 2 technical reports (Stillwater Sciences 2011, Carollo Engineers 2014) as well as responses to comments on the November 1, 2017 Draft report are summarized in this Final report. Detailed responses to comments on the November 1st Draft report received through January 2018 will be provided in March 2018 under a separate cover.

Suggested citation:
Stillwater Sciences. 2018. City of Ventura Special Studies – Phase 3: assessment of the physical and biological conditions of the Santa Clara River Estuary, Ventura County, California. Final Report. Prepared by Stillwater Sciences, Berkeley California for City of Ventura, California. February.

Cover photo: Aerial view of the Santa Clara River Estuary, September 1, 2015

Table of Contents

Tables

Figures

Appendices

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 waterbody named the Santa Clara River Estuary (SCRE), a lagoon type estuary [1] that is formed by a berm that periodically forms across the mouth of the Santa Clara River during low river flow conditions. The SCRE is located just south of the City, and is the interface between the Santa Clara River and the Pacific Ocean. Under the California Enclosed Bays and Estuaries Policy ("EBE 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." (SWRCB Res. No. 74-43 [May 16, 1974], readopted, as amended, SWRCB Res. No. 95-84 [Nov. 16, 1995]). In 1976, the City submitted a plan for effluent utilization that 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 (RWQCB) in May 1977, granted the City an exception to the discharge prohibition and allowed continued discharge of the VWRF effluent into the SCRE.

In discussions prior to the 2008 renewal of the City's National Pollutant Discharge Elimination System (NPDES) discharge permit (CA0053651; NPDES Permit), several questions arose regarding the definition of enhancement, the benefits that the discharge provides to the Estuary and adjacent subwatershed, and how discharge practices could be modified over time to protect and enhance habitat and water quality of the portion of the SCRE directly affected by the VWRF discharge. To address these issues the RWQCB ordered the City to complete a series of Special Studies under Order R4-2008-0011: the Phase 1 Estuary Subwatershed Study, the Treatment Wetlands Feasibility Study, and the Recycled Water Market Study. Following the completion of these studies, stakeholder feedback indicated the need for additional studies (the Phase 2 Studies) to: (1) develop additional information (more hydrologic and water quality data) to improve the understanding of SCRE functioning and help assure a discharge regime that protects the sensitive wildlife and aquatic resources and habitats within the SCRE; and (2) integrate the conclusions of all three of the Phase 1 Studies into a process for selection, environmental review, and design of a preferred VWRF discharge/diversion alternative or combination of alternatives to create a discharge regime that best realizes beneficial uses of the SCRE. These needs were addressed through the Phase 2 Studies, which were completed and a final report was submitted to the Regional Board in 2014.

At the conclusion of the Phase 2 studies, several stakeholders expressed concerns about identified data gaps and the study findings. In response to these concerns, the Regional Board adopted requirements in the City's current NPDES Permit, Order R4-2013-0174 for VWRF discharges (Ventura NPDES Permit) mandating the following additional special studies: the *Phase 3 Estuary Studies* (Phase 3 Study), the *Nutrient, Dissolved Oxygen and Toxicity Special Study*, and the *Groundwater Special Study*. Order R4-20013-00174, § VI.c.2.b.; Attachment F, *Fact Sheet*, §§

[1] In discussing the SCRE in this report we use the terms "estuary" and "lagoon" interchangeably, but note that the SCRE is considered an "estuary" as it relates to the Enclosed Bays and Estuaries Policy of 1974, which specifically states that "the mouths of streams which are temporarily separated from the ocean by sandbars shall be considered as estuaries." (see EBE Policy pg. 9).

III.D.10.; VI, B.2. As discussed below, this report is intended to address the requirements for the Phase 3 Estuary Studies. The *Nutrient, Dissolved Oxygen and Toxicity Special Study* will be provided to the RWQCB under separate cover to meet RWQCB requirements, although it will rely on the data collected and analysis done in this Phase 3 Study.

Purpose

This Phase 3 Study Report provides an updated assessment of the effects of VWRF tertiary treated discharge on the ecological functions and beneficial uses of the SCRE. The results of the study will be used to make three interrelated discharge recommendations, which may be stated as the following study goals:

1. As required by the Ventura NPDES Permit, determine the levels of VWRF discharge, if any, that likely provide fuller realization of beneficial uses of the SCRE and its watershed relative to the absence of all discharge, defined as the "Enhancement Discharge Levels". The analyses supporting recommendations on Enhancement Discharge Levels examine a series of alternative VWRF discharge scenarios to identify the discharge levels that most fully support the 10 designated beneficial uses of the SCRE in the Basin Plan, as well as the potential for additional municipal water supply (MUN) uses within the SCRE subwatershed.
2. As allowed by the Ventura NPDES Permit and required by the Tertiary Treated Flows Consent Decree and Stipulated Dismissal among the Wishtoyo Foundation Ventura Coastkeeper (Wishtoyo), Heal the Bay (HTB) and the City (Consent Decree), determine the "maximum ecologically protective diversion volume" (MEPDV), or the maximum level of VWRF discharge that could be diverted to water reclamation uses rather than discharged to the SCRE, particularly during closed-mouth, dry-weather conditions, while still protecting ecological functions of the SCRE.
3. As required by the Ventura NPDES Permit, determine the minimum level of continued VWRF discharge, if any, needed to protect the SCRE's existing ecological resources, particularly native listed species and critical habitats, as well as related SCRE beneficial uses during closed-mouth, dry-weather conditions, defined here as the "Continued Discharge Level".

The three discharge recommendations in this report are interrelated and rely upon the same analyses. Recommendations for the MEPDV and the Continued Discharge Level are the converse of one another.

Study Approach

In accordance with the City's *Combined Workplan for Phase 3 Estuary, Nutrient and Toxicity, and Groundwater Special Studies* (October 2014) approved by Wishtoyo and HTB prior to submittal, and conditionally approved by the RWQCB on December 12, 2014 (Workplan), the Phase 3 Study relies upon updates to previously developed conceptual models and modeling tools, historical summaries of data used in the Phase 1 and 2 assessments, NPDES monitoring data collected since 2012, as well as data collected pursuant to the Work Plan as part of this study during 2015–2016, and recently available information from the scientific literature.

As in the Phase 1 and Phase 2 Estuary Studies, the Phase 3 Study focuses on ecological beneficial uses of the SCRE, with emphasis on support of native species listed for protection and critical habitats designated under the state and federal Endangered Species Acts. As was the case in the Phase 1 and Phase 2 Studies, the Phase 3 Studies employs a focal species approach, in which effects of the alternative VWRF discharge scenarios on listed species and their habitats are assessed in detail, and then used as a conservative indicator of ecosystem health for other aquatic and wildlife species. Lastly, as identified in the data gaps analysis of the Phases 1 and 2 studies, Phase 3 assessments integrate the effects of discharge, groundwater, and surface water quality data more directly into the analysis of beneficial uses.

In order to provide discharge recommendations outlined above (see *Purpose*), eleven alternative VWRF discharge scenarios, ranging from current conditions to 100% diversion (i.e., no discharge to the SCRE) are assessed to determine the degree to which they support each of 10 designated beneficial uses of the SCRE, as well as municipal water supply (MUN) uses in the SCRE subwatershed. These assessments are based on quantitative modeling and qualititative analysis of conditions in the SCRE that can be affected by VWRF discharge. Weights are assigned to each factor considered in the analysis in order to balance potentially competing beneficial uses. Weights for each factor are assigned using a rigorous multiple-criteria decision-making framework based on the Analytic Hierarchy Process (AHP) (Saaty 1980, 2008).

Geomorphology

Historically, the SCRE contained an unconstrained river channel and a periodically forming 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 SCRE by at least 75%, decreased the effective flow width of the river during flood events, and essentially “locked” the SCRE into its present-day location and extent. The frequency and magnitude of storms that can cause substantial geomorphic change within the SCRE were elevated during the late-1960s to mid-2000s, associated with a wet period of the El Niño Southern Oscillation (ENSO). During these major storm events, geomorphic features (e.g., depositional bars, back channels) within and adjacent to the river mouth and 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. Since 2011, the most significant changes to the SCRE have been the steady landward-migration of the beach berm, and prolonged periods of closed-mouth conditions between 2011 and 2015. Unauthorized manual trenching of the beach berm by unknown parties near the VWRF outfall channel’s confluence with the lagoon in recent years, including throughout 2016, also affect geomorphology and have been considered.

Hydrology and Hydraulics

Analysis of current and historical data shows that the relative rate of flows into and out of the SCRE varies seasonally, inter-annually, and over longer timescales, due to both natural and anthropogenic influences. Typically, the Santa Clara River flow is the dominant inflow to the SCRE in wet periods, typically occurring from the fall through the spring, while VWRF effluent is the dominant inflow during dry periods, typically the summer months. Long-term daily averages show Santa Clara River inflow to the SCRE is typically low (<80 cfs 90% of the time; in 2014-2016 <4 cfs 90% of the time), but is punctuated by large storm events. During these low-flow conditions, the SCRE fills gradually, typically over a period of weeks. VWRF effluent discharge can maintain the SCRE at a quasi-equilibrium “full” stage that can be maintained for extended periods. When storm-induced river flows enter the SCRE during closed-mouth periods, the filling rate is much more rapid and the stage associated with berm breaching can be higher than the breaching stage during low-flow conditions.

Long-term trends show relatively high frequency of mouth berm breaching and subsequent open-mouth conditions from November through June (i.e., the mouth is open >50% of the time on average), with low breach frequency during summer months (mouth closed >50% of the time on average). However, the persistence of dry conditions in recent years resulted in extended closed mouth periods throughout 2015 and 2016. From December 2014 through November 2016, 13 breach events occurred. Of those, only four were naturally induced by river discharge during rainfall events; nine breach events resulted from unauthorized manual trenching of the beach berm by unknown parties. The Phase 3 monitoring also evaluated the potential influence of McGrath Lake surface water flows into the SCRE. However, no evidence of a surface water connection to the SCRE was found and surface water flows from McGrath Lake are assumed not to influence conditions in the SCRE.

Water Quality

Water quality conditions have both direct and indirect impacts on beneficial uses in the SCRE, and both the VWRF discharge, and ambient water quality within inflows contributing to the SCRE, including Santa Clara river flows, surface water runoff, groundwater from banks around the SCRE, wave overwash, and/or tidal exchanges. Water quality within the SCRE has been intensively studied for the past decade. Data synthesized from a number of sources, including prior Phases 1 and 2 Estuary Studies, NPDES monitoring data, and surface and groundwater data collected as part of the Phase 3 Study have been synthesized and used to characterize physical and chemical water quality conditions in the SCRE, including spatial and temporal patterns and the relative contributions of various sources to observed conditions.

Water quality conditions vary on annual, seasonal, and daily timescales. Variability in water quality is driven by climatic conditions as well as the predominant source of SCRE water inputs. SCRE water temperatures vary from wintertime lows near 8ºC (46ºF) to summertime highs approaching and sometimes exceeding 28ºC (82ºF), although median temperatures varied from 17 ºC (63ºF) to 26 ºC (78ºF) during 2015–2016. During open-mouth conditions, tidal exchange exerts strong control on water quality conditions. Salinity is temporally variable, with levels during closed mouth periods typical of freshwater or oligohaline brackish environments, and periods of higher salinity driven by tidal exchange during open mouth conditions. Upon mouth closure, mineral salts and observed nutrient levels approach those of the dominant water source as the SCRE fills, which varies seasonally as described above.

Water column profiles as well as limited spatial analysis of *in situ* SCRE water quality indicate generally uniform conditions with some spatial variation in temperature, dissolved oxygen, and conductivity during some periods. The VWRF outfall channel exhibited generally lower temperatures and conductivity from winter into early summer with some evidence of warmer surface temperatures in summer and early fall. Despite substantial reductions in nitrate concentrations in VWRF effluent starting in late 2011, nutrient concentrations in the SCRE remain above the saturation level for algal production, resulting in high algal biomass in the SCRE. The Phase 3 monitoring data indicated elevated concentrations of nitrate in upstream groundwater wells, indicating that upstream sources also contribute to nitrate loading in the SCRE. The SCRE is eutrophic, resulting from high nutrient loading from groundwater, VWRF effluent, and riverine and local runoff. The resultant high algal production drives observed patterns in dissolved oxygen (DO) and pH. High primary productivity results in periods of DO saturation, while algal die offs can lead to periods of near anoxia due to the oxygen demand of bacterial decomposition of algal detritus.

Toxicity results from the Phase 3 quarterly testing during 2015–2016 as well as NPDES permit monitoring results indicated no significant toxicity effects on test organisms of water collected in SCRE or VWRF sites. Trace metal analysis shows some periods of elevated zinc and copper concentrations, with concentrations generally higher in the outfall channel than in the SCRE (though elevated copper concentrations were observed in an upstream SCRE location, suggesting additional contribution from an upstream source).

Analysis of over 100 constituents of emerging concern (CECs), which encompass many unregulated chemicals that occur in air, water, sediment, and biota, showed four constituents found at concentrations exceeding the predicted no-effect concentration (PNEC), each of which occurred at the VWRF effluent transfer station (ETS) and (in one case) at the outlet from the Wildlife Water/Quality ponds, and only one of which (Sulfamethoxazole antibiotic) also occurred in the SCRE. The ratio of Maximum Effluent Concentration (MEC)/aquatic PNEC, which is the metric used to determine environmental risk posed by CECs, for these 4 compounds ranged from 1.1 to 1.9 in effluent locations, as was 1.2 when measured in the SCRE. Ratios approaching 1 are in very close proximity the "no aquatic effects level." Although no guidelines are provided by federal of California regulatory agencies for Sulfamethoxazole concentrations, published studies showed only limited effects upon algal reproduction. Based upon the cumulative NPDES Permit and Phase 3 Studies CECs monitoring results to date, the potential risks of CECs is low for aquatic species within the SCRE.

Vegetation and Wetlands

As with the SCRE geomorphology, there have been considerable changes to the vegetation assemblage of the lower Santa Clara River over the past 150 years. General vegetation/habitat types currently comprising the SCRE in order of dominance include riparian, open water, mudflats, foredune, ocean, developed/disturbed, open beach, and wetland (including freshwater wetland and a small amount of salt marsh). There has been a recent shift in vegetation successional stage upstream of Harbor Blvd., reflecting the recovery of riparian vegetation after the 2005 flood that resulted in large areas of bare riverwash. In general, open water habitats of the main SCRE lagoon area have relatively little aquatic vegetation due in large part to the regular occurrence of scouring flood flows. Comparing current vegetation conditions with historical records dating back to the mid-1980s 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, including *Arundo donax*. A number of rare plants have been documented in the SCRE in each of the habitat types.

Aquatic Habitat and Species

Aquatic habitat types present in the SCRE include open water, freshwater wetlands, and a minimal amount of salt marsh. Depending upon river flows and mouth closure status, the largest aquatic habitat type is open water characterized by shallow sand-bedded flats with scattered emergent vegetation. The SCRE aquatic community consists of a number of native and non-native fishes and amphibians, as well as benthic macroinvertebrates (BMI). The BMI community in the SCRE is composed of organisms common to freshwater or estuarine environments, and tends to be dominated by species that are tolerant of the naturally dynamic salinity and sediment conditions in the SCRE in most years and seasons. BMI abundance and diversity was particularly low during Phase 3 sampling, which might be due in part to the persistence of drought conditions in recent years, resulting in extended periods of closed-mouth, low salinity conditions, as well as the lack of scouring of SCRE sediments that typically occurs during high winter flows.

A number of non-native freshwater fishes are abundant in the SCRE and may prey upon or compete with native fish. Two federally endangered fish species occupying or using the SCRE were selected as focal species for analysis of aquatic habitat related beneficial uses: southern California steelhead (*Oncorhynchus mykiss*) and tidewater goby (*Eucyclogobius newberryi*). California has also designated the tidewater goby as a species of special concern.[2] In addition, the NMFS designated the SCRE as critical habitat for the Southern California steelhead under the ESA. (70 Fed. Reg. 52,488 (Sept. 2, 2005) [designating, among other areas, the SCRE as critical habitat for steelhead].) The U.S. Fish and Wildlife Service (USFWS) has designated the SCRE as critical habitat for the tidewater goby under the ESA. (73 Fed. Reg. 5920 (Jan 31, 2008), and also published a Recovery Plan for the endangered fish,[3] which is a plan required by Section 4(f) of the ESA that delineates reasonable actions that are believed to be required to recover and/or provide future protections for a listed species.

A data and literature review was conducted to identify habitat needs of focal fish species, including primary constituent elements (PCEs – also referred to as physical or biological features, or PBFs) included in critical habitat designations of the listed species above. The following briefly summarizes the factors affecting each of the focal fish species in the SCRE as well as their relationship with VWRF discharge:

- Southern California steelhead use of SCRE habitat is not well documented. The SCRE provides a migratory corridor for upstream adult steelhead spawners and outmigrant smolts, as well as potential rearing habitat for subadults. Rearing steelhead require moderately low salinity, relatively high dissolved oxygen, refuge from excessive water temperatures, and cover to avoid avian predation. Stranded subadult steelhead were documented in the SCRE following an unauthorized third-party breach in 2010, indicating that steelhead may potentially use the SCRE. After review of general life history requirements for southern California steelhead, PCEs/PBFs related to spawning and early rearing, as well as floodplain rearing were determined to be unaffected by VWRF discharge. Depth provides the predominant source of cover for rearing steelhead, since the SCRE lacks large woody debris and other structural components that comprise natural cover in some other systems. Water quality was determined to be an important factor for rearing steelhead, and is affected by VWRF discharge, predominantly through the effects of nutrient loading on dissolved oxygen. The effects of VWRF discharge on breach duration and the risks of unseasonal breaching were also considered important for rearing steelhead and were evaluated accordingly.
- Tidewater goby use aquatic habitats within the SCRE for their entire life-cycle. They have been one of the most abundant fish species in the SCRE during past surveys, but were relatively scarce during Phase 3 surveys. Tidewater goby require shallow habitat with sandy substrate for spawning burrow construction. They are relatively tolerant of salinity

[2] Cal. Dept. of Fish and Game, *Species of Special Concern in California* 79, 235 (2d ed. 1995). Species of special concern are those with low, scattered, or highly localized populations and require active management to prevent them from becoming threatened or endangered. (*Id*. at p. 3.)
[3] Fish and Wildlife Service, Pacific Region, *Recovery Plan for the Tidewater Goby (Eucyclogobius Newberry)* (Recovery Plan) (issued Jan. 7, 2005) (*available at* http://www.fws.gov/pacific/ecoservices/endangered/recovery/documents/TidewaterGobyFinalRecoveryPlan.pdf).

fluctuations, although low to moderate salinity is listed as a PCE of critical tidewater goby habitat. Very high water temperatures or extended periods of low DO may be unsuitable for rearing and spawning tidewater goby. Major threats to goby in the SCRE include dispersal due to storm flows, dewatering of spawning burrows due to unauthorized third-party breaches, and predation by or competition with native and introduced species. After review of general life history requirements tidewater goby, PCEs related to spawning and rearing were reviewed to establish assessment criteria based upon depth, subtsrate, DO conditions, and limiting unseasonal breaching.

Wildlife Habitat and Species

The wildlife species habitats within and adjacent to the SCRE 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 and a number of bird species also use habitats provided by the VWRF Wildlife/Polishing Ponds. The SCRE, Wildlife/Polishing Ponds, and surrounding area also provide valuable rearing and foraging habitat for a number of mammals and herpetofauna. Two federally listed avian species were selected as species for analysis of wildlife habitat related beneficial uses: western snowy plover (*Charadrius alexandrinus nivosus*) and California least tern (*Sternula antillarum browni*). The western snowy plover is federally listed as threatened. (58 Fed. Reg. 12864 (Mar. 5, 1993) [listing the Pacific Coast population as threatened throughout its range].) Further, USFWS has designated the SCRE as critical habitat for wester snowy plover under the federal Endangered Species Act. (64 Fed Reg. 68,508 (Dec. 7, 1999); 77 Fed. Reg. 36,727 (June 19, 2012).) In addition, the western snowy plover is designated as a California Species of concern (2008). The California least Tern is a bird species that is federally listed (35 Fed. Reg. 8,491 (June 2, 1970) and state listed (June 27, 1971) as endangered throughout its range. The California least tern is also designated for protection as a California Fully Protected Species. Although the SCRE is designated as critical habitat for the southwestern willow flycatcher (*Empidonax traillii extimus*) and surveys have identified suitable habitat and habitat use upstream of the SCRE, no suitable habitat or habitat use has been idientified in the SCRE.

California least lern and western snowy plover have been monitored in the vicinity of the SCRE for over 30 years. A data and literature review of habitat needs, including PCEs/PBFs, was conducted to define physical habitat suitability criteria for both nesting and foraging habitat for these species. After examining the extent to which these PCEs are realized in the SCRE,the following briefly summarizes the factors affecting each of the focal fish species in the SCRE as well as their potential relationship with VWRF discharge:

- Western snowy plover may be found near the SCRE during both the summer (nesting) and winter seasons. Nesting activity typically peaks in May or June. 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 due to factors ranging from invasive plant species (e.g., ice plant, giant reed), urban development, and recreational use of beaches. Human interference and predation are both common causes of nest failure in the SCRE and vicinity. Based upon review of existing information on PCEs/PBFs, suitable habitat relationships with SCRE stage were developed to evaluate nesting, forage habitat, and the potential for nest disturbance.

- California least tern is found near the SCRE only during the summer (nesting) season. Nesting activity typically peaks in June or July. 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 invasive plants, urban development, and recreational use of beaches. Human interference, nest abandonment, and predation are common causes of nest failure in the SCRE. Based upon review of existing information on PCEs/PBFs, suitable habitat relationships with stage were developed based upon the amounts of open water as well as open beach habitat for foraging and nesting, respectively.

Water Balance and Water Quality Modeling Analyses

A suite of modeling tools was developed for the SCRE from observed SCRE data and well-established mathematical principles as part of the Phase 3 Study to quantitatively evaluate how conditions in the SCRE change over time in response to a range of environmental conditions. A water balance model was developed to evaluate changes in the volume and water surface elevation. Water quality in the SCRE was analyzed with three models that built on the results of the water balance model: a salinity model to estimate changes in the SCRE salinity concentration, a nutrient balance model to estimates changes in the nitrogen and phosphorous (N, P) concentrations in the SCRE, and a heat balance model to assess the magnitude of the various heat fluxes influencing the SCRE water temperature. All four models were designed specifically for the processes critical to characterizing SCRE conditions, calibrated with observed SCRE data, evaluated against the observed SCRE data, and refined based on feedback from a technical review team (TRT) selected by Wishtoyo and HTB.

The first model developed to evaluate conditions in the SCRE is a water balance, which is a quantitative method to account for the inflows, outflows, and change in water stored in a specific water body over a given period. The water balance developed for this study uses observed data as inputs for most model parameters and determines empirical relationships from the observed data to estimate groundwater inflows or outflows that cannot be measured directly. The Phase 3 Study water balance updated the Phases 1 and 2 water balance by incorporating new hydraulic and hydrologic data collected at previously established and new monitoring locations to better define modeled parameters with the Phase 3 model period spanning from January 13, 2015 through December 6, 2016. Updates included:

- Continued measurements of surface water inflows to the SCRE, including VWRF discharge
- Improved estimates of groundwater inflows and outflows between the SCRE and the underlying Semi-perched aquifer
- Improved estimate of water balance unmeasured flow by incorporating known, but unmeasured hydrologic processes into unmeasured flow equation
- Improved estimates of berm breach/closure dynamics
- Updated lagoon morphology
- Estimates of McGrath Lake inflow[4]

[4] The SCRE has been historically observed to extend a narrow arm southward at high water surface elevations and appeared to connect with the outfall channel of McGrath Lake (Stillwater Sciences 2016). No evidence of a surface water connection was observed during Phase 3 monitoring, so McGrath Lake surface inflows to the SCRE were assumed be negligible.

After inputting the available observed SCRE data, the model was calibrated to the SCRE conditions. Groundwater flows cannot be fully defined by measured data due to the naturally occurring spatial and temporal variability of groundwater conditions, so the water balance groundwater flows were calibrated by selecting reasonable hydraulic conductivity values for the various mixtures of soil types (i.e., relative amounts of clay, silt, and sand) around the SCRE perimeter. An unmeasured flow equation representing the contribution of unmeasured base flow, wave overwash, and drainage of bank storage groundwater was calibrated to maximize the agreement between modeled and observed SCRE water surface elevation and volume during the modeling period. A final calibration step involved adjustments of the assumed breaching elevation thresholds to match observed water surface elevation data. Model performance statistics comparing the calibrated water balance and observed data over the entire modeling period showed very good performance for the water balance overall, but the water balance did not agree well with observed data during several periods when man-made influences altered SCRE conditions or exceptionally high tidal conditions occurred. The water balance was not designed to account for unmeasured changes in the SCRE berm due to dredge deposits from Ventura Harbor, manual trenching of the beach berm that causes breaching, or wave overwash along McGrath State Beach during closed-mouth berm conditions due to exceptionally high tidal conditions during fall/winter 2015. Overall, the calibrated water balance model well represents closed-mouth conditions in the SCRE.

The calibrated water balance and the flow relationships defined within it are used to predict typical SCRE conditions resulting from hypothetical modeling scenarios, recognizing that infrequently occurring man-made influences from dredge deposits or manual trenching and wave overwash from exceptionally high tidal conditions would not be represented by the model results. The calibrated water balance model can be run for a range of SCRE conditions, including dry, normal, and wet water year types, using observed data when available and assumptions of alternative VWRF discharge scenarios. This model served as one basis for evaluating the effects of alternative VWRF management scenarios on beneficial uses of the SCRE (Section 5) along with water quality modeling, stage/habitat area modeling, and AHP pairwise comparisons. While the range in quality of data sources used and the need to develop estimates for components not directly measured introduces uncertainty, the calibrated water balance is considered suitable for use in examining the relative changes in SCRE volume, berm breaching frequency and duration of open-mouth conditions under a range of water year types, SCRE water quality, and open water habitat quantity associated with modifications to the VWRF discharge.

As a means of understanding the relative changes in water quality under current versus alternative future VWRF discharge scenarios, the water balance results were used with the water quality models to evaluate potential conditions in the SCRE. The salinity and nutrient balance models were two separate models, but both utilized a similar mass balance approach to model the either salinity or nutrient changes in the SCRE assuming the total of all material inflows and outflows balance over the modeling time-step (i.e., mass in equals mass out) and the lagoon is well-mixed due to the shallow SCRE lagoon depth, consistent onshore winds, and water quality measurements in the SCRE during the monitoring period. A simplified heat balance model was also developed utilizing a heat balance approach to model the heat fluxes from the various inputs and outputs to the SCRE and the relative influence of the heat fluxes on the SCRE water temperature assuming the total of heat inflows and outflows balance over the modeling time-step (i.e., heat in equals heat out) and the lagoon is well-mixed due to the shallow SCRE lagoon depth, consistent onshore winds, and water temperature measurements in the SCRE during the monitoring period. Assumptions for the water quality models are detailed in Appendix A, while water quality data supporting the well-mixed lagoon assumption is provided in Appendix D.

Modeled and measured salinity and nutrient concentrations were compared to evaluate model performance and its applicability to estimating the relative change in SCRE conditions under alternative VWRF discharge scenarios. The salinity model well represented SCRE salinity concentrations during equilibrium, closed-mouth conditions with a model prediction error of +/-4%, but tended to overpredict salinity during open-mouth or recently closed-mouth conditions. The nutrient balance assuming no algal uptake tended to over-estimate nutrient levels, but agreement between the modeled and observed nutrient concentrations were significantly improved by modeling the nutrient balance assuming rates of algal uptake. Modeling results indicated algal uptake is an important component of the nutrient balance in the SCRE. The heat balance showed magnitudes of the heat fluxes with solar radiation, longwave radiation, inflow through the open-mouth, and evaporation being the dominant influence on water temperature.

Despite the limitations in the water quality modeling used here, the models do allow for the assessment of the relative magnitude of changes in SCRE conditions from contributing sources of flow. However, it should be noted that the dynamic nature of the SCRE may result in large departures of estimated concentrations from actual conditions even during closed-mouth conditions due to several factors related to hydraulics (e.g., rapid changes in SCR flows, ocean exchanges), mixing (e.g., wind, salinity), or seasonal algal growth dynamics. Biological uptake of nutrients or losses of nutrients from the system (e.g., through burial in sediment or through bacterially mediated denitrification) are likely a significant component of nutrient cycling in the SCRE. During dry periods when other nutrient exchanges are limited (i.e., scouring of lagoon sediments), biological uptake or other losses of nutrients are particularly important. The biological uptake was estimated in the nutrient balance model using nutrient uptakes rates from literature, but the uncertainty associated with the uptake rates means the model does not well represent the actual nutrient concentrations in the potential VWRF discharge alternatives. The approach appears to characterize seasonal and flow-related changes well enough to represent relative changes in nutrient levels associated with alternative discharge scenarios (Section 5), but uncertainties associated with the nutrient balance, especially algal uptake, should be considered in subsequent analysis using the results.

Comparison of Discharge Scenarios

Eleven (11) VWRF discharge scenarios were selected to represent the range of discharge alternatives considered as management options. Scenario 1 (VWRF discharge of 4.7 MGD) represents current conditions. Scenarios 2–11 represent 10% incremental reductions in VWRF discharge (0.47 MGD) from Scenario 1, such that Scenario 11 represents a 100% diversion or complete elimination of VWRF discharge (0 MGD) from the SCRE. The degree to which each VWRF discharge scenario would support or adversely affect each of 10 identified beneficial uses was assessed using the tools and information summarized in the prior section of this Report, including the compiled water balance and other modeling tools, as well as metrics for weighted factors relevant to individual beneficial uses of the SCRE. This entailed establishing the following:

1. Changes in equilibrium stage for idealized closed SCRE mouth berm conditions:
    - The calibrated water balance model was used to estimate equilibrium stage under idealized closed mouth conditions
2. Changes in vegetation communities and overall habitat types within the assessment boundary:

- A spatially explicit GIS model was developed that predicted changes to mapped habitat types based on a set of rules relating previous habitat type and predicted water depth.
- GIS rules were established using scientific literature and available site-specific elevation data on plant presence and absence of common plants found in the SCRE. Using best professional judgment and considering input from the TRT, different rules were established for different regions of the SCRE (e.g., riverine, lagoon, and campground).
- Water depths were determined using the equilibrium stage estimate for each scenario (Item 1 above) and the digital elevation model (DEM) surface of the SCRE.

3. Changes in water quality conditions within open water habitats of the SCRE:
    - Changes is salinity were evaluated quantitatively by modeling a time series of conductivity following a simulated breach event and subsequent mouth closure and estuary refilling.
    - While nutrient uptake and removal rates were estimated using the scientific literature values and input from the TRT, changes in dissolved oxygen (DO) potentially affecting aquatic species were qualitatively evaluated on the basis of relative nutrient loading from VWRF discharge and the likelihood of algae blooms under each discharge scenario.
    - Changes in water temperature were evaluated using a simplified heat balance model that incorporated local and site-specific climatic data, as well as water balance estimates of the relative contributions of various water sources under each VWRF discharge scenario.
4. Changes in SCRE mouth berm breach frequency and the duration of open mouth conditions under representative wet, normal, and dry water years:
    - Water years from 1928 to 2016 were analyzed to determine water year types, with representative year types (Dry, Normal, Wet) selected using best professional judgment.
    - Hydrologic parameters measured during those years were used as input to the water balance model.
    - VWRF discharge flows corresponding to each of the discharge scenarios were evaluated for each of the water year types.
    - The number of days of open mouth conditions was counted for each of the 33 model runs.
    - Other VWRF discharge scenarios were estimated by linear interpolation.
5. Changes in habitat area specific to focal species as well as individual beneficial uses.
    - Equilibrium stage estimates (Item 1 above) were used in conjunction with developed stage-habitat curves for focal species or other habitat criteria to estimate habitat area specific to factors affecting a given beneficial use.
    - Equilibrium stage estimates (Item 1 above) were used in conjunction with vegetation/habitat type estimates (Item 2 above) to estimate areas of open water, wetland, and other habitat types.

This analysis described above provided a quantitative basis for evaluating the extent to which discharge scenarios support each of the beneficial uses of the SCRE.

To further build upon the initial quantitative analysis, VWRF discharge scenarios were scored by the AHP assessment factors summarized for each beneficial use below. In addition, because competing factors contribute to the 11 beneficial uses, the AHP was used to conduct value-based weighting of these competing demands to determine the three recommendations required of this report: a determination of Enhancement Discharge Levels, the MEPDV, and the Continued Discharge Level (see *Purpose* section). The sections below summarize the predicted effects of VWRF discharge on the factors contributing to each beneficial use, as well as the relative weighting of those factors in the analysis (Section 5):

Commercial and Sport Fishing (COMM):

COMM was assessed on the basis of physical habitat area for bait and sport fish in the SCRE, as well as dissolved oxygen (DO) conditions. Habitat area was given greater weight (~83%) in the analysis because many of the COMM fish species are tolerant of low DO conditions. Physical habitat (i.e., open water area) for sport and bait fish decreases with increasing reductions in VWRF discharge. Up to 60% reductions in VWRF discharge maintain ~80% of open water habitat area. Greater than 60% reductions in discharge result in considerable reductions in habitat area. Because available data did not allow for predictive modeling of future DO conditions under alternative VWRF discharge scenarios, DO conditions were assessed on the basis of nutrient loading from VWRF discharge. VWRF nutrient loads decrease linearly with decreasing discharge, so DO conditions are predicted to improve under reduced VWRF discharge scenarios. On balance, realization of the COMM beneficial use is greatest under current VWRF discharge, with only minor decreases in Scenarios 1–5.

Estuarine Habitat (EST)

Estuarine habitat was assessed on the basis of open water habitat area for estuarine species (~55% weighting), DO conditions for estuarine species (~28% weighting), wetland habitat area (~12% weighting), and salinity conditions for native estuarine species (4%). Open water and wetland habitat areas decrease with decreasing VWRF discharge. DO conditions are expected to improve as VWRF discharges are reduced. Because salinity conditions rapidly approach freshwater levels under all scenarios, this factor was assigned a low weighting relative to other factors. Only 100% reductions in VWRF discharge are predicted to result in salinities high enough for a long enough period of time to favor native estuarine species over non-native, predominantly freshwater-adapted species. On balance, Scenario 1 (current VWRF discharge) results in the fullest realization of the EST beneficial use.

Marine Habitat (MAR)

Marine habitat, as supported by the SCRE, was considered to be represented by other beneficial uses (EST, WILD, and RARE) and was not evaluated separately here.

Migration of Aquatic Organisms (MIGR)

MIGR was evaluated on the basis of changes in the duration of open mouth conditions providing access to the SCRE during the steelhead and lamprey migration window (November–May), as well as on the basis of DO conditions for steelhead and lamprey. Although DO conditions are predicted to improve as VWRF discharges are reduced, because low DO conditions in the SCRE are episodic and related to algal blooms under closed mouth conditions, they were not thought to limit upmigration or outmigration opportunities, which generally occur under open mouth conditions when Santa Clara River flows and ocean exchanges primarily control water quality.

Open mouth conditions were given 89% of the weighting for the MIGR analysis because of the dependence of migration opportunities upon this factor. The number of days of open mouth conditions decreases with increasing reductions in VWRF discharge, but only marginally so. Greater than 90% of the maximum modeled number of open mouth days are maintained under all VWRF discharge scenarios. MIGR is most fully realized under Scenario 1 (current VWRF discharge), but all discharge scenarios 1-11 are comparable within the range of error of the analysis.

Municipal and Domestic Use (MUN)

MUN was evaluated based on the amount of current VWRF discharge diverted to municipal re-use (100% weighting) relative to current conditions. Water diverted to municipal reuse is equal to the reduction in VWRF discharge. Under current conditions, MUN is minimally supported, since the majority of VWRF tertiary treated water is discharged to the estuary, while a small portion is used for irrigation of commercial facilities and two municipal golf courses. MUN is most fully realized with a 100% reduction in VWRF discharge. Greater than 60% reductions in VWRF discharge are required to meet the minimum target of 2.6 MGD, required for feasible development and operation of the proposed VenturaWaterPure municipal reuse project, though smaller reductions may provide some support for MUN relative to current conditions.

Navigation (NAV)

The SCRE is not thought to be used extensively for navigational purposes, and stakeholders identified NAV as a very low priority consideration. Recreational boating opportunities are assessed under REC-2 below, but NAV was not further considered in the Phase 3 Studies.

Rare, Threatened, or Endangered Species (RARE)

Assessment of support of RARE was a focal point of the Phase 3 Studies. Support provided by VWRF discharge scenarios for the RARE beneficial use was assessed on the basis of habitat conditions, including physical habitat area, water quality, and other factors affecting federally listed aquatic (southern California steelhead and tidewater goby) and avian species (California least tern, western snowy plover), as well as special status plant species. Aquatic species were given ~75% of the weighting for the RARE beneficial use, avian species were given ~18%, and rare plant species were given ~6%. Briefly, the rationale for this assignment was that aquatic species in the SCRE are less mobile than avian species, and are thus more dependent on conditions in the SCRE. Avian species can more readily access alternate habitat in the vicinity of the SCRE. Lastly, habitat types associated with special status plants in the SCRE are present throughout the Santa Clara River corridor and are considered less sensitive to SCRE conditions than federally listed fish and bird species, and their habitats.

Within the aquatic species assessment, physical habitat area (based on depth and substrate criteria) for tidewater goby spawning and rearing were the highest priority considerations (~33% and ~22%, respectively), followed by DO conditions affecting for tidewater goby and steelhead (~17%), and the potential for unseasonal breaching impacts on tidewater goby spawning and tidewater goby and juvenile steelhead rearing (~17%). The resulting discharge scenario scores indicate that tidewater goby spawning habitat area is greatest for Scenario 5 with 40% reductions in VWRF discharge, with only minor differences across a broader range (0–50%) of discharge reductions. Steelhead rearing physical habitat area is greatest under current discharge, but remains largely constant with up to 30% reductions in VWRF discharge, after which point rapid declines in area are apparent with increasing reductions in discharge. DO conditions were assessed on the basis of nutrient loading from VWRF discharge as described above, and are expected to improve with increasing reductions in VWRF discharge. The likelihood of unseasonal breaching was assessed based on the difference between equilibrium water surface elevation (WSE) and berm height because increasing difference in WSEs and berm height increase the difficulty of accomplishing manual unseasonal breaches and reduce the likelihood of breaches from small, unseasonal precipitation events. Because WSEs are highest under current conditions and the berm height is not expected to change as a function of VWRF discharge, the likelihood of breaching is greatest under Scenario 1. The risk of unseasonal breaching decreases with increasing reductions in VWRF discharge. As with EST, salinity conditions were assigned a low weighting (~4%) because predicted variability in salinity is limited across all VWRF discharge scenarios. Because of the broad salinity tolerances of many introduced species, salinity conditions are expected to potentially benefit tidewater goby only at the greatest discharge reduction modeled (100%, Scenario 11). On balance, support for aquatic RARE species is greatest under Scenarios 3-5 (20–40% reductions in VWRF discharge), but comparable over the broader 0–60% range in VWRF discharge reductions associated with Scenarios 1–7.

Support for RARE avian species was assessed on the basis of open water foraging habitat area for California least tern (~62% weighting) and foraging habitat for western snowy plover (~21% weighting). Limiting flooding risk for California least tern and western snowy plover nests was given ~9% weighting. Habitat for riparian-associated special status bird species, which are not federally listed, were given ~8% weighting. California least tern were given highest priority consideration because (a) they are endangered and fully protected species under federal and state law, while western snowy plover are listed as threatened under federal law; (b) the SCRE provides high quality foraging habitat for least tern by concentrating prey, relative to open ocean habitat, while western snowy plover have extensive high quality foraging habitat along the Pacific coast line. Nest flooding is not a high priority consideration based on limited recorded instances of nesting in high risk habitat areas in the SCRE. Foraging habitat for California least tern is comprised of open water area and is greatest under current VWRF discharge. Western snowy plover foraging habitat was represented by the length of the beach berm, since they forage in sandy habitat along the water line. Western snowy plover foraging habitat is largely constant with reductions in VWRF discharge up to 90%, after which point it declines precipitously. Riparian habitat is lowest under current conditions and increases with increasing reductions in VWRF discharge up to 90%. The risk of nest flooding (as estimated by area of high risk nesting habitat) is highest under current conditions and lowest at 90–100% reductions in VWRF discharge. On balance, Scenario 1 (current VWRF discharge) provides the greatest support for RARE bird species primarily due to prioritization of open water foraging habitat for the least tern, with only minor declines in weighted assessment scores up to 70% VWRF discharge reduction (Scenarios 1–8).

Support of special status plant species was evaluated on the basis of the area of riparian and wetland habitat types, with which RARE plants are associated. Wetland habitat was given greater weight (~67%) because it is associated with a greater number of special status plants, and represents a locally rare habitat type, while riparian habitat occurs throughout the Santa Clara River corridor. Riparian habitat increases with increasing reductions in VWRF discharge, while wetland decreases. On balance, Scenario 1 (current VWRF discharge) provides the greatest support for RARE plants.

When the weights of the aquatic, avian, and plant species are applied to evaluate RARE for all three categories of species, Scenarios 1–7 (up to 60% reductions in VWRF discharge) provide the greatest support for RARE species on balance.

Water Contact Recreation (REC-1)

REC-1 was assessed on the basis of open water area for recreational activities involving water contact, potentially including wading or swimming. These activities are not thought to be common in the SCRE. Open water area decreases with increasing reductions in VWRF discharge as described above for other beneficial uses. Scenario 1 (current VWRF discharge) provides the greatest realization of the REC-1 beneficial use.

Non-Contact Water Recreation (REC-2)

REC-2 was assessed on the basis of modeled predictions of inundation of the McGrath State Beach campground (~52% weighting), open water habitat area for viewing waterfowl and wading birds (~26% weighting), nutrient loading from VWRF discharge as a measure of visual and olfactory aesthetics (~17% weighting), and boatable water area (open water area >1ft depth) (~4% weighting). Camping is popular at the McGrath State Beach campground and can be severely affected by campground flooding, particularly since destruction of the protective campground berm during extraordinary storm events in 2005. Birding is also a popular activity. All REC-2 activities can be impacted by aesthetics. Campground inundation is modeled to occur with <30% reductions in VWRF discharge. Although boating uses of the SCRE are not common, boatable water area as well as open water representing birding opportunities both decrease with increasing reductions in VWRF discharge. Aesthetic quality is predicted to improve under reduced VWRF discharge scenarios, as the reduced nutrient loading from VWRF discharge can reduce the frequency and duration of algal blooms. On balance, REC-2 is most fully realized by Scenarios 4–7 (30 – 60% reductions in VWRF discharge).

Spawning, Reproduction, and/or Early Development (SPWN)

SPWN was assessed on the basis of physical habitat area for spawning (open water area), potential changes in DO conditions with decreased VWRF discharges, as well as changes in the likelihood of unseasonal breaching. Because tidewater goby spawning is also assessed separately under RARE, the likelihood of unseasonal breaching was given ~70% of the SPWN weighting since breaches can dewater spawning nests and displace eggs or larvae of other native fish species. DO conditions were given ~21% of the weighting, as eggs and larval fish can be sensitive to low DO conditions. Open water area for spawning of native fish species was given ~8% weighting, separate from the assessment of tidewater goby spawning conducted for RARE. The likelihood of unseasonal breaching was assessed based on the difference between equilibrium WSE and berm height. The likelihood of unseasonal breaching is highest under Scenario 1 (current VWRF discharge), and decreases with increasing reductions in discharge associated with Scenarios 2–11. As with other beneficial uses, DO conditions are expected to improve with increasing reductions in VWRF discharge. Open water area is greatest under Scenario 1 (current discharge conditions) and decreases under reduced VWRF discharge scenarios 2–11. On balance, Scenarios 5–11 (40–100% reductions in VWRF discharge) most fully support the SPWN beneficial use.

Wetland Habitat (WET)

The WET beneficial use was assessed on the basis of wetland habitat area (freshwater marsh and saltmarsh), riparian habitat area, localized flood control capacity, and nutrient conditions favoring native wetland species. Because freshwater marsh and saltmarsh serve a number of important ecological functions and are a locally rare habitat type, they were given ~51% of the WET weighting. Localized flood control capacity, measured as the amount of wetland habitat available to buffer areas in the vicinity of the SCRE during storm events, was given 28% weighting. It has been increasingly recognized by the scientific community and by regulatory agencies in California that riparian habitats also serve a number of same ecological functions typically ascribed to wetlands such as stream bank stabilization, erosion control, and water quality enhancement. As such, they were assigned 17% of the WET weighting. The competitive ability of the invasive *Arundo donax* is enhanced under high nutrient conditions. However, because *Arundo* remains competitively dominant under all nutrient conditions relative to native wetland species, nutrient conditions supporting native species were given low weighting (~4%). Wetland habitat area decreases with increasing reductions in VWRF discharge, while riparian habitat area increases. Nutrient loads from the VWRF decrease with increasing reductions in discharge. On balance, WET is most fully supported under Scenario 1 (current VWRF discharge). This is largely due to reductions in freshwater marsh area associated with reduced VWRF discharge scenarios.

Wildlife Habitat (WILD)

The WILD beneficial use was assessed on the basis of riparian habitat area, mudflat habitat area, and open water habitat area. Riparian habitat was given the majority of the weighting for WILD (~69%) as it supports the widest variety and greatest abundance of wildlife species using the SCRE. Mudflat habitat, which is generally absent under equilibrium closed mouth SCRE conditions, was assigned ~22% of the WILD weighting, as it can serve as valuable foraging habitat for a variety of wildlife species during open mouth conditions when the estuary is drained. Open water habitat can support waterfowl, amphibians, and other wildlife, but was assigned low weighting (~9%) since open water is considered for multiple other beneficial uses, and is more crucial to aquatic species than to wildlife species. As described for other beneficial uses above, riparian habitat increases with increasing reductions in VWRF discharge, while open water habitat decreases with increasing reductions. Mudflat habitat area decreases with increasing reductions in VWRF discharge based on the predicted encroachment of vegetation with lower equilibrium stage. On balance, with all three factors considered, there is only negligible difference between the support provided by any of the VWRF discharge scenarios for the WILD beneficial use.

Beneficial Use Synthesis and Recommendation on Enhancement

As described above, we use the AHP as a framework for assigning weights to each factor contributing to each beneficial use, and scoring each VWRF discharge scenario for each weighted factor. The weighted scores are then compiled to determine overall support for each beneficial use in isolation (see sections above). The AHP was also used to weight each of the beneficial uses relative to each other such that ecological beneficial uses, particularly those supporting native and threatened and endangered species, were prioritized. Considering input provided by the TRT, Resources Agencies, and other stakeholders, beneficial uses were ranked and weighted as follows:

1. Rare, Threatened, or Endangered Species (35%)
2. Spawning, Reproduction, an/or Early Development (16%)
3. Migration of Aquatic Organisms (15%)
4. Estuarine Habitat (10%)
5. Wildlife Habitat (7%)
6. Wetland Habitat (7%)
7. Municipal and Domestic Use (5%)

8. Non-Contact Water Recreation (2%)
9. Commercial and Sport Fishing (1%)
10. Water Contact Recreation (1%)
11. Marine Habitat (N/A)
12. Navigation (N/A)

Based on this relative weighting, and on the results for individual beneficial uses, VWRF discharge scenarios were compared based on the extent to which they support the balance of all beneficial uses of the SCRE. Intermediate reductions in VWRF discharge (Scenarios 5–7; 40-60% reductions in VWRF discharge, respectively) provide the fullest realization of beneficial uses of the SCRE. Scenarios 8, 4, and 1 (70%, 30% and 0% reductions in VWRF discharge, respectively) support greater than 90% of the maximum score. Scenarios 2, 3, and 9–11 (80–100% reductions in VWRF discharge) result in relatively lower potential for beneficial use realization.

Recommended Enhancement Discharge Levels, the Maximum Ecologically Protective Diversion Volume (MEPDV), and the Continued Discharge Level

To provide a recommendation for Enhancement Discharge Levels pursuant to the EBE Policy, alternative VWRF discharge scenarios (Scenarios 1–10) were compared to Scenario 11, which constitutes the absence of VWRF discharge. On balance, the results of the beneficial use analyses above indicate that continued treated tertiary effluent discharges for Scenarios 5 through 7 in the amounts ranging from 1.9–2.8 MGD (i.e., 40–60% of current levels) constitute Enhancement Discharge Levels, providing fuller realization of the balance of beneficial uses important to protection of the SCRE relative to the absence of all VWRF discharge. Although beyond the scope of EBE Policy requirements, the results of the analysis support the conclusion that continued discharge of tertiary treated effluent during closed-mouth, dry-weather conditions in amounts ranging from 1.9 to 2.8 MGD optimize realization of beneficial uses as compared to all other VWRF discharge scenarios assessed, including as compared to Scenario 1, continued discharge of 4.7 MGD..

For the purposes of identifying the maximum ecologically protective diversion volume (MEPDV), the beneficial use assessment conducted for this study indicates that the balance of all ecological beneficial uses (as determined by weighted AHP scores for COMM, EST, MIGR, RARE, SPWN, WET, and WILD) is maximized by Scenarios 5 through 7, with Scenarios 1, 4 and 8 exhibiting intermediate scores, and Scenarios 2, 3, 9, 10 and 11 exhibiting the lowest scores (Table 5-24). Based upon review of these results and the underlying ecological considerations reviewed in this report, including review of the PCEs/PBFs for the listed species in the SCRE, we recommend that an MEPDV of 60% in compliance with the Consent Decree, corresponding to a diversion of 2.8 MGD under current conditions.

Corresponding to the MEPDV recommendation above, we recommend that a minimum Continued Discharge Level of 1.9 MGD to the SCRE during closed-mouth, dry-weather conditions will not result in take of listed species.

It should be noted that if additional tertiary-treated effluent becomes available in the future, the diversion flow volume would be expected to increase above the 2.8 MGD MEPDV recommendation in this report, as long as the Continued Discharge Level of 1.9 MGD to the SCRE is maintained during closed-mouth, dry-weather conditions.

Caveats and Uncertainty

The analysis presented in this report relies upon data and literature review to develop conceptual models and modeling tools for the assessment of existing as well as potential future conditions under alternative VWRF discharge levels. Uncertainties discussed in this report include limitations in modeling tools and factors affecting habitat suitability, quantitative and qualitative assessments of beneficial uses using the Analytic Hierarchy Process, as well as the assessment of future conditions.

Because of complex interactions of abiotic, biotic, and anthropogenic processes affecting the beneficial uses of the SCRE, the ability to accurately and precisely predict future conditions under hypothetical future VWRF discharge scenarios is limited by a number of factors, including modeling constraints and data limitations. While we have made every to incorporate the most important factors into the quantitative assessments, conceptual models and modeling tools developed in this report, each have limitations and associated uncertainty. Uncertainties associated with the water balance model include representation of the lagoon bathymetry, groundwater seepage rates into the SCRE, and assumptions regarding berm height and length affecting seepage rates out of the SCRE as well as breach dynamics, among others. The nutrient balance, subject to the same uncertainties as the water balance, has additional uncertainties regarding assumptions of fuly mixed conditions, nutrient uptake by algae as well as internal sediment sources and sinks that were not mechanistically represented. Lastly, potential variations in water temperature were represented by a combination of existing data review as well as a simplified heat balance model based upon a limited number of monitoring locations. While we are confident that the VWRF discharge has little influence upon lagoon temperatures, it should be noted that water temperature monitoring included in the Workplan was not designed for use in water temperature modeling, so the model predictions could not be validated based upon data not used in the model calibration. Despite these limitations, the models were parameterized based on site-specific observations, extensive data collection, and literature review. However, as with any model, there is some uncertainty resulting from predictions of future conditions based on past observations; this is particularly true in a geomorphically and hydrologically dynamic system like the SCRE.

The quantitative comparison of VWRF discharge scenarios in relation to designated beneficial uses of the SCRE was based upon application of the Analytic Hierarchy Process (AHP) and included factors that met three criteria: (a) the factor was important in the realization of a beneficial use; (b) the factor was affected by VWRF discharge; and (c) the factor could be feasibly modeled or otherwise estimated under alternative VWRF discharge scenarios. There were a number of factors that met the first criterion, but were excluded from the analysis because they did not meet one or both of the last two criteria. For example, food conditions may be important determinants of habitat quality for the fish and wildlife evaluated. However, only limited data were available to assess current food conditions in the SCRE, so determining future conditions under alternative VWRF discharge scenarios would result in highly uncertain predictions. Similarly, because of the complex interactions controlling algal growth in surface waters, some water quality parameters such as nutrient or dissolved oxygen levels could not be accurately modeled using available data. Thus, it should be acknowledged that some features potentially affecting designated beneficial uses could not be incorporated into the assessment for this study.

The AHP was used to provide a transparent framework for comparing and discharge alternative upon a wide range of factors considered in this study. Recognizing, however, that the potential that discharge recommendations may be more subjective, a sensitivity evaluation was conducted to compare individual weighting and scenario rankings. The results show general agreement over discharge scenarios optimizing realization of beneficial uses and suggest robust conclusions regarding enhancement of the beneficial uses of the SCRE and the recommended MEPDV. It should be noted that the AHP is intended as a tool to aid in decision making, rather than as the final decision-maker itself. The recommendations of this study should be viewed holistically, and best professional judgement should ultimately remain central to decision-making process.

In making future VWRF discharge recommendations in this study, it is important to note that the VWRF discharges as well as other future flow contributions to the SCRE will vary at seasonal and annual time scales based on changes in hydrologic conditions, geomorphic change, and human population. Other than potential variations in VWRF discharge and simplified assumptions regarding evolution of vegetation types in the SCRE in the future, no other future changes in lagoon morphology or habitat conditions were considered as part of this study.

Lastly, to ensure that the VWRF discharge recommendations do not result in potentially adverse conditions, adaptive management is recommended as part of future permitting actions by the Resources Agencies in their evaluation of the analysis, information and discharge recommendations herein.

Conclusions

Based on the comprehensive analysis of the effects of 11 different VWRF discharge scenarios on 11 beneficial uses protecting the SCRE and its watershed as presented in this Report, the following primary conclusions were reached:

- As required by the NPDES permit and EBE Policy, continued VWRF discharges ranging from 1.9 to 2.8 MGD during closed-mouth, dry-weather conditions provide a fuller realization of the balance of beneficial uses of the SCRE and MUN uses within the SCRE watershed relative to the absence of all VWRF discharge, and therefore provide “enhancement”.
- The maximum ecologically protective diversion volumne (MEPDV) recommendation of 60%, corresponding to a diversion of 2.8 MGD under current conditions, will be protective of the ecological functions of the SCRE while providing for improvement of local water supply by diversion of VWRF effluent to water reclamation uses. This recommended diversion of 2.8 MGD is expected to increase as additional wastewater flow is available, provided that the Continued Discharge Level of 1.9 MGD (see below) is maintained during closed-mouth, dry-weather conditions.
- A Continued Discharge Level 1.9 MGD during closed-mouth, dry-weather conditions would be protective of estuary native species and habitats, particularly those that are listed for protection under the state and federal Endangered Species Acts.

In addressing questions detailed in Order R4-2013-0174, § VI.c.2.b.ii and § VI.c.2.b.iii regarding potential causes of nutrient, dissolved oxygen and toxicity impairments in the SCRE to be addressed as part of the *Nutrient, Dissolved Oxygen and Toxicity Special Study*, it should be noted that although the Phase 3 Study results do not indicate any toxicity impairments, the VWRF discharge has been shown to contribute to elevated nutrient levels in the SCRE. Because the Consent Decree between Wishtoyo, HTB and the City included provisions for treatment wetlands to reduce nutrient loading to the SCRE for any VWRF discharge that is not able to be diverted for recycled water purposes, it should be expected that the recommended 60% MEPDV and discharge recommendation of 1.9 MGD in conjunction with treatment wetlands or other measures adopted by the City for any continued discharges to the SCRE will result in additional reductions in nutrient loading to the SCRE, thus improving water quality above that associated with VWRF flow reductions alone.

The recommendations in this Report regarding the Enhancement Discharge Levels, the MEPDV, and the Continued Discharge Level will be further considered and peer reviewed by three scientific experts appointed to the “Scientific Review Panel” (SRP). The SRP will prepare a report of their findings regarding these levels, and will recommend an alternative MEPDV if they disagree with the recommendation set forth in this Report. The RWQCB, NMFS, USFWS, and CDFW, as Resources Agencies with jurisdiction to environmentally review, consult with respect to, certify, approve, condition, or otherwise permit continued VWRF discharges to the SCRE and/or modifications to existing VWRF discharges, may then use this Report and the SRP Report to inform their regulatory determinations.

1 INTRODUCTION

1.1 Background

The City of San Buenaventura (City; also known as Ventura) owns and operates the Ventura Water Reclamation Facility (VWRF), which discharges some of its recycled, tertiary treated municipal wastewater to the Santa Clara River Estuary[5] (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. R4-2013-0174, most recently adopted and renewed by the California Regional Water Quality Control Board, Los Angeles Region (Regional Board), on November 7, 2013 (National Pollution Discharge Elimination System [NPDES] Permit No. CA0053651). In accordance with the Order, the City is required to complete data collection and special studies to address the effects of the VWRF discharge to the SCRE.

Under Section A of Chapter 1 of the 1974 Water Quality Control Policy for the Enclosed Bays and Estuaries of California ("EBE Policy"; originally adopted by Resolution No. 74-43 [May 16, 1974], as amended RWRCB Resolution No. 95-84 [Nov. 16, 1995]; interpreted by Memorandum from Bill Dendy, former Executive Officer of the SWRCB, to Dr. David Joseph, Executive Officer of the RWQCB, North Coast Region [Oct. 21, 1974]; Water Quality Order 79-20 [May 17, 1979]), discharges of municipal wastewater to enclosed bays and estuaries are to be phased out except "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 estuary, such as fish and wildlife habitat and non-contact water recreation, were more fully realized 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.

In discussions prior to the adoption of Order R4-2008-0011, a number of questions arose regarding the definition of enhancement, the benefits that the discharge provides to the Estuary and adjacent subwatershed, and how discharge practices could be modified over time to protect and enhance habitat and water quality of the portion of the SCRE directly affected by the VWRF discharge. To address these issues regarding a finding of enhancement and other information needs in the Order, the Regional Board required the City to complete a series of three Special Studies as a condition of the City's NPDES discharge permit, which are summarized below:

- **Estuary Subwatershed Studies** (September 2011) (i.e., Phase 1 Estuary Studies) **–** Provided a synthesis of information regarding the SCRE ecosystem functioning under

[5] The SCRE is a lagoon-type estuary that is periodically disconnected from the Pacific Ocean by a sandbar. We use the terms "estuary" and "lagoon" largely interchangeably in this report when referring to the SCRE. Note that the SCRE is considered an "estuary" as it relates to the Enclosed Bays and Estuaries Policy of 1974, which specifically states that "the mouths of streams which are temporarily separated from the ocean by sandbars shall be considered as estuaries." (see EBE Policy pg 9)

existing conditions (characterized by tertiary treated VWRF flows discharged to the Wildlife/Polishing Ponds and then to the SCRE) to determine if VWRF discharge results in fuller realization of beneficial uses within the SCRE as compared to the absence of discharge, and compared the level of realization of beneficial uses for a range of representative potential future VWRF effluent discharge alternatives and management measures that could be implemented to achieve further improvement in quality and/or beneficial uses using water balance and water quality predictive tools developed with existing and newly-collected data.

- **Treatment Wetlands Feasibility Study** (March 2010) – Provided an evaluation at a planning concept level of the feasibility of implementing a constructed treatment wetland to achieve additional reductions in nutrients, copper and other metals in the VWRF tertiary treated discharge to further promote improvements in receiving water for beneficial uses. Depending upon flow volume requirements of one or more of the VWRF discharge alternatives developed under the Phase 1 Estuary Study, additional nutrient reductions were identified through a combination of process upgrades at the VWRF plant and a wetland design accommodating a hydraulic residence time of 4–12 days.
- **Recycled Water Market Study** (March 2010) – Provided an evaluation and quantification at a conceptual planning level of the feasibility of expanding the City's existing reclaimed water system through evaluation of potential users within a five-mile radius of the VWRF for purposes of providing an alternative to discharging VWRF effluent flow to the SCRE. Depending on the flow diversion requirements, this study determined that recycled water projects could be implemented for the purpose of diverting the VWRF discharge on a seasonal basis, provided that diversion requirements take into account technical constraints on diversion, such as public health and safety, design and capacity, and/or operational constraints that may make diversions at certain times infeasible or inappropriate to implement.

Following completion of these three Phase 1 Studies and receipt of stakeholder feedback, a number of additional data collection and analysis needs were identified by the City, including the need to: (1) develop additional information (more hydrologic and water quality data) to improve the understanding of SCRE functioning and help assure protection of the sensitive wildlife and aquatic resources and habitats within the SCRE; and (2) integrate the conclusions of all three of the Phase 1 Studies into a process for selection, environmental review, and design of a preferred VWRF discharge/diversion alternative or combination of alternatives to create a discharge regime that further improves beneficial uses of the SCRE. These needs were addressed through Estuary Special Studies Phase 2: Facilities Planning Study for Expanding Recycled Water Delivery (March 2013) (Phase 2 Studies), the amended final report for which was submitted to the Regional Board in 2014.

At the conclusion of the Phase 2 Studies, several stakeholders expressed concerns about identified data gaps and the report findings. In response to these concerns, the City's NPDES Permit, Regional Board Order R4-2013-0174, for VWRF discharges requires the City to conduct the following additional "special studies":

- **Phase 3 Estuary Studies (Phase 3 Study) –** "The Discharger shall perform additional estuary studies to provide sufficient information to allow the Regional Water Board to determine whether or not the continued discharge of effluent enhances the Estuary. The study will clarify the water budget analysis for the Santa Clara River Estuary, to determine whether any effluent discharge is needed to sustain the SCRE native species, and if so how much." Order R4-20013-00174, § VI.c.2.b.i.

- **Nutrient, Dissolved Oxygen, and Toxicity Special Study –** "The Discharger must perform a special study to identify the cause of nutrient, dissolved oxygen and toxicity impairments in the Estuary. The Dissolved Oxygen Study will include sufficient monitoring, including diurnal monitoring, to determine the suitability of DO levels for the Estuary's aquatic life. If it is determined that the effluent from the Facility is causing the impairments, the Facility must propose a plan for reducing nutrient loading, including ammonia, nitrogen and phosphorus loading and toxicity impairments." Order R4-20013-00174, § VI.c.2.b.ii.
- **Groundwater Special Study –** "The Discharger must perform a special study to document the interaction between the estuary, discharge and groundwater and determine if the beneficial use of MUN [*any water designated as municipal or domestic supply, MUN, in a Regional Water Board Basin Plan*] applies to the water impacted by the discharge." Order R4-20013-00174, § VI.c.2.b.iii.

The Fact Sheet for the City's NPDES Permit, Regional Board Order R4-2013-0174, recognized that the City has entered into the Tertiary Treated Flows Consent Decree and Stipulated Dismissal with the Wishtoyo Foundation Ventura Coastkeeper, Heal the Bay (Consent Decree), and that the consent Decree requires a determination, through scientific analysis, of the Maximum Ecologically Protective Diversion Volume (MEPDV) with the assistance of a Scientific Review Panel, The Consent Decree defines the MEPDV the maximum annual average volume or flow of effluent appropriate to divert from the discharge to the SCRE that is ecologically protective of the SCRE, the Estuary's aquatic species, and the surrounding subwatershed. The Regional board specified in the Fact Sheet:

> "The special studies described in this Order may provide the scientific analysis used to define the MEPDV, but the special studies must provide sufficient and meaningful information to determine if discharge enhances the Estuary. The MEPDV analysis may be used by the Regional Water board staff in its evaluation of Estuary enhancement during the next revision of this Order, projected to take place in November of 2018."

Order R4-2013-0174, Attachment F-Fact Sheet, Section III.C.10.d (p. F-19).

The City submitted work plans reviewed and approved by the Wishtoyo Foundation's Ventura Coastkeeper Program (Wishtoyo) and Heal the Bay to the Regional Board for each of these studies on October 27, 2014 outlining the elements needed to address the information needs of the NPDES permit. The Regional Board approved the Workplans with some additional requirements on December 12, 2014. In approving of the Workplans, the Regional Board required that a separate Data and Alternative Evaluation Workplan be developed to set a framework for providing "a finding on estuary enhancement and a recommendation of the effluent discharge flow rate needed to sustain the estuary's native species." This Data and Alternative Evaluation Workplan was submitted to the Regional Board on December 30, 2016. These approved Workplans form the foundation for this report, and for the final reports for the other two required studies.

1.2 Purpose

This report has been prepared in fulfillment of the "Phase 3 Estuary Studies" (Phase 3 Study) requirement of Order R4-2013-0174,§ VI.c.2.b.i. and provides an updated assessment of the

VWRF tertiary treated discharge on the ecological functions and beneficial uses of the SCRE. The results of the study will be used to make three interrelated discharge recommendations, which may be stated as the following study goals:

1. As required by the Ventura NPDES Permit, determine the levels of VWRF discharge, if any, that likely provide fuller realization of beneficial uses of the SCRE and surrounding subwatershed relative to the absence of all discharge ("Enhancement Discharge Levels"). To derive recommended Enhancement Discharge Levels, this Report evaluates the effects of a "no-discharge" scenario in comparison to the effects of 10 other VWRF discharge scenarios on the basis of both quantitative and qualitative assessments to determine which of the discharge scenarios, are likely to more fully support the 10 designated beneficial uses of the SCRE in the Basin Plan, as well as the potential for realizing additional municipal water supply (MUN) uses within the SCRE subwatershed. This analysis complies with requirements of the NPDES Permit, and facilitates the RWQCB's determination of "enhancement" (as defined by EBE Policy) likely to be associated with different levels of discharge for purposes of the next renewal of the VWRF NPDES Permit.
2. As allowed by the Ventura NPDES Permit and the Consent Decree, determine the MEPDV, or the maximum level of VWRF discharge that may be diverted from the SCRE to water reclamation uses, while still protecting ecological resources of the SCRE. To derive the MEPDV, this Report evaluates the degree to which a range of VWRF diversions—from 0% diversion (current discharge conditions) to 100% diversion (no discharge)—may either better support or potentially adversely impact existing ecological conditions and related beneficial uses of the SCRE, with particular weight and emphasis upon habitat use by native listed species and designated critical habitat. This analysis is anticipated by the NPDES Permit, complies with the Consent Decree, and may also facilitate environmental review under NEPA and CEQA of potentially significant environmental impacts to existing SCRE conditions that may be associated with diversions of VWRF tertiary treated flow to augment and increase reliability of local municipal (MUN) water supplies. In addition, this analysis facilitates review, consideration and future permitting determinations by NMFS, USFWS and CDFW with respect to significant adverse impacts of diversions, if any, on listed species and associated critical habitats within the SCRE.
3. As required by the Ventura NPDES Permit, determine the minimum level of continued VWRF discharge, if any, needed to protect the SCRE's existing ecological resources, particularly native listed species and critical habitats, as well as related SCRE beneficial uses during closed-mouth, dry-weather conditions (the "Continued Discharge Level"). To derive a recommended Continued Discharge Level, if any, needed to protect the SCRE's existing ecological resources, this Report evaluates the threshold at which potential impacts to existing ecological resources of the SCRE and related beneficial uses are likely to result from reducing current VWRF discharges of tertiary treated effluent to (i.e., increasing diversions of effluent from) the SCRE, taking into account existing conditions and current discharge levels. This analysis complies with requirements of the NPDES Permit, and facilitates a determination by the RWQCB regarding an appropriate average annual flow that may be permitted or required by the next VWRF NPDES Permit.

Figure 1-1. Study area location and surrounding features of the Ventura Wastewater Reclamation Facility and the Santa Clara River Estuary.

1.3 Summary of Phases 1 and 2 Estuary and Subwatershed Studies

To compare the degree to which SCRE conditions support designated beneficial uses under existing and alternative operational scenarios of the VWRF, contemporary and historical data for geomorphological, hydrological, chemical, and biological processes were synthesized and reviewed in the Phase 1 Study. This information synthesis was used to develop a conceptual model linking ecosystem functioning of the estuary (Figure 1-2) to biological outcomes of representative focal species. As an idealized model of ecosystem function, 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 community composition. Modification of any of these key inputs, including VWRF inflows, may affect physical processes and, subsequently, plant communities, fish and wildlife populations, and ultimately beneficial uses of the SCRE.

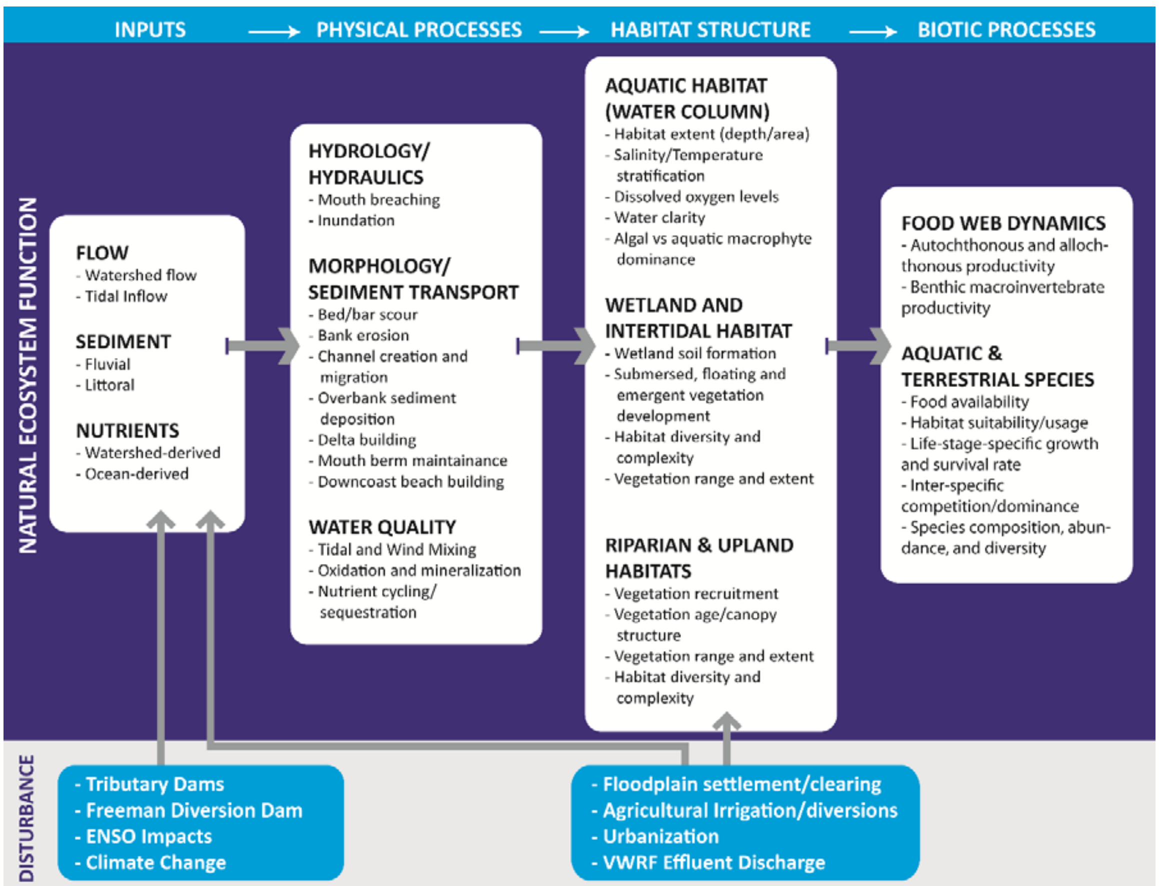


Figure 1-2. Conceptual model of SCRE ecosystem function.

1.3.1 Selection and use of focal species

To address ecological linkages to broader terrestrial and aquatic species and habitat related beneficial uses, a focal species approach was adopted for the Phase 1 and Phase 2 Studies. Focal species are often used to understand broader ecosystem linkages so that management actions directed to benefit these species and their habitats will also benefit the larger ecosystem. This approach can be used when the depth of data collection, analysis, or review required to properly characterize the habitat requirements and predict changes in those requirements makes evaluating all conservation target species in a given ecosystem infeasible. In the case of the SCRE, evaluating the effects of VWRF discharge requires identifying the habitats used by multiple lifestages of a given species, the ecological processes that create and maintain those habitats, and how management actions (i.e., VWRF discharge) influences those ecological processes and habitat conditions. This requires extensive literature review, analysis, and predictive modeling that is not feasible for all conservation targets in the SCRE. To overcome this challenge, two focal fish species were used to evaluate effects of ecosystem inputs upon aquatic species, habitats, and related beneficial uses, and two avian species will be used to evaluate effects on terrestrial species, habitats, and related beneficial uses (see Sections 3.6.3 and 3.7.2 for complete species profiles):

- southern California steelhead (*Oncorhynchus mykiss*),
- tidewater goby (*Eucyclogobius newberryi*),
- western snowy plover (*Charadrius alexandrinus nivosus*[6]), and
- California least tern (*Sternula antillarum browni*).

Each of these selected focal species is federally listed as either threatened or endangered, indicating they are sensitive and thus a conservative indicators of ecosystem health. Furthermore, the possibility of “take” of listed species through management alternatives can be evaluated under this framework. In the conservation literature, smaller-bodied organisms than those selected herein are typically used as indicator species for evaluating environmental health, as they tend to be more sensitive to factors such as environmental pollutants (Caro and O’Doherty 1999). While such organisms were used for toxicity testing in this study (see Section 2.2.2), the scope and goals of the study require evaluating organisms that are directly relevant to designated beneficial uses, rather than the *most* sensitive species in the ecosystem, so our selected species suite does not strictly fit the definition for health indicator species presented by Caro and O’Doherty (1999). Rather, our approach more closely aligns with the focal species approach described by Lambeck (1997) in which a suite of species is chosen whose combined needs are intended to encapsulate the needs of other conservation target species in that ecosystem. In addition to their sensitivity and listed status, our four focal species were selected based on variable life history traits intended to encapsulate or, in part, represent the requirements of other species in the SCRE. Steelhead, for example, are migratory and transient users of the SCRE, while tidewater goby are full-time residents. The two species have different breeding and feeding patterns and they require somewhat different abiotic conditions. Similarly, least tern and snowy plover have different feeding patterns and foraging habitat requirements, though their breeding patterns show a high degree of overlap. California least tern eat mostly fish, while western snowy plover forage for invertebrates. California least tern are only seasonal residents while many western snowy plover are year-round residents of the SCRE. This focal species approach has been criticized for failing

[6] 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.

to represent all wildlife species of conservation concern (e.g., Lindenmayer and Fischer 2003). Certainly, some wildlife species are better represented by selected focal species than others. Inherently, a select number of focal species will fail to fully represent at least some of the wide range of aquatic and terrestrial plants, reptiles, amphibians, mammals, invertebrates, fishes, and birds in a given ecosystem. Nonetheless, this approach has utility in assessing management options when comprehensive evaluation of the full suite of species is not feasible. The current study requires an evaluation of relative VWRF discharge impacts on designated beneficial uses of the SCRE and its watershed. Overall, the selected focal species represent well the potential relative effects of the VWRF discharge scenarios on beneficial uses related to SCRE ecological resources (e.g., MIGR, RARE, SPWN, WILD). For some beneficial uses (e.g., COMM, EST, WILD), our in-depth evaluation of the selected focal species was supplemented with additional research on species not well-represented by the focal species. In other cases, the focal species approach was not relevant to a given beneficial use (REC-1, REC-2), and was thus not used. Some beneficial uses rely on assessments of habitat characteristics associated with broad groups of species, such as habitat type (e.g. open water, riparian, wetland) or salinity conditions (fresh vs. salt water). However, stakeholder input, including input from the Resources Agencies, during the Phase 1 and Phase 2 Studies indicated that habitat for native, and particularly listed species is the primary concern, and so the focal species approach was considered central to the overall study approach.

1.3.2 Phases 1 and 2 Study findings

The Phase 1 Study evaluated the predicted effects of six scenarios (including current operations and 100% reduction in VWRF discharge) on sensitive species habitat areas and water quality as they relate to designated beneficial uses, with emphasis on beneficial uses protecting focal fish, wildlife species, and their habitats listed above and, secondarily, upon recreational use of the SCRE. Because the SCRE water contributions are dominated by river flow during rainfall events (primarily in the winter) when the estuary mouth is typically open, the Phase 1 Study identified low-flow, closed-mouth, dry-weather conditions as the critical period for assessing discharge effects on physical and chemical water quality, as well as on species, habitat extent and quality in the SCRE. This focus on critical conditions has been retained in the Phase 3 Studies.

Physical and chemical water quality conditions were assessed using historical averages under various discharge conditions, as well as by a nutrient balance using components of the water balance model. Habitat conditions for the four-focal species above were assessed both quantitatively and qualitatively based on stage-habitat relationships. Water quality was considered in qualitative assessments of habitat conditions and effects on the focal species, primarily related to nutrient and dissolved oxygen levels, as well as on historical data summarized to compare open and closed mouth conditions in the SCRE. Finally, recreational opportunities were assessed based on the frequency and duration of flooding at McGrath State Park, as well as potential losses of Wildlife/Polishing Pond-related recreational opportunities.

The Phase 1 Study found that, on balance, VWRF discharge (as operated at the time of the study) provided fuller realization of designated beneficial uses relative to the absence of all VWRF discharge to the estuary, but that additional treatment for nutrients or some level of discharge reduction might better optimize beneficial uses. The then existing VWRF discharge was found to provide additional habitat for tidewater goby and southern California steelhead, unique off-channel habitats such as the VWRF outfall channel, as well as fuller realization of wildlife related uses from the VWRF Wildlife/Polishing Ponds that would not be present under a zero-discharge alternative.

Nevertheless, the results of the Phase 1 Study indicated that additional treatment of the VWRF discharge or some level of discharge reduction to reduce nutrient loads might further improve or optimize beneficial uses in SCRE by reducing the frequency, duration and/or severity of algal blooms, and thereby reducing periods of adverse dissolved oxygen conditions. The Phase 1 Study concluded that reducing periods of low dissolved oxygen would benefit tidewater goby and steelhead, resulting in further increased realization of the RARE, WILD, EST, and SPWN beneficial uses. The Phase 1 Study also concluded that discharge reductions might also further optimize realization of SCRE beneficial uses by (i) potentially decreasing the frequency of unseasonal breaching by unknown parties attempting to reduce high water conditions in areas neighboring the SCRE during closed berm conditions, as well as potentially reducing habitat for freshwater predators (thereby increasing habitat quality for tidewater goby), and (ii) potentially dry weather flooding of the campground (improving REC-2 beneficial uses), conditions that have become more frequent since the loss of the campground's protective berm during 2005. The Phase 1 Study concluded that management actions involving tertiary treated flow reductions and/or nutrient reductions would provide further identifiable benefits to SCRE conditions, though only during dry weather, closed-mouth conditions normally occurring during summer months. 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), the improved nutrient treatment and volume reduction alternatives evaluated in the Phase 1 Study would not be expected to result in detectable improvements in the realization of beneficial uses because all flows, from VWRF discharges and all other sources, flow out to the ocean during those open mouth conditions.

The Phase 2 Study used the predictive tools developed as part of the Phase 1 Study to evaluate a number of discharge alternatives vis-à-vis each other and existing discharge volumes, each of which included treatment improvements for denitrification and reduced VWRF discharge volumes as compared to the existing condition (i.e., variants of Alternative 5 from the Phase 1 Study). Based on stakeholder feedback received prior to the Phase 2 Study indicating that the campground might be moved and/or the campground area might be restored to a natural condition, campground flooding was not explicitly reconsidered, and the Phase 2 evaluation of VWRF discharge alternatives focused on the effects on fish- and wildlife-related beneficial uses. The Phase 2 Study found that habitat quantity would likely be greatest at VWRF discharge volumes of 8 MGD, but that total inorganic nitrogen (TIN) would decrease with decreasing VWRF discharge as well as under improved denitrification scenarios, suggesting that habitat quanity and quality would be improved at VWRF discharges of 4–5 MGD, relative to higher discharge scenarios analyzed.

Following the conclusion of the Phase 1 and 2 studies and Stakeholder processes, denitrification and other treatment technologies were upgraded at the VWRF. In addition, a data gaps analysis was conducted to address stakeholder comments and remaining information needs regarding assessment of the amounts of VWRF tertiary treated flows discharged to the SCRE under present-day and alternative discharge scenarios that would optimize realization of beneficial uses, and best protect native species and habitats within the SCRE, particularly those listed for protection. Additionally, planning and analysis was initiated to identify alternative projects that should be further planned, designed, engineered, permitted, environmentally reviewed and implemented to potentially divert VWRF discharges (i.e., up to the MEPDV) from the SCRE to water reclamation uses. *See* Diversion Infrastructure Projects Study (March 2016). Based on planning and analysis for VWRF diversion infrastructure, combined with planning and analysis of projects to address local water supply in increasingly dry and drought conditions, the City updated its Comprehensive Water Resources Report (Ventura Water 2017) and Urban Water Management

Plan (City of Ventura 2015) to incorporate Ventura local Water Supply Projects that will augment and improve the quality and reliability of local water supplies. These Water Supply Projects include a project to divert VWRF tertiary treated flows from discharge to the SCRE to a new water purification plant to produce recycled water for potable reuse (the “VenturaWaterPure Project”). The information in this Phase 3 Studies report, including the recommendations regarding the MEPDV and Continued Discharge Level, will be used to plan and design the VenturaWaterPure Project, and to review the potential environmental impacts of the project under CEQA and, as applicable, NEPA.

1.4 Data Gaps Analysis

Based on stakeholder feedback on earlier studies of the SCRE (Entrix 2002a,b, 2004, 2007a,b; Kamman Hydrology 2007; Kennedy/Jenks 2007; Nautilus 2005) as well as the Phase 1 and 2 Estuary Studies (Stillwater Sciences 2011a, Carolllo Engineers 2014), including the feedback at the February 2013 stakeholder workshop and written comments on the Phase 2 Study, some stakeholders, including United States Fish and Wildlife Service (USFWS), National Marine Fisheries Service (NMFS), and California Department of Fish and Wildlife (CDFW) (Resources Agencies), agree with the findings that some VWRF flow into the SCRE is necessary to protect endangered and threatened species. However, other stakeholders disagreed with the findings of the Phase 1 and 2 Studies, including HTB and Wishtoyo.

Many stakeholders indicated the need for additional collection of groundwater quality data, additional data collection to confirm dominant groundwater sources for the SCRE, additional testing of the VWRF discharge for trace levels of constituents of emerging concern (CECs), as well as more intensive characterization of habitat suitability based on water quality parameters, and use patterns likely to be exhibited by tidewater goby and southern California steelhead, rather than primary reliance on comparisons of stage-habitat relationships, to better reflect the complexity of habitat suitability for the listed species occupying the SCRE.

1.4.1 Additional information needed to determine the degree to which SCRE Beneficial Uses continue to be enhanced by VWRF discharges

As identified in the Phase 2 Study and the stakeholder comments thereon, the following additional information and analysis were required to determine the degree to which beneficial uses are not only more fully realized and enhanced as compared to the absence of discharge, but actually improved to the maximum extent possible:

- Evaluation of optimal berm height and stability relative to breaching risk. This would include additional stage and discharge data collection in comparison to mouth closure status and berm height.
- Collection of additional groundwater quality data at existing and potentially new locations. This information would help identify dominant groundwater nutrient source contributions to the SCRE and potential water quality effects in the event VWRF discharge reductions are implemented.

1.4.2 Additional information needed to determine the volume of discharge, if any, required to protect SCRE native species, including Endangered Species

As identified in the Phase 2 Study and the stakeholder comments thereon, the following additional information and analysis were required to determine the degree to which VWRF discharges are necessary to protect endangered, threatened and sensitive species occupying or using the SCRE as required by the NPDES Permit, and the ESA and CESA.

- Collection of additional flow, water quality and species monitoring data corresponding to SCRE sources under seasonal and mouth berm conditions not encountered during monitoring conducted pursuant to the Phases 1 and 2 Estuary Special Studies.
- Evaluation of SCRE habitat use data for endangered species, and in comparison to stage-habitat relationships and water quality.
- Collection of reliable, long-term data regarding presence of CECs in support of potential future evaluations of biological effects upon tidewater goby, southern California steelhead, or selected sentinel species to be determined by the SWRCB.

Using the information collected in prior phases of study, the Phase 3 Studies seek to not only determine the degree to which the VWRF discharges provide enhancement of the SCRE as compared to the absence of discharge, but also to determine and confirm (a) the minimum volume of VWRF discharge to the Ponds and the SCRE that is necessary protect native species and habitats, particularly those protected by state and federal laws and-related designated beneficial uses of the Estuary, and (b) the maximum volume of VWRF discharge that can be diverted away from the SCRE to improve municipal water supply within the SCRE watershed and to provide water supply benefits to the community (e.g., use of recycled water to augment water supply), without adversely impacting species, habitat, and-related designated beneficial uses of the SCRE (i.e., the MEPDV). The minimum required discharge recommended to protect the SCRE’s native species and habitats (i.e., the Continued Discharge Level), and, on the flip side, the MEPDV recommendation of this Phase 3 Studies report will also allow the Resources Agencies to determine the degree to which discharges may be reduced to improve SCRE water quality and related aquatic habitat conditions, to augmenting local water supply, and by reducing the potential for localized flooding and unseasonal breaching, all while avoiding, minimizing and fully mitigating adverse impacts to other sensitive species using the SCRE, adverse modification of designated critical habitat, and incidental “take” of listed species (as defined in the state and federal Endangered Species Acts), and precluding jeopardy to their survival and recovery.

1.5 Phase 3 Approach and Methods

The Phase 3 Studies rely upon additional Phase 3 monitoring data and updates to previously developed conceptual models and modeling tools, historical summaries of data used in the Phase 1 and 2 assessments, NPDES monitoring data collected since 2012, as well as data collected during 2015–2016 pursuant to the approved Phase 3 Study Workplan, which was developed in consultation with Ventura Coastkeeper and Heal the Bay. As identified in the data gaps analysis of the Phase 1 and 2 Studies, Phase 3 assessments integrate the effects of discharge, groundwater, and surface water quality data more directly into the analysis of all beneficial uses protective of SCRE ecology. In this regard, the Phase 3 assessments have relied upon conceptual modeling in combination with available data and quantitative models to distill ecosystem functioning of the SCRE into a manageable set of key processes with defined relationships. This process is reductionist, and as such, necessitates reliance on assumptions. Every effort has been made to ensure that the assumptions made herein are transparent, valid, and that they are supported by the best available science. For cases in which only limited data were available, assumptions were made based on best professional judgment. See Appendix A for a list of assumptions made and the information on which they were based.

As in Phases 1 and 2, because the SCRE water contributions and water quality conditions are dominated by river flow during wet weather and significant rainfall events (typically in the winter) when the estuary mouth is generally open. Closed-mouth dry weather conditions (typically in the spring and summer) are the critical period for assessing discharge effects on physical and chemical water quality, as well as the habitat extent and quality for listed native species and their habitats in the SCRE. The following quantitative factors are considered during the dry weather closed berm condition:

- Equilibrium stage and estuary volume estimates based on VWRF discharge using the calibrated water balance;
- Physical and chemical water quality conditions affecting beneficial uses for fish and wildlife, focusing on the four listed native species using the SCRE;
- Estimates of suitable habitat area for four focal species used to represent fish and wildlife in the SCRE;
- Potential for flooding impacts on recreational opportunities in the McGrath Beach State Park campground;
- Estimates of the amounts of aquatic, beach, fore dune, riparian, wetland and upland habitat types as a function of stage; and
- Comparisons of open-mouth conditions and breaching dynamics for representative water year types based on the calibrated water balance model.

For the purposes of determining Enhancement Levels, as well as for full assessment of the greatest diversions that might constitute the MEPDV, the effects of existing VWRF discharge are evaluated and compared to the effects of a "no-discharge" alternative, as well as all other alternative discharge scenarios based on both quantitative and qualitative assessments to determine whether they more fully realize each of nine (9) designated beneficial uses of the SCRE in the Basin Plan, as well as domestic and municipal water supply (MUN) uses that may potentially benefit from alternatives that include increased diversion for potable reuse. A framework based upon the Analytic Hierarchy Process (AHP) (Saaty 1980, 2008) is used to evaluate how VWRF discharge scenarios support each factor and attribute assigned to each beneficial uses, as well as a balance of all potentially competing beneficial uses. The AHP is a mathematical approach to multiple criteria decision making that has been applied to a range of applications in solving large and complex decision problems. The AHP uses an analytical hierarchy to group factors and attributes related to designated beneficial uses, pairwise comparison between factors and attributes using specific metrics to determine relative importance, and weighting of those factors and attributes to develop scores for that synthesize the overall realization of each beneficial use in isolation under alternative discharge scenarios. The framework provides a rational and repeatable approach to understanding whether alternative VWRF discharge scenarios enhance the SCRE. In addition to being used as the basis of a recommended finding on estuary enhancement, this assessment approach is also used to provide the recommended Continued Discharge Level, i.e., the effluent discharge flow rate needed to sustain the estuary's native species listed for protection, as required by the RWQCB and for use in permitting by other Resources Agencies, as well as the maximum amount of discharge that can be diverted from the estuary while protecting its ecology, and particularly its listed species and habitats).

1.5.1 Use of models

Based upon review of existing information (Stillwater Sciences 2011), it is apparent that the SCRE is a highly dynamic ecosystem that is characterized by the complex interactions of myriad abiotic, biotic, and anthropogenic processes. Large seasonal and inter-annual variations in precipitation and inflows to the SCRE are associated with the geomorphology of the SCRE, including lagoon bathymetry, berm position, berm breaching behavior, as well as variations in habitat conditions, and habitat use of the SCRE by the selected focal species. For this reason, assessing the effects of VWRF discharges upon beneficial uses of the SCRE is not easily separable from other factors that influence ecosystem functioning. Predicting the effects of changes in one of these processes (i.e., alternative VWRF management scenarios under predicted future climatic conditions) on the emergent properties of the system (i.e., beneficial uses) requires the use of models.

The primary modeling tool originally developed for the Phase 1 and 2 studies and updated during Phase 3 is a water balance model (Section 4) which has been updated to quantify water level (“stage”), volume, and wetted area as a function of inflow in various seasons. We have used the water balance in combination with other information and quantitative assessment tools to examine and create estimates for assessing the relative effects of each VWRF discharge scenario on habitat amounts and quality for focal listed species, as well as other species occupying the SCRE; risk of unseasonal breaching; the duration of open mouth conditions; nutrient and salinity dynamics; and assessment of temperature conditions in the SCRE, all in relation to the life stage timing and habitat needs of spawning, feeding, rearing, and migratory life stages of focal fish and bird species.

Several point and non-point discharge sources affect water quality in the lower Santa Clara River and SCRE, including VWRF inflows, agricultural and storm water runoff, groundwater inputs, as well as tidal exchanges and wave overwash. In addition to the dynamic nature of these inputs as well as spatial and temporal variations in biological as well as geochemical uptake and release of nutrients and contaminants to the SCRE, dynamic modeling of water quality conditions was not considered feasible for the Phase 3 Study. As in the Phases 1 and 2 studies, we rely upon a simplified mixing model approach to assess the relative contributions of these sources in combination with semi-quantitative and qualitative assessments of water quality suitability for focal and other species and habitats. To evaluate temperature within the lagoon, we have relied upon a simplified heat balance model (Section 4.4) to examine the relative influence of VWRF discharges to primary factors affecting water temperatures in the SCRE (e.g., evaporation, solar insolation, radiation, etc.).

1.5.2 Habitat availability

The effects of VWRF discharge upon habitat amounts and quality are predicted either quantitatively using modeling, or qualitatively based on general principles when quantitative analysis is not feasible. Recognizing that additional physical, chemical, and biological factors affect life history outcomes of the species considered here, the most direct linkages with discharge and other flows entering the SCRE are those related to water depth, areas of open water, and bare sands, as well as estimates of the amounts of hiding cover. To more broadly examine habitat suitability for the focal species selected for the Phase 3 Study, we rely upon long established ecosystem "niche theory" that assumes individual species are adapted to a range of environmental conditions (Grinnell 1917). In the past decades, a range of habitat suitability models have been used to predict the likelihood of occurrence and abundance using habitat attributes considered important for its survival, growth and reproduction of fish and wildlife species (Jowett, 1987; Orth 1987; Rushton et al 2004). For aquatic species, water quality parameters are essential habitat attributes determining habitat suitability. For both aquatic and bird species, habitat successional rules have been developed to predict the long-term effects of modeled equilibrium stage upon the establishment of vegetation and various habitat types. Relationships between SCRE stage, general habitat types, and estimates of suitable habitat area for the focal species are discussed in Sections 3.5 through 3.7. For aquatic species, habitat area estimates based on physical habitat attributes such as water depth and substrate are qualified on the basis of water quality data and suitability criteria established through extensive monitoring and review of scientific literature. For the bird and wildlife species, habitat area estimates and other metrics for assessing habitat quality, including water quality, are considered in the analysis..

1.6 Outreach and Consultation

In accordance with the Phase 3 Study Workplan, outreach and consultation was conducted through a series of workshops to present and discuss the scope, progress, status, and findings of the Phase 3 Studies, as well as to provide opportunities for comment on the underlying work products. Beginning with the November 19, 2014 project kickoff meeting, the City held two annual progress workshops on November 3, 2015 and November 17, 2016. Participants in these workshops included representatives from the RWQCB, Resources Agencies, environmental groups (NGOs), local water agencies, agricultural interests, City of Ventura rate payers, and other interested parties. Additional study progress meetings were held with Heal the Bay and the Ventura Coast Keeper/Wishtoyo Foundation during on February 15, 2017, August 31, 2017, and October 12, 2017. The purpose of these meetings was to provide input on study approaches, critical review of data quality, and review of study assumptions (Appendix A). Review comments were received on two in-progress Drafts of the Phase 3 Study distributed on May 16, 2017 and a second internal draft distributed in two parts on August 29, 2017 and September 15, 2017. Lastly, following publication of the complete Draft Phase 3 Study Report on November 1, 2017, three additional workshops were held on November 8, 2017, November 15, 2017, and December 20, 2017 to provide an opportunity for additional public comment.

In total, this Final Phase 3 Study Report integrates stakeholder input across six workshops and three review drafts over a three-year period and reflects a wide range of public and resource agency perspectives regarding enhancement of designated beneficial uses associated with existing VWRF discharges to the estuary as well as the potential benefits of reduced discharges.

1.6.1 Comments on the Draft Phase 3 Report

Comments on the November 1, 2017 Draft Phase 3 Study report were received both verbally from agency and NGO representatives, other individuals attending the November and December 2017 workshops, as well as in written form from the individuals listed below.

1. National Marine Fisheries Service (NMFS)
2. Wishtoyo Foundation, Ventura Coastkeeper, and Heal the Bay Technical Review Team (TRT)
3. California Department of Fish and Wildlife (CDFW)
4. California State Lands Commission (CSLC)

5. California State Water Resources Control Board (SWRCB)
6. United Water Conservation District (UWCD)
7. Dr. Eric Todd, City of Ventura resident
8. Surfrider Foundation

We summarize the primary issues raised in the comments below, with detailed responses provided as a separate memorandum to this Final Phase 3 Study Report. In addition to lists of workshop attendees, written comment responses will be presented in a tabular matrix format organized by author in the order they appear in the original letters from which they were extracted.

1.6.1.1 Hydrology / Water balance

Comments from workshop participants, including particularly the TRT, on the hydrologic data analysis center upon uncertainties in prediction of future conditions, uncertainties in the water balance modeling conducted, as well as representation of transient filling conditions following closure of the berm at the mouth of the SCRE. This final draft of the Report fully acknowledges comments regarding the variable hydrology of the Santa Clara River watershed discussed in the Report, including both historical hydrology as well as potential changes in future hydrology under climate change scenarios, making predictions regarding future lagoon stage, water surface elevations, and beach berm morphology uncertain. Morphologically, because differing berm positions and berm lengths affect actual as well as modeled seepage rates across the SCRE mouth berm, river flood scour events can be expected to alter those conditions in the future resulting in some changes in the amounts and distribution of habitat types presented in the report. Additionally, changing berm heights can be expected to affect breaching dynamics. The Report fully discloses these uncertainties and assumptions necessary to develop predictive relationships between VWRF discharge scenarios and habitat area and water quality conditions within the SCRE. Uncertainties in the water balance modeling may affect estimates of selected assessment metrics (e.g., depth, open water areas, distribution among habitat types, breaching thresholds, duration of open mouth conditions). However, since many of these uncertainties are well bounded and all the mechanisms represented are simple (i.e., monotonic) functions of discharge and stage, we have confidence in the relative changes in habitat conditions predicted based on VWRF discharge variations are valid, even though we expect modeled conditions (e.g., stage, breach occurrence, water quality) may not neccesarily match modeling results.

1.6.1.2 Water quality / Estuary Mixing Model

Workshop participants, including especially the TRT, highlighted uncertainties in the relative contributions of nutrients from surface water and groundwater sources, as well as unquantified nutrient uptake and removal rates, and nutrient inputs/sinks related to algal and bacterial communities associated with the surficial sediments of the SCRE. The Report has been updated to fully acknowledge all uncertainties related to mixing assumptions as well as sources of nutrient updtake in the nutrient balance predictions provided. For example, although no sediment nutrient analyses were conducted as part of the Phase 3 Studies due to the relatively high frequency of scour events, the report has been updated to provide estimates of nutrient removals and uptake due to benthic algae.

In addition, study assumptions regarding vertically and spatially uniform water quality conditions in open water portions of the SCRE were questioned based upon differences between the recorded surface and bottom water temperatures over several months at a site with a vertically paired instrument ("Sonde") deployment during 2015. However, a more detailed inspection of the underlying data indicates that apparent temperature differences between the (Sonde) instruments persist due to issues with the instrument settings even during the periods when they were returned to the climate-controlled laboratory for servicing. We have updated the report to include additional quality assurance discussion of these issues, and note that any future use of the temperature data should be reserved without a more detailed data validation. We are confident in the well mixed assumption for the SCRE because *in situ* vertical profiles of water temperature from recently calibrated instruments strongly support the well mixed assumption and do not support the apparent thermal stratification suggested in the comments.

Lastly, several comments addressed the potential influences of cooler groundwater inputs upon the availability of thermal refugia for steelhead and other aquatic species. Localized thermal refugia may be expected to develop in the SCRE based on depth, shading, and any potential influence from surface water-groundwater exchanges such as hyporheic flow. Nevertheless, available groundwater temperature data collected incidentally at the groundwater wells installed as part of the Phase 3 Study as well as relatively low groundwater exchanges modeled in the water balance does not suggest such influences.

1.6.1.3 Vegetation and Wetlands

Comments related to vegetation and wetlands highlighted the importance of riparian habitat as well as locally rare wetlands discussed in the draft Report, consideration of localized flood control functions of wetlands in the overall beneficial use assessment, as well as uncertainties regarding habitat succession modeling used to predict future habitat conditions under alternative VWRF discharges. In response to these comments, riparian habitat was more fully evaluated in terms of habitat, as well as wetland related functions that it can perform. As discussed in the Report, long-term variations in plant communities following flood events, water depth relationships of plant communities within the study area, as well as available local mapping of existing habitat conditions were all considered in the simplified habitat succession modeling used in the Report. Recognizing the predicted amounts of the habitats presented may differ from future empirical observations, we are confident in the relative changes in habitat predicted for each VWRF discharge scenario, such as reduced open water and accompanying increases in riparian habitats within the SCRE, are valid for purposes of the study.

1.6.1.4 Aquatic Habitat and Species

Comments related to aquatic habitats and species included a discussion of the relative importance of habitat area for the selected focal species (steelhead and tidewater goby) in comparison to other conditions, and particularly water quality conditions and conditions conducive to non-native species competition and predation, affecting these species. This Report acknowledges comments regarding the apparent over-simplification of the wide-ranging factors affecting aquatic species, its habitat, and the abundance and diversity of both. , The Report discloses factors that cannot be adequately assessed and/or that do not exhibit a sufficient relationship to discharges to vary when it comes to the support of SCRE species, suitability of habitats, or other factors contributing to beneficial uses. . Comments discussing the potential threats to aquatic species related to unauthorized (i.e., human induced) and unseasonal breaches confirmed the importance of this factor, and the Report directly considers the risk of breaching presented by each VWRF discharge scenario. Other comments discussed the stabilizing influence of VWRF flows upon water levels and reduced salinity conditions, which may favor non-native species that compete with or prey upon native species. This Report directly assesses and weighs effects of VWRF discharge on water levels and the duration of elevated salinity following breach events, as well as other factors affecting aquatic habitat and species.

1.6.1.5 Wildlife Habitat and Species

Comments regarding wildlife habitats and species highlighted nest monitoring work by California State Parks biologists have documented the loss of two western snowy plover nests and one American avocet nest due to rising SCRE waters (California State Parks. 2017 pers. comm.). Because shorebird nesting can also be threatened by bank erosion and washouts associated with berm breaching events, an explicit factor related to nest flooding has been included in this final Phase 3 Report.

1.6.1.6 Analytic Hierarchy Process

Several comments on the selected assessment approach incorporating the Analytic Hierarchy Process (AHP) by Saaty (1980, 2008) were related to known criticisms of multi-criteria decision-making approaches discussed in the Phase 3 Studies. Workshop comments discussed at the November and December 2017 Workshops identified additional factors and attributes contributing to realization of the SCRE beneficial uses that had not been included in the draft Report, and the need to provide a greater weight to certain factors and attributes. For example, comments included:

- Add an evaluation of factors related to risk of nest flooding within mudflat and margin habitats of the SCRE for focal bird species;
- Include habitat considerations for riparian associated bird species within RARE and WILD beneficial uses;
- Add an evaluation of localized flood control capacity factor under the "Wetland Habitat (WET)" beneficial use;
- Increase weighting for the "Migration of Aquatic Organisms (MIGR)" beneficial use;
- Increase weighting for factors related to unseasonal breaching under the RARE and SPWN beneficial uses.

Commenters also raised the need to disaggregate the AHP assessment for relative effects of VWRF discharge scenarios on RARE, in order to consider both: the factors important to each of 3 component attributes of RARE, i.e., rare aquatic species, rare wildlife species, and rare plant species, in isolation; as well as the relative overall importance of those 3 component attributes to RARE. Further, workshop comments suggested assessment of MUN uses to assure proper consideration of municipal water supply benefits resulting from increased VWRF diversions. All of these comments are addressed in this final Report by inclusion of an updated AHP weighting and scoring procedure addressing each of the foregoing areas.

Lastly, Workshop as well as written comments highlighted the need for an assessment of uncertainty or sensitivity to alternative AHP assumptions, as well as a description of the component results to provide more understanding of how individual factors and attributes of each beneficial use, and how each different beneficial uses was prioritized and weighed to derive the final scoring and ranking of the support provided to each of the beneficial uses, as well as an optimized balance of all beneficial uses by each alternative VWRF discharge scenario. This final Report includes expanded results discussing the relative influences of individual assessment metrics, beneficial factors and attributes, and the overall balance of beneficial uses, as well as a sensitivity analysis reflecting variability in discharge scenario ranking from individual TRT and Resources Agencies participants in an updated AHP weighting process.

1.6.1.7 Enhancement and MEPDV Recommendations

Several comments were received discussing study recommendations regarding Enhancement Discharge Levels providing fuller realization of beneficial uses as compared to the absence of discharge, as well as the maximum ecologically protective diversion volume (MEPDV). Based upon the uncertainties in the underlying modeling tools, in the amounts of habitat needed to sustain the native species of the SCRE, in the the effects of VWRF as well as other nutrient loads to the SCRE, in the effects of decreased duration of elevated salinity following breach events associated with greater discharge scenarios, as well as the metrics and factors selected for use in the AHP assessment approach, several commenters recommended a greater reduction in VWRF discharges than was recommended in the draft Report. While the additional recommended factors, attributes and beneficial uses incorporated into the and revised weighting incorporated into the AHP assessment were intended to, and do result in lower Enhancement Discharge Levels and Continued Discharge Level recommendations and an increased MEPDV recommendation, few if any of the comments provided an alternative or updated decisional framework recommendation to serve as the basis for such determinations. Although reduced VWRF discharge will be associated with some decreases in habitat area for aquatic species, the Enhancement Levels and Continued Discharge Level recommendations are expected to optimize beneficial uses overall, and to be protective of the SCRE's ecological resources, particularly its listed species and their habitats. Further, the MEPDV recommendation in this Report, including implementation of planned treatment wetlands for any continued discharges to the SCRE, is expected to improve water quality conditions for aquatic species, to increase the variability in salinity conditions that currently favors many non-native species, to reduce the potential for unseasonal breaching, and to reduce the potential for inundation of the McGrath State Beach campground (given the absence of the campground's protective berm), improve local water supply, all while protecting the ecology of the SCRE.

Lastly, several comments were received regarding the need for adaptive management in the future depending upon whether anticipated habitat quality or quantity conditions predicted in this report are realized in the future. This final Report has been updated to include adaptive management recommendations for consideration as part of future permitting actions by the Resources Agencies in their evaluation of the analysis, information and discharge recommendations herein.

1.6.2 Revisions included in the Final Phase 3 Study Report

Based upon comments received on the November 1, 2017 Draft Phase 3 Study report, this Final Phase 3 Study report responds to Stakeholder comments received during the extended comment period. In addition to minor editorial updates, udpates to study assumptions (Appendix A), udpates clarifying the Study Purpose (Section 1.2) in making future VWRF discharge recommendations (Section 5), the primary analysis updates made to the Draft report are intended to establish technically defensible linkages between discharge scenarios, SCRE water quality, habitat area, and the support of each scenario for beneficial uses, particularly in the following areas.

- **Continuous in situ water temperature analyses**. Based upon updated QA review of continuous water temperature monitoring data collected during instrument ("Sonde") deployment during portions of 2015, several discrepancies were identified between the continuously monitoring Sondes and instantaneous "spot" water temperature measurements collected at the Sondes during recalibration and servicing events (Section 2.2.3.2). Because discrepancies between spot and continuous measurements were corroborated by comparison of the Sonde measurements to one another when co-located in the laboratory (i.e., should have measured close to the same temperature), a decision was made to exclude these data from the monitoring results presented in this report (updated presentation of continuous in situ water temperature in Section 3.4.1.3 and Appendix D).

- **Mixing model estimates of equilibrium nutrient concentrations**. Based upon comments on the Draft Study report highlighting differences between measured and modeled predictions of SCRE nutrient concentrations assuming no biological removal discussed in Section 4.3, estimates of the nutrient concentrations for the 2015-2016 sampling period in Table 4-4 and supporting discussion have been updated to include realistic biological rates of removal within the ranges found in literature sources. We have retained the more conservative estimates of nutrient removal and uptake used in Section 5.3.2.2 to assess SCRE response to reduced VWRF discharge scenarios.

- **Habitat needs of threatened and endangered fish and wildlife used as focal species**. In order to identify the physical, chemical, and biological characteristics of the SCRE that are important for species behavior and long-term success, we reviewed the ESA critical habitat designations for federally listed fish and wildlife species occurring in the SCRE study area to evaluate the extent to which Primary Constituent Elements (PCEs)[7] of habitat are realized in the SCRE and to determine whether these factors might be affected by VWRF discharge (Sections 3.6.3, 63.6.4, and 3.7.2). Updated discussions of habitat needs of the selected focal species have also been included in Sections 5.5.7 and 5.6.2.

- **Updated beneficial use assessment using the Analytic Hierarchy Process (AHP)**. Based upon comments received on the Draft report, we have updated the beneficial use assessment that utilizes an adaptation of the AHP (Saaty, 1980, 2008) to include additional factors, attributed and beneficial uses, including factors and attributes related to risk of nest flooding within mudflat and margin habitats of the SCRE, inclusion of habitat considerations for riparian associated bird species within the RARE beneficial use (Section 5.5.7), addition of a flood control capacity factor under the WET beneficial use (Section 5.5.11), increased weighting for the MIGR beneficial use (Section 5.5.4), as well as increased weighting for factors related to unseasonal breaching under the EST, RARE, and SPWN (Sections 5.5.2, 5.5.7 and 5.5.10) beneficial uses. An updated AHP weighting and scoring procedure including the factors above has been included this Final report (Appendix H, Sections 5.1 and 5.5). In addition, in response to comments, an uncertainty evaluation including comparison of VWRF discharge scenario rankings under alternative weighting assumptions has been included in Appendix H and Section 5.6.

- **Recommendations regarding enhancement of existing beneficial uses under the EBE Policy**. Based upon updated beneficial uses, contributing factors and attributes, relative prioritization and weighting, and the scoring used in the beneficial use assessment (Appendix H, Section 5.5), alternative VWRF discharge scenarios are compared to provide updated recommendations regarding enhancement of beneficial uses under EBE Policy (Section 5.6). These updated recommendations also include refinements to the draft Report

[7] Referred to as "physical or biological features" (PBFs) established in updated Critical Habitat regulations (81 FR 7414); however, we retained the PCE since that was convention at the time of designation for the SCRE

recommendations related to the Continued Discharge Level and its converse, the MEPDV, that areas considered protective of the SCRE's native species, are required by the Consent Decree and the VWRF NPDES Permit (Order R4-2013-0174), respectively.

1.7 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 SCRE subwatershed conditions and an overview of known changes to SCRE physical and biological conditions over the past 150 years. This assessment integrated historical records, scientific literature, existing data, and data collected as part of the targeted monitoring conducted for the Phase 3 Study (see Section 2), providing the scientific grounding necessary to quantify the processes and conceptual linkages illustrated in Figure 1-2. On the basis of the geomorphic and hydrologic information synthesized, a water balance was developed to model the relative contributions of both measured and unmeasured water inputs to and outputs from the SCRE in order to estimate estuary stage, volume, and inundated area. These inputs and outputs can be modified to represent alternative discharge volume or climatic conditions. The water balance model was used in combination with water quality data and a nutrient balance model to assess water quality conditions under alternative discharge volume and/or climatic conditions. The synthesized ecosystem information is then used to evaluate the effects of alternative VWRF discharge scenarios on beneficial uses of the SCRE.

The following describes the structure of this report by major section heading:

- **Section 2: Phase 3 Data Collection**
    - Physical data
        - Describes physical data used in updating the water balance model, including, flow, surface water and groundwater elevations, topography and bathymetry data, as well as mouth breaching status.
    - Water Quality Data
        - Describes water quality data collected as part of Phase 3;
        - Includes surface water sampling of in situ conditions for DO, Temp, pH, Specific Conductivity, and Chlorophyll;
        - Includes surface water sampling results for nutrients, Chlorophyll-a, trace metals, constituents of emerging concern as well as toxicity testing; and
        - Includes groundwater sampling of nutrients.
    - Habitat Data:
        - Includes water quality, substrate, and cover conditions and relative fish species abundance during seine sampling conducted in 2015–2016;
        - Includes habitat mapping based upon CDFW habitat types within the SCRE subwatershed; and
        - Includes benthic macroinvertebrate (BMI) sampling within the SCRE during 2015.
- **Section 3: Physical and Biological Setting**
    - 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;
        - 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.
    - 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; and
        - 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.
    - Water Quality
        - Describes SCRE water quality conditions based on historical data (1997–2008) and data collected as part of this study (2009–2016).
    - Vegetation and Wetlands
        - Provides a description of contemporary and historical vegetation conditions including plant community composition, habitat types, and vegetation response to change or disturbance.
    - Aquatic Habitat and Species
        - Describes current and historical aquatic habitat conditions;
        - Describes current and historical data on the aquatic community, including fish and benthic macroinvertebrates;
        - Discusses species' status, distribution, life history information, and habitat requirements in the SCRE for the two focal fish species, steelhead and tidewater goby; and
        - Discusses inter-species interactions and their effects on native fish.
    - Wildlife Habitat and Species
        - Describes contemporary and historical wildlife habitat conditions; 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.
- **Section 4: Water Balance Evaluation**
    - Presents the current estuary water balance 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 to the SCRE.
    - Develops an estuary mixing model (Section 4.3) to predict nutrient (N, P) and salinity (conductivity) as well as transient responses of SCRE salinity following breaching.
    - Develops simplified heat balance model (Section 4.4) to examine the relative influence of VWRF discharges to primary factors affecting water temperatures in the SCRE (e.g., evaporation, solar insolation, radiation, etc.).

- **Section 5: Beneficial Use Assessment**
  - Uses information the water balance model and information developed in Section 2.3.1 to evaluate the extent to which each of the eleven beneficial uses of the SCRE is realized under each alternative VWRF management scenario;
  - Compares the results to determine whether VWRF discharge enhances beneficial uses of the SCRE relative to the absence of all VWRF discharge;
  - Compares the results to determine which volume of VWRF discharge to the estuary results in the fullest realization of designated beneficial uses; and
  - Identifies the maximum diversion volume at which fish- and wildlife-related beneficial uses are not adversely affected.

2 PHASE 3 DATA COLLECTION AND QA/QC

The Phase 3 Monitoring Plan (2015), developed pursuant to the approved Workplan, included targeted data collection to address the data gaps described above and to provide up-to-date information on which to base the beneficial use assessment (Section 5). The Phase 3 data collection and monitoring efforts provide physical, water quality, and habitat data to support the three primary study components on which the beneficial use assessments are based: the water balance evaluation, water quality evaluation, and habitat suitability evaluation. The following sections describe the Phase 3 monitoring conducted for each these components (see Section 3 for data reporting, analysis, and synthesis of data from other sources, including prior monitoring efforts). Figure 2-1 shows the Phase 3 monitoring locations.

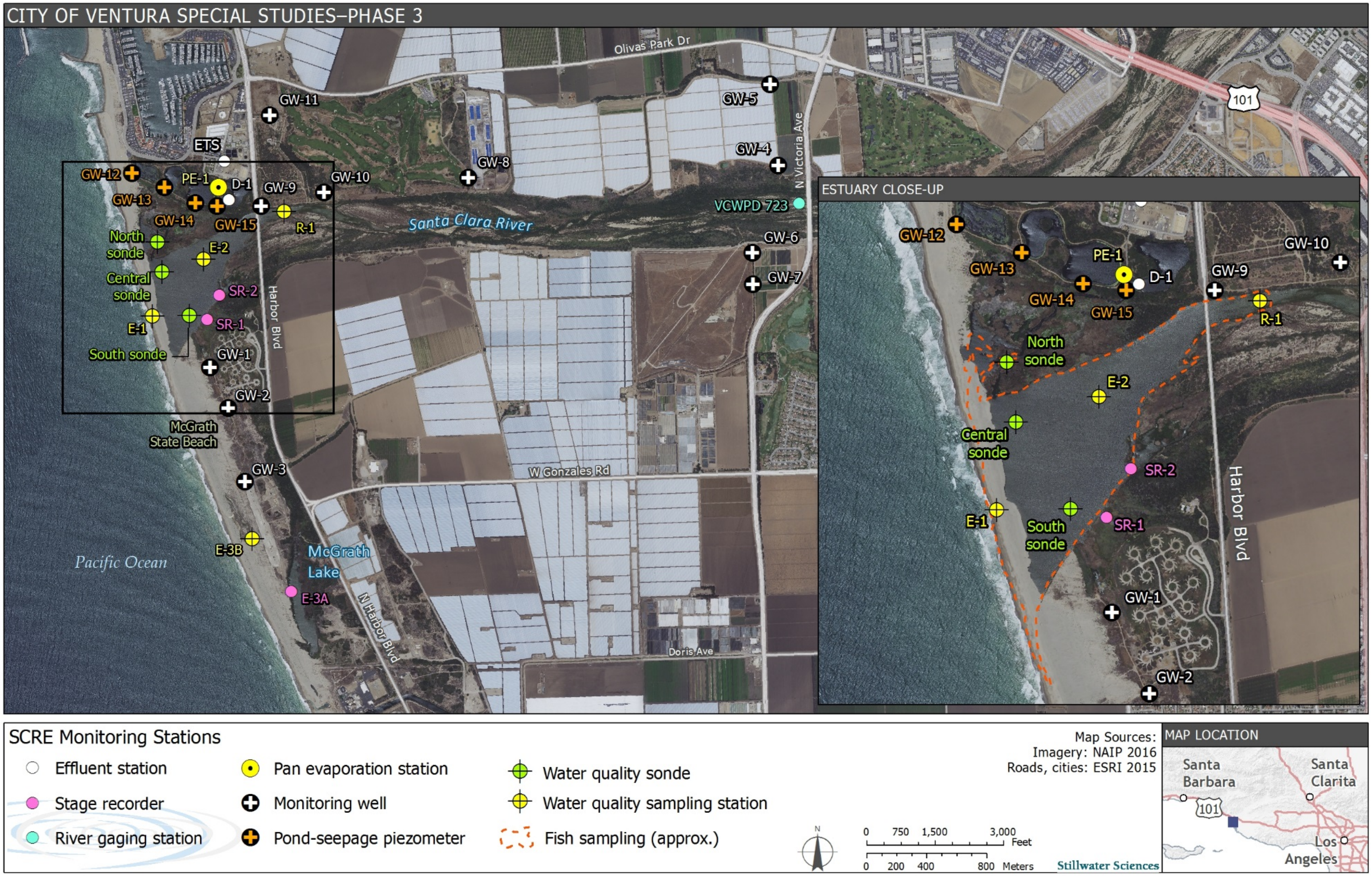


Figure 2-1. Locations of monitoring stations around the SCRE for the Phase 3 Special Studies.

2.1 Physical Data

The Phase 3 water balance evaluation included collection of additional hydrologic and geomorphic data to better characterize water balance model parameters. Table 2-1 shows monitoring locations, frequency, and duration for the Phase 3 data collection. Data collection included continued operation of existing monitoring sites from Phases 1 and 2, updated data from other sources, as well as the addition of new monitoring sites (Table 2-1). Locations of the new monitoring stations (GW-8 through GW-15, and E-3) were intended to improve estimates of groundwater and surface water contributions to the SCRE water balance. Field data was collected from January 2015 through December 2016. The timing of some monitoring was dependent on the formation and seasonal timing of breach events of the SCRE mouth berm along McGrath State Beach. The following sections summarize data collections methods and quality assurance procedures for each of the physical data parameters.

Table 2-1. Summary of physical data collected for the Phase 3 water balance evaluation.

Water balance component	Update data from other source(s)	Existing monitoring station(s)	New monitoring station(s)	Phase 3 monitoring location	Parameter	Frequency	Duration
Precipitation	✓			VCWPD Stn 66E	(+) Volume	Event (open/closed)	2 water years
Tidal Elevation	✓			NOAA Stn 9411340	(+/-) Water Level	Continuous	
Evaporation	✓	✓		VCWPD Stn 239, PE-1	(-) Volume	Daily	
SCR Surface Water Inflow	✓			VCWPD Stn 723	(+) Flow	Continuous	
SCRE Stage		✓		SR-1, SR-2	(+/-) Water Level	Continuous	
VWRF Effluent	✓			ETS, D-1	(+) Flow	Continuous	
VWRF Pond Seepage		✓		Calculated (Pond stage, D-1)	(+) Flow	Continuous	
			✓	GW-12, GW-13, GW-14, GW-15	(+/-) Water Level	Continuous	
Berm Seepage (Hyd. properties unchanged from original study)	✓			Calculated (SCRE stage, NOAA 941130)	(+/-) Flow	N/A	N/A

Water balance component	Update data from other source(s)	Existing monitoring station(s)	New monitoring station(s)	Phase 3 monitoring location	Parameter	Frequency	Duration
South Side Groundwater Stage		✓		GW-1, GW-2, GW-3, GW-6, GW-7	(+/-) Water Level	Continuous	
North Side Groundwater Stage		✓		GW-4, GW-5	(+/-) Water Level	Continuous	
			✓	GW-8, GW-9, GW-10, GW-11			
McGrath Lake Inflow			✓	E-3	(+/-) Water Level, Pumping Activity	Continuous	2 water years
Berm Status		✓		City of Ventura Monitoring	Open/ Closed	Daily/Event	
Berm Height			✓	City of Ventura Monitoring [1]	(+/-) Elevation	Quarterly/ Event (open/closed)	
Lagoon Morphology			✓	2000 and 2012/2014 SCRE Bathymetry [2]	(+/-) Volume	Event/As Available	N/A

[1] Berm-height fluctuations affect SCRE capacity and breach dynamics and, therefore, require monitoring to update estuary water balance.

[2] Lagoon morphology adjustments affect SCRE capacity and, therefore, require consideration in the update to the estuary water balance. The original water balance utilized a 2005/2009 bathymetric surface. We have since compiled surfaces for 2000 and 2012/2014 that can be readily applied to the analysis. (Sources: McGrath State Park, CSU Channel Islands, and cbec, Inc.).

2.1.1 Pressure logger data

2.1.1.1 Atmospheric pressure

Atmospheric pressure levels near the SCRE were determined using a pressure transducer (Solinst Barologger) installed within the vented, PVC-casing used as part of the stage-recorder station SR-1. The purpose of this data was to compensate the water-level readings made at the monitoring stations used to monitor SCRE stage (SR-1 and SR-2), pan evaporation (PE-1), groundwater levels (GW-1 through GW-15), and McGrath Lake stage (E-3A). The installed pressure transducer recorded pressure head (i.e., water equivalent in feet) and air temperatures at a 30-minute time interval between January 2015 and December 2016. Data were downloaded quarterly during the monitoring period. Each instance the data were downloaded, the equipment were assessed for any natural and/or man-made disturbances. The downloaded data were also assessed for irregularities. The data were combined with data recorded since 2009 as part of the Phases 1 and 2 Studies. Erroneous or missing readings were identified based on deviations from daily mean trends, and then removed and noted in the data file. The equipment appeared to function properly throughout the duration of the monitoring period. The data file also presents a comparison to air pressure recorded nearby at the Oxnard Airport (NOAA NCDC GHCND: USW00093110). The data sets were found to compare well to one another, offset consistently by only the difference in altitude of each station.

2.1.1.2 McGrath Lake levels

Potential surface water inflow/outflow between the SCRE and McGrath Lake was evaluated by establishing a new monitoring station (E-3) near the point of outflow from McGrath Lake. This section describes the monitoring procedure conducted in accordance with the Monitoring Plan. However, note that visual inspections conducted quarterly showed no evidence of any surface water connection between the SCRE and McGrath Lake and the data collected by the procedures described below were not used in the Phase 3 Study analysis. Water-level stage was recorded continuously in the lake (E-3A) and pumping activity in the outfall pipes (E-3B)—both locations situated on State Park property. Stage was recorded in the lake (E-3A) with a pressure transducer (Solinst Levelogger) placed in a protective PVC housing. The outfall pond station (E-3B) was installed in each of the three outfall pipes leading into the linear pond that receives pumped water from the lake. Continuously recording water-temperature loggers (HOBO DataLogger) were installed in the pipes near their openings. The installed pressure transducer and water-temperature loggers recorded data at a 30-minute time interval between January 2015 and December 2016. Data were downloaded quarterly during the monitoring period. At each instance the data were downloaded, the equipment was inspected for any damage or natural and/or man-made disturbances. The downloaded data were also assessed for irregularities. Erroneous or missing readings were identified based on deviations from daily mean trends, and then removed and noted in the data files. All equipment appeared to function properly throughout the duration of the monitoring period. The elevation of the stage recording station (E-3A) was determined from a topographic survey conducted in December 2016 using RTK-GPS surveying equipment.

As described above, per the requirements of the Workplan, surface water output from McGrath Lake was visually inspected during quarterly data downloads, and any evidence of a surface water connection with the SCRE was recorded. These visual inspections yielded no evidence of a surface water connection between McGrath Lake and the SCRE. Accordingly, surface water flows from McGrath Lake are assumed not to influence conditions in the SCRE and the data related to McGrath Lake surface water described above were not used in the Phase 3 Study analysis.

2.1.1.3 Groundwater levels

Existing and new groundwater monitoring wells were utilized during the Phase 3 Study to monitor groundwater levels in the vicinity of the SCRE (GW-1 through GW-15; Table 2-1). Water levels were continuously recorded in each of the wells with a pressure transducer (Solinst Levelogger). The locations and elevations of the wells were surveyed at the start and end of the Phase 3 Study. The installed pressure transducers in GW-1 through GW-15 recorded groundwater levels at a 30-minute time interval between January 2015 and December 2016. Data were downloaded quarterly during the monitoring period. Each instance the data were downloaded, the wells and associated equipment were assessed for any natural and/or man-made disturbances. The downloaded data were also assessed for irregularities. The data were combined with data recorded since 2009 as part of the Phases 1 and 2 Studies, where applicable. Erroneous or missing readings were identified based on deviations from daily mean trends, and then removed and noted in the data files. All equipment appeared to function properly throughout the duration of the monitoring period. Readings in well GW-7 (at the Bailard Landfill) often indicated dry conditions, which were verified in the field during the quarterly download events. The wellhead and ground elevations at each of the wells were surveyed in December 2016 using RTK-GPS surveying equipment.

2.1.1.4 Evaporation rates

A pan-evaporation station was established by the City near the VWRF Wildlife Ponds (near groundwater-monitoring well GW-15) to provide estimates of local evaporation rates. The station (PE-1) consisted of a Class A evaporation pan (10 in [25 cm] deep, 42.7 inches [120.7 cm] in diameter) constructed of stainless steel installed in an open space near the Pond outlet Parshall flume gaging station (D-1) with easy access and away from adjacent, large, overhanging vegetation and local heat sinks (e.g., large paved areas). Operation of the pan entailed filling it daily (i.e., once every 24-hours on a fixed schedule) with a known water level near the rim and continuously monitoring the water level with a pressure transducer (Solinst Levelogger) in protective PVC housing anchored in the center of the pan. The pressure transducer recorded daily changes in water depth from evaporation (and precipitation when present) at 30-minute time intervals from January 2015 through December 2016. Data recorded between January 2015 and December 2016 was downloaded quarterly and subsequently compiled in a single spreadsheet. Daily evaporation rates using the continuous water-level data entailed computation of water loss in the pan between discrete moments of pan re-filling less any contribution from rainfall. A standard “pan factor” with a coefficient of 0.7 was applied to all data, per advisement of the pan manufacturer. The evaporation results were evaluated by comparing to evaporation rates recorded at two local atmospheric stations—VCWPD’s station #239 at the El Rio-United Water Spreading Grounds near Saticoy and California Irrigation Management Information System’s (CIMIS’s) station #156 in Oxnard—between January 2015 and September 2016. However, some problems arose with the maintenance of water quantity and quality at evaporation station PE-1, leading to unreliable data: QA/QC procedures identified some significant deviations from the seasonal trends observed in data from the other two stations (i.e., El Rio VCWPD station #239 and CIMIS station #156) (see Section 3.3.2). Data from the El-Rio station (VCWPD station #239) were used in the Phase 3 analysis.

2.1.1.5 SCRE stage

The water-level monitoring stations (SR-1 and SR-2) previously established in the SCRE during the Phases 1 and 2 Studies were re-established during the Phase 3 Study to record water-surface levels in the SCRE at a 30-minute time interval from January 2015 through December 2016. Water level data was recorded with a pressure transducer (Solinst Levelogger) secured within protective PVC housing anchored to the bed of the SCRE. The locations and elevations of the stations were surveyed at the start and end of the Phase 3 Study. The data were downloaded quarterly during the monitoring period. At each instance that stage data were downloaded, the equipment were assessed for proper functioning and elevational shifts. The downloaded data were also assessed for irregularities. Data recorded between January 2015 and December 2016 were compiled in a single spreadsheet. The spreadsheet also contained data recorded since 2009 as part of the Phases 1 and 2 Studies. Erroneous or missing readings were identified based on deviations from daily mean trends, and then removed and noted in the data file.

Initial QA/QC reviews indicated that readings from SR-1 were progressively incorrect since about September 2016 due to instrument error. Water-level readings were found to be about 2-feet too deep compared to manual measurements recorded in December 2016 with this difference initially attributed to fine sediment accumulation in the well screen. Because two stage recoding locations were included in the Workplan to provide a level of redundancy during the Phase 3 Study in the event that either station was disturbed by natural or man-made causes, the erroneous readings made by the pressure transducer installed at the SR-1 station between September and December 2016 were not utilized in the water-balance model (Section 4) and associated analyses. The modeling and analyses will largely rely on the SR-1 readings through August 2016 and then the SR-2 readings between September and December 2016.

The elevations of the stage recording stations (SR-1 and SR-2) were determined from a topographic survey conducted in December 2016 using RTK-GPS surveying equipment. The data were compiled in a single spreadsheet.

2.1.2 Berm status

The status of the beach berm separating the SCRE from the Pacific Ocean continued to be monitored daily by the City for the duration of the Phase 3 Study. Hand-drawn sketches of the SCRE berm were reviewed for information on inundation extent, mouth status and position, and discharge passing through the mouth (inward or outward). These data were compiled in a single spreadsheet. The spreadsheet also contained similar data recorded by the City since 1984. Erroneous or missing data were identified based on deviations from longer-term trends, and then either removed or noted in the data file.

Several manmade breaches by unknown parties were identified during 2015 and 2016, affecting the ability to establish breaching threshold elevations in the water balance model (Section 4). In order to estimate the changes in breaching elevation as well as monitor the evolution of the beach berm’s position and contemporary morphology, the topography was surveyed in December 2016 using RTK-GPS surveying equipment between the north and south ends of the lagoon. The data were compiled in a single spreadsheet.

2.1.3 Precipitation

The volume of direct precipitation onto the SCRE was derived from rainfall amounts recorded hourly in downtown Ventura at Ventura County Watershed Protection District’s (VCWPD) rainfall gage (Station 66E). Data reported between water years 2009 through 2016 (i.e., October 1, 2008 through September 30, 2016) were compiled in a single spreadsheet. This period of record spans the entirety of the Phase 1, 2, and 3 Studies. The data were downloaded and reviewed periodically during the Phase 3 Study duration.

2.1.4 Surface flows from river channel

The volume of surface water flowing into the SCRE from the river channel was determined primarily from daily mean discharge estimates approximately 1.5 mi (2.5 km) upstream of the SCRE eastern boundary at the Victoria Ave. bridge (Ventura County Watershed Protection District [VCWPD] Station 723). Available data collected at the station, including the former USGS gaging station near Montalvo (USGS 11114000), through September 30, 2015 were compiled. No information regarding the source(s) of these flows was collected.

Because flow data recorded at the Victoria Ave. station since September 30, 2015 was not available from VCWPD in time for use in the Phase 3 Study water-balance modeling and associated analyses, data from UWCD detailing the flow immediately below the Freeman Diversion were used for water year 2016 (October 2015–September 2016). Percolation of 50–150 cfs is commonly measured in the reach of the Santa Clara River that overlies the Oxnard Forebay (the 4.5 miles of broad sandy river channel and floodplain between approximately Ellsworth Barranca and the Hwy. 101 bridge). Percolation may be greater than 150 cfs at flows greater than 500 cfs, but higher flows such as these are not commonly measured (D. Detmer, pers. comm., 2017). To provide an order of magnitude estimate of summertime base flow discharge in the river upstream of the SCRE, surface flow measurements were collected on July 13, 2017. Flow ranged from 0.1 to 0.4 cfs in the reach immediately upstream of the SCRE with isolated pools found in the channel immediately downstream of Victoria Ave.

2.1.5 Tidal elevation

Tide elevation data were obtained from routinely downloading continuously recorded tide information at the NOAA station at Santa Barbara, CA (NOAA Station 9411340). Data reported between water years 2009 through 2016 (i.e., October 1, 2008 through September 30, 2016) were compiled in a single spreadsheet file. This period of record spans the entirety of the Phase 1, 2, and 3 Studies.

2.1.6 VWRF discharge

The VWRF Wildlife/Polishing Ponds outfall channel Parshall flume continued to be monitored by flow gaging station D-1 previously installed by the City in September 2010 (NPDES station M-001A). The gage is located approximately 75 feet from the discharge point into the SCRE outfall channel. The City continued to oversee recording of data, transferring of data, and equipment maintenance. The daily mean data recorded at the Pond outlet Parshall flume gaging station (D-1) and the VWRF effluent data recorded at the Effluent Transfer Station (“ETS”; site of VWRF discharge into the Wildlife/Polishing Ponds; “M-001” in NPDES permit R4-2013-0174) through December 2016 were compiled into a single spreadsheet. This period of record spans the entirety of the Phase 1, 2, and 3 Studies. Erroneous or missing readings were identified based on deviations from longer-term trends, and then removed and noted in the data files.

2.2 Water Quality Data

The City initiated water quality data collection for the Phase 3 Study in late January. The overall goal of data collection for the Phase 3 water quality evaluation was to improve characterization of SCRE water quality including spatial variation in SCRE surface water and groundwater quality, as well as the quality of other inputs/outputs to the SCRE. To address data gaps identified at the conclusion of the Phases 1 and 2 studies, Table 2-2 shows monitoring locations and parameters for the Phase 3 water quality evaluation. The following water quality data types are provided as an accompanying electronic data appendix (Appendix B):

- Analytical Water Quality Laboratory Testing Results
- Toxicity Testing Results
- Continuous In Situ Water Quality Data

Analytical testing of nutrient contributions of groundwater and subsurface inputs (including seepage from the wildlife/polishing ponds) was used to assess nutrient conditions, including the SCRE through the installation and monitoring of eight additional groundwater wells (GW-8–GW-15). Spatial and temporal characterization of water quality in the SCRE was addressed by deploying an additional sonde (central sonde) for continuous monitoring of temperature, DO, pH, and conductivity. An additional monitoring location was established to characterize the contribution of potential McGrath Lake surface input to SCRE water quality (E-3) since in previous years a narrow arm of the SCRE had extended southward and appeared to connect with the outfall channel from McGrath Lake (Stillwater Sciences 2016). A surface-water connection between the two water bodies was never observed during any of the quarterly observation events in this study and no surface water samples were collected for laboratory analysis. Finally, additional water quality constituents, including CECs were analyzed from existing monitoring locations to better characterize chemical water quality in the SCRE.

Table 2-2. Summary of data collected for the Phase 3 water quality evaluation.

Water quality source	Update data from other source(s)	Existing monitoring station(s)	New monitoring station(s)	Monitoring location[1]	Parameter	Frequency over 2 water years
SCR Inflow		✓		R-1 (R-005)	Temp, DO, pH, EC, Algae, NH_4, TKN, NO_3, PO_4, TP	Monthly/Event (open/closed)
					Trace Metals, Toxicity	Quarterly/Event (open/closed)
					CECs	Annually
VWRF Effluent		✓		ETS (M-001), D-1 (M-001A)	NH_4, TKN, NO_3, PO_4, TP	Monthly/Event (open/closed)
					Trace Metals, Toxicity	Quarterly/Event (open/closed)
					CECs	Annually
VWRF Pond Seepage			✓	GW-12, GW-13, GW-14, GW-15	NH_4, TKN, NO_3, PO_4, TP	Quarterly
South Side Groundwater Inflow/Outflow		✓		GW-1, GW-2, GW-3, GW-6, GW-7	NH_4, TKN, NO_3, PO_4, TP	Quarterly
North Side Groundwater Inflow		✓		GW-4, GW-5	NH_4, TKN, NO_3, PO_4, TP	Quarterly
			✓	GW-8, GW-9, GW-10, GW-11		
McGrath Lake Inflow			✓	E-3	NH_4, TKN, NO_3, PO_4, TP	Monthly/Event
					Trace Metals, Toxicity	Quarterly/Event (open/closed)

Water quality source	Update data from other source(s)	Existing monitoring station(s)	New monitoring station(s)	Monitoring location[1]	Parameter	Frequency over 2 water years
SCRE		✓	✓	near-bottom at North Sonde (R-004), Central Sonde, and South Sonde (R-002) locations,	Temp, DO, pH/ORP, EC	Continuous/ Monthly
		✓		Near surface at South Sonde (R-002) location	Temp, DO, pH, EC, Chlorophyll-a	Continuous
		✓		E-1 (R-003), E-2	Temp, DO, pH, EC, Algae, NH_4, TKN, NO_3, PO_4, TP	Monthly/Event (open/closed)
					Trace Metals, Toxicity	Quarterly/Event (open/closed)
					CECs	Annually/Event (closed)

[1] Monitoring locations sampled under the routine receiving water monitoring for the 2013 NPDES Permit shown in parentheses.

2.2.1 Analytical water quality laboratory data

Data from analytical testing of SCRE receiving water and VWRF discharge water quality samples collected 2012–2016 as part of the VWRF's required NPDES reporting as well as the 2015–2016 as part of the Phase 3 Study data collection efforts were used for the Phase 3 Study pursuant to the approved Workplan. QC review of laboratory testing data centered on the accuracy and precision of the reported results. Accuracy is an expression of the degree to which a measured or computed value represents the true value and is assured by adhere nce to standard sample collection procedures, laboratory analysis of "spiked" samples with known standards (surrogates, laboratory control samples, and/or matrix spike) and measurement of percent recovery. Precision measures the reproducibility of measurements under a given set of conditions and is generally evaluated using field and laboratory duplicates against quantitative relative percent difference (RPD) performance criteria.

For the nutrient, Chlorophyll-a, and metals data provided, QA/QC issues during 2015 included the following:

1. Metals analysis during February 2015 was conducted on unfiltered samples only and so dissolved metals were not reported.
2. Metals analysis for samples collected during May 2015 were analyzed using an incorrect method (EPA 200.7). Additional samples were collected in June and analyzed under EPA Method 200.8 in accordance with the Workplan.

3. Metals analysis for the ETS (NPDES M-001) location in August 2015 was conducted on unfiltered samples only and so dissolved metals were not reported.
4. With the exception of the July 2015 sampling event, field duplicates were inadvertently omitted from sample collections during other sampling events during 2015.
5. For the July 2015 sampling event, Phosphate was inadvertently omitted on the analysis request (COC) for sites D-1 (NPDES M-001A), R-1, E-1 (NPDES R-003) and E-2 and only total phosphorus (TP) was reported.

Samples with analytical concentrations that were below the method detection limit (MDL) were assumed to equal the MDL for the purposes of the water quality evaluation (Section 3.4.1.5). In addition to the nutrient, Chlorophyll-a, and metals data collected in accordance with the Workplan, laboratory reports for NPDES required annual Contaminants of Emerging Concern (CEC) monitoring (2015, 2016) from the ETS location were evaluated. It should be noted that there are instances in which daily sampling results used for the Phase 3 Study are in excess of the monthly average results reported under the NPDES permit.

2.2.2 Toxicity testing

Pursuant to the Workplan, toxicity testing of the VWRF effluent (ETS or NPDES M-001), Pond outlet Parshall flume gaging station (D-1 or NPDES M-001A), upstream control location (NPDES R-005), and VWRF receiving waters in the SCRE (E-1 [NPDES R-003], E-2) were performed as part of the Phase 3 Study data collection, in addition to the acute and chronic toxicity testing conducted pursuant to requirements in the NPDES Permit.

Aquatic Bioassay & Consulting Laboratories (ABC Labs) is a California ELAP certified laboratory (#1907), which follows rigorous QA/QC procedures including annual inspections with review of testing procedures, internal QA/QC results, and reports. Each year, ABC Labs participates in a nationwide DMR survey for each test species, including analysis of EPA-provided control samples and verification that results are within a range of the mean of other certified laboratories nationwide. Separate from certification requirements, each toxicity test must pass test acceptability criteria established in the EPA protocols, including testing organisms using laboratory controls, ensuring the variability in the control is low, and ensuring control water quality meets certain criteria (temperature, pH, etc.). Lastly, each toxicity test or batch of tests are accompanied by an associated Reference Toxicant test to ensure the organisms being used perform to a standard of past lab performance. A new test must fall within ±2 SD of the mean of the past 20 reference toxicant tests for an organism. No QA/QC issues affecting the usability of the toxicity testing results were identified.

2.2.3 Continuous *in situ* water quality data

Continuous water monitoring was conducted by ABC Labs at three locations since January 2015 using 3–4 unattended multi-parameter sondes (YSI EXO2, Yellow Springs Instruments, Yellow Springs, OH). At the South sonde location, a second sonde was deployed nearer the water surface and fitted with a Chlorophyll-a sensor to allow assessment of algal bloom conditions that may occur. Sondes were removed from the SCRE in anticipation of high flows from storm events as well as during extended periods of open-mouth status to prevent the damage or vandalism. To address the resulting data gaps additional spot checks and in situ profiles were collected during daylight and bottom values were presented with the continuous data (see Appendix B for profile data). The following general procedures were followed to ensure representative water quality data were used in subsequent analyses.

At the initial deployment, all sondes were pre-calibrated in the laboratory, with periodic spot checks of sonde calibration using an independently calibrated sonde. Accuracy of each unit was verified by instrument calibrations at standard conditions (e.g., oxygen in fully saturated air) and the use of standard solutions (e.g., pH, ORP, conductivity, or turbidity standards). Laboratory calibration logs were maintained for all units and no units were deployed that did not pass pre-deployment calibration checks. Using a recently calibrated unit, field spot checks were made to validate the post-deployment calibration of the operating sondes.

Upon return to the laboratory and prior to analysis of data collected from the sondes, data were downloaded into MS Excel. Data quality reviews included identification of periods when the sondes were deployed in the SCRE as well as identification of results that were unexpectedly higher or lower than spot check samples, as well as identification of other anomalies related to sensor fouling or other issues. Data records varied slightly by sampling site and deployment over the sampling period, with the primary difference being the number/type of variables collected at the South (surface) sonde versus other (bottom) sondes. Below we discuss QA/QC reviews and data exclusions applied.

2.2.3.1 General data exclusions based upon deployment status and site conditions

The primary data exclusion was applied to time periods when the sondes were not physically deployed in the SCRE. In other words, records were excluded for dates and times prior to when the sondes were deployed at the sampling site as well as records from periods after the sonde was retrieved from the site. These records can be roughly determined by comparing the deployment/retrieval times that were provided (noting that sonde records are standardized at 15-minute recording intervals) and by examining the pressure variable and depth variable which provides an indication when the sonde is positioned in the water column. Both of these methods were used to determine which records to exclude.

Although post deployment spot checks were used as the basis of additional data exclusions below, only data from predominantly open-mouth conditions in Fall 2016 were eventually excluded from the data record for the following reasons:

1. Subsequent to the initial deployment of the sondes in late January 2015, it was noted that high DO readings during February may be associated with sonde membrane cleaning techniques being employed. Increases to sonde wiper frequency as well as membrane cleaning appeared to address these issues. Data exclusions on the basis of spot checks are discussed in the next section.
2. Subsequent to the sonde retrieval on 10/29/2015, it was noted that the wiper malfunctioned at the Central Sonde on 10/19/2015 at 17:45. No other data concerns were detected and the wiper was functioning during calibration. The sonde was sent to YSI and the sonde from the South (bottom) site was used to replace it at the Central sampling site, resulting in a short data gap at the South (bottom) sampling site. No data records were excluded.
3. All sondes were removed from the estuary intermittently for brief periods during January to March 2016 in anticipation of high flow events associated with predictions of large runoff events. Sondes were also removed between 03/22/2016 and 06/14/2016 due open berm conditions that resulted in low water levels and a concern of low submergence as well as potential vandalism.

4. Due to low water level resulting due to a breached berm condition, the North (bottom) and South (surface) sondes were re-deployed from 06/15/2016 to 07/21/2016 in a horizontal position. Following increases in depth, sondes were redeployed in a vertical position on 06/23/2016 the sondes were moved to a vertical position. Although some anomalous readings in the depth and pressure readings from the North Sonde were identified, no other data appeared to be impacted and no data records were excluded.
5. All four sondes were re-deployed from 07/21/2016 to 08/15/2016 and then removed due to low water levels until 09/21/2016.
6. Beginning on the 09/21/2016, extended open mouth conditions at the berm mouth persisted and it was determined that sonde submergence could not be maintained the original vertical deployment. Although an attempt was made to collect shallow water data using a horizontal deployment, following the breach in the estuary berm on 10/5/2016, the data showed a combination of data drift and extended periods of zero DO with no diel variation, indicating fouling by sediment. Accordingly, a decision was made to exclude all data collected during the horizontal deployment (the 09/21/2016 to 11/09/2016). This represents the only data exclusion from the record unrelated to deployment dates.

2.2.3.2 Data exclusions following QA/QC review

Water temperature data from several periods during summer/fall 2015 were flagged for exclusion in the QA/QC review because temperature sensor drift was identified by comparing water temperature data from multiple sondes and *in situ* instantaneous water temperature measurements. During approximately monthly calibrations of the DO, specific conductivity, and pH sensors, the sondes were removed from the water and kept together under the same environmental conditions. Temperature sensors were not calibrated during these periods since the temperature sensor is only designed to be calibrated by the manufacturer. The temperature recorded by the sondes during the calibration period provide QA/QC data because the sondes are kept together under the same environmental conditions and should record the same temperature within the accuracy of the sensor. During most calibration periods, the temperature data recorded by the sondes agreed, but there were several periods between May and November 2015 when the one or more sondes disagreed when they were out of the SCRE and stored under the same environmental conditions.

The qualification of water temperature data was undertaken in two steps: 1) evaluate the temperature difference between sondes when kept together under the same environmental conditions and 2) compare the sonde water temperature recorded during deployment in the SCRE with *in situ* instantaneous SCRE vertical profile water temperature measurements at each sonde location measured at the same time the sonde was recording. In the first step, the water temperature data recorded by sondes was flagged as having apparent temperature sensor drift if the temperature recorded by the sondes deviated by more than 1^{o}C when kept under the same environmental conditions. In the second step, the water temperature data recorded by a sonde was compared with an instantaneous water temperature measurement made by a more recently calibrated instrument at the sonde location. If the difference between the recorded sonde water temperature and the instantaneous water temperature measurement at the depth of the sonde was greater than 1^{o}C, sonde data was flagged for exclusion due to temperature sensor drift/poor temperature sensor performance in the sonde. The period of water temperature data exclusion extended from the last time it agreed with other sondes during a calibration period within 1^{o}C or the last time it agreed with an instantaneous water temperature measurement within 1^{o}C until it again agreed with other sondes during a calibration period within 1^{o}C or agreed with an instantaneous water temperature measurement at that sonde location within 1^{o}C. While the sonde temperature sensor drift probably was not more than 1ºC during this entire period, there was no objective way to establish exactly when sonde temperature sensor drift exceeded 1ºC between the calibration events and instantaneous water temperature measurements given the available temperature data. Table 2-3 details the periods that were excluded from analysis of the temperature data record based on the QA/QC review. While there were temperature data exclusion periods for the Central, North, and South Bottom Sondes, the South Surface Sonde data passed all QA/QC steps so no temperature data was excluded from that location.

Table 2-3. Temperature data exclusions from the continuous *in situ* water quality record.

Sonde location	Begin date	End date
Central (bottom)	6/10/2015	7/23/2015
	8/28/2015	11/24/2015
North (bottom)	5/13/2015	8/27/2015
South (bottom)	5/27/2015	10/29/2015
South (surface)	--	--

The temperature sensor performance did not have an influence on the performance of the other sonde sensors or quality of the recorded conductivity or dissolved oxygen data, but it would influence the calculation of specific conductivity from recorded conductivity data, dissolved oxygen saturation estimates, and pH. A ±5ºC error in water temperature within the range of temperatures measured in the SCRE would result in an approximately 10% error in the calculation of specific conductivity from measured conductivity and an approximately 10% error in the calculation of dissolved oxygen saturation. The sensitivity of pH to variations in the recorded temperature within the temperature range experienced in the SCRE indicates a ±5ºC error in water temperature would result in an approximately ±0.02 error in the pH value (Barron et al. 2006).

Based upon comparisons with *in situ* DO readings from the deployed sonde and spot checks from a recently calibrated unit, all DO data showing deviations greater than 2 mg/L at retrieval were considered out of range and flagged for exclusion. In general, large deviations between the deployed sonde and the recently calibrated unit were apparent following recent breaches and open mouth conditions, making sonde data credibility questionable. Whether this was related to sensor drying, fouling, or algal growth on/accumulation around sensors is unknown. Table 2-4 shows the periods that were excluded from analysis of the continuous data record based on post-deployment spot checks. In these cases, *in situ* DO conditions were based on spot check readings alone. Because only a few data records excluded on the basis showed differences between *in situ* and spot measurements of water temperature, pH and Conductivity, a decision was made to retain these data for subsequent analysis.

Table 2-4. DO data exclusions from the continuous *in situ* water quality record.

Sonde location	Begin date	End date
Central (bottom)	2/12/2015	3/2/2015
	6/10/2015	6/29/2015
	8/12/2015	8/27/2015
	10/30/2015	11/23/2015
	12/17/2015	1/5/2016
	2/10/2016	3/16/2016
North (bottom)	2/12/2015	3/2/2015
	7/24/2015	8/12/2015

	10/30/2015	11/23/2015
	12/17/2015	1/5/2016
	2/10/2016	3/16/2016
	6/15/2016	7/12/2016
South (bottom)	2/12/2015	3/2/2015
	8/12/2015	8/27/2015
	9/2/2015	9/28/2015
	10/12/2016	11/8/2016
South (surface)	2/12/2015	3/2/2015
	6/23/2016	7/20/2016

2.2.4 Supplemental water quality sampling at site R-1 in 2016

As requested by VCK in fall 2015, an evaluation of the surface water monitoring location for R-1 was carried out to ensure that the sample was representative of riverine as well as potential hyporheic (groundwater) influences upstream of the SCRE (i.e., downstream of Victoria Avenue bridge and upstream of the Santa Clara River Estuary). As described in the Workplan for the Phase 1 studies and continuing through Phase 2 and the current (Phase 3) Workplan, sample point R-1 was intended to capture riverine flows unaffected by VWRF discharge and is used to provide an upstream control for open water sampling locations in the SCRE. The location was selected by evidence of flowing water (i.e., visible disturbance of the water surface by river flow and or hyporheic upwelling).

Detailed review of field notes indicated that the R-1 sampling location used in 2015 was just upstream of the Harbor Blvd. bridge, corresponding to NPDES sample location R-005. Because this location did not correspond to the farthest upstream extent of SCRE inundation during the extended periods of inundation occurring during 2015, additional data collection was initiated in 2016 that included sampling at both the R-005 location as well as the Phase 3 “R-1” location farther upstream as detailed in the Workplan. Results of these events were included in the Phase 3 water quality data compilation. It should be noted that although riverine surface water flows were apparent during all sampling events in 2016 and surface flows were measured in July 2017 (Section 2.1.4), no information regarding the source(s) of these flows was collected and hyporheic upwelling was not observed.

2.3 Habitat Data

The overall goal of the Phase 3 habitat data collection for the habitat suitability evaluation was to improve characterization of habitat suitability in the SCRE under varying VWRF discharge alternatives. Table 2-4 shows the Phase 3 habitat suitability monitoring plan. Fish and benthic macroinvertebrate (BMI) sampling was conducted, and habitat attributes were recorded to characterize the abundance and composition of the aquatic community spatially and temporally. This data is used to refine, where applicable, habitat suitability relationships developed during the Phase 1 Study. Additionally, mapping of general habitat types was refined using updated aerial imaging from the National Agriculture Imagery Program (NAIP), as well as imagery from unmanned aircraft system (i.e., drone) flights conducted during the December 2016 field effort to provide higher resolution data. Sediment maps were constructed based on field data collected during fish and BMI sampling, as well as high resolution aerial imagery. Spatial maps of water quality conditions at the time of sampling were also constructed based on spot measurements collected during the habitat surveys (3.4.1.4).

Table 2-5. Summary of data collected for the Phase 3 habitat suitability evaluation.

Habitat suitability component	Phase 3 monitoring location	Parameter	Frequency	Duration
Fish Sampling	SCRE perimeter	Species Composition and Abundance[1], and Water Quality (Temp, DO, pH, EC)	Quarterly/Event (open/closed)	≥1 water year
Habitat Mapping	SCRE (entire)	Depth, Substrate, vegetation, and Water Quality (Temp, DO, pH, EC)	Event (open/closed)[2]	
Benthic Macro-invertebrate (BMI) Sampling	4 locations in SCRE: near R-1, E-1, E-2, and outfall channel (near North Sonde)	Species Composition and Abundance, Depth, Substrate (grain size), and Water Quality (Temp, DO, pH, EC)	Quarterly/Event (open/closed)	

[1] The proposed and permitted method of fish sampling will be using beach seines, and will include sub-sampling of fish weight and length of any focal fish species captured (goby, steelhead).
[2] No high-flow events occurred during the Phase 3 monitoring period that warranted event-based habitat mapping.

2.3.1 Quarterly benthic macroinvertebrate sampling data

Benthic macroinvertebrates (BMI) were sampled quarterly in 2015. BMI samples were collected in the proximity of four of the fish-sampling locations (near R-1, E-1, E-2, and the outfall channel [near North Sonde]) using the existing sampling protocol for the SCRE developed by ABC Labs (2008). Individual BMI were identified to Standard Taxonomic Effort (STE) level 1 specified by Southwest Association of Freshwater Invertebrate Taxonomists (SAFIT; Richards and Rogers 2011), and removed from the sample, using forceps, and sorted into separate 20 mL sample vials based upon the different taxa identified. This process was repeated until all organisms were removed from the whole sample collected. Data was compiled into summary reports for each sampling event including species richness and diversity metrics, as well as a tolerance value (TV) assigned to each group determined by the STE.

It should be noted that in order to comply with requirements for the protection of tidewater goby by USFWS, BMI sampling could not be conducted using standard sweep or kick nets. Instead, sediment grab samples at each sampling location were collected using a 0.02 m^2 petite Ponar sediment grab sampler. The implications of sampling methods are discussed in Section 3.6.1.

2.3.1.1 Sorting QA/QC

Once all invertebrates had been removed from a BMI sample, the remaining material was transferred from the Petri dish into a clean container labeled “grunge.” The sample matrix remaining after sorting was evaluated to determine elutriation efficiency. Approximately 10% of the grunge from each sample was placed into a Petri dish and observed under a microscope at 10 times magnification to verify that no BMIs had been missed during the sorting process.

2.3.1.2 Taxonomic QA/QC

All organisms removed during the sorting process were identified using aforementioned STE level one, specified by the SAFIT. Voucher specimens were retained for all unique taxa. The identified taxa from the processed portion of each sample were placed in separate vials and preserved with 70% ethanol. Approximately 10% of the samples underwent QA/QC by other Aquatic Bioassay Laboratory taxonomists and all samples met SAFIT data quality objectives.

2.3.2 Quarterly fish sampling data

Quarterly fish sampling was conducted in the SCRE pursuant to USFWS permit #TE198917-1 for tidewater goby. Sampling activities utilizing seine and dip net methods were conducted at varying locations in the SCRE during March, June, and September in 2015, targeting habitats to provide representative results. A similar sampling effort was conducted in September 2016.

During field efforts, standardized data sheets were used that mirror the project electronic database created for data entry. Upon return from the field, all physical data sheets were photocopied and archived, and entries compiled into a MS Excel database. Data entries were double checked to identify any inconsistencies. All fish species captured were enumerated and identified to species and, where possible, life stage. Additional habitat parameters and water quality data were also collected along with substrate, as well as GPS coordinates and site photographs. It should be noted that water quality and habitat information collected for this study were intended to identify spatial patterns and habitat associations and should not be used to infer compliance with NPDES permit conditions.

3 PHYSICAL AND BIOLOGICAL SETTING

3.1 Study Area

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 areal extent of the SCRE has been reduced by approximately 75–90% from historical conditions (Swanson et al. 1990, ESA 2003, Nautilus Environmental 2005). 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 11 feet (vertical datum: North American Vertical Datum of 1988 [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 0.8 mile 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 subbasin, which is part of the larger Santa Clara River Valley groundwater basin (CDWR 2006a). The Mound subbasin consists of several aquifers composed of sand and gravel separated by low permeability clay layers. The aquifer located directly below the SCRE (referred to as the "Semi-perched aquifer") is shallow and underlain by a clay layer, thereby disconnecting the SCRE from the deeper subbasin aquifers (Hanson et al. 2003). The 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 2005) and currently has an area of approximately 30 acres and an average depth of approximately 2 ft (Kennedy/Jenks Consultants 2002). The lake receives runoff from the floodplain south of the SCRE and river via a network of agriculture drainage ditches (URS 2005). When water levels in the lake reach a certain level, the lake is partially pumped delivering water to an outfall pond where the water then seeps through or flows over the beach and reaches the ocean (see Figure 1-1). Note that pumped water or other surface outflows do not enter the SCRE.

Land use within the floodplain adjacent to the SCRE includes open beach/dunes, agriculture, park land (McGrath State Beach), oil exploration, and the VWRF. The SCRE and adjacent floodplain host numerous animal species throughout the year, including federally listed fish and birds whose habitat is highly dependent on the physical conditions of the SCRE.

3.2 Geomorphology

The geomorphology of the Santa Clara River is influenced primarily by periodic short duration, high intensity floods rather than by the moderate discharges frequently used to characterize channel form response in temperate climates (Stillwater Sciences 2007b). The dynamic nature of the river has a direct effect on morphology and habitat conditions in the SCRE. As such, this section focuses on the morphodynamics and primary influences on the physical form of the estuary. Overall, the SCRE is formed at the downstream end of the river, and is composed of a main lagoon impounded by a seasonally closed-mouth beach berm. The main lagoon contains a network of channels and bars that are formed and reworked during storm events and subsequent tidal exchange while the river 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 about 180 acres. During closed-mouth periods, the lagoon fills causing the ponded water to rise and extend upstream of the Harbor Blvd bridge.

Much of the information presented here builds upon material originally presented in the Phase 1 Study report (Stillwater Sciences 2011a) and the watershed-wide geomorphology assessments conducted on behalf of the Coastal Conservancy (Stillwater Sciences 2007) and VCWPD (Stillwater Sciences 2011b).

3.2.1 Historical geomorphic change

The entire lower river corridor 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

(Beller et al. 2011). 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 Niño Southern Oscillation (ENSO) has resulted in a wet-period ENSO cycle in southern California between the mid-1960s and mid-2000s, resulting in a higher frequency and duration of large storms (see Section 3.3.1 Precipitation below).

Changes to the lower river corridor and SCRE extent since the mid-19th century were assessed to highlight the drivers for morphologic change. Data sources used 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, Beller et al. 2011, Stillwater Sciences 2011a,b), and orthorectified topographic maps and aerial photographs from 1855 through 2014. These data were compiled and then used to assess morphologic changes approximately every few decades since 1855. Building upon previous assessments, six distinct morphologic periods were identified (Table 3-1). Historical maps and aerial photographs of the SCRE produced between 1855 and 2014 are presented in Appendix C, along with narrative summaries of the SCRE planform changes.

A secondary analysis was performed to demarcate the SCRE beach berm position during recent years: 2000, 2005, 2009, 2012, and 2016 (Figure 3-1). As discussed below, the berm has been steadily migrating landward since the last major river flood in 2005, equating to a maximum migration length of 1,000 feet between the 2005 and 2016 berm positions. The landward migration is associated with the recent drought period whereby tidal-wave forces have overcome riverine forces due to the paucity of significant runoff events. Further, as is discussed in greater detail below, the percentage of time the SCRE mouth was open has steadily decreased since the 1990s, with the more recent drought period between water years 2012 and 2015 experiencing significantly fewer days with open-mouth conditions (see Section 3.3.5 Mouth Breaching Dynamics below).

Overall, the dominant planform geometry, or areal extent and shape, of the SCRE during the Phase 3 Study has differed from the SCRE morphology occurring prior to and during the Phase 1 Estuary Study.

Table 3-1. SCRE morphologic periods since the early 19th century through 2016.

Time period	Description	Major storms in a water year ($\geq$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)[e] **Feb 1969** (152,000)[e]
Full Build-out and ENSO wet-period Early 1970s–2005	Development in the estuary subwatershed peaks to contemporary levels. Estuary very confined; levees contain large 2005 floods.	**1973** (58,200) **Feb 1978** (98,600) **Mar 1978** (102,200) **1980** (81,400) **1983** (100,000) **1992** (104,000) **1995** (110,000) **1998** (84,000) **Jan 2005** (136,000) **Feb 2005** (82,200)
Present day (Post-2005)	Estuary in quasi-stable state following 2005 floods.	No flows >50,000 cfs; Highest flows in: 2008 (33,000) 2011 (44,000)

[a] Known ENSO years since 1950 are shown in bold, based on NOAA's Climate Prediction Center (2016).
[b] The 5-year flood is used as the threshold flow because of 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). Peak flows in water years 2005–2013 from station #723 maintained by VCWPD (2016).
[d] The St. Francis Dam failure; peak flood estimate from Begnudelli and Sanders (2007).
[e] The flooding of 1969 destroyed Ventura Harbor, and was a major basis for subsequent implementation of additional flood protections

Figure 3-1. Contemporary views of the SCRE during its maximum extent in 2005 (top) and minimum extent in 2016 (bottom), with combined overlay of beach berm positions since 2000 (bottom) (source imagery from NAIP; analysis by Stillwater Sciences) [See Appendix C for additional information].

3.2.2 Contemporary morphology

In recent years, the SCRE has been responding to morphological changes induced by two high magnitude storm events in 2005 (with peak flows of 136,000 and 82,000 cfs) and the more recent drought period. The 2005 flood event resulted in significant expansion and oceanward-migration of the SCRE lagoon. It was noted during the Phase 1 Estuary Study that the lagoon had been steadily contracting as the beach berm migrated landward in the years following the flood event. Routine inspections of the SCRE by VWRF staff, Stillwater Sciences, and ABC Laboratories since 2011 found that the most significant changes to the SCRE have been the steady landward-migration of the beach berm, in addition to prolonged periods of closed-mouth conditions between 2011 and 2015, and manual trenching of the beach berm near the VWRF outfall channel's confluence with the lagoon throughout 2016. Additional details on SCRE hydrology and mouth status are presented below.

An updated elevational surface of the SCRE (circa 2016) was employed in the present study, which was accomplished by merging two high-resolution digital elevation model (DEM) surfaces of the SCRE lagoon bathymetry and surrounding beach and floodplain topography (Figure 3-2). Much of the merged DEM surface, which adequately represented conditions observable before the start of the Phase 3 Study in fall 2014, was sourced from topographic and bathymetric data compiled by cbec, Inc. on behalf of the Wishtoyo Foundation (cbec 2015). This dataset compiled other topographic and bathymetric data generated using Light Detection and Ranging (LiDAR) topography technology employed during 2009–2011 by the California Coastal Conservancy, multi-beam bathymetry technology in 2012 by Dr. Sean Anderson of California State University at Channel Islands, and field surveys conducted in 2014 by cbec. Because the berm position and profile have continued to evolve, Stillwater Sciences collected high-resolution topographic data along the entire beach berm in December 2016 using a combination of real-time kinematic global positioning system and unmanned aerial system equipment. These newer data were subsequently used to update that portion of the available DEM surface provided by cbec, Inc (see Figure 3-2).

Bed elevations within the SCRE range between approximately 1 and 10 ft NAVD88 (Figure 3-2). The beach berm elevation currently varies between 14 and 17 ft NAVD88, which is approximately between 8.6 and 11.6 ft above mean higher high water (MHHW: 5.4 ft NAVD88 during WY 2014–2016). These conditions are similar to those reported during the Phase 1 Estuary Study (Stillwater Sciences 2011a). The SCRE bathymetry results generated from use of the 2016 DEM surface reveal an overall increase of water-surface area and volume as stage increases in the SCRE (Figure 3-3). Two distinct inflection points at approximately 7 and 9 ft NAVD88 are apparent in the plotted curve for the stage versus water-surface area occurs (see top plot in Figure 3-3). The bends in the curve are a result of the rising water in the SCRE becoming increasingly confined within the main lagoon shorelines starting around 7 ft NAVD88, then decreasingly confined above 9 ft NAVD88 upon the surrounding floodplain areas. The relationship between the water-surface stage and water volume follows a curve having a fairly constant slope (see bottom plot in Figure 3-3).

The effective distance, or longitudinal length, of the beach berm has varied considerably over time as the SCRE morphology has responded to episodic flood events and long-term natural and anthropogenic factors. Several recent morphologic states of the beach berm length, as represented by different DEM surfaces—2005, 2009, 2012, and 2016—that were evaluated for the present study, are displayed together in Figure 3-4. The estimation of berm lengths running along the western water-side boundary of the wetted lagoon for each of the DEM surfaces reveals different relationships as a function of rising water-surface stage. The most significant change has been a reduction in the total length since the post-2005 flood event morphology where the 2016 length is approximately 2,000 linear feet less than the 2009 and 2012 lengths. The stage-length relationship in 2016 also has a steeper slope at lower stage as a result of the berm position having migrated into the deeper (and broader) lagoon areas. It should be noted that the accuracy of any comparison of the current 2016 DEM surface to older surfaces is limited by the differences in data resolution and vertical completeness of those older surfaces.

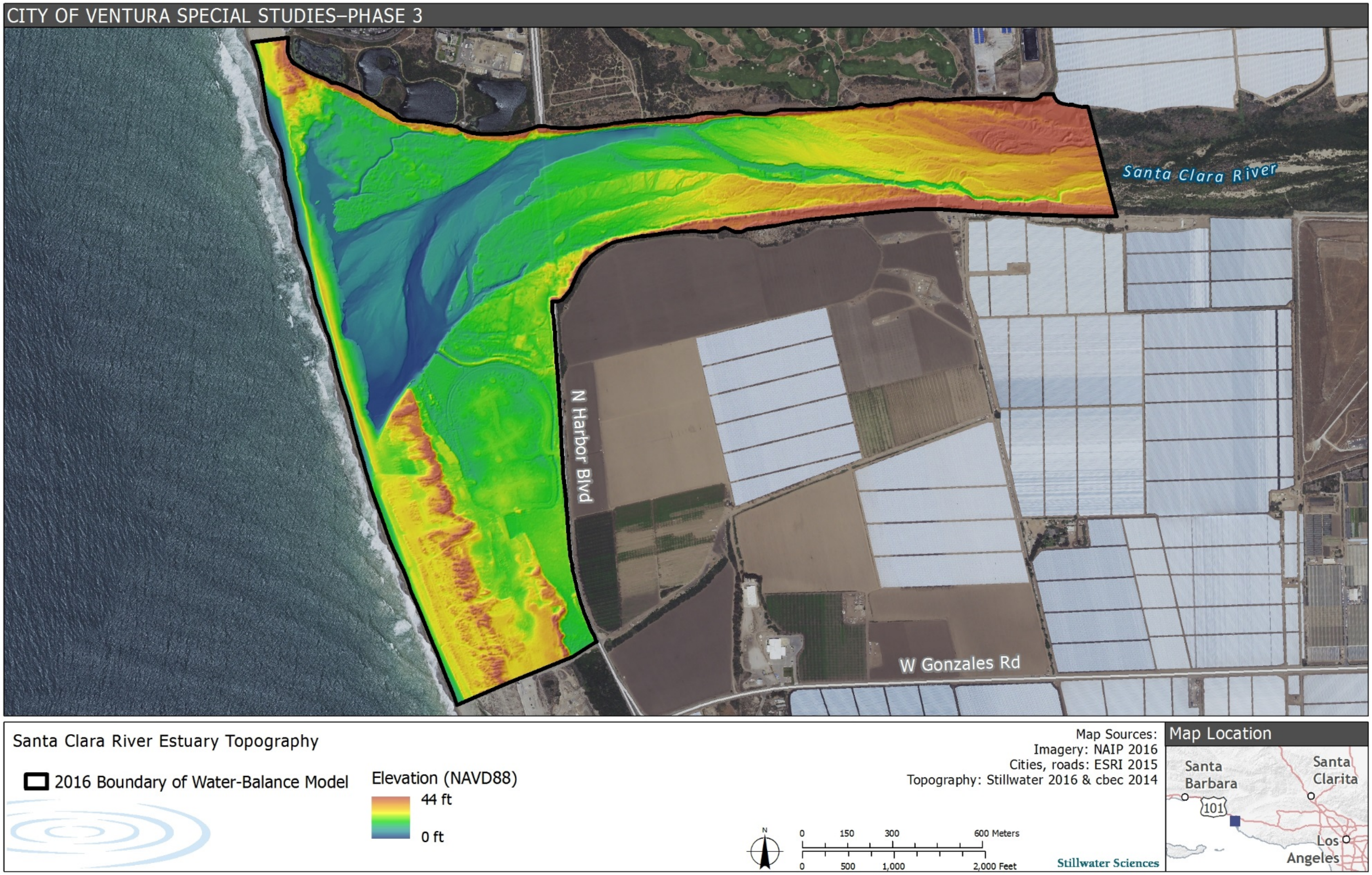


Figure 3-2. SCRE and surrounding floodplain topography based on 2016 conditions (Source: Stillwater Sciences and cbec).

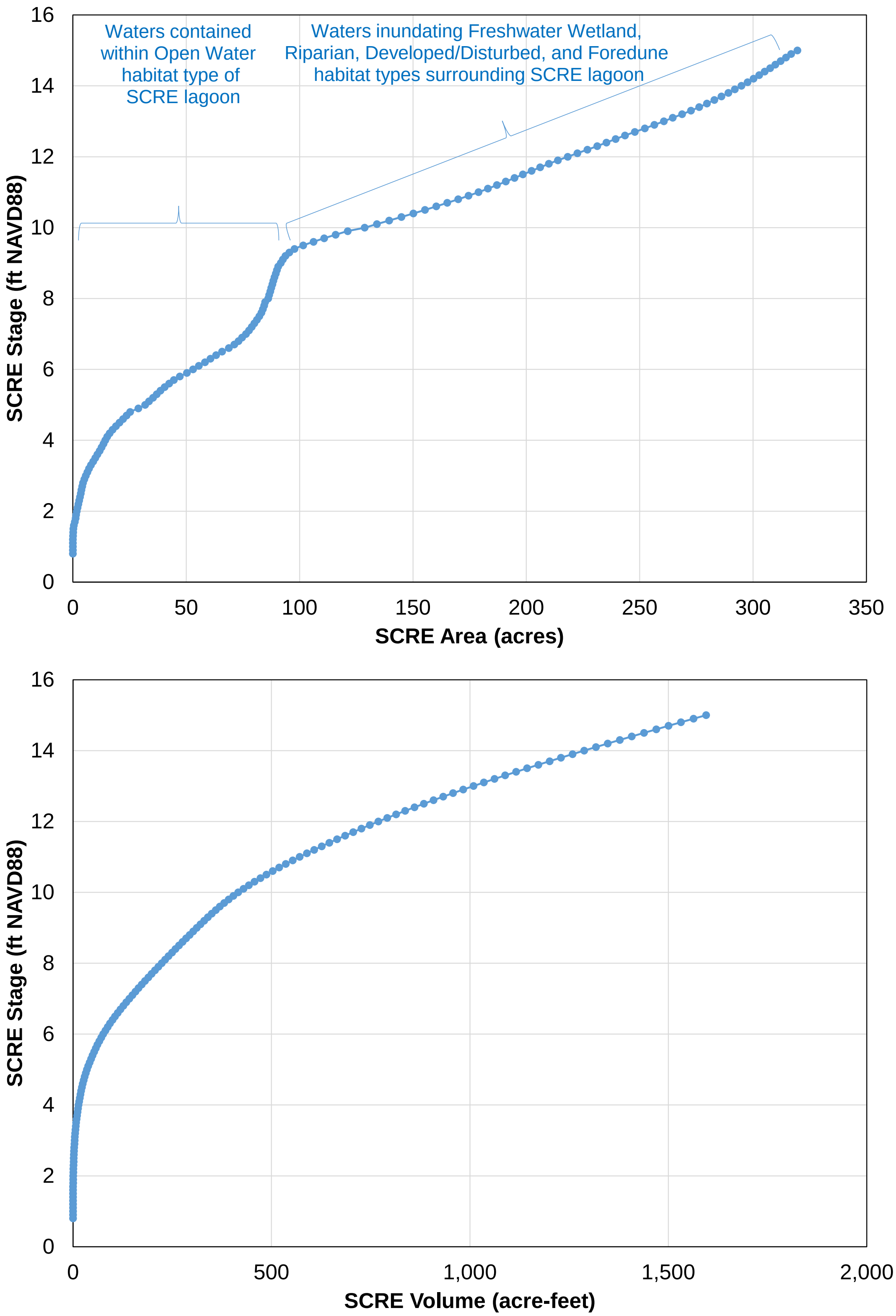


Figure 3-3. Relationship between SCRE stage and inundated area (top) and volume (bottom) based on 2016 conditions.

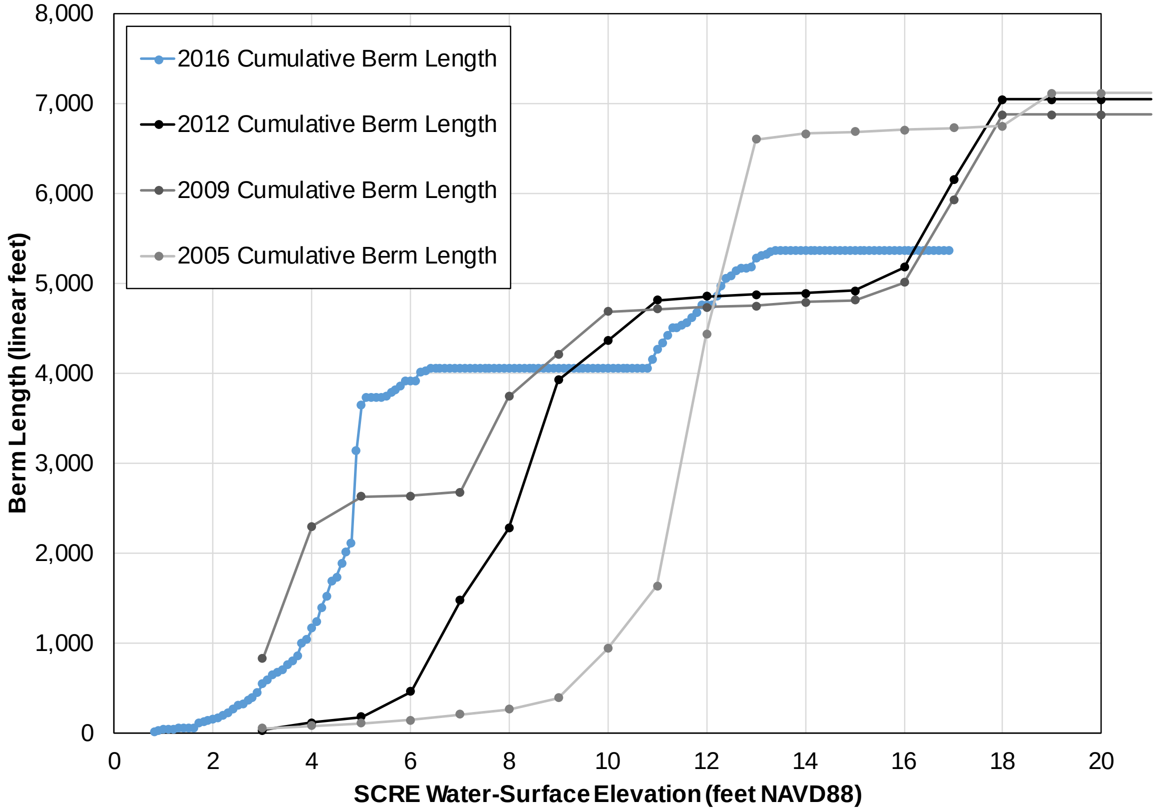


Figure 3-4. Relationship between SCRE stage and beach-berm length based on 2016 and past conditions.

3.2.3 Sediment characteristics

As part of the watershed's setting within the tectonically active Transverse Mountain ranges of southern California, extreme yet episodic rainfall events directly contribute to dramatically high sediment-production rates in many of its tributary basins (Stillwater Sciences 2011b, Warrick and Farnsworth 2009). The river thus transports a considerable amount of sediment to its lower reaches and, eventually, discharges offshore into the Santa Barbara Channel. In general, the coarser sediment (>0.065 mm, or coarse sand and larger) transported 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 SCRE mouth berm during periods of lower river discharge.

Bed sediments within the SCRE are characterized by stratified layers of coarse sand and range in size from clay- to cobble-sized particles (Figure 3-5). 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. Approximately 330 ft upstream of the Harbor Blvd bridge, there is a distinct break in the main channel slope and associated transition from coarser to finer bed sediment going upstream to downstream (see Figure 3-5). Temporal variation in SCRE sediment particle size is attributable to seasonal flow variations in the Santa Clara River as well as large storm events. Wet winter periods with higher flow scour fine sediments from the bed and increase average particle size. Quiescent conditions in lower flow periods result in fine sediment deposition and decrease average particle size.

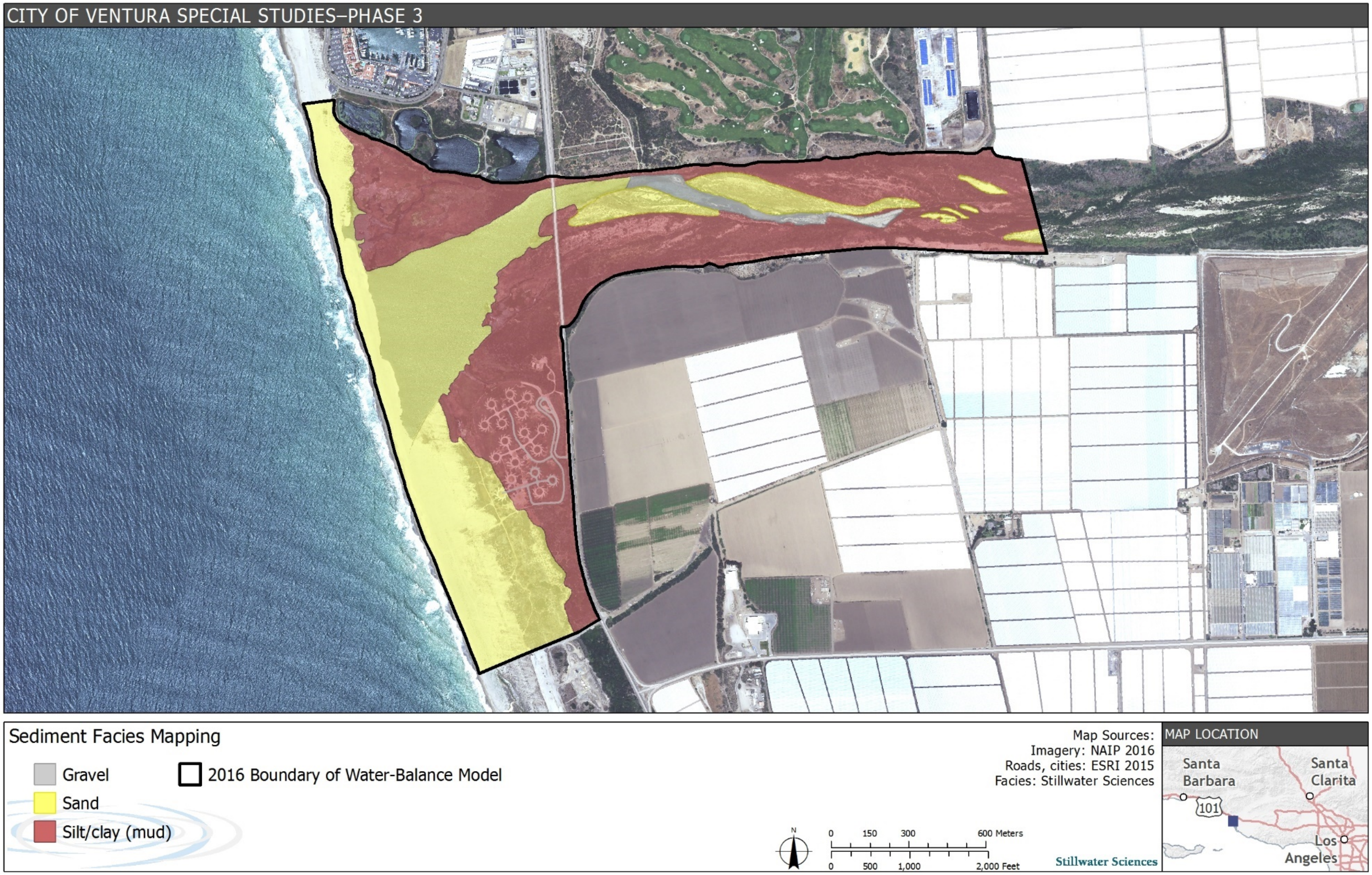


Figure 3-5. SCRE generalized sediment facies based on 2015–2016 conditions.

3.3 Hydrology and Hydraulics

The SCRE receives water from groundwater, precipitation, and three surface water sources: Santa Clara River flow and local runoff, VWRF discharge, and tidal flow. During normal and wet water-year types, river flows typically dominate water inputs to the SCRE from fall to spring, while VWRF discharge (~5–10 cfs) is the predominant source during closed mouth conditions in the summer. During periods of high in-river flow, the mouth of the SCRE breaches allowing for tidal flow exchange with the estuary. Water outputs from the SCRE occur through outflow to the ocean during open-mouth conditions as well as evaporation, subsurface outflow through the mouth berm, and groundwater seepage to the semi-perched aquifer.

This section summarizes hydrologic (e.g., surface and subsurface inflows and outflows) and hydraulic (e.g., mouth breaching and estuary stage/inundation) processes in the estuary over the past several decades. This information was utilized to update the estuary water balance evaluation (see Section 4: Water Balance Evaluation).

3.3.1 Precipitation

The southern California coast experiences highly variable annual rainfall depending on each storm's frequency and magnitude and the landscape relief. In the 1,623 mi^2 Santa Clara River watershed rainfall and air moisture both tend to decrease with greater distance from the coast. A clear pattern of increased rainfall with elevation is expressed across the watershed, as the lowland areas, such as the Oxnard Plain, receive less than half of the rainfall received in the headwaters of Santa Paula, Sespe, and Piru creeks (PRISM 2017). Total annual precipitation, as depicted graphically in Figure 3-6 based on long-term rainfall recordings in downtown Ventura (VCWPD 2017), exhibits substantial variability over time, ranging over a factor of 7. Longer-term cyclic patterns of precipitation in the lower watershed can be visualized when plotting the cumulative departure, or difference, of each annual total from the long-term average of annual totals (black line in Figure 3-6). The time series of cumulative departure of annual precipitation at the downtown Ventura station reveals decadal-scale patterns of generally drier than average (i.e., hydrologically losing trend) and wetter than average (i.e., hydrologically gaining trend). A wetter period between the mid-1990s and mid-2000s followed by a drier period through water year 2016 can be inferred from the plotted trends.

Periodicity in the pattern of wet/dry years in southern California is correlated to the El Niño–Southern Oscillation (ENSO) climatic phenomenon. ENSO is characterized by warming and cooling cycles (oscillations) in the waters of the eastern equatorial Pacific Ocean having 1–1.5-year duration and 3–8-year recurrence interval (Inman and Jenkins 1999, NOAA 2017a). ENSO-induced climate change occurs on a multi-decadal time scale that is consistent with the hydrologically gaining and losing trends occurring in the lower Santa Clara River watershed (see Figure 5), with a relatively wetter climate through 2006 followed by a relatively drier period through 2016.

The Phases 1 through 3 Estuary Studies have all been conducted during the recent drier period, though the annual total rainfall amounts have been significantly different. As depicted in Figure 3-6, the annual rainfall totals since the start of the Estuary Studies have been (per water year): 14.1 (2008), 10.4 (2009), 16.2 (2010), 19.7 (2011), 8.9 (2012), 6.6 (2013), 6.2 (2014), 8.4 (2015), and 8.4 (2016). Overall, the annual rainfall totals during the Phase 3 Study have been about half of those during the Phase 1 study. Hourly rainfall data recorded in downtown Ventura are presented in Figure 3-7.

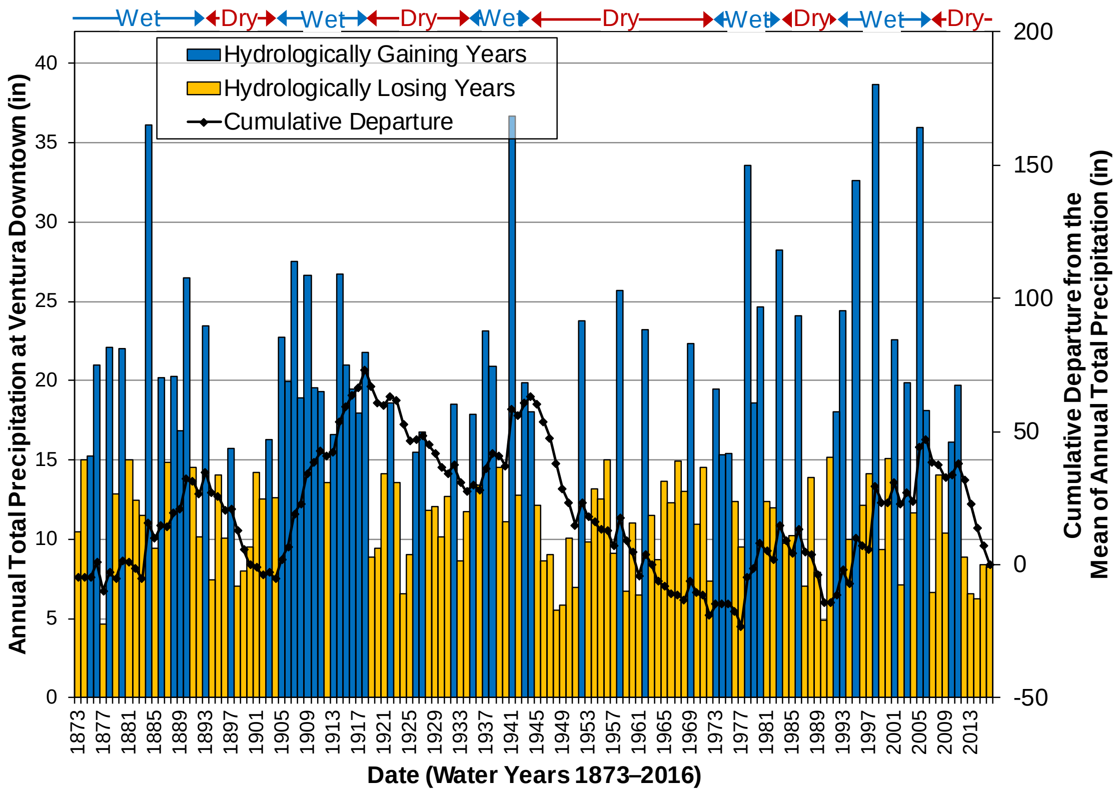


Figure 3-6. Long-term annual total precipitation amounts and patterns of variation based on the cumulative departure from the average of annual total precipitation amounts recorded at the Downtown Ventura atmospheric monitoring station (VCWPD Station 066) northwest of the SCRE (source data from VCWPD; analysis by Stillwater Sciences).

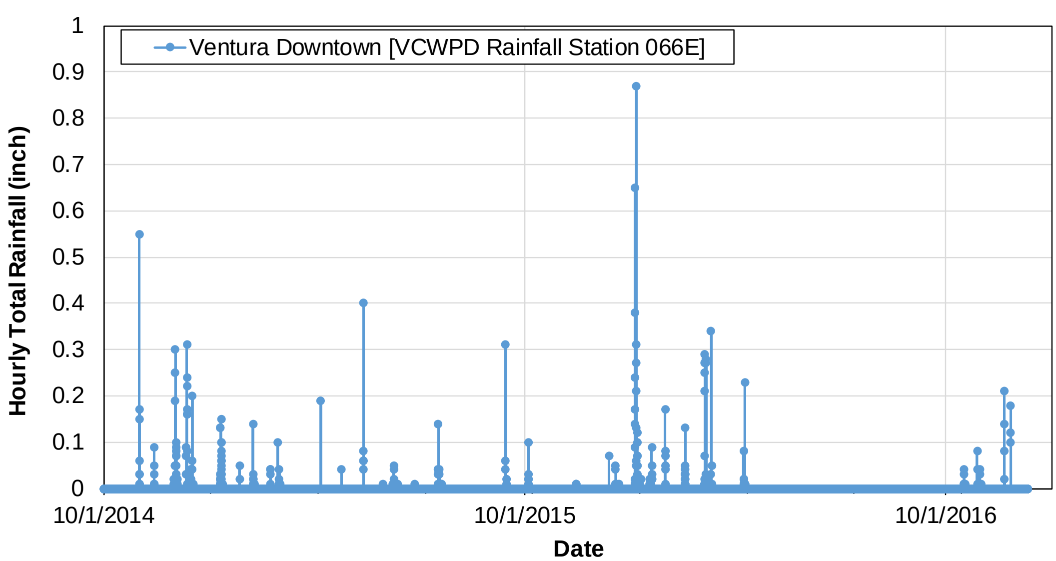


Figure 3-7. Hourly precipitation amounts recorded at the Downtown Ventura atmospheric monitoring station (VCWPD Station 066) northwest of the SCRE during water years 2015–2016 (source data from VCWPD; analysis by Stillwater Sciences).

3.3.2 Evaporation

Evaporation of water from the SCRE is most sensitive to local air temperatures which vary each day. Local open water evaporation rates were determined from an existing atmospheric monitoring station at the El Rio-United Water Conservation District (UWCD) Spreading Grounds (VCWPD Station #239), located approximately 6 miles east of the SCRE. This monitoring station records evaporation rates on an hourly basis. A secondary, more local, station was established for the Phase 3 Study near the VWRF Wildlife/Polishing Ponds, which consisted of an evaporation pan and a data logger recording changes in water levels at 30-minute time steps (station PE-1; see Figure 2-1). Finally, a third monitoring station, located in Oxnard, was utilized to provide an additional check on daily evaporation rates (CIMIS station 156).

Some problems arose with the maintenance of water quality and quantity at the evaporation station PE-1 during the Phase 3 Study. QA/QC procedures found that daily mean evaporation data from station PE-1 exhibited some significant deviations from the other two monitoring stations during 2016 (Figure 3-8). The data recorded at the El Rio-UWCD Spreading Grounds and Oxnard stations exhibit trends driven by seasonal changes in air temperature, with the greatest evaporation rates occurring in the summer months and the lowest rates occurring in the winter months. Based on this information, it was decided that the data from the El Rio-UWCD Spreading Grounds (VCWPD station #239) was better suited to represent evaporation rates in the SCRE.

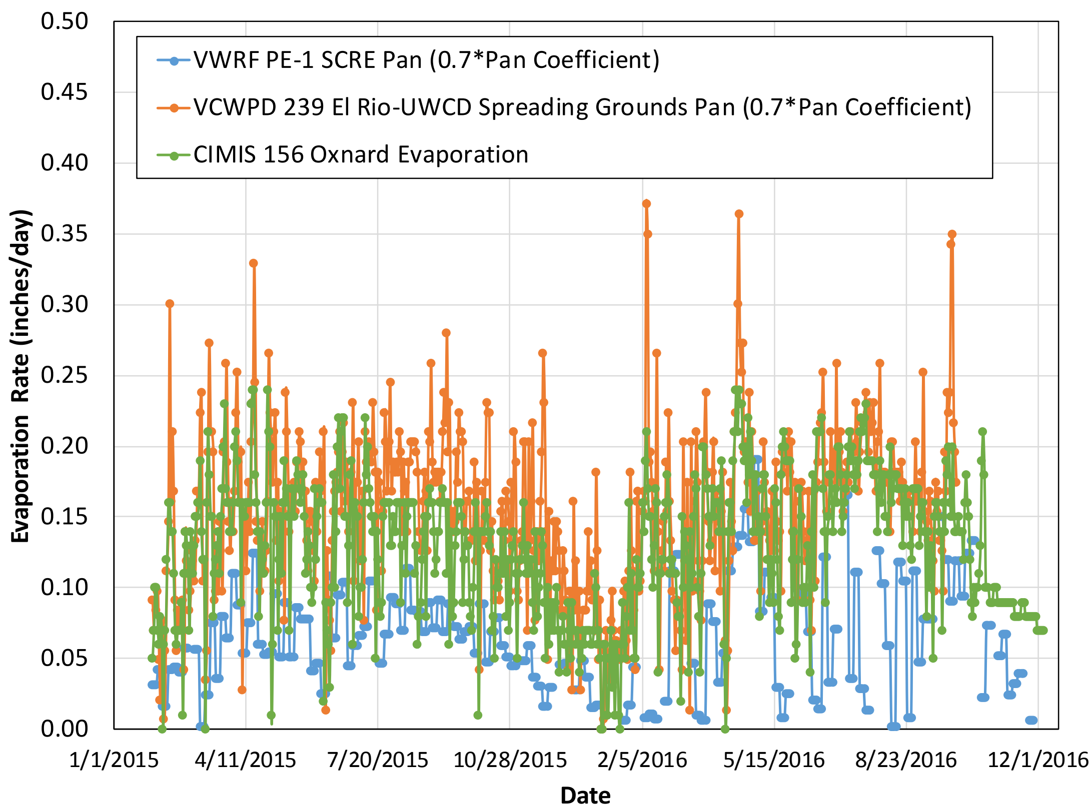


Figure 3-8. Daily evaporation amounts recorded at the VWRF (PE-1), El Rio-UWCD Spreading Grounds (VCWPD 239), and City of Oxnard (CIMIS 156) monitoring stations near the SCRE during 2015–2016 (source data from VCWPD, CIMIS, City of Ventura; analysis by Stillwater Sciences).

3.3.3 Surface water

3.3.3.1 River flow

The semi-arid climate in the region is interrupted by periods of intense winter storms that often cause high-water flow in the Santa Clara River and its many smaller tributaries (barrancas) draining to the lower valley reach. The contemporary river hydrology is dominated by floods. Between flood periods, flows through the lower river reach reaching the SCRE are intermittent with some sub-reaches, such as the Oxnard Forebay reach between highways 118 and 101 bridges, going dry for much of the year, while the lowermost reach leading into the SCRE supports perennial, albeit low volume, flow during most water-year types. This baseflow, which is driven by inputs from the semi-perched aquifer, is partly enhanced by seasonal agricultural runoff, particularly on the northern floodplain.

The watershed's seasonal hydrologic regime is apparent in examination of monthly mean discharge recorded at four locations in the lower watershed (Figure 3-9). Over the long-term record, February has experienced the highest monthly flows (~750 cfs in the lower river) while August and September have experienced the lowest flows (~1 cfs in the lower river). As expected, these patterns closely follow seasonal variability of precipitation (see above).

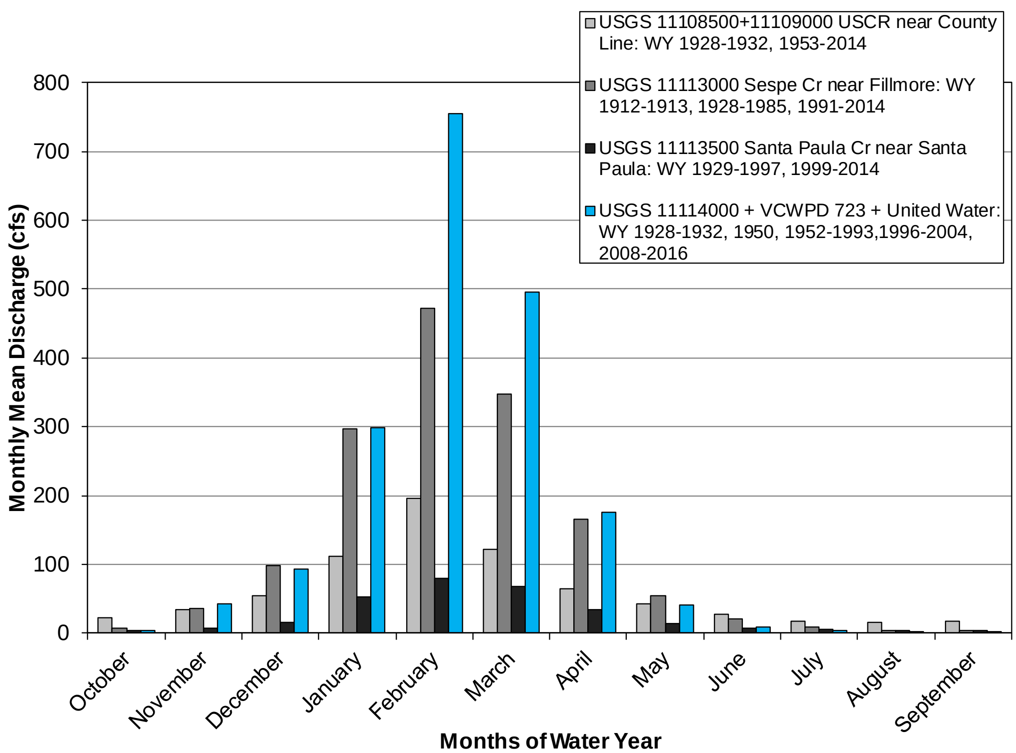


Figure 3-9. Monthly mean discharge characteristics of the lower Santa Clara River watershed based on compilation of available long-term gaging records through water year 2016 (source data from USGS, VCWPD, and UWCD; analysis by Stillwater Sciences).

Daily mean flows have averaged about 150 cfs in the lower river over the past century (Figure 3-10). During the Phase 3 Study period, daily mean flows peaked at approximately 640 cfs in water year 2015 (December 12, 2014) and 560 cfs in water year 2016 (January 6, 2016), which were both relatively dry years (Figure 3-11). Flows for 2016 were measured immediately downstream of the Freeman Diversion, so the peak flow into the SCRE may be lower than 560 cfs due to uncertainty about the amount of flow percolating in the Oxnard Forebay reach. Overall the long term, for 90 percent of the time, daily mean flows in the lower river have been less than about 80 cfs (Figure 3-12). During water-years 2014–2016, daily mean flows have averaged only about 4 cfs, indicative of the drought conditions.

Annual peak flows have been considerably larger in comparison with the daily mean flows and usually span only a few hours to days, indicating the flashy nature of the river. The largest flood events exceeding 50,000 cfs are listed above in Table 3-1. Since water year 2011, the largest event recorded in the lower river (Victoria Ave bridge) occurred on March 20, 2011 (~44,000 cfs). The recent drought period between water-years 2012 and 2016 have yielded very few high-flow events; the annual peak flows have all been less than 4,000 cfs. Thus, the present study period experienced relatively few high-flow events capable of flooding the lower river corridor and naturally breaching the SCRE mouth berm (see additional details in Section 3.3.5 Mouth Breaching Dynamics).

Overall, the river and SCRE naturally experience a wide variation of flows, punctuated episodically by short-duration but intensive channel-/lagoon-adjusting flood events. These traits are common to large, semi-arid riverine systems that periodically experience dramatic geomorphic change resulting from their flashy discharge dynamics (Graf 1978, Warrick and Mertes 2009, Downs et al. 2013). And while climate change models predict warmer air temperatures, increased variability and frequency of intense storms are expected to make southern California rivers, especially the large Santa Clara River watershed, more susceptible to flooding (USGCRP 2009, TNC 2013, IPCC 2014).

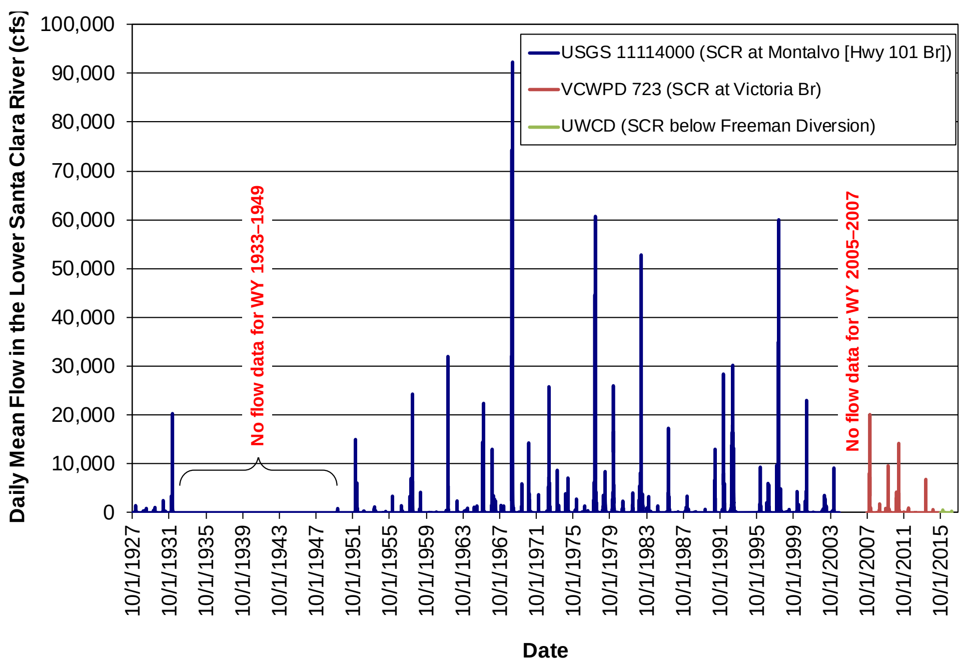


Figure 3-10. Lower Santa Clara River daily mean discharge measured at Montalvo, CA (USGS 11114000, WY 1927–2004), Victoria Avenue bridge (VCWPD 723, WY 2008–2015), and immediately below Freeman Diversion (UWCD, WY 2016) (source data from USGS, VCWPD, UWCD; analysis by Stillwater Sciences).

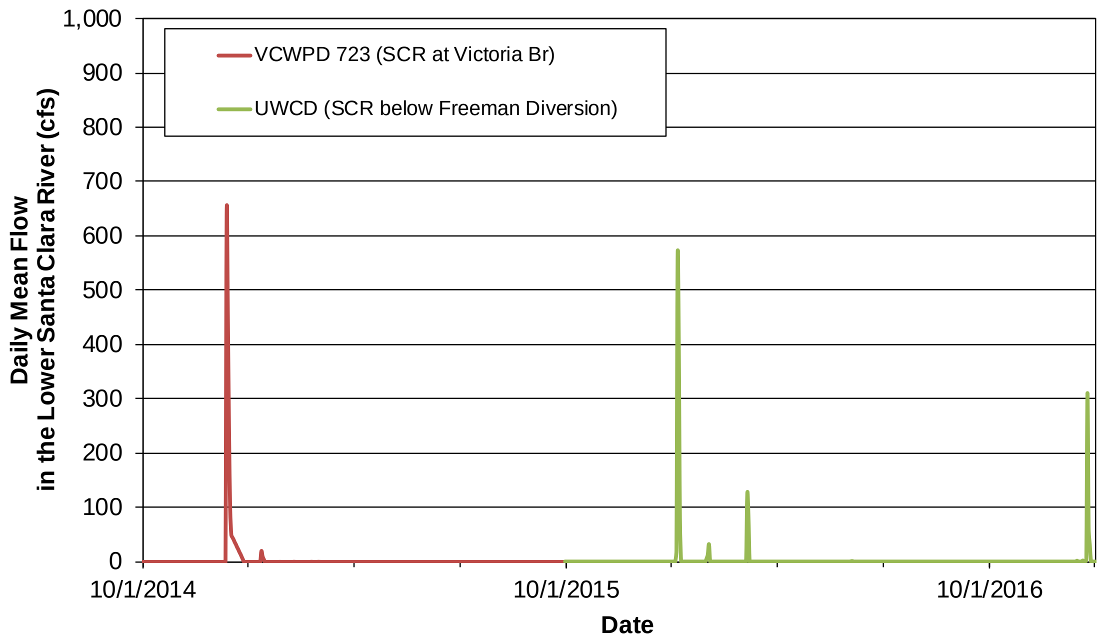


Figure 3-11. Lower Santa Clara River daily mean discharge measured at Victoria Avenue bridge and immediately below Freeman Diversion during water years 2015–2016 (source data from VCWPD and UWCD; analysis by Stillwater Sciences).

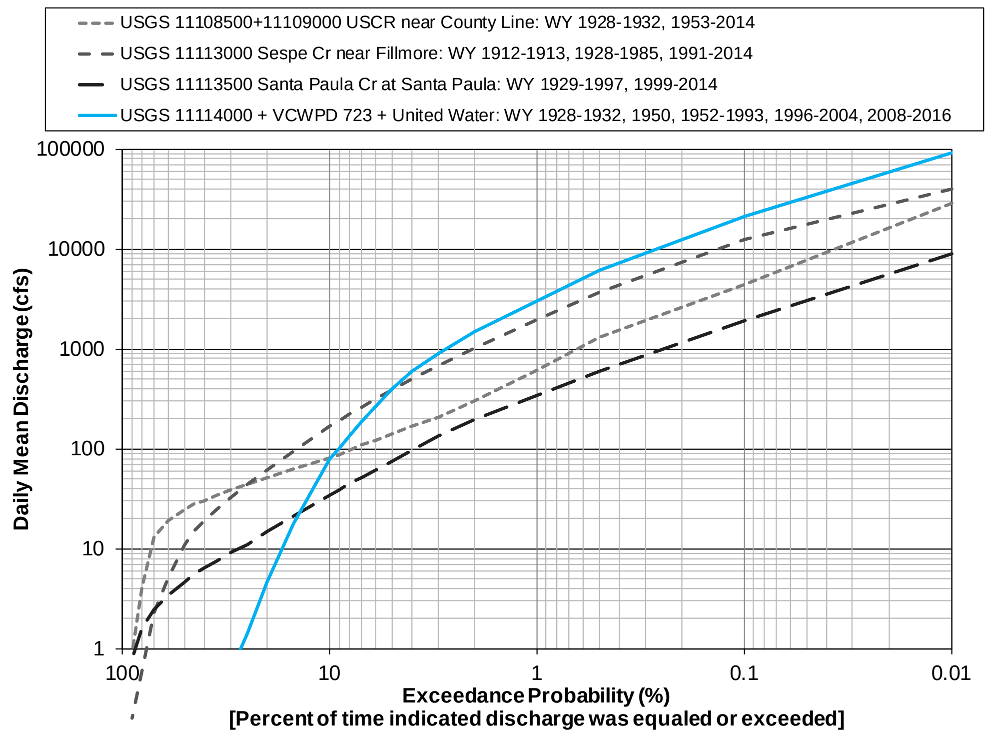


Figure 3-12. Daily mean flow characteristics (flow exceedance curves) of the lower Santa Clara River watershed based on compilation of available long-term gaging records through water year 2016 (source data from USGS, VCWPD, and UWCD; analysis by Stillwater Sciences).

3.3.3.2 VWRF discharge

Effluent is discharged from the VWRF via the effluent transfer station (ETS) to the Wildlife/ Polishing Ponds and then to the SCRE via the Parshall flume (Site D-1 or NPDES M-001A). The daily mean inflows to the Wildlife/Polishing Ponds at the ETS (NPDES M-001) and daily mean outflows to the SCRE outfall channel (gaged D-1) through December 2016 are presented in Figure 3-13. Flows recorded at both stations during the Phase 3 Study period are presented in Figure 3-14. Although problems associated with the ETS flow meter resulted in anomalously low ETS flows for several months, ETS and D-1 flows typically differ by approximately 2 cfs, which is attributed to water loss from the ponds via evaporation and groundwater seepage to the SCRE and semi-perched aquifer.

Comparison of monthly mean VWRF discharge with Santa Clara River discharge reveals a similar seasonal pattern with the greatest flow occurring in February and the least in the summer months (Figure 3-9 and Figure 3-14). In the wetter months, the effluent discharge (at D-1) has been about one-tenth the river flow. Whereas, during the late summer (August–September) the monthly mean VWRF discharge into the SCRE (at D-1) has been an order of magnitude greater than the river discharge into the SCRE. Thus, the effluent discharge is the dominant surface water source to the SCRE.

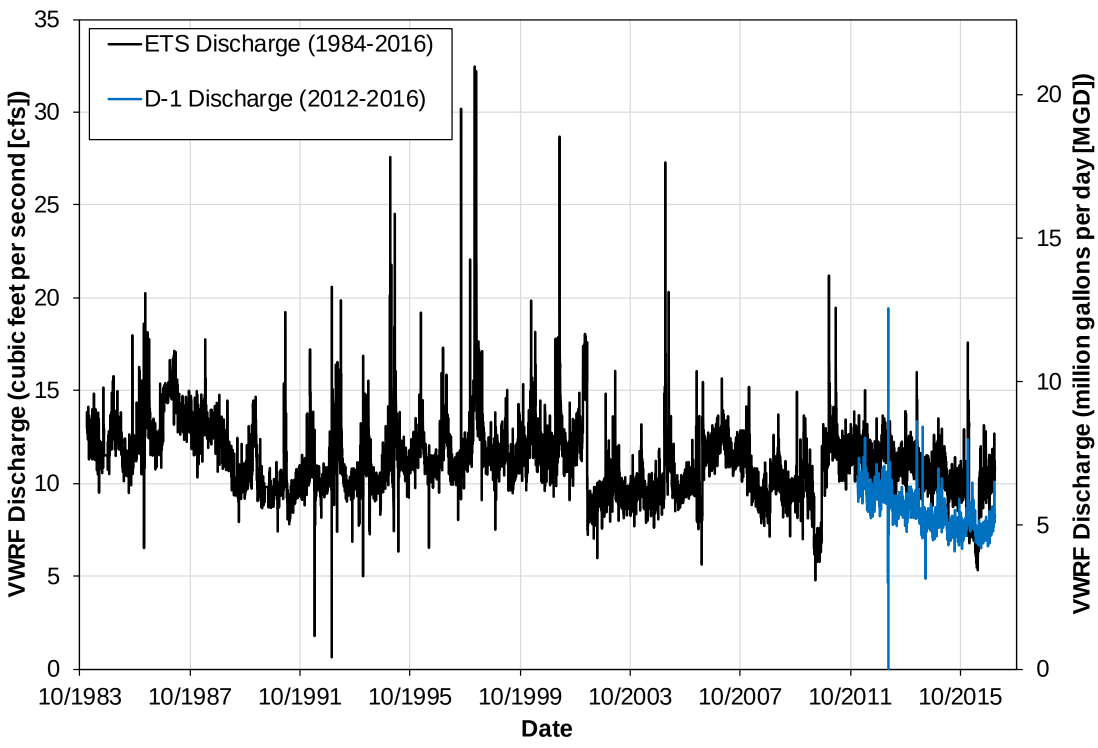


Figure 3-13. VWRF daily mean discharge measured at the ETS (1984–2016) and D-1 (2012–2016) monitoring stations (source data from VWRF; analysis by Stillwater Sciences).

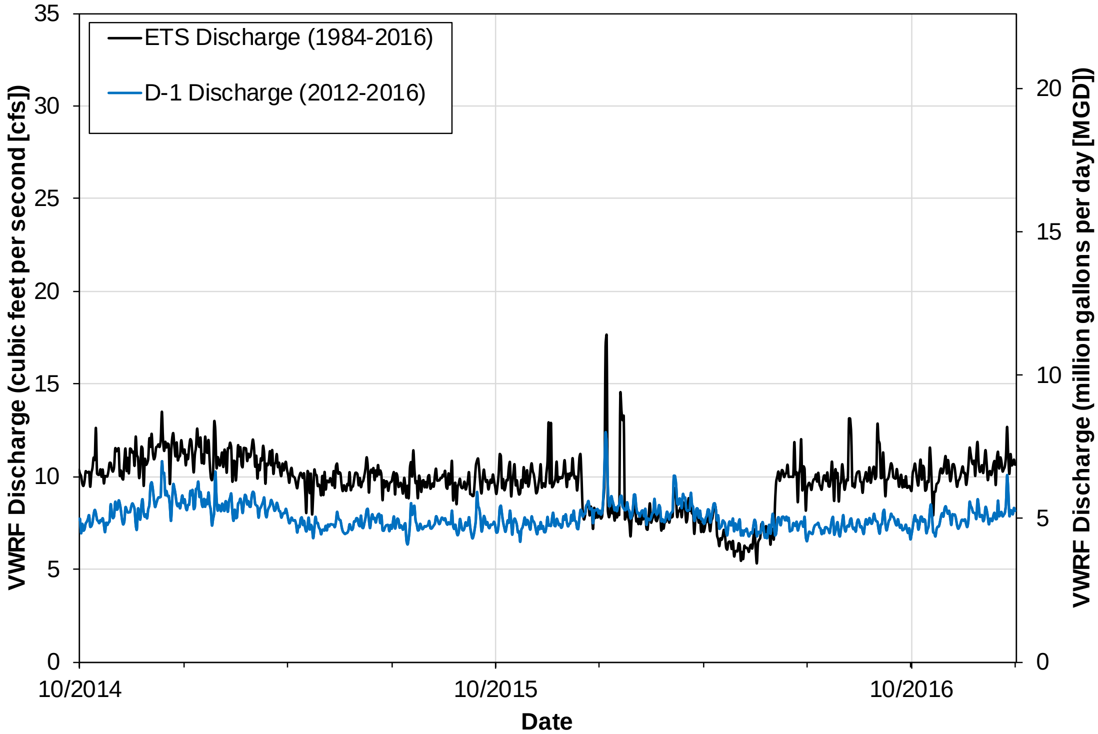


Figure 3-14. VWRF daily mean discharge measured at the ETS and D-1 monitoring stations during water years 2015–2016 (source data from VWRF; analysis by Stillwater Sciences).

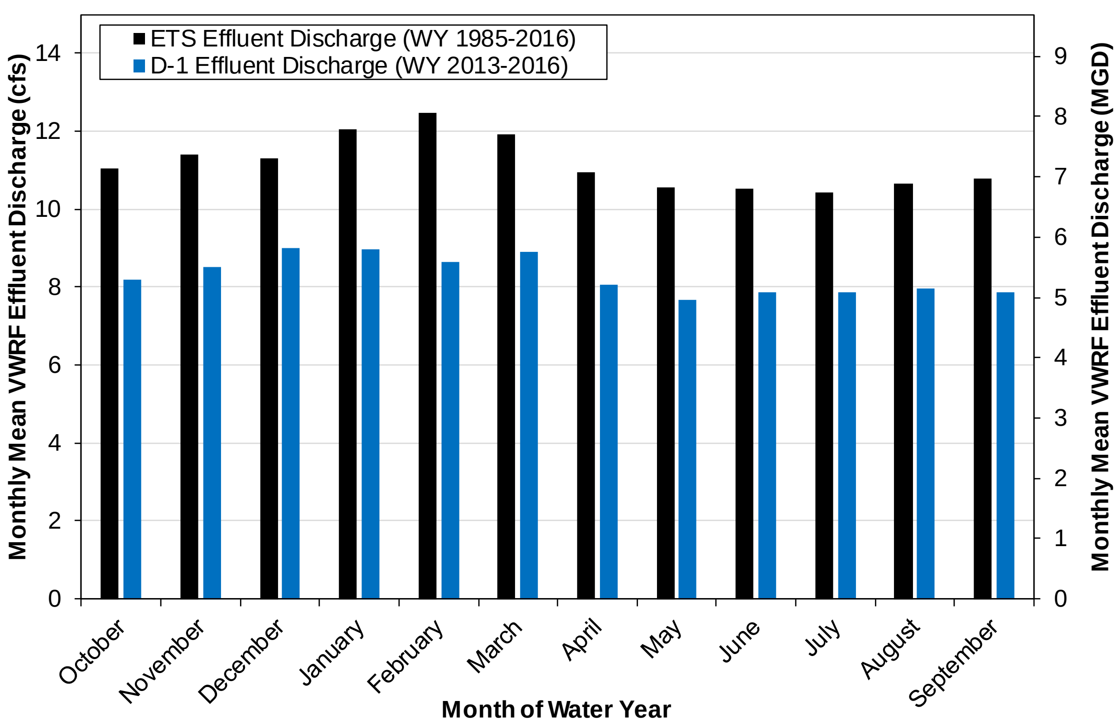


Figure 3-15. Monthly mean discharge characteristics at the ETS (1984–2016) and D-1 (2012–2016) monitoring stations (source data from VWRF; analysis by Stillwater Sciences).

3.3.3.3 Tidal flow

Tidal forces along the coastline strongly influence conditions within the SCRE during both open-mouth and closed-mouth periods. Tidal inflow to the SCRE during open-mouth periods can account for a large portion of the total SCRE water volume when the lagoon has drained following a storm-induced or other breaching event. The tides adjacent to the SCRE are mixed semi-diurnal consisting of two high tides (a lower high and a higher high) and two low tides (a higher low and a lower low) typically occurring each day (O'Hirok 1985). The closest continuously operating tide gauge that can be used to reflect conditions adjacent to the SCRE is near Santa Barbara, CA (NOAA Station 9411340). Between water-years 2009 and 2016, mean tide range (the difference between mean high water [MHW] and mean low water [MLW]) has been 3.5 ft and mean diurnal range (the difference between mean higher high water [MHHW] and mean lower low water [MLLW]) has been 5.1 ft during the most recent tidal epoch (1983–2001) (Table 3-2). 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).

Using the published MHHW (water surface elevation of 5.3 ft NAVD88 in Table 3-2) as the maximum tidally-induced water surface elevation in the SCRE, the tide can currently extend approximately 1,300 ft inland from the beach berm and inundate approximately one-quarter of the SCRE area. 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). Hourly tidal elevations recorded during the Phase 3 Study period are presented in Figure 3-16.

Table 3-2. Tidal elevations at the Santa Barbara monitoring station during water-years 2009–2016.

Tidal datum	Water-level elevation (ft NAVD88)		
	WY 1983–2001	**WY 2009–2016**	**WY 2015–2016**
Extreme High	7.14	7.38	7.25
Mean Higher High Water (MHHW)	5.31	5.28	5.43
Mean High Water (MHW)	4.55	4.53	4.74
Mean Diurnal Tide Level (MDTL)	2.61	2.73	2.91
Mean Tide Level (MTL)	2.72	2.80	2.98
Mean Sea Level (MSL)	2.70	2.78	2.98
Mean Low Water (MLW)	0.89	1.07	1.23
Mean Lower Low Water (MLLW)	-0.09	0.18	0.40

Source data from NOAA; analysis by Stillwater

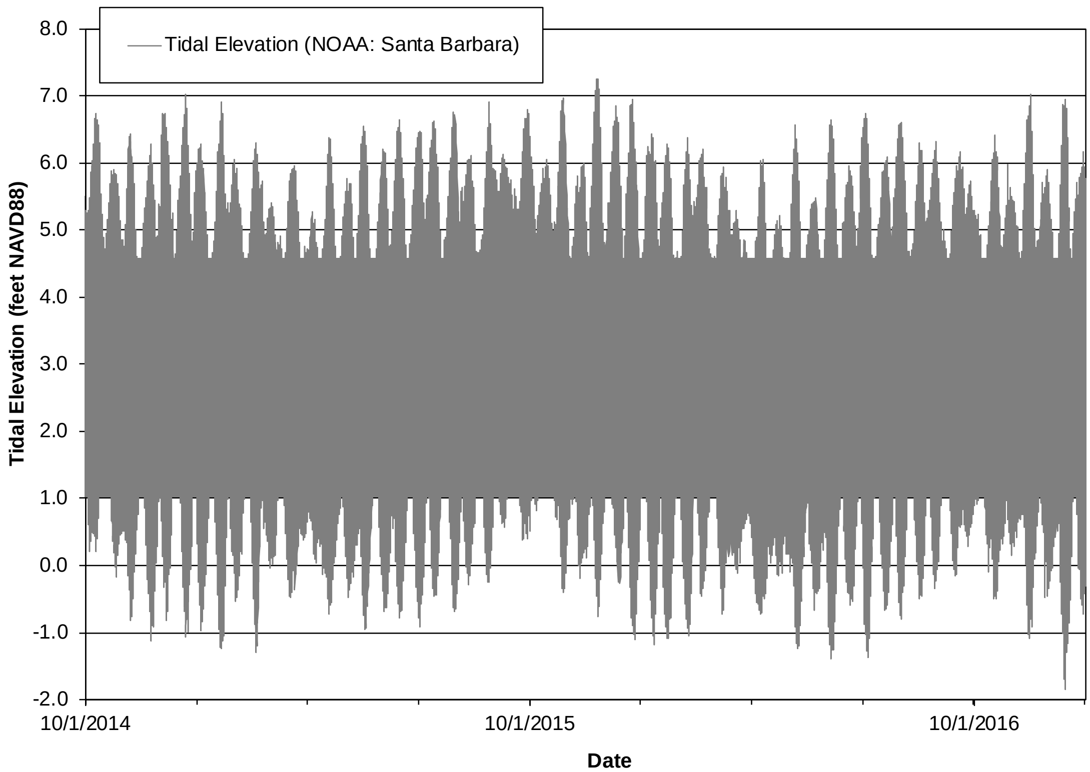


Figure 3-16. Hourly tidal elevation at the Santa Barbara monitoring station during water years 2015–2016 (source data from NOAA; analysis by Stillwater Sciences).

3.3.3.4 McGrath Lake connectivity

Hydrologic connectivity between the SCRE and McGrath Lake was addressed qualitatively during the Phase 1 and 2 studies. It is known that McGrath Lake, which lies approximately 5,000 ft to the south of the SCRE, is supplied by irrigation runoff routed through a network of drainage ditches installed throughout the floodplain area south of the river east of Harbor Blvd (see Figure 1-1). The lake is also supplied by groundwater sourced from the contributing watershed (URS 2005). Groundwater inflow has been estimated to contribute to 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 fields east of the lake to flood, with the flooding being exacerbated by groundwater flow out of the SCRE during closed-mouth, filling conditions (Nautilus Environmental 2005) (see Section 3.3.3 Groundwater Flows below). 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 2005). Under normal operations, the lake is pumped so that the water surface elevation remains between 2.7–3.6 ft above MSL (5.3–6.3 ft NAVD88) (Kennedy/Jenks Consultants 2002). The lake is pumped from its northern end via a series of pipes that discharge to a smaller outfall channel positioned approximately 900 ft to the west. This outfall channel is situated in the foredune complex of McGrath State Beach about midway between the shoreline and the oil and gas pumping facility operated by Venoco, Inc. (see Figure 1-1). Water pumped to the outfall channel eventually seeps through the beach berm reaching the ocean.

Between the Phase 1 and 3 study periods, it was observed that the southern end of the ponded lagoon of the SCRE would connect with the McGrath Lake outfall channel during closed-mouth conditions (see right-hand panel of Figure C-6 in Appendix C). To explore this surface-water connection in the Phase 3 Study, two new monitoring stations were established, including a water-surface stage recorder in McGrath Lake (E-3A) to continuously record the fluctuating water levels in the lake, and temperature sensors in the three discharge pipes in the outfall channel (E-3B) to detect occurrences of pumping activity. Potential surface-water connectivity was also monitored visually during the Phase 3 Study using field-based observations and aerial photography published by Google Earth and Planet Explorer online services.

The most recent surface-water connection between the SCRE and McGrath Lake's outfall channel was observed in aerial imagery captured on January 22, 2014. No surface-water connection was observed in the field or in aerial imagery at any time during the Phase 3 Study period. The mean water-surface elevation of the lake fluctuated between 5.6 and 9.3 ft NAVD88, and had an average stage of 6.5 ft NAVD88 between the monitoring period of January 28, 2015 and December 6, 2016 (Figure 3-17). Discrete moments of drawdown of the lake stage are apparent in the record which are indicative of pumping activity. The stage was at its highest levels between January and March 2016 in response to local precipitation events and high stage in the SCRE. The ocean-side of the outfall channel breached forming a tie-channel to the ocean in mid-March 2016. Active pumping of the lake into the outfall channel and continuous flow to the ocean was observed during this period. Given the overall landward migration of the shoreline near the SCRE over the past several years (see Section 3.2 Geomorphology), this phenomenon was likely a contributing factor to the breaching of the outfall channel as the distance between the channel and the ocean diminished to a point where the beach berm separating these features became vulnerable to erosion when ponding in the outfall channel and storm-induced wave wash coincided in March 2016. The tie-channel remained generally intact through the remainder of the Phase 3 Study period but surface flow there was only observed in association with pumping from the lake. The Phase 3 monitoring found no evidence that this flow connected with the SCRE, and Phase 3 analysis assumed no surface water connection between McGrath Lake and the SCRE.

Additional discussions on SCRE stage dynamics and the interactions of McGrath Lake on local groundwater connectivity with the SCRE are presented below.

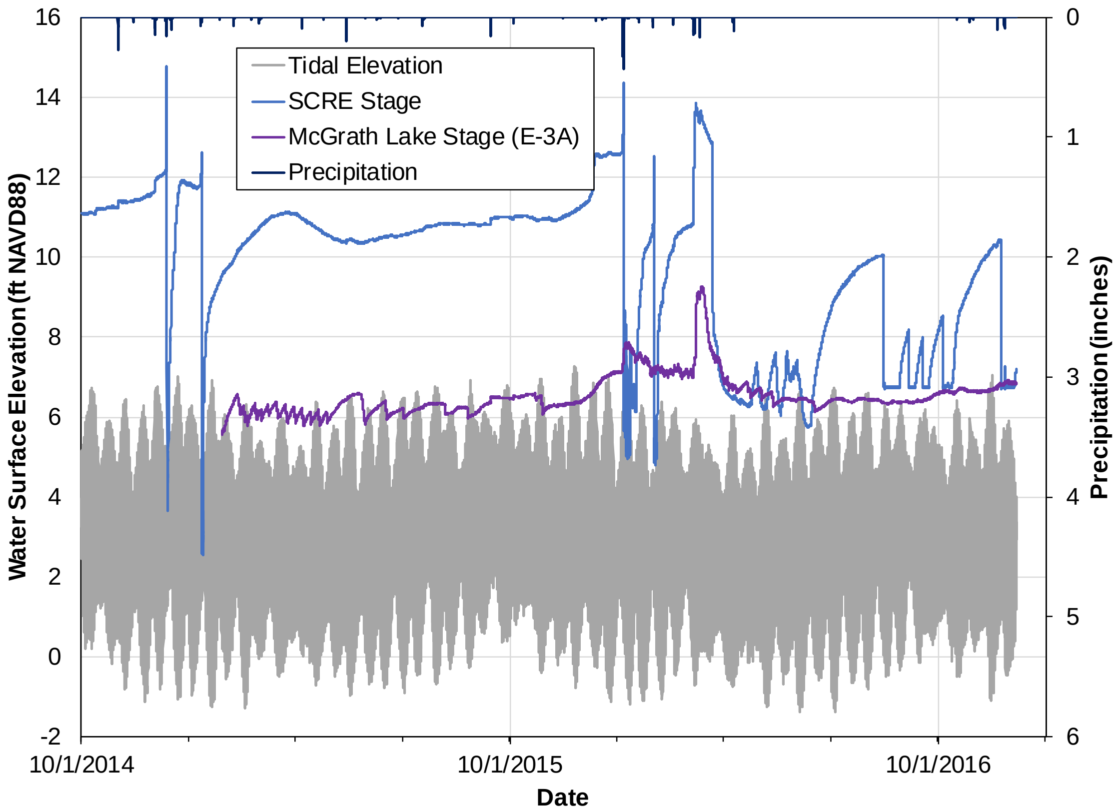


Figure 3-17. McGrath Lake water level, SCRE water level, Santa Barbara tidal elevations, and downtown Ventura precipitation amounts recorded during water years 2015–2016 (source tide data from NOAA, precipitation data from VCWPD, McGrath Lake and SCRE stage data from Stillwater Sciences; analysis by Stillwater Sciences).

3.3.4 Groundwater flows

The SCRE lies atop the Oxnard Plain—a broad alluvial, coastal plain formed at the lower reaches of the Santa Clara River and Calleguas Creek before each waterway flows into the Pacific Ocean. As part of the CDWR-designated Santa Clara River Valley Groundwater Basin, the Oxnard Plain comprises several subbasins. The SCRE and the lowermost reach of the Santa Clara River (extending 0.7 mile upstream of the Harbor Blvd bridge), or that area generally considered in the present study, overlie the Mound Subbasin (CDWR 2006a), while the river reach situated immediately upstream overlies the Oxnard Plain Subbasin (CDWR 2006b). Beneath the SCRE and lower river are several aquifers that together extend to depths of approximately 2400 feet before contacting non-water-bearing bedrock (Hanson et al. 2003). The aquifer located directly below the SCRE (referred to as the "Semi-perched aquifer") is shallow (<100 ft) and underlain by a clay layer, thereby disconnecting the SCRE from the deeper subbasin aquifers (Turner 1975, Hanson et al. 2003, CDWR 2006a). The Semi-perched aquifer consists of geologically young (i.e., Holocene age), fine-to-medium sand with interbedded clay layers (Turner 1975, Hanson et al. 2003). The low storage capacity of this relatively thin aquifer reportedly leads to frequent saturation by infiltrated precipitation and agricultural irrigation in the surrounding floodplain areas (Hanson et al. 2003).

The following characterizes the local groundwater dynamics based upon monitoring activities conducted throughout the SCRE and vicinity during the Phase 1 through 3 studies.

3.3.4.1 Groundwater monitoring

The Phase 1 Study relied upon information on local groundwater conditions derived from published literature and data monitored in three monitoring wells installed south of the SCRE: GW-1, GW-2, and GW-3 (see Figure 2-1; Table 3-3). The present study continued to record groundwater levels at these three wells, in addition to two additional wells installed during Phase 2 on lands owned by The Nature Conservancy (GW-4 and GW-5), and two existing wells at the Bailard Landfill operated by the Ventura Regional Sanitation District (GW-6 and GW-7). The original three wells were installed along a north-south transect within McGrath State Beach through the foredune complex between the lower SCRE sough bank and McGrath Lake. These wells were installed within the surface Semi-perched aquifer to a depth of approximately 25 feet on October 26, 2009 and were each equipped with an electronic water-level sensor to continuously record groundwater depths below the ground surface. The newer Phase 2 monitoring-well transect was installed on August 14, 2012 upstream of the SCRE within the Semi-perched aquifer on the northern floodplain downstream of the Victoria Ave. bridge. One well (GW-4) is located adjacent to the river channel and the other (GW-5) is located farther north near Olivas Park Dr. The Bailard Landfill site on the southern floodplain has a network of monitoring wells in the Semi-perched aquifer, two of which (GW-6 and GW-7) were used during the Phase 2 Study. These two wells, located on the eastern side of the landfill, were modified by the landfill operators in fall 2014. Wells GW-1 through GW-5 were inspected by Stillwater Sciences on November 20, 2014 and their electronic water-level sensors (Solinst Levelogger pressure-transducers) were restarted to initiate continuous recording of water-level data for the Phase 3 Study. The modified wells on the Bailard Landfill site were inspected and equipped with new electronic water-level sensors by Stillwater Sciences on January 28, 2015. All sensors were programmed to record continuously at 30-minute intervals.

To further improve understanding of local groundwater inflow/outflow from the northern floodplain, groundwater levels were recorded at four new wells (GW-8, GW-9, GW-10, and GW-11) installed on January 21–24, 2015 upstream of the SCRE around the Olivas Links golf course property (see Figure 2-1; Table 3-3). These wells were installed within the Semi-perched aquifer down to depths of approximately 25 feet below ground surface. One well (GW-8) was installed near the river channel on the upstream (east) side of the golf course, and a group of three wells (GW-9, GW-10, and GW-11) were installed between the golf course and Harbor Blvd. in a triangular arrangement to better allow for estimation of local groundwater-flow directions and hydraulic gradients. These four wells were inspected and equipped with new electronic water-level sensors by Stillwater Sciences on January 28, 2015. All sensors were programmed to record continuously at 30-minute intervals.

Four additional groundwater monitoring wells were installed along the SCRE side of the VWRF’s Wildlife/Polishing Ponds to better estimate pond seepage into the SCRE (see Figure 2-1; Table 3-3). These new wells, GW-12 through GW-15, were positioned along a west-east transect and installed within the subsurface materials composing the man-made berms of the ponds. The wells were completed to depths of approximately 15 feet below ground surface. These four wells were inspected and equipped with new electronic water-level sensors by Stillwater Sciences on January 28, 2015. All sensors were programmed to record continuously at 30-minute intervals.

Subsurface materials encountered during installation of the monitoring wells ranged from clay to gravel-sized particles, as logged by state-licensed geologists with Hopkins Groundwater Consultants, Inc. (Table 3-3). Wells GW-1 through GW-3 were located primarily within fine to coarse-grained, loose sands, which have relatively high hydraulic conductivities of about 30 to 200 feet per day. This estimate of hydraulic conductivity is consistent with that previously estimated for the sand beach dunes around McGrath Lake (URS 2005). On the north floodplain, substrates were found to be vertically mixed. The stratigraphy of the area near GW-4 and GW-5 included silty sand layer overlying a gravelly sand layer overlying a clay layer. The stratigraphy near GW-8 through GW-11 was variable, consisting on some amount of clayey, silty, gravelly sands. Hydraulic conductivities estimated for these substrates vary approximately between 1 and 100 feet per day, based on published values for clayey sands ($2.8x10^{-3}$–$2.8x10^{-1}$ ft/day) and gravelly sands ($2.8x10^{0}$–$2.8x10^{2}$ ft/day) (Fetter 2001).

All 15 groundwater monitoring wells were visited on a quarterly schedule during the 2-year duration of the Phase 3 Study, during which data were downloaded from the water-level sensors and manual measurements of groundwater levels were recorded to verify sensor readings. The wells and monitoring equipment were also inspected and, as necessary, corrected for impairment. The compiled data were subsequently used to create a continuous time-series of groundwater elevations for the entire period of record (see below). The sensors were programmed to continue monitoring groundwater levels following the final download event in early December 2016.

Table 3-3. Summary of groundwater monitoring wells utilized for the Phase 3 water balance evaluation.

Monitoring well No.	Well installation date [a]	Well location [b]				Well depth (feet below ground surface) [b]	Generalized lithological description [a]	Water-level recording periods [b]
		General area	Northing (feet, CA State Plane Zone 405)	Easting (feet, CA State Plane Zone 405)	Ground surface elevation (feet, NAVD88)			
GW-1	10/26/09	McGrath Beach south of SCRE	1907393.064	6180714.357	15.3	26.9	Fine to coarse-grained, loose sand	10/26/09–4/8/14, 11/20/14–12/6/16
GW-2	10/26/09		1906482.767	6181077.055	13.6	26.7	Fine to coarse-grained, loose sand, with some gravel	
GW-3	10/26/09		1904861.303	6181353.422	17.7	26.5	Fine to coarse-ground, loose sand with some gravel	
GW-4	8/14/12	North Floodplain downstream of Victoria Ave. (TNC Property)	1911186.427	6193368.577	44.6	25.1	Silty sand over gravelly sand over clay	8/31/12–4/23/14, 11/20/14–12/6/16
GW-5	8/14/12		1912965.499	6193285.989	43.3	11.4		
GW-6	Unknown	South Floodplain downstream of Victoria Ave. (Bailard Landfill)	1909313.212	6192715.541	42.6	29.6	Unknown	6/13/12–4/23/14, 1/28/15–12/6/16
GW-7	Unknown		1908621.946	6192695.108	45.3	18.5	Unknown	
GW-8	1/24/15	North Floodplain upstream of Harbor Blvd. (Olivas Links)	1911251.146	6186579.351	27.0	25.9	Silty sand over gravelly sand	1/28/15–12/6/16
GW-9	1/22/15		1910859.609	6182012.199	24.3	26.3	Silty sand over coarse sand over clayey sand over sandy clay over mixed sand	
GW-10	1/21/15		1911093.599	6183402.291	17.1	25.8	Sand over silty clay over mixed sand over coarse sand	
GW-11	1/23/15		1912833.278	6182298.339	21.0	25.9	Silty sand over gravel over clayey sand over gravel over mixed sand over clayey sand over mixed sand	
GW-12	1/17/15	Wildlife/Polishing Ponds berm (VWRF)	1911716.082	6179219.648	24.6	20.8	Fine to coarse, loose silty sand	
GW-13	1/16/15		1911370.010	6179918.619	23.3	16.1		
GW-14	1/20/15		1911001.473	6180574.632	23.0	15.6		
GW-15	1/21/15		1910903.360	6181043.831	21.7	15.8		

[a] Construction of wells GW-1 through GW-5 and GW-8 through GW-15 and their lithologic characterization conducted by Hopkins Groundwater Consultants, Inc.
[b] Well-head survey, well measurements, and groundwater-level recording conducted by Stillwater Sciences.

3.3.4.2 Groundwater-flow dynamics

Groundwater levels near the SCRE have been observed during the Phase 1 through 3 studies to fluctuate in response to changing water surface levels in the SCRE and pumping of McGrath Lake into its outfall channel. The measurements taken since 2009 along the transect south of the SCRE (GW-1, GW-2, GW-3) reveal that groundwater flow is predominately directed towards the SCRE during open-mouth conditions and then directed towards the south when the SCRE becomes full following mouth closure. The groundwater levels recorded in the south bank wells and water-surface levels recorded in the SCRE since 2009 are presented in Figure 3-18. The groundwater levels, SCRE stage, and McGrath Lake stage data recorded during the Phase 3 Study period are presented in Figure 3-19a.

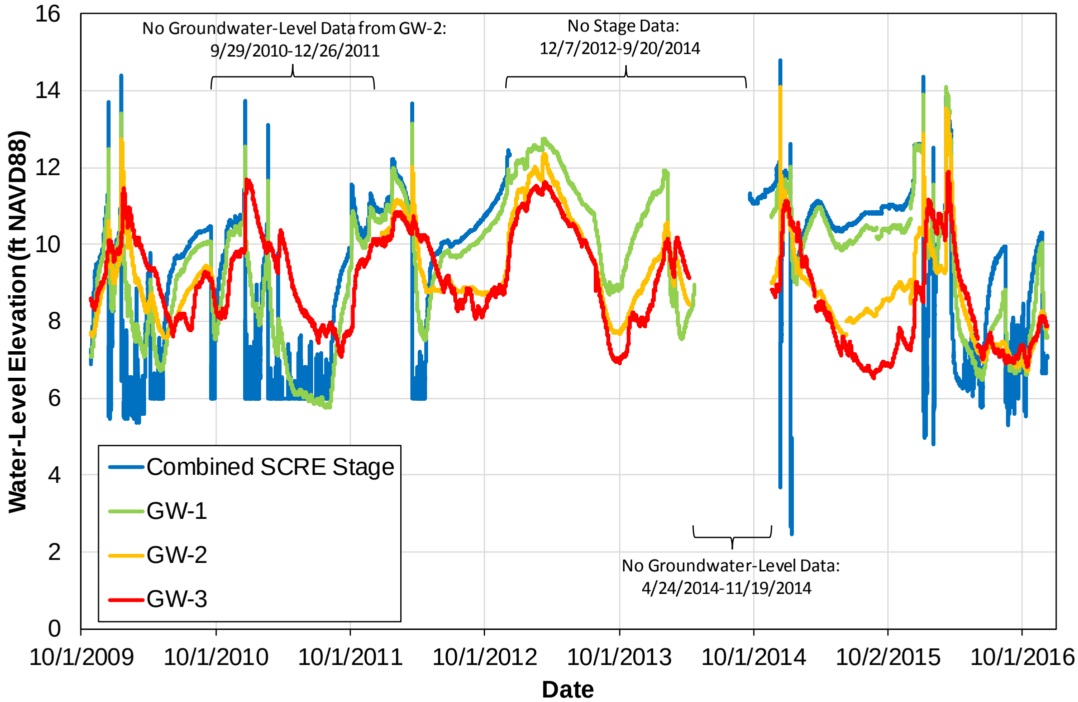


Figure 3-18. Response of groundwater levels on the south bank of the SCRE (GW-1 through GW-3) to changes in SCRE stage during water years 2009–2016 (source data from Stillwater Sciences; analysis by Stillwater Sciences).

Groundwater levels recorded closer to McGrath Lake at GW-3 were found to always remain above the water-surface level of the lake. Pumping of the lake into its outfall channel was observed to influence the local water table elevation near the nearby monitoring well (GW-3; see Figure 3-19a). The computed groundwater-flow direction and gradient using the water-surface elevations recorded in the SCRE and three monitoring wells (GW-1, GW-2, GW-3) were found to further support the previous finding that groundwater flows towards the SCRE during open-mouth conditions, and vice-versa. The groundwater-flow gradient generally has an inverse relationship to SCRE stage, whereby the gradient becomes increasingly negative (i.e., steeper in the direction away from the SCRE) when stage in the SCRE rises above approximately 10 ft NAVD88, and becomes increasingly positive (i.e., steeper in the direction towards the SCRE) when stage in the SCRE lowers below approximately 10 ft NAVD88 (Figure 3-19b). Overall, groundwater flow was directed north toward the SCRE for only 29% of the monitoring period between January 28, 2015 and December 6, 2016 because the SCRE was closed for most of this period.

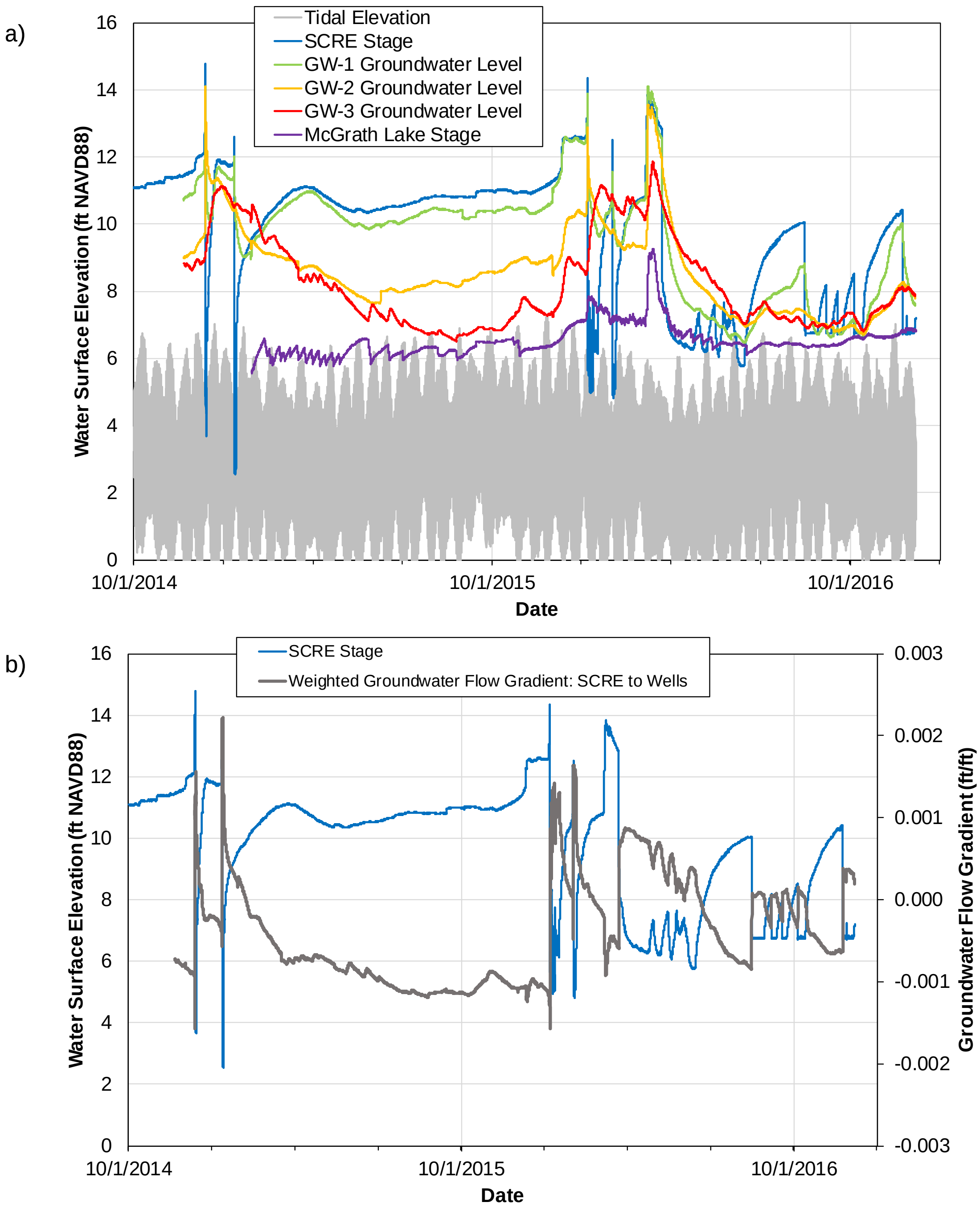


Figure 3-19. Response of groundwater levels on the south bank of the SCRE (GW-1 through GW-3) to changes in SCRE stage and McGrath Lake stage during the Phase 3 Study (2014–2016) (a) and computed weighted-average groundwater-flow gradient between the SCRE and wells (b). [Source tide data from NOAA, SCRE and McGrath Lake stage and groundwater-level data from Stillwater Sciences; analysis by Stillwater Sciences.]

Groundwater additionally enters the SCRE on the north side from the VWRF Wildlife/Polishing Ponds. The surface-water level in the ponds is controlled by a weir near the effluent discharge point in the Parshall Flume (near station D-1 and GW-15; see Figure 2-1) that maintains a relatively constant head in the ponds of approximately 19 feet NAVD88. Groundwater levels recorded continuously during the Phase 3 Study in wells GW-13, GW-14, and GW-15 fluctuated little compared to groundwater levels recorded elsewhere near the SCRE, with their values ranging between 16.2 and 17.6, 18.3 and 19.8, and 17.4 and 18.7 ft NAVD88, respectively (Figure 3-20a). These levels appear to have responded primarily to fluctuations in effluent discharge (see Figure 3-11) and direct rainfall (see Figure 3-7) based on the coincident timing of peak values apparent in the data records. Groundwater flow from the ponds, based on data recorded in GW-13, GW-14, and GW-15, was consistently directed south towards the SCRE based on their levels consistently remaining higher than the SCRE stage. As such, the computed groundwater-flow gradient was found to remain positive throughout the monitoring period and, further, the gradient had an inverse relationship to SCRE stage (Figure 3-20b), as found for the south bank groundwater conditions (see above).

Groundwater levels recorded closest to the ocean in GW-12, however, were considerably lower relative to the approximate water-surface level in the adjacent pond (see red-colored data plotted in Figure 3-20a). The groundwater levels were also lower than SCRE stage during closed-mouth conditions. These findings indicate the groundwater levels in the area near GW-12 were weakly influenced by water levels in the western pond and in the SCRE. Instead, the groundwater levels exhibited a close relationship to the diurnal tidal elevations based on their coincident fluctuations over time. Therefore, groundwater flow near this monitoring location is directed west towards the ocean rather than south towards the SCRE, indicating that some portion of water infiltrated to the subsurface from the western pond is delivered directly to the ocean.

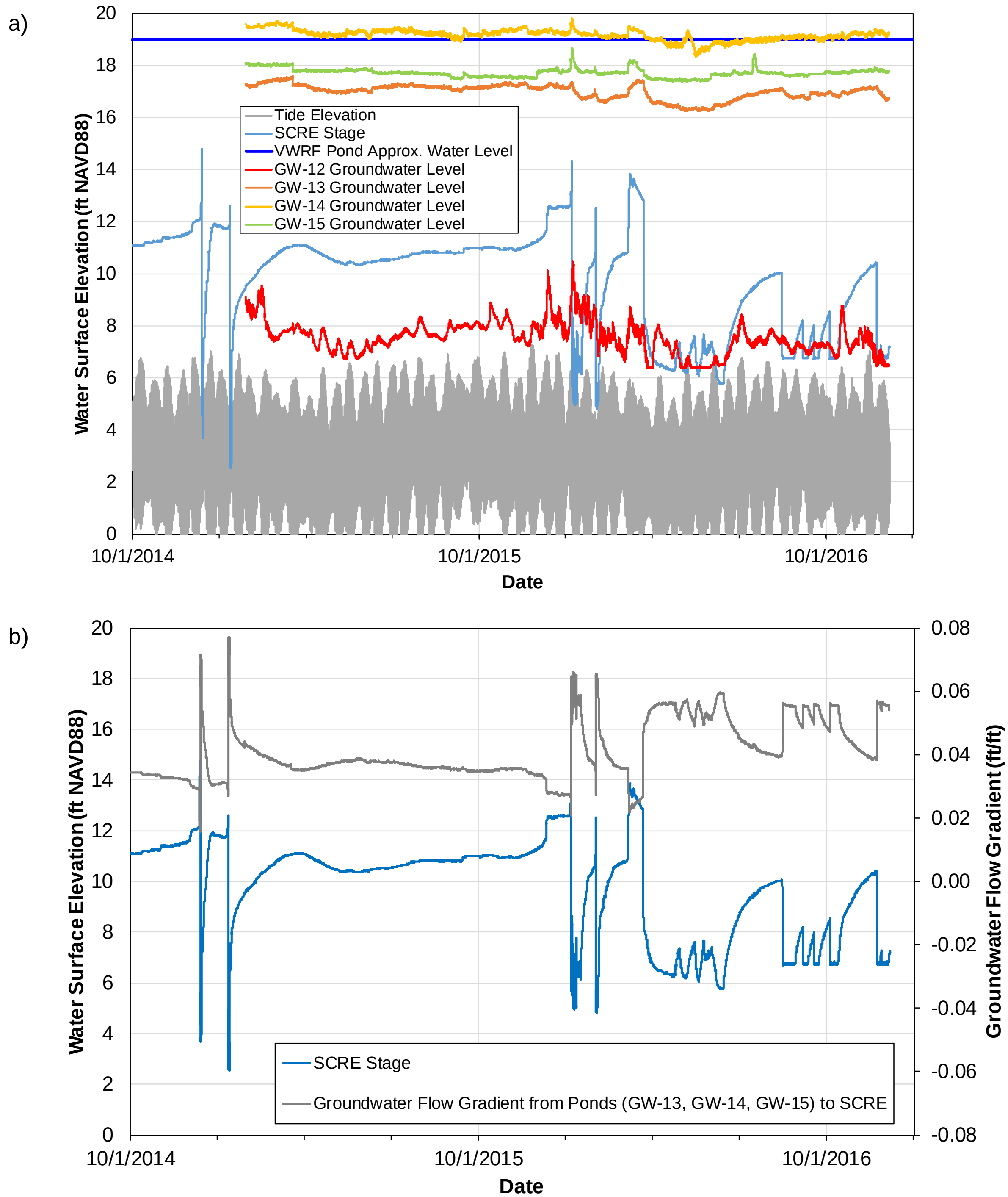


Figure 3-20. Response of groundwater levels on the north bank of the SCRE along the VWRF Wildlife/Polishing Ponds (GW-12 through GW-15) to changes in SCRE and VWRF-pond stage (a) and computed weighted-average groundwater-flow gradient between the SCRE and wells (b) during water years 2014–2016. [Source tide data from NOAA, SCRE stage and groundwater-level data from Stillwater Sciences; analysis by Stillwater Sciences.]

Groundwater input to the SCRE and lower river from the north-bank floodplain between Harbor Blvd and Victoria Ave. bridges may be differentiated into a western and eastern half based on the flow directions observed in the 8 monitoring wells installed upstream of Harbor Blvd bridge (GW-4 through GW-11) (see Figure 2-1). The western half, or that portion of the north-bank floodplain situated closer to the SCRE, had groundwater levels responding to changes in SCRE stage in a similar fashion as that observed in the south bank wells along McGrath State Beach. In these four wells (GW-8 through GW-11), groundwater levels typically fluctuated with changes in SCRE stage, river flow, and direct precipitation (Figure 3-21a), with groundwater-levels in wells positioned closest to the SCRE (i.e., GW-9 and GW-10) coinciding the most with SCRE stage. Groundwater levels were consistently higher than the nearby portion of the river bed elevation (or "thalweg") by approximately 2 to 8 feet, and were consistently lower than the approximate water-surface levels in the VWRF Wildlife/Polishing Ponds located to the west. The correlation between groundwater levels and SCRE stage did not appear to deviate during the presumed irrigation season in the drier months, indicating that irrigation activities at Olivas Links golf course and the nearby agricultural fields either: (1) do not cause seasonal changes in the local water table, or (2) do not influence the water table to the same degree as does the SCRE water volume.

The direction of regional groundwater flow from the western half of the north-bank floodplain, based on data recorded in GW-1 through GW-11 along with stage measurements in the SCRE and McGrath Lake, varied between west-southwest and north depending on the SCRE stage. Figure 3-22a presents typical regional groundwater elevation contours under closed-mouth (berm in place) conditions while Figure 3-22b presents typical regional groundwater elevation contours under open-mouth (berm breached) conditions. Data from the monitoring wells along the southern edge of the VWRF wildlife ponds (GW-12, GW-13, GW-14, and GW-15) showed the recharge mound in the vicinity beneath the ponds remains above the SCRE stage, but there were only minor changes in the well levels due to the influence of changes in SCRE stage. The plotted contours in Figure 3-22 along the with arrows represent the directions of groundwater flow in the vicinity of the SCRE and McGrath Lake. Data from GW-12, GW-13, GW-14, and GW-15 are not shown in Figure 3-22 since it was only a strong local condition immediately next to the ponds that did not vary with SCRE stage and the regional groundwater elevation contours would be unaffected by the local condition. Additionally, there was limited information on the rate of change between the VWRF wells and the nearest wells (GW-9 and GW-11), so plotting groundwater elevation contours around the VWRF wells would have had very high uncertainty.

Similar to observations on the south bank near McGrath State Beach, analysis of groundwater flow computed from the well triplets of GW-8–GW-10–GW-11, and GW-9–GW-10–GW-11 at 30-minute time-steps over the course of the monitoring period also show groundwater flow was consistently directed toward the SCRE during open-mouth periods and away from the SCRE during closed-mouth periods (see Figure 3-20a). As such, the groundwater-flow gradient generally has an inverse relationship with SCRE stage (Figure 3-21b). Overall, groundwater flow was directed south to southwest toward the river and SCRE for approximately 23% of the monitoring period between January 28, 2015 and December 6, 2016 because the SCRE was closed for most of this period.

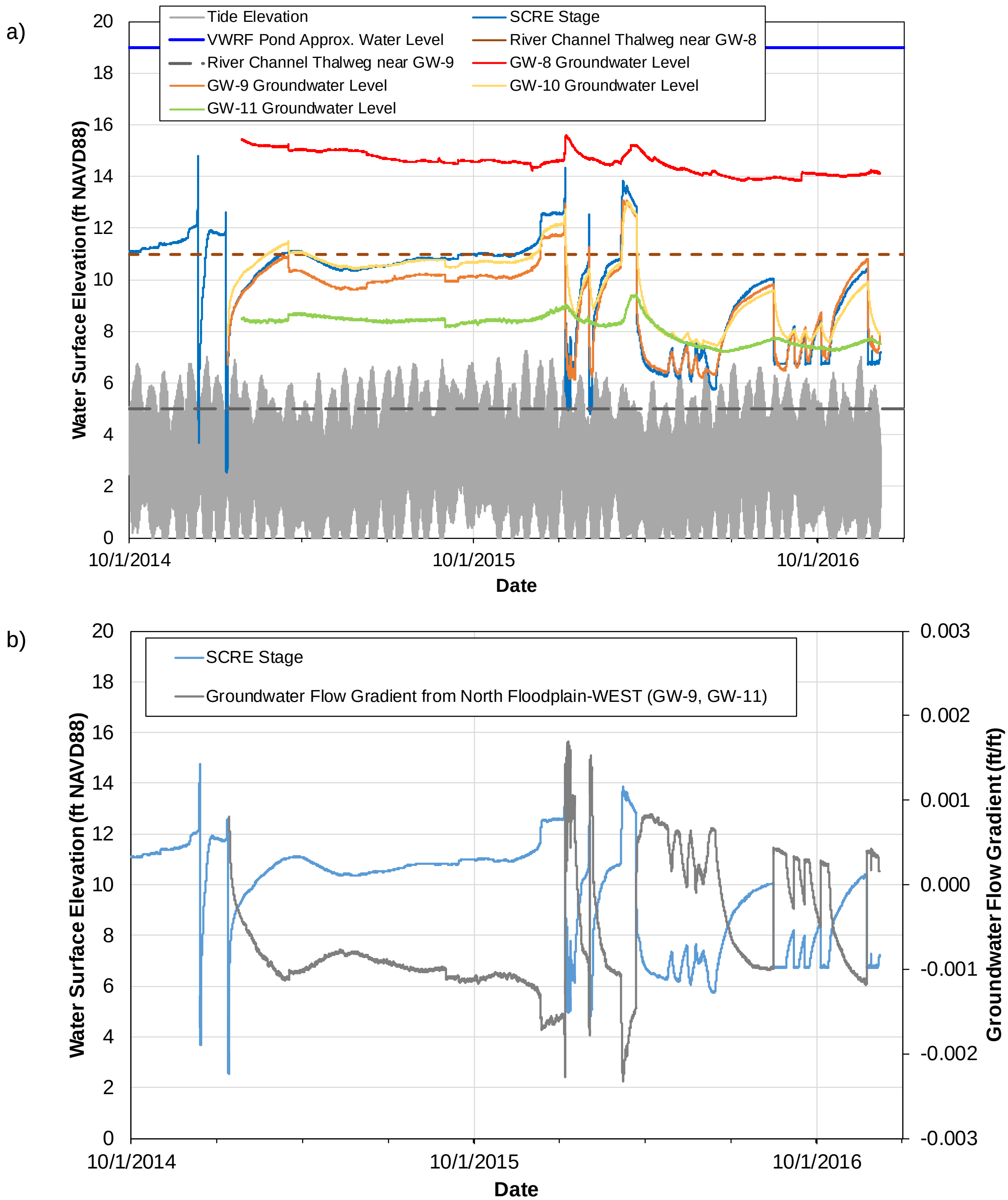
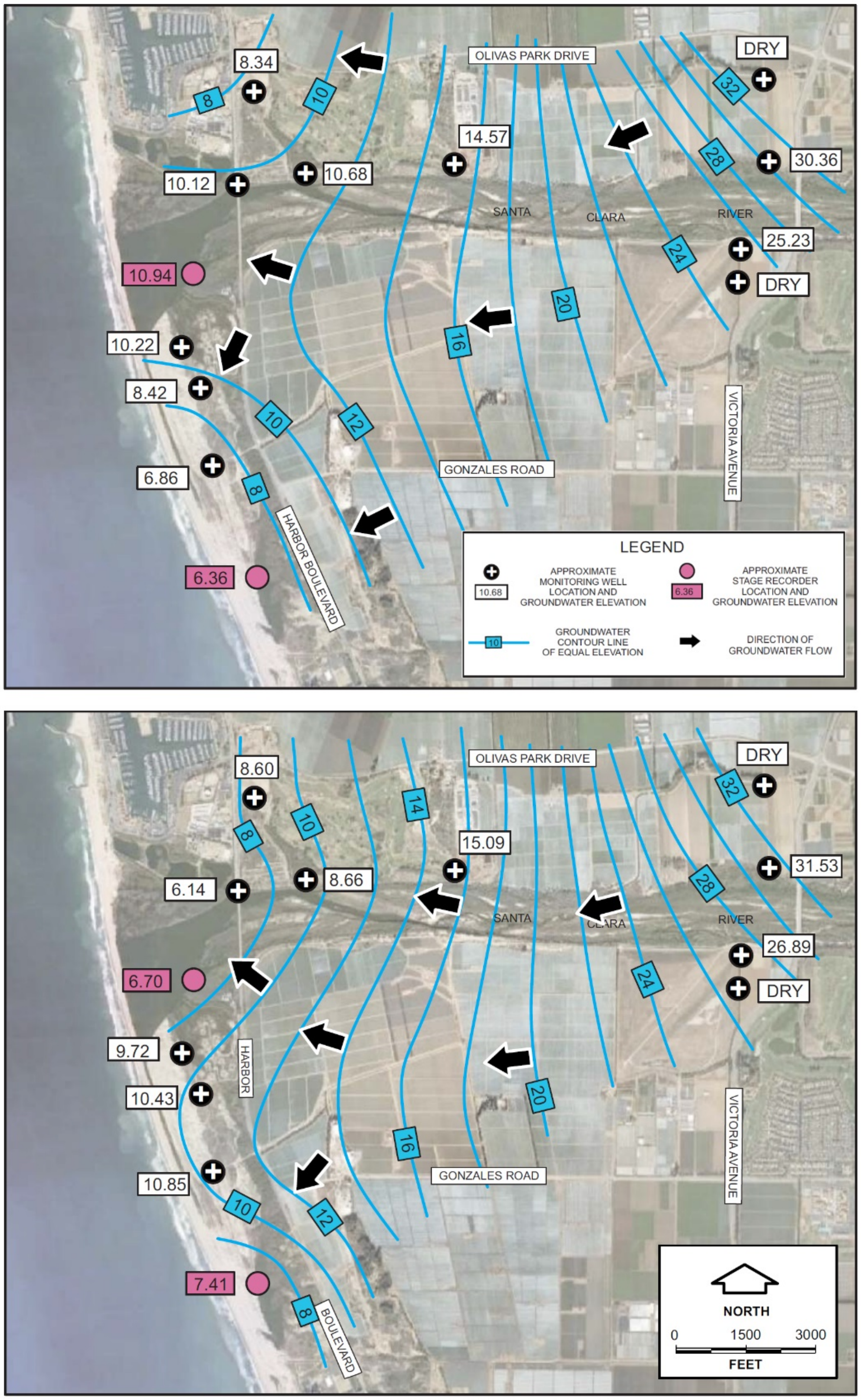


Figure 3-21. Response of groundwater levels on the north bank of the SCRE and lower river upstream of Harbor Blvd (GW-8 through GW-11) to changes in SCRE stage (a) and computed weighted-average groundwater-flow gradient between the SCRE and wells (b) during water years 2014–2016. [Source tide data from NOAA, SCRE stage and groundwater-level data from Stillwater Sciences; analysis by Stillwater Sciences.]

a)
b)



[Source: Hopkins 2018]

Figure 3-22. Groundwater elevation contour map (a) October 1, 2015 (Berm in place) (b) January 17, 2016 (Berm breached).

Groundwater input from the eastern half of the north-bank floodplain, as determined from water levels recorded in GW-4 and GW-5 near the Victoria Ave. bridge (see Figure 2-1), fluctuated in response to runoff events in the river and gradually declines over the course of the water year (Figure 3-23a). The water table here was consistently higher than the river bed elevation along this transect, thus indicating that groundwater flow was directed consistently south towards the river channel. Additionally, these wells are positioned upstream of the hydraulic backwater created in the lower river during closed-mouth periods in the SCRE. Groundwater levels recorded south of the river along this transect at GW-6 and GW-7, located on the east side of the Bailard Landfill (see Figure 2-1), reveal that groundwater flow is similarly directed toward the south but, instead, is direct away from the river. Further, groundwater levels at the Bailard Landfill wells were consistently lower than the river bed elevation along this transect.

There were two notable periods during the Phase 3 Study when the wells located farthest from the river channel—GW-5 on the north bank and GW-7 on the south bank—were observed being nearly or completely dry (see Figure 3-23a). The first of these periods occurred between June 2015 and February 2016. The second periods for GW-5 and GW-7 occurred during October–December 2016 and May–December 2016, respectively. Because the groundwater-level trends of these two wells closely related to levels recorded in the nearby and adequately deeper wells of GW-4 and GW-6 during months outside of the dry-well periods, it is assumed that the water levels in GW-5 and GW-7 continued to follow patterns similar to GW-4 and GW-8, respectively, during the dry-well periods. Specifically, it is assumed that levels in GW-5 on the north bank were consistently higher than those in GW-4, thus indicating a consistent flow direction towards the river having a flow gradient steepest during periods when the difference in water levels were greatest (Figure 3-23b). Similarly, it is assumed that levels in GW-6 on the south bank were consistently higher than those in GW-7 (despite the data record depicted in Figure 3-23a suggesting otherwise during the dry-well periods), thus indicating a consistent flow direction away from the river. A groundwater gradient was not computed for this pairing of wells, however, due to the prolonged duration of the dry-well periods in GW-7.

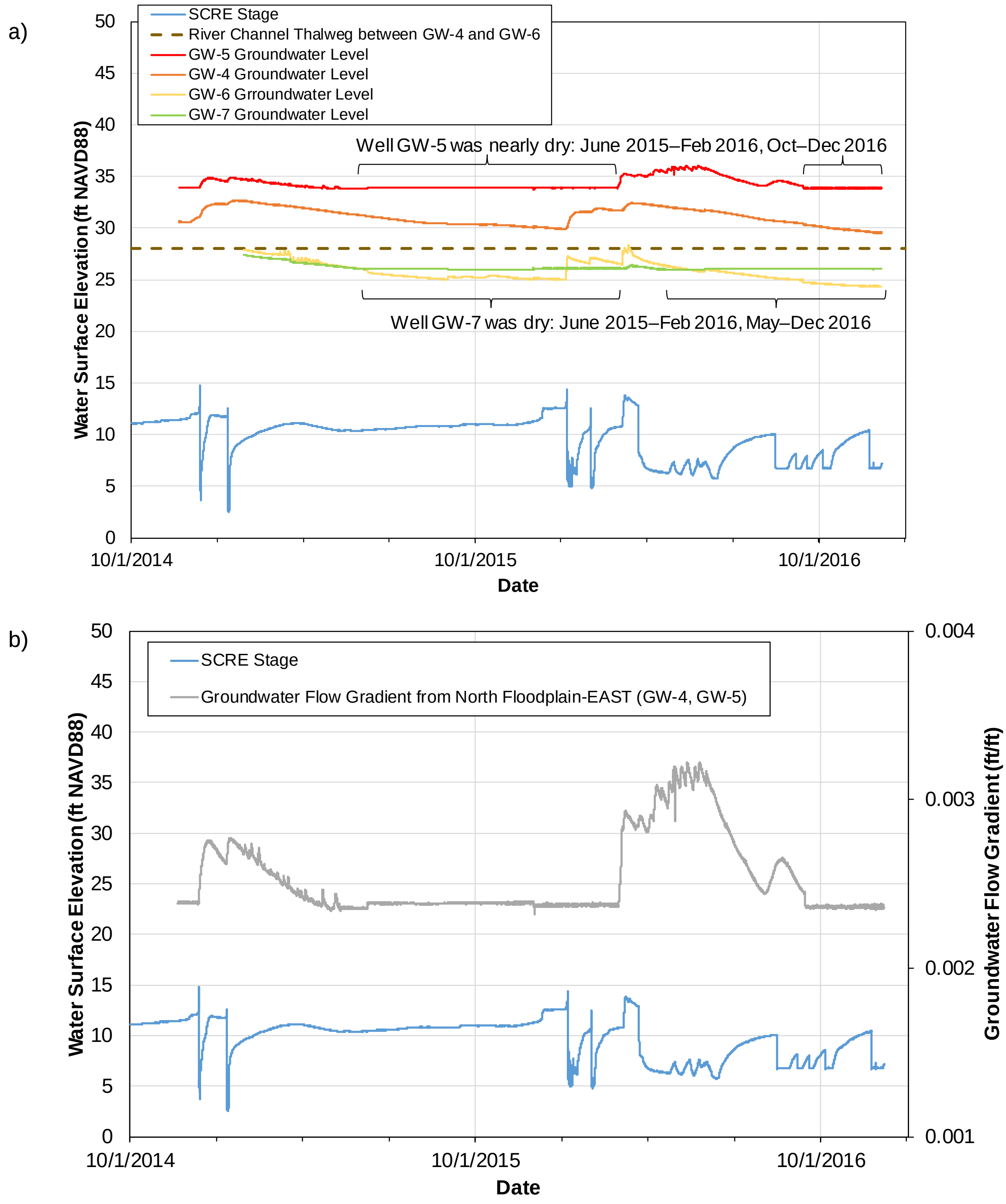


Figure 3-23. Response of groundwater levels on the north and south banks of the lower river downstream of Victoria Ave (GW-4 through GW-7) to changes in SCRE stage (a) and computed weighted-average groundwater-flow gradient on the north bank between the SCRE and wells (b) during water years 2014–2016. [Source SCRE stage and groundwater-level data from Stillwater Sciences; analysis by Stillwater Sciences.]

3.3.4.3 Beach berm seepage

Subsurface water seepage through the beach berm controls water volume in the SCRE during closed-mouth, low-flow conditions and, therefore, mediates mouth-breaching frequency. The length of the beach berm as it runs along the western margin of the wetted lagoon varies annually in response to the region's natural flood-drought cycle and on a decade-scale in response to human land-use activities, such as urban encroachment on the river corridor. Overall, because the berm is composed of uniform substrate materials (i.e., sand), the amount of lagoon water that may be transmitted through the beach berm is a function of the berm length. The groundwater-flow gradient through the berm may be computed as a function of SCRE stage relative to tidal elevation, where the gradient would be steeper when the SCRE stage exceeds tidal elevations (Figure 3-24). During the Phase 3 Study period, the estimated flow direction was consistently directed west toward the ocean except during brief moments of open-mouth periods with tidal elevations exceeding SCRE stage.

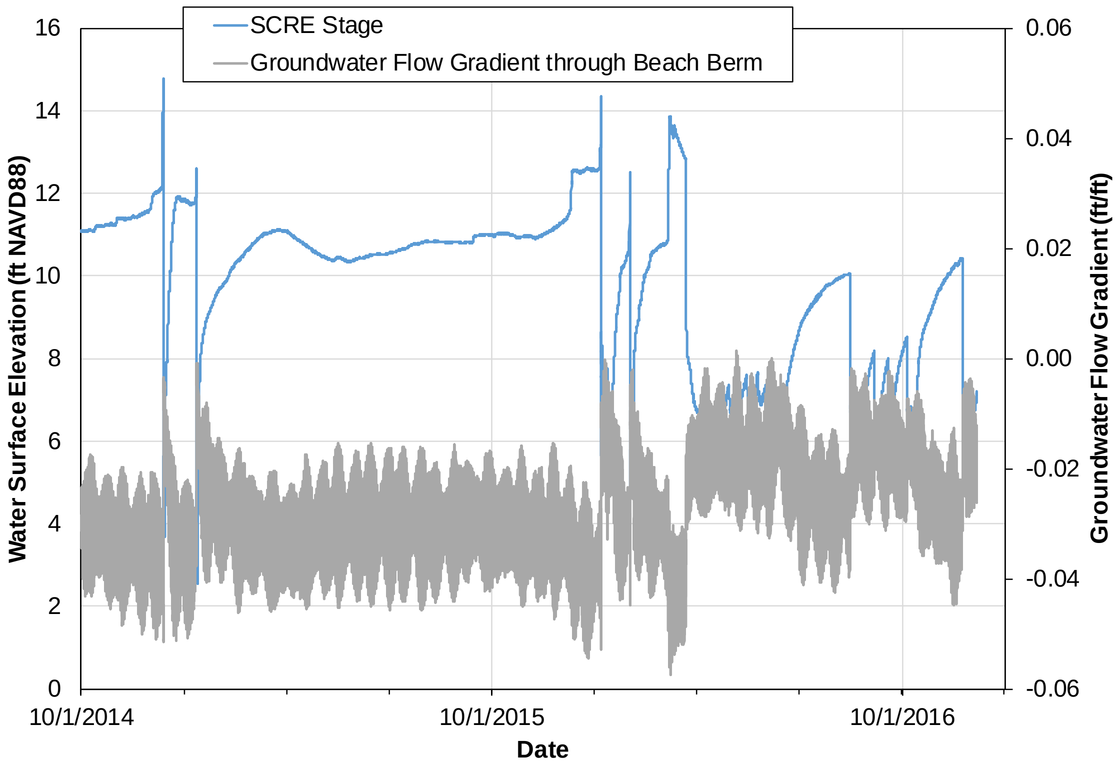


Figure 3-24. Computed groundwater-flow gradient through the beach berm between the SCRE and ocean during water years 2014–2016. [Source SCRE stage data from Stillwater Sciences; analysis by Stillwater Sciences.]

3.3.5 Mouth breaching dynamics

Breaching of the SCRE-mouth berm along the beach occurs episodically in response to both natural and anthropogenic influences. In many southern California estuaries, high river discharge during wetter months causes extended open-mouth periods, while low river discharge during drier months results in extended closed-mouth periods where the estuary lagoons fill with both river baseflow and subsurface water sourced from groundwater, ocean-water seepage, and wave overwash. Flow-induced breaching depends on the ability of the SCRE to store the high river flow and, therefore, is a function of flow magnitude and duration, and the pre-storm (antecedent) water volume in the SCRE. In addition to storm-induced wave impacts, wave overwash into the SCRE when the mouth is closed during any season can both add to the SCRE water volume and initiate the formation of a breach-mouth channel, the latter process causing rapid draining of the lagoon when the tide recedes. Prior to 2005, the nearly constant VWRF effluent discharge was observed to contribute to berm breaching when SCRE water surface elevation reached approximately 10 ft NAVD88 (Nautilus Environmental 2005).

Authorized and unauthorized breaching of the SCRE beach berm has also occurred in the past to alleviate risks of flooding adjacent to the lagoon. 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 SCRE mouth to prevent flooding of the campground. Due to natural resource considerations, this practice ended by 1985 (ESA 2003). Unauthorized third-party breaches have occurred on several occasions during the Phase 1 through 3 Study periods. Evidence of manual trenching of the beach berm near the VWRF outfall channel's confluence with the lagoon was observed on several occasions over the course of the Phase 3 Study period. The occurrences of sizeable trenching were first noted in March 2016 and several times thereafter through the remainder of the year (see photographic examples in Figure 3-25). It was believed by VWRF and State Parks staff that the unauthorized trenching may have been undertaken by homeless individuals residing near the SCRE to drain the lagoon and limit flooding of their encampments. The trenching location was consistently located on the north side of the lagoon near the confluence with the VWRF outfall channel that meanders through a riparian and wetland area.

The combination of the breaching influences has resulted in a SCRE mouth that has been open most consistently during wetter years, but also during drier years due to man-made activities. Daily observations of the SCRE beach berm made by VWRF staff from 1984 to present (99.9% of all days having observations) indicate the mouth has been open approximately 52% of observed days (Figure 3-26a). On a seasonal basis, the daily observations show that the mouth has been open with the highest daily frequency during March (74% of time) and the lowest daily frequency during September and October (36% of time), and open on average less than 50% of time from summer into the fall (July through October) (Figure 3-26b). The variability in the amount of time the mouth has been 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. 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 between 1984 and 2011, and the influence of VWRF discharge on mouth breaching during drier months of the year.

During the Phase 3 Study period, there were 13 distinct mouth-breaching events documented, which are listed in Table 3-4 below. The first four events that occurred between December 2014 and February 2016 were associated with storm-induced, albeit low-magnitude, runoff events that caused the SCRE stage to rise to a point that destabilized the beach berm and caused the SCRE mouth to open. These open-mouth events were relatively short-lived, lasting only a few days before river flows waned and wave energies reformed the beach berm closing the mouth. The following nine breach events were not associated with high river flows, but rather with manually trenching of the beach berm by unauthorized individuals. These open-mouth periods persisted many days longer than those formed during the storm-induced periods.

Figure 3-25. Photographic examples of a manually dug trench through the north end of the SCRE beach berm taken on March 21, 2016 (top) and July 20, 2016 (bottom). [Source photographs from ABC Laboratories.]

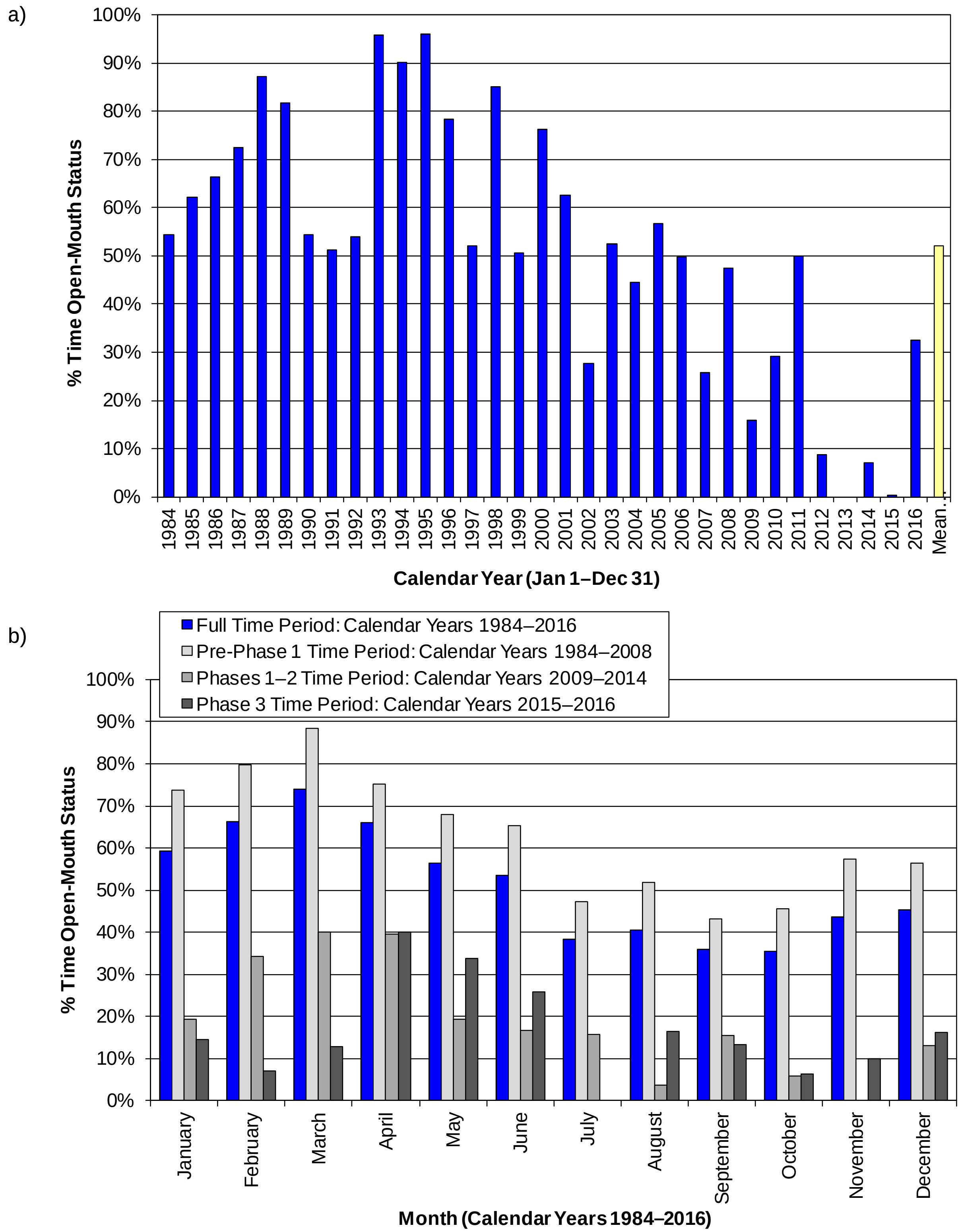


Figure 3-26. Percentage of time the SCRE mouth was open on an annual basis (a) and on a monthly basis (b) during 1984–2016. [Source mouth-status observations from City of Ventura; analysis by Stillwater Sciences.]

Table 3-4. SCRE mouth breaching and closure characteristics recorded during Phase 3 Study period.

Breach event	Date of initial open-mouth, approx. [a]	Peak SCRE stage immediately prior to breach event (ft NAVD88) [b]	Daily mean SCR discharge at time of breach (cfs) [c]	Assumed trigger for breach event	Date of initial closed-mouth, approx. [a]	Daily mean SCR discharge at time of breach (cfs) [c]	Tidal range at time of breach (ft) [d]
1	12/13/2014	14.8	639	Rainfall/runoff	12/15/2014	250	3.4
2	1/12/2015	12.6	20		1/14/2015	5	2.9
3	1/7/2016	14.3	559		1/15/2016	0	3.6
4	2/2/2016	12.5	13		2/6/2016	0	4.9
5	3/22/2016	12.8	0	Man-made disturbance to beach berm observed prior to event and measured moment of SCRE-water level lowering coincides with night-time hours	4/25/2016	0	4.9
6	4/30/2016	7.4	0		5/8/2016	0	7.5
7	5/16/2016	7.6	0		5/23/2016	0	6.1
8	5/25/2016	7.7	0		6/16/2016	0	4.6
9	8/16/2016	10.0	0		8/27/2016	0	4.7
10	9/3/2016	8.2	0		9/8/2016	0	2.8
11	9/19/2016	8.0	0		9/23/2016	0	4.8
12	10/5/2016	8.5	0		10/11/2016	0	3.6
13	11/25/2016	10.4	0		12/3/2016	0	5.0

[a] Estimated first day of open-mouth period based on visual observations made by VWRF staff.
[b] SCRE stage based on compilation of recorded water-surface elevations at monitoring stations used in the present study.
[c] SCR discharge from WY 2014–2015 provided by VCWPD from gaging station #723 at Victoria Avenue bridge [note: data were preliminary and are subject to change] and WY 2016 provided by UWCD from field measurements taken downstream of Freeman Diversion [note: data may not accurately represent inflows to the SCRE].
[d] Tidal range computed from tidal elevations provided by NOAA from the Santa Barbara tide gage.

3.3.6 SCRE stage and inundation dynamics

The water-surface elevation, or stage, in the SCRE fluctuates in response to various hydrologic inputs and outputs, and berm breaching frequency. SCRE stage recorded intermittently over the past decade exhibit typical patterns of rapid (<1-hr duration) lowering at the onset of mouth breaching and gradual (weeks) rising after initial mouth closure. During open-mouth periods, tidal exchange typically causes the SCRE stage to range between approximately 2 and 7 ft NAVD88 (Nautilus Environmental 2005, Stillwater Sciences 2011a). Surface water typically flows freely through the open mouth and there is very little lag time between a change in tidal elevation and associated change in SCRE stage during both filling and draining. Once closed, refilling of the SCRE has taken over a month. During low-flow conditions, berm breaching by overtopping has occurred at approximately 10 ft NAVD88, based on recordings made in 2004 (Nautilus Environmental 2005). When storm-induced river flows enter the SCRE during closed-mouth periods, the filling rate is much more rapid and the stage associated with berm breaching can be higher than the breaching stage during low-flow conditions. During the Phase 1 Study period, the mouth was observed breaching when the SCRE stage was between 10.7 and 13.5 ft NAVD88. And, during the storm-induced breaching events of the Phase 3 Study period, the associated SCRE stage ranged between 12.4 and 14.7 ft NAVD88. Together, these findings indicate a gradual increase in the stage associated with mouth breaching. Recordings of SCRE stage made intermittently since October 2009 by Stillwater Sciences and cbec, Inc. are presented in Figure 3-27. A synthesized record of stage during the Phase 3 Study period is presented in Figure 3-28.

Additional discussion on the inflows and outflows that together influence the water volume in the SCRE is presented in Chapter 4 Water Balance Evaluation below.

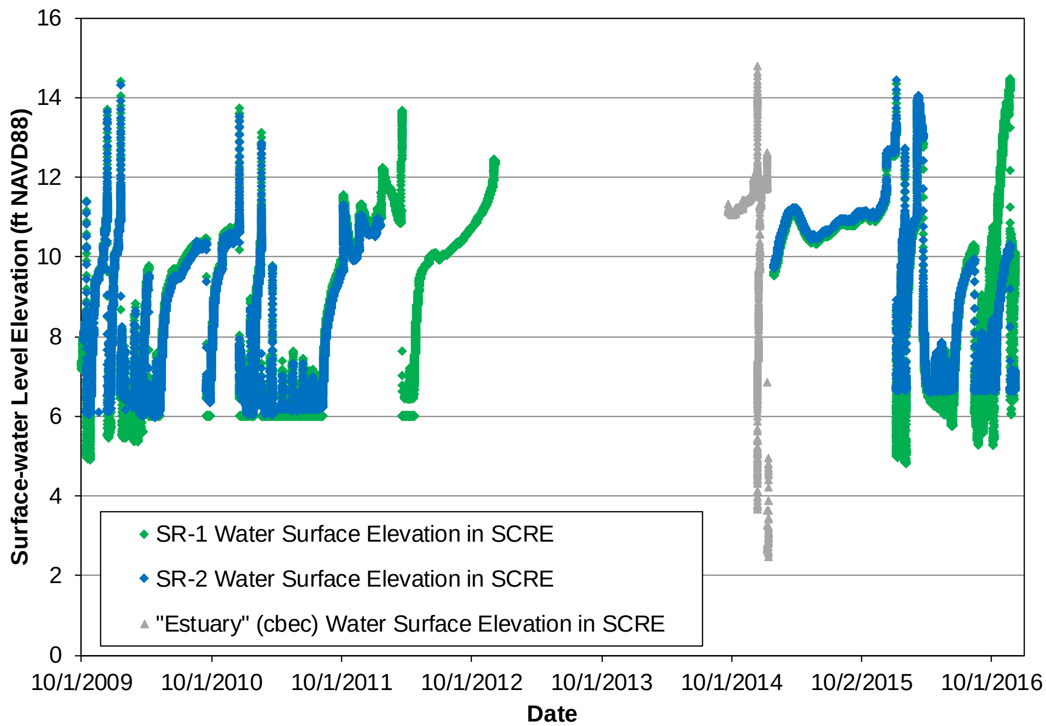


Figure 3-27. SCRE stage during water years 2010–2016. [Source SCRE stage data from Stillwater Sciences and cbec; analysis by Stillwater Sciences.]

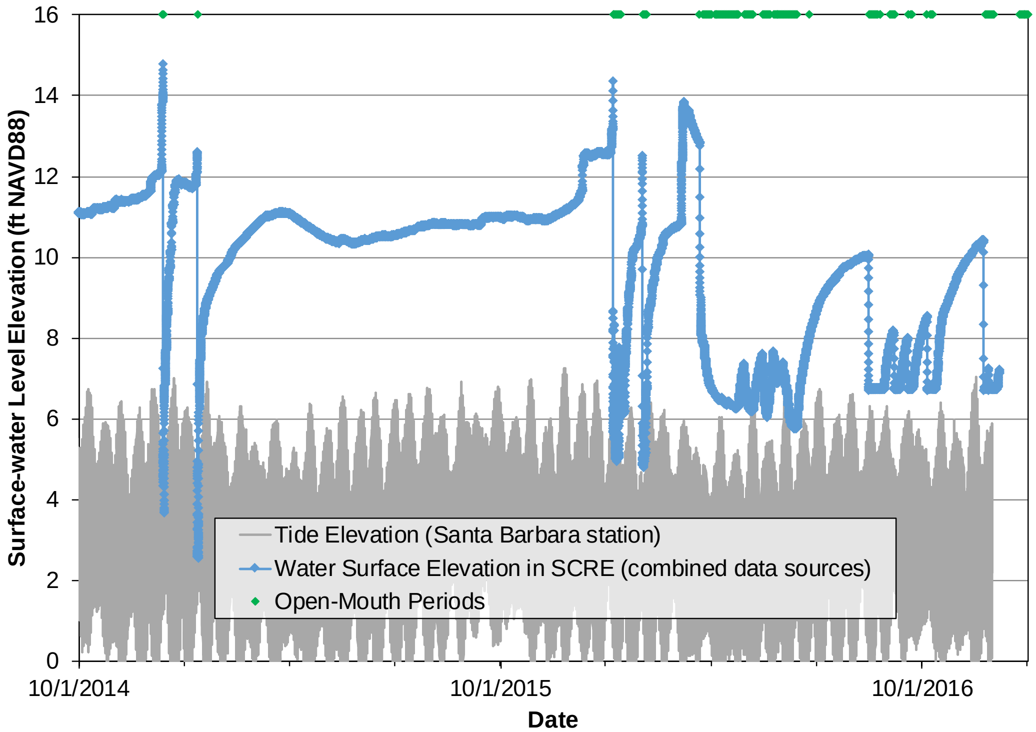


Figure 3-28. SCRE stage, tidal elevations, and open-mouth periods during water years 2015–2016. [Source tidal data from NOAA, SCRE stage data from Stillwater Sciences and cbec, mouth-status observations from City of Ventura; analysis by Stillwater Sciences.]

3.4 Water Quality

Physical and chemical water quality conditions within the SCRE are highly variable on an annual basis due to the combination of surface and groundwater inflows, seasonal variability in the SCRE mouth conditions, ocean exchanges, as well as seasonal meteorological variations. To improve understanding of the functioning of the SCRE, this section includes: assessment of contemporary SCRE physical and chemical water quality conditions based on recent data collected since VWRF facility upgrades were completed in 2011. Although a discussion of historical (1997–2010) water quality conditions is provided in the Phase 1 study, this section includes analysis of SCRE *in situ* and analytical water quality, trophic status, toxicity, and CEC results based on recent surface water and groundwater data (2012–2016). SCRE water quality data analyzed in this report for the SCRE were compiled from two sources including:

1. 2015–2016 data collected in accordance with the Phase 3 Monitoring Plan (Stillwater Sciences 2015); and
2. 2012–2016 data collected as required under the NPDES permit (#CA0053651) (Ventura 2012, 2013, 2014, 2015, 2016).

For the purposes of the analyses in this report, all surface water quality data (Table 2-2) were grouped by their general location within the SCRE, including VWRF sites (ETS [NPDES M-001], Wildlife Pond discharge [D1 or NPDES M-001A], and outfall channel [North Sonde and NPDES R-004]); Lower Estuary sites generally located between the eastern end of the McGrath State Beach campground and the open water areas east of Harbor Blvd. (South Sonde and

NPDES R-002, Central Sonde, E-1 and NPDES R-003); Upper Estuary and Riverine sites generally located east of Harbor Blvd. (E-2, NPDES R-005, and R1). Groundwater quality data, principally nutrient data, were also grouped by their general location within the SCRE and lower Santa Clara River, including “McGrath Beach” (GW1, GW2, GW3), “VWRF Wildlife Ponds” (GW 12, GW13, GW14, GW15), “North Bank West” including wells between Harbor Blvd. and the Olivas Park Golf Course (GW8, GW9, GW10, GW 11), “North Bank East” including wells east of the Olivas Park Golf Course (GW4, GW5), and “South Bank” including wells located within the Bailard Landfill (GW6, GW7).

3.4.1 Surface water quality

3.4.1.1 303(d) list of water quality limited segments of the Santa Clara River

Historically SCRE waters have exceeded Basin Plan objectives for several parameters including ammonia, nitrate, toxicity, bacteria, DO, and pH (RWQCB 2014). While the VWRF discharge currently meets NPDES discharge effluent limitations, there are several historical water quality impairments for indicator bacteria[8], nitrate, Toxaphane, and ChemA being addressed by a completed total maximum daily load (TMDL). The most recent Draft California 2016 Integrated Report for the 303(d) List/305(b) Report[9] recommends listing of the SCRE for apparent exceedances of water quality objectives for ammonia and pH, as well as maintaining the impaired status for toxicity (RWQCB 2017). In March 2017, the City submitted detailed comments on the proposed listings for ammonia, pH, toxicity, and indicator bacteria, requesting recalculation of exceedances and consideration of appropriate determinations based on water quality conditions that are typical and biologically attainable within estuaries, and using recent data that is more reflective of current SCRE conditions. The city also requested delisting of nitrogen and nitrate based on recalculation using appropriate data and correct averaging periods.

3.4.1.2 In situ Water Quality during grab sampling events

Summary statistics of *in situ* data collected during daylight hours since 2012–2016 are shown in Table 3-5 through Table 3-8 for both open and closed-mouth conditions during winter, spring, summer, and fall time periods. See Section 2.2 for data collection and QA/QC procedures, including information on data exclusions.

Water temperature

Water temperatures during grab sampling events exhibit expected seasonal increases from winter to summer. SCRE water temperatures vary from wintertime lows near 8ºC (46ºF) with the mouth open and 10.5 ºC (51ºF) with the mouth closed (Table 3-5) and summertime highs of near 26ºC (79ºF) during open condition, and nearer 28ºC (82ºF) with the mouth closed (Table 3-7).

Dissolved oxygen

During grab sampling events, which were conducted during daylight hours, DO concentrations were highly variable throughout the SCRE but generally higher during open mouth conditions than closed mouth conditions (Table 3-5 through Table 3-8). Although, DO was generally greater than 7 mg/L and suitable for aquatic life, some areas of the SCRE frequently do not meet the Basin Plan water quality objective for dissolved oxygen of >5mg/L (RWQCB 2014). During closed mouth conditions, the outfall channel location (R-004) was below the 5 mg/L water quality objective in 22% and 47% of the samples for the spring and summer periods, respectively. Though DO is highly variable, it appears consistently lower in the outfall channel relative to other

[8] Previously described as Coliform Bacteria
[9] The Draft report was under review in March 2017 during the completion of this report.

sampling locations. In lower estuary sites, SCRE DO in grab sample data was below the minimum objective more often when the estuary was open than closed (Table 3-9).

pH

On an annual basis, pH in the SCRE was relatively high with pH values ranging from 7.1–9.5 within the SCRE sites. Monitored pH levels commonly exceeded 8.0 (Table 3-5 through Table 3-8). In general, pH in daytime grab samples was higher due to photosynthetic uptake of dissolved CO_2 by algae, which is consistent with the wide daily variations in pH in other estuaries (Park et al 1958). pH concentrations were lower during open-mouth conditions, likely due to flushing with tidal exchanges.

Table 3-5. Summary of SCRE in situ water quality by mouth closure status during winter grab sampling events (2012–2016).

Winter		Temperature (Celsius)		Dissolved oxygen (mg/L)		Specific conductivity (us/cm)		Salinity (ppt)		pH (s.u.)	
Site[1]	Mouth	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel											
ETS (M-001)	All	22.7	20–28.5	7.4	5.4–8.4	NA	NA	NA	NA	7.0	6.6–8.9
D-1 (M-001A)	All	18	13.1–22	15.20	1–21.6	2,508	1,752–5,701	1.3	0.9–3.1	8	7.3–8.7
North Sonde (R-004)	Open	16.2	12.6–19.5	12.2	0.6–20.2	10,866	1,939–50,922	6.7	1.0–33.4	7.7	7.1–8.6
	Closed	14.8	9.1–20.8	13.4	1.1–35.4	6,111	989–29,756	3.5[2]	NA	8.4	7.3–9.3
Lower estuary											
South Sonde (R-002)	Open	14.3	11.8–17.1	9	1.1–12.9	37,985	11,276–50,106	24.3	6.4–32.8	7.6	7.3–8.0
	Closed	14.9	9.9–20.4	15.2	0.8–34.2	11,201	2,865–44,188	6.8	NA	8.6	7.4–9.5
Central Sonde (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	17	11.9–20.1	13.2	0.8–31.1	10,432	3,455–29,542	6.2	NA	8.7	7.7–9.3
E-1 (R-003)	Open	14.4	10.5–19.0	10.3	6.9–14.4	31,996	2,126–43,492	20.3	1.1–28.0	7.7	7.3–8.0
	Closed	14.5	8.2–20.3	14.2	1.1–28.9	10,054	1,752–41,813	6	0.9–26.8	8.5	7.5–9.2
Upper estuary and riverine											
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	17	12.2–20.1	13.8	0.9–26.6	10,440	3,425–30,627	6.1	1.8–19.1	8.7	7.7–9.2
(R-005)	Open	14.4	11.0–19.6	9.4	6.6–12.3	5,992	0–30,050	4	0.8–18.6	7.4	7.2–7.6
	Closed	15.2	10.2–20.5	13.7	0.8–37.1	8,843	2,497–33,548	5.2	1.3–21.0	8.4	7.2–9.1
R1	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

2 Because value based upon a single reading, no range is reported.

Table 3-6. Summary of SCRE in situ water quality by mouth closure status during spring grab sampling events (2012–2016).

Spring		Temperature (Celsius)		Dissolved oxygen (mg/L)		Specific conductivity (us/cm)		Salinity (ppt)		pH (s.u.)	
Site[1]	**Mouth**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**
VWRF, wildlife ponds and outfall channel											
ETS (M-001)	All	24.9	21–28.9	7	5.7–7.9	NA	NA	NA	NA	7.1	6.7–9.1
D-1 (M-0001A)	All	22.1	18.5–24.2	14.7	6.9–20.3	2,374	2,126–2,497	1.2	1.1–1.3	8.3	7.5–9
North Sonde (R-004)	Open	19.1	14.2–21.6	9.4	4.5–12.4	4,968	2,126–24,552	2.8	1.1–14.9	8	7.4–8.6
	Closed	20.3	16.0–22.9	8.7	2.8–25.1	3,546	2,126–12,578	1.9	1.1–7.2	8.1	7.4–9.4
Lower estuary											
South Sonde (R-002)	Open	17.9	12.0–22.1	8.9	2.6–14.0	11,918	1,182–50,515	7.2	0.6–33.1	8	7.6–8.8
	Closed	20.5	14.5–23.2	11	3.5–32.9	6,444	2,682–40,265	3.7	1.4–25.7	8.4	7.7–9.2
Central Sonde (NA)	Open	17.4 [3]	NA	11.2 [3]	NA	4,848 [3]	NA	2.6 [3]	NA	8.7 [3]	NA
	Closed	20.3	19.9–20.9	9.2	7.6–11.9	2,889	2,792–3,059	1.5	1.5–1.6	8.3	8.1–8.8
E-1 (R-003)	Open	17.6	12.4–21.1	8.4	1.7–14.8	12,736	2,312–48,741	7.7	NA	8	7.5–8.6
	Closed	20	14.5–23.0	10	1.1–27.8	4,649	2,126–35,711	2.6	NA	8.4	7.6–9.4
Upper estuary and riverine											
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	20.5	20.0–21.0	10	7.1–15.2	2,919	2,811–3,055	1.5	1.5–1.6	8.3	8.1–8.8
(R-005)	Open	16.3	11.9–17.8	8.2	1.0–13.8	4,360	1,182–7,441	2.3	0.6–4.1	7.5	7.1–8.4
	Closed	20.2	14.0–23.3	10.2	0.8–22.0	4,396	2,497–21,826	2.4	1.3–13.1	8.2	7.6–9.3
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	18.8 [3]	NA	8.4 [3]	NA	3,707 [3]	NA	2.2 [3]	NA	7.3 [3]	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

[2] Because value based upon a single reading, no range is reported

Table 3-7. Summary of SCRE in situ water quality by mouth closure status during summer grab sampling events (2012–2016).

Summer		Temperature (Celsius)		Dissolved oxygen (mg/L)		Specific conductivity (us/cm)		Salinity (ppt)		pH (s.u.)	
Site[1]	Mouth	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel											
ETS (M-001)	All	27.4	25.4–30.8	6.7	4.4–7.8	NA	NA	NA	NA	7.1	6.9–7.5
D-1 (M-0001A)	All	24.8	22.5–27.1	11.1	3.9–17.7	2,319	2,126–2,682	1.2	1.1–1.4	8.1	7.5–8.9
North Sonde (R-004)	Open	21.7	21.2–22.0	7.4	3.5–10.6	7,981	3,412–11,440	4.5	1.8–6.5	8.2	8.0–8.4
	Closed	23.1	20.9–25.4	5.7	0.7–16.9	3,051	1,182–29,609	1.6	0.6–18.3	7.9	7.1–8.8
Lower estuary											
South Sonde (R-002)	Open	21.6	21.2–22.2	6.5	0.9–14.6	12,446	4,131–18,129	7.2	2.2–10.7	8.3	8.0–8.7
	Closed	23.6	20.0–26.0	9.9	0.5–20.9	3,733	2,497–18,129	2	1.3–10.7	8.8	8.0–9.4
Central Sonde (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	23.9	21.8–25.9	9.8	6.2–14.5	4,066	2,595–9,074	2.2	NA	8.7	8.5–9.0
E-1 (R-003)	Open	22.1	21.0–23.8	7.2	1.4–12.0	13,423	2,865–19,833	7.8	1.5–11.8	8.3	8.1–8.6
	Closed	23.4	19.8–28.1	9.2	1.7–17.7	3,441	1,373–17,035	1.8	NA	8.6	7.9–9.1
Upper estuary and riverine											
E-2 (NA)	Open	23.7	20.1–26.1	10.4	4.9–14.9	4,379	2,605–9,260	2.4	1.3–5.2	8.7	8.1–9.0
	Closed	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
(R-005)	Open	19.2	17.6–20.6	6.5	4.7–9.7	4,576	4,488–4,665	2.5	2.4–2.5	7.4	7.3–7.5
	Closed	23.5	17.2–26.6	9.2	1.0–19.6	3,648	2,126–18,284	2	1.1–10.8	8.4	7.1–9.1
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	20.7 [2]	NA	12.9 [2]	NA	3,548 [2]	NA	2 [2]	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

[2] Because value based upon a single reading, no range is reported

Table 3-8. Summary of SCRE in situ water quality by mouth closure status during fall grab sampling events (2012–2016).

Fall		Temperature (Celsius)		Dissolved oxygen (mg/L)		Specific conductivity (us/cm)		Salinity (ppt)		pH (s.u.)	
Site[1]	Mouth	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel											
ETS (M-001)	All	25.7	20.2–29.4	7	5.4–8.6	NA	NA	NA	NA	7.1	6.6–7.9
D-1 (M-0001A)	All	19.6	12.4–26	13.7	6.8–22.8	2,402	2,126–2,682	1.2	1.1–1.4	8.1	7.6–9
North Sonde (R-004)	Open	15.8	13.4–18.1	11.6	11.0–12.1	14,130	3,257–25,003	8.4	1.6–15.2	7.9	7.8–8.1
	Closed	18.1	9.3–24.7	10.9	0.8–25.6	4,404	2,126–29,609	2.4	1.1–18.3	8.2	6.7–9.2
Lower estuary											
South Sonde (R-002)	Open	13.8	12.9–14.7	9	7.6–10.5	33,913	19,221–48,604	21.6	11.5–31.7	7.7	7.5–7.9
	Closed	18.3	9.6–25.4	16.2	5.7–30.1	5,168	2,126–27,392	2.9	1.1–16.8	8.8	7.7–9.4
Central Sonde (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	18.7	11.8–25.2	8.4	6.2–10.9	6,287	2,588–14,778	3.5	1.3–8.6	8.6	8.1–9.1
E-1 (R-003)	Open	14.8 [2]	NA	10.4 [2]	NA	46,956 [2]	NA	30.5 [2]	NA	7.9 [2]	NA
	Closed	18	9.1–24.9	15	2.4–29.4	4,301	1,939–20,141	2.3	1.0–12.0	8.7	7.7–9.5
Upper estuary and riverine											
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	17.6	11.8–24.2	11.7	6.4–31.9	6,539	2,596–15,246	3.5	1.3–9.0	8.6	8.4–9.0
R-005 (R-005)	Open	13.3 [2]	NA	14.3 [2]	NA	4,665 [2]	NA	2.5 [2]	NA	NA	NA
	Closed	18.1	9.6–25.9	13	2.7–25.1	4,417	1,752–20,755	2.4	0.9–12.4	8.3	7.1–9.1
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	14 [2]	NA	7 [2]	NA	3,071 [2]	NA	2.1 [2]	NA	7.2[2]	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

[2] Because value based upon a single reading, no range is reported

Table 3-9. Frequency of analytical grab samples found below the 5 mg/L water quality objective for DO (2012–2016).

Site	Winter		Spring		Summer		Fall	
	Closed	Open	Closed	Open	Closed	Open	Closed	Open
VWRF, wildlife ponds and outfall channel								
ETS (M-001)	0 of 900		0 of 917		6 of 912 (<1%)		0 of 906	
D-1 (M-0001A)	1 of 42 (2%)		0 of 39		0 of 42		0 of 43	
North Sonde (R-004)	5 of 62 (8%)	1 of 8 (13%)	12 of 54 (22%)	1 of 15 (7%)	35 of 75 (47%)	1 of 4 (25%)	12 of 73 (16%)	0 of 2
Lower estuary								
South Sonde (R-002)	3 of 45 (7%)	2 of 19 (11%)	3 of 45 (7%)	3 of 20 (15%)	6 of 46 (13%)	5 of 31 (16%)	0 of 55	0 of 21
Central Sonde (NA)	1 of 5 (20%)	NA	0 of 3	0 of 1	0 of 10	NA	0 of 7	NA
E-1 (R-003)	4 of 63 (6%)	0 of 8	5 of 55 (9%)	3 of 15 (20%)	10 of 72 (14%)	1 of 5 (20%)	1 of 72 (1%)	0 of 1
Upper estuary and riverine								
E-2 (NA)	1 of 6 (17%)	NA	0 of 3	NA	1 of 11 (9%)	NA	0 of 7	NA
R-005 (R-005)	4 of 62 (6%)	0 of 7	6 of 54 (11%)	3 of 14 (21%)	11 of 70 (16%)	1 of 4 (25%)	5 of 68 (7%)	0 of 1
R1 (NA)	NA	NA	0 of 1	NA	0 of 1	NA	0 of 1	NA

Specific conductivity/salinity

The SCRE is a brackish system with highly variable salinity due to ocean exchanges of saltwater during all seasons and estuary mouth conditions. Specific conductivity ranged from 1,939–50,106 uS/cm during open-mouth conditions and 989–44,188 uS/cm during closed-mouth conditions within the SCRE (Table 3-5 through Table 3-8). Using calculations presented in Schemel (2001), corresponding salinity values at the above ranges of specific conductivity were 0.6- 33 ppt and 0.6–27 ppt during open and closed-mouth conditions, respectively.

Salinity levels were greatest during winter at sites nearest to the ocean (Table 3-5) when there may be higher contributions of saltwater during open mouth conditions, and lowest during the spring when the SCR may be contributing greater amounts of runoff from the watershed (Table 3-6). However, in the summer and fall 2015 higher specific conductivities were observed in the uppermost SCRE reaches compared to the estuary near McGrath State Beach, likely due to the naturally high salinity in the SCR watershed (Table 3-7 and Table 3-8).

3.4.1.3 Continuous In situ water quality

Four continuously recording multi-parameter water quality sondes were deployed at three locations in the SCRE to characterize *in situ* water quality parameters (temperature, DO, pH, specific conductivity, Chlorophyll-a) and estimates of chlorophyll-a at approximately 15-minute intervals to provide a near-continuous data record. Figure 3-29 through Figure 3-32 present continuous *in situ* water quality data from the sondes deployed in the north, central and south SCRE locations between March 2015 and November 2016, with periods corresponding to open and closed-mouth conditions shown by grey or white shading.[10] The variations in *in situ* water quality were characterized using order statistics (median, lower and upper quartiles) by season for the closed-mouth condition only (Table 3-10), analysis was not completed for the open-mouth condition due to the inability to collect continuous data at the low stage levels encountered during 2016.

Water temperature

Continuous records of water temperature collected during closed mouth conditions show median temperatures between 17.1°C to 25.5°C (Table 3-10) and followed the expected seasonal pattern of higher and lower temperatures with longer and shorter periods of sunlight (Figure 3-29 through Figure 3-32). 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 three sonde locations.

In addition to comparisons of diurnal temperature variations at the three sonde locations (Appendix D, Figures D-1 through D-8), Figure 3-34 shows the differences of average daily temperature at the North, Central, and South sondes compared to the average daily temperature in the SCRE during the closed-mouth conditions. For closed mouth conditions when all the Sondes were deployed, the average daily water temperature at the North Sonde in the outfall channel were generally between 0ºC and 0.5ºC cooler than the average daily temperature in the SCRE during late-winter/spring 2015 (March to late-May 2015), but it appeared to transition from being around between 0ºC to 0.5ºC warmer in winter 2016 to being around 0ºC to 0.5ºC cooler than the average daily temperature in the SCRE in early spring 2016. The average daily water temperature at the North Sonde was typically around 0.5ºC warmer during summer (July to August 2016). The maximum difference between the North Sonde and the overall estuary average daily water temperature was approximately 1ºC in January 2016. The maximum difference between the South Bottom Sonde and the average estuary water temperature was approximately 0.7ºC in January 2016 with the South Bottom Sonde cooler than the average estuary water temperature. The Central Sonde and the average estuary water temperature had a maximum temperature difference of approximately 0.7ºC in February 2016.

On an hourly basis (Appendix D, Figures D-1 through D-8), the North Sonde deployment showed VWRF discharges were generally warmer than the open water sites between midnight and noon in winter (January – March), but the VWRF discharges were cooler or similar temperatures in spring (April – June), summer (July – September), and fall (October – December). During summer 2016, North Sonde temperatures were similar to other open water sites during the afternoon, but warmer during other periods. The diurnal pattern in summer 2016 may be related to changes in morphology following the manual breaches at the north end of the SCRE berm in summer 2016. During fall 2016, the North Sonde temperatures tended to be slightly warmer during the entire day compared to other open water sites.

Dissolved oxygen

Continuous records collected during closed mouth conditions show DO generally above saturation at the South and Central locations (Figure 3-29 through Figure 3-32; Appendix D, Figures D-1 through D-8). Following a breach event in January 2015, the SCRE exhibited very high DO during the initial sonde deployment with the notable drop-off in DO across all

[10] For additional sensor data (depth, pH and chlorophyll-a), see accompanying electronic data (Appendix B)

monitoring locations for approximately 1-week during March 2015. Because the data is consistent with in situ spot checks using an independently calibrated sondes and a similar pattern was seen at all locations, this suggests a broad introduction of un-oxidized organic matter (i.e., BOD) perhaps from an algal bloom or upstream sources.

Other than the North Sonde deployment in the VWRF outfall channel, which showed extended periods of low DO during 2015–2016 (Figure 3-32) and median DO below Basin Plan objectives (Table 3-10), diurnal variations in DO at other open water sites generally show high concentrations during daylight hours and lower DO at night (Appendix D, Figures D-1 through D-8). As found in previous phases of study (Stillwater Sciences 2011), although surface waters were generally above saturation during 2015–2016 (Table 3-10) this pattern is consistent with periods of higher solar insolation and photosynthetic activity (daytime) followed by lower DO concentrations during low light levels (night) when bacterial and phytoplankton respiration are expected to exceed surface re-aeration at the water surface.

pH

Continuous records collected during closed mouth conditions show median pH at open water sites ranged from 7.8 to 9.1 (Table 3-10), in excess of the water quality objectives in the Basin Plan (6.5-8.5) from spring through fall 2015 as well as during summer 2016 (Appendix D, Figures D-1 through D-8). pH levels found at the North Sonde deployment in the VWRF outfall channel were consistently below Basin Plan objectives (Table 3-10), however.

Specific conductivity/salinity

Continuous records of specific conductivity generally were higher following breach events (Figure 3-29 through Figure 3-32) and rapidly approached SCR background levels during closed mouth conditions. Short term increases at one or more locations (e.g., May and July 2015 at the South Sonde location; July and August 2015 at the North Sonde location) suggest that periodic wave overwash events may introduce ocean water even in the absence of a documented breach event.

Other than the North Sonde deployment in the VWRF outfall channel, which showed increased specific conductivity during nighttime hours (generally 3 am – 9 am) in 2015 when VWRF flows are typically low, no general diel patterns in specific conductivity were apparent at other Sonde locations (Appendix D, Figures D-1 through D-8). It is noted that the specific conductivity for the period April 1st through June 30th, 2016 lacks a typical diurnal pattern, likely because of one or more discrete breaching events during this period.

Chlorophyll-*a*

To examine variations in algae production with time, a Chlorophyll-a sensor was fitted to the surface deployment of the South Sonde location. Over the 2015–2016 monitoring period, Chlorophyll-a levels were generally higher in spring and summer (Figure 3-33). Although comparisons between grab samples analyzed from the SCRE at nearby Site E-1 generally showed a positive relationship with continuously monitored Chlorophyll-a, results were variable and seasonal variations in phytoplankton were assessed using analytical sampling (Section 3.4.1.5).

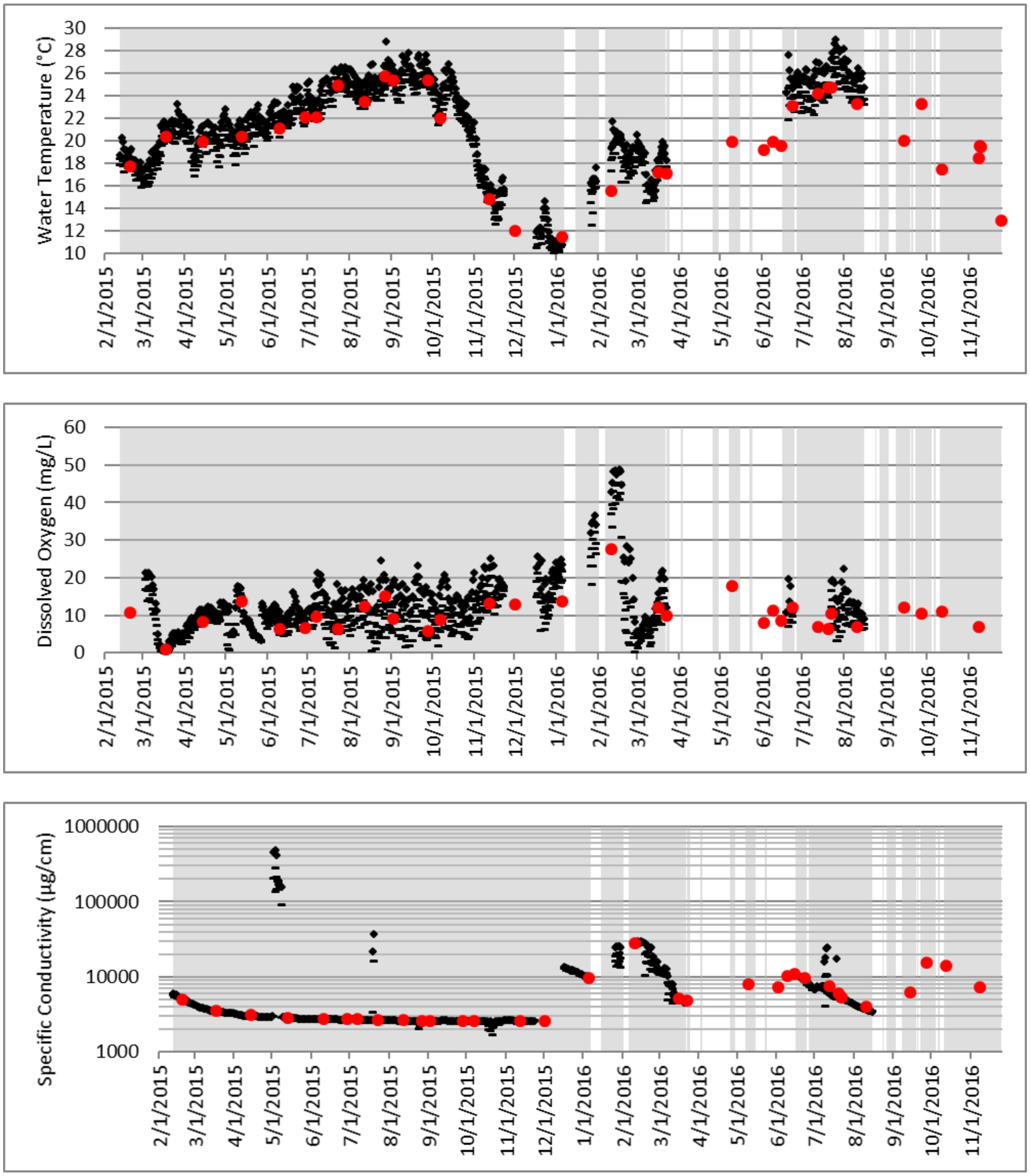


Figure 3-29. South Sonde surface: 2015–2016 continuous (black symbols) and instantaneous (red symbols) in situ water quality for selected parameters at the south surface sonde in the SCRE during closed-mouth (grey shading) and open-mouth (no shading) conditions.

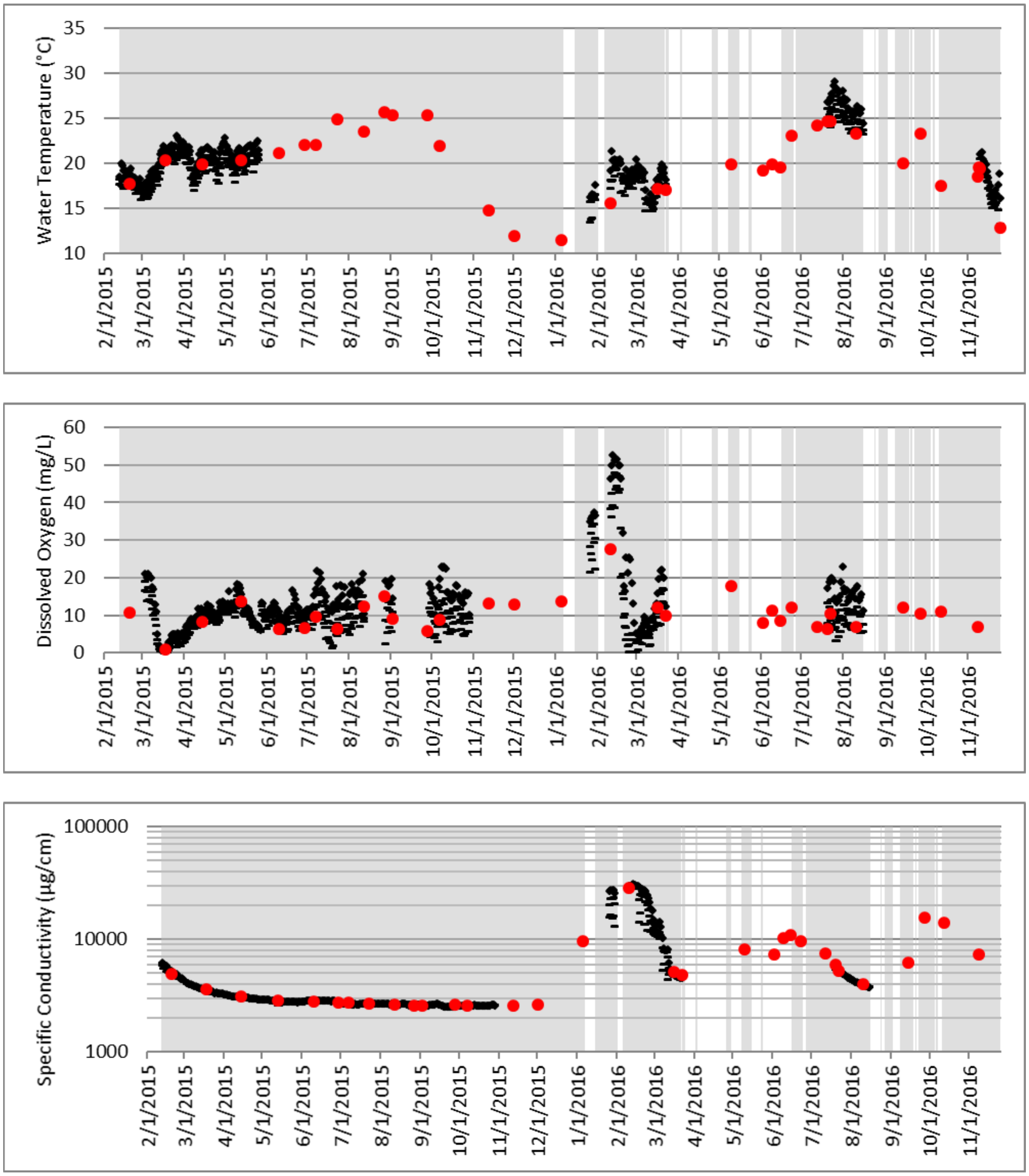


Figure 3-30. South Sonde bottom: 2015–2016 continuous (black symbols) and instantaneous (red symbols) in situ water quality for selected parameters at the south bottom sonde in the SCRE during closed-mouth (grey shading) and open-mouth (no shading) conditions. See Section 2.2.3.2 for a discussion of the apparent temperature sensor drift identified.

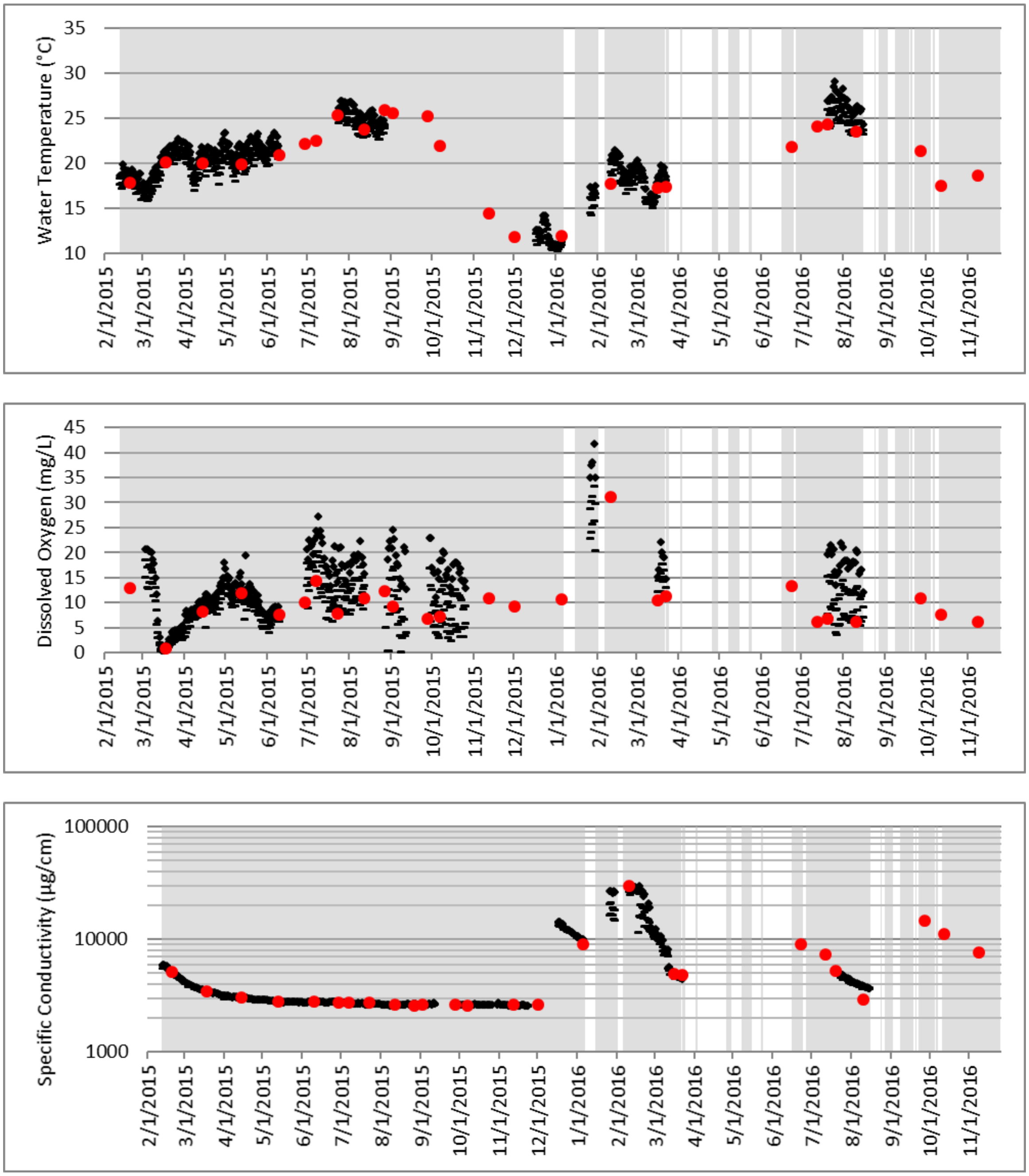


Figure 3-31. Central Sonde: 2015–2016 continuous (black symbols) and instantaneous (red symbols) in situ water quality for selected parameters at the central bottom sonde in the SCRE during closed-mouth (grey shading) and open-mouth (no shading) conditions.

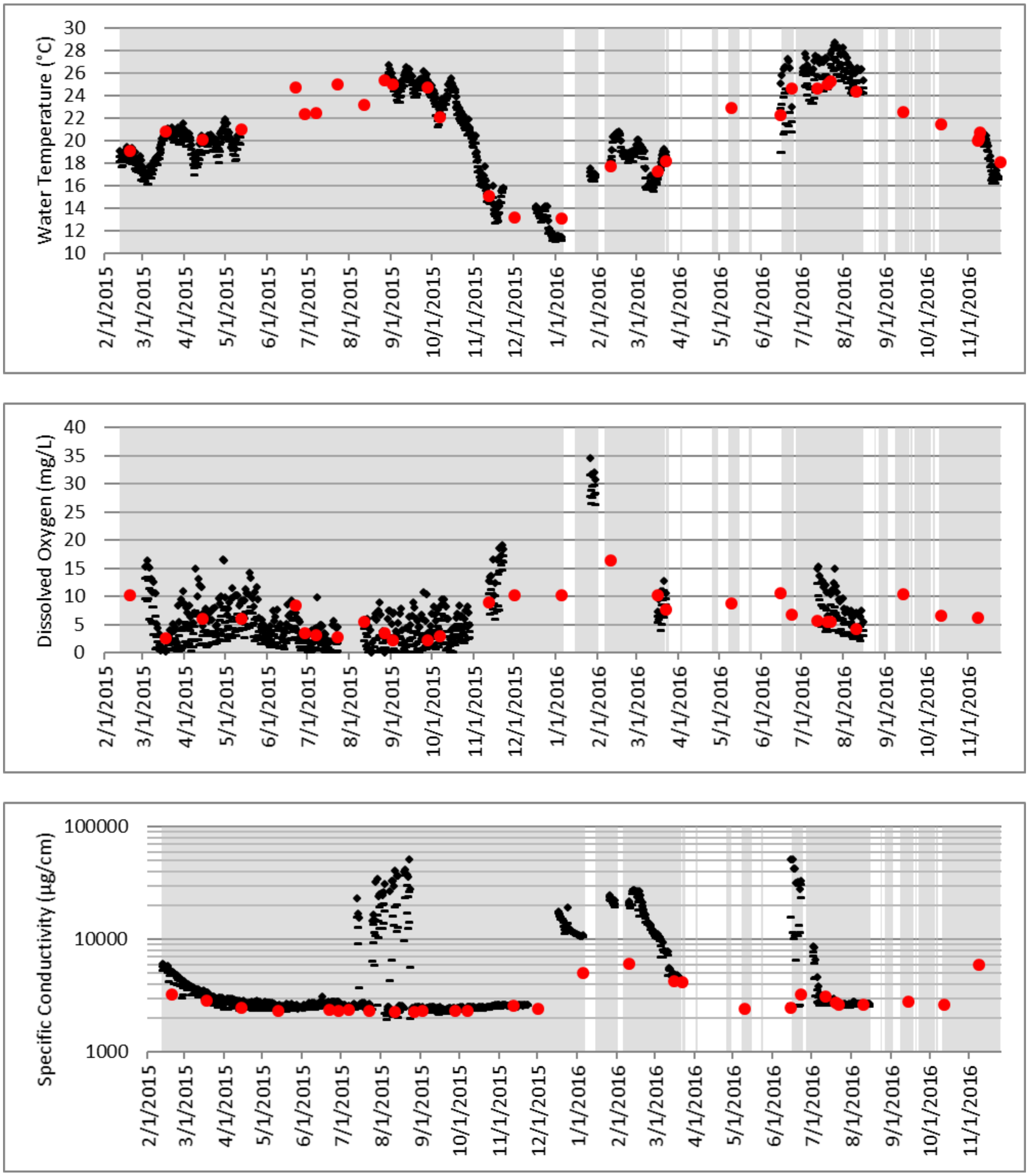


Figure 3-32. North Sonde: 2015–2016 continuous (black symbols) and instantaneous (red symbols) in situ water quality for selected parameters at the north bottom sonde in the SCRE during closed-mouth (grey shading) and open-mouth (no shading) conditions. See Section 2.2.3.2 for a discussion of the apparent temperature sensor drift identified.

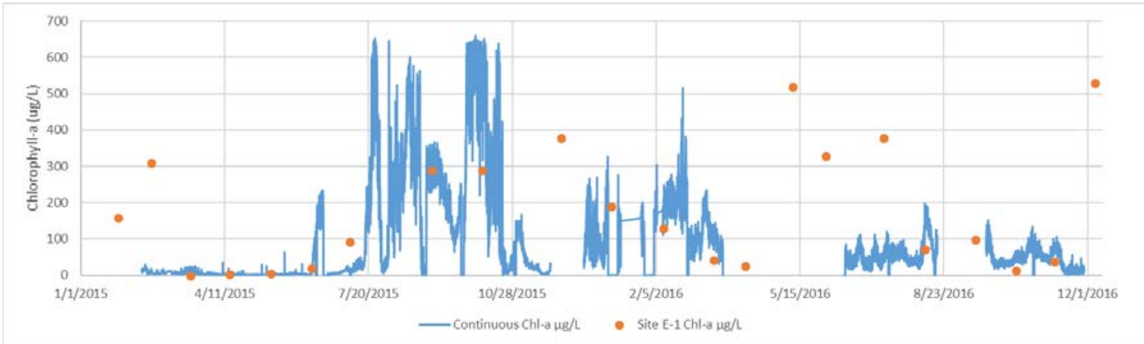


Figure 3-33. 2015–2016 continuous (solid line) and instantaneous (red symbols) measurements of Chlorophyll-a at the south surface sonde and site E-1 in the SCRE.

Table 3-10. Summary of SCRE *in situ* water quality parameter quartiles by season during 2015–2016.

Location	North bottom			Central bottom			South bottom			South surface		
Quartile	**Median**	**Q1**	**Q3**	**Median**	**Q1**	**Q3**	**Median**	**Q1**	**Q3**	**Median**	**Q1**	**Q3**
Temperature (°C)												
Winter	18.1	13.5	18.8	17.8	12.5	18.7	18.1	17.6	18.8	17.6	12.1	18.4
Spring	19.5	18.1	20.3	20.1	18.6	21.0	19.9	18.5	20.9	20.0	18.5	20.9
Summer	25.5	24.6	26.3	24.6	23.6	25.5	25.4	24.4	26.3	24.2	23.2	25.2
Fall	22.4	17.5	24.6				17.1	15.9	19.0	23.8	18.7	25.4
pH												
Winter	9.0	8.8	9.2	9.1	9.0	9.2	9.1	8.8	9.2	9.1	9.0	9.2
Spring	7.9	7.6	8.2	8.4	8.0	8.9	8.5	7.9	9.2	8.5	8.0	9.1
Summer	7.8	7.6	8.1	8.8	8.6	9.0	8.8	8.7	8.9	8.9	8.7	9.0
Fall	7.8	7.6	8.1	8.7	8.6	8.9	8.6	8.4	8.7	8.8	8.7	8.9
Dissolved oxygen (mg/L)												
Winter	29.1	28.3	30.5	30.7	27.9	33.9	27.6	5.3	44.9	19.4	12.6	27.8
Spring	4.8	2.6	6.9	9.5	5.9	12.0	8.9	5.2	11.4	8.0	4.5	10.6
Summer	3.1	1.5	5.1	12.9	9.1	16.2	10.1	8.1	12.8	9.8	8.0	12.3
Fall	4.0	2.1	6.1	10.6	7.4	13.5	8.9	7.5	11.7	11.3	8.3	14.8
Dissolved oxygen (% sat)												
Winter	325	315	339	332	301	374	304	61	546	189	129	305
Spring	49	25	73	106	66	135	95	56	127	92	52	125
Summer	66	50	89	144	100	186	130	97	172	117	94	150
Fall	48	25	68				81	77	90	129	98	166
Specific conductivity (uS/cm)												
Winter	11,987	5,721	17,972	11,824	5,804	16,843	15,333	5,218	25,665	11,795	5,676	16,766
Spring	2,759	2,517	3,612	3,090	2,871	3,760	3,167	2,891	3,829	3,244	2,935	4,423
Summer	2,587	2,490	2,768	2,708	2,629	2,810	2,698	2,651	2,860	2,725	2,637	4,848
Fall	2,461	2,368	2,569	2,590	2,567	2,604	2,580	2,546	2,626	2,578	2,559	2,597
Chlorophyll-a (ug/L)												
Winter	No Chlorophyll-a sensor on 3 of 4 Sondes									168.3	112.5	198.4
Spring										2.1	1.5	6.7
Summer										55.9	22.8	204.9
Fall	Split deployment for Sonde maintenance (See note) >						6.3	4.9	11.5	61.0	41.2	90.2

Note: Sonde maintenance during October 2015 required South sonde (bottom) to be redeployed at Central Sonde location and South sonde (surface) to be redeployed as South sonde (bottom).

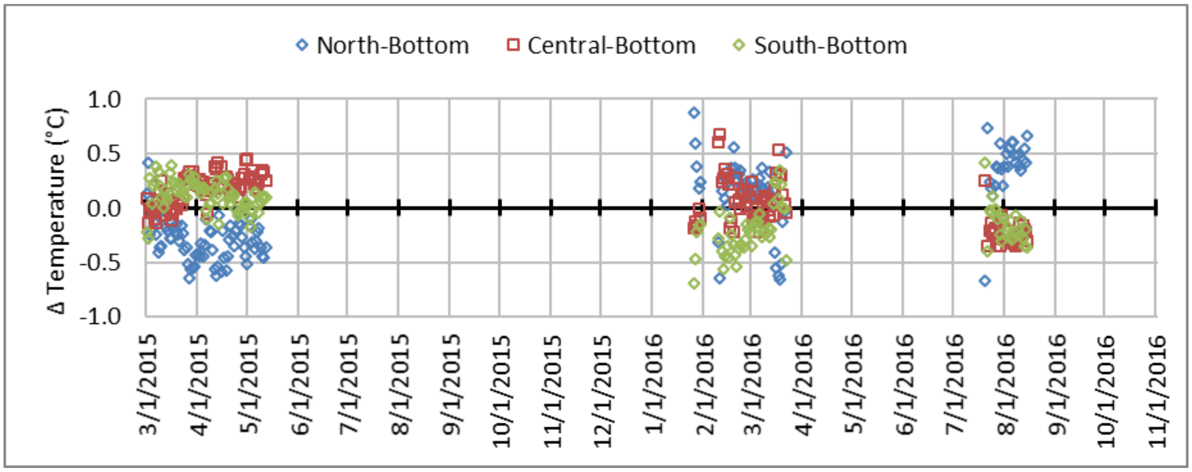


Figure 3-34. The difference of recorded water temperature records from the average estuary water temperature at the north, central, and south bottom sites (2015–2016).

3.4.1.4 Spatial variations of SCRE In situ water quality

To better characterize whether SCRE water quality conditions support existing beneficial; uses such as habitat suitability for focal fish species, spatial variations of in situ water quality were assessed through two methods, vertical profiling and contour mapping.

Vertical water column profiles

In accordance with the Monitoring Plan, monthly profile measurements of Temp, DO, pH, and EC were collected from October through May at R-1, E-1, E-2, and the three sonde locations and are presented in Appendix D (Figures D-9 through D-42). In general, although some density stratification of the water column was apparent following breaching (e.g., January 2015, January-March 2016, July 2016) and in the outfall channel, *in situ* water quality did not appreciably vary with depth at any profile location. There were some exceptions to this pattern but in situ profiles show only two periods of estuary-wide density stratification in February 2016 and August 2016 following breaches in the preceding months (Appendix D, Figure D-18 and Figure D-20). Although there was near bottom DO depletion observed in some samples (e.g., July-August 2016), surface waters were generally above saturation and the SCRE was vertically mixed during most sampling events.

Spatial maps of *in situ* water quality

To identify whether areas of the SCRE were associated with DO depletion or elevated water temperatures, spatial maps were constructed from available *in situ* measurements of water quality collected during habitat surveys (Appendix D). Spot measurements of in situ water quality were collected around the SCRE perimeter over four sampling events during Winter through Fall 2015 and one additional event in Fall 2016. The general methods used to construct the water quality isopleths shown in Appendix D are to accumulate data collected at the same date and time ("synoptic") with spatial coordinates from field GPS and then perform trend surface analysis (Ripley, 2001) in GIS which relies upon least squares regression to interpolate between the known data points. However, because several parameters (Temperature, DO, pH) all vary with sunlight and thus time of day, some normalization of this data was required prior to plotting.

In order to normalize the *in situ* water quality measurements collected during the habitat surveys, we assume water quality varied similarly from one location to the next, we used the 30-minute

Sonde data to standardize the observed spot measurement values to a common "reference value" at a common time. That is, the observed spot measurement value y_t of a water quality parameter at time t replaced by an imputed value from the Sonde data record

$$y_{\text{reference}} = y_t + \varepsilon(t)$$

where the "adjustment factors" $\varepsilon(t)$ were constructed to behave well at one or more of the Sonde locations over the relevant time period. Specifically, $\varepsilon(t)$ was calculated as

$$\varepsilon(t) = Y_{\text{reference}} - Y(t)$$

where $Y(t)$ is an idealized (smoothed) version of the Sonde value of the metric, and $Y_{\text{reference}}$ is its "reference value". These adjustment factors are calculated separately for each of the three surveys (March 17–18 2015, June 9–10 2015, and September 2 2015) with $Y_{\text{reference}}$ defined as $(Y(t_1) + Y(t_2))/2$, where t_1, t_2 are two nearby noons (2015-03-18 12:00 and 2015-03-19 12:00 for the March survey; 2015-06-09 12:00 and 2015-06-10 12:00 for the June survey; 2015-09-02 12:00 and 2015-09-03 12:00 for the September survey). Correspondingly, $y_{\text{reference}}$ is interpreted as an estimate of the average of the true values of the metric y at these two times.

Lastly, $Y(t)$ is a smoothed version of the metric at the South Surface Sonde time series. The South Surface Sonde was selected as the preferred reference in the end because it is the only surface station and less complicated by VWRF inflows, and because it has good coverage over the periods of the surveys; however, we calculated adjustment factors for all four of the fixed stations, and confirmed that they yielded consistent results.

To illustrate the process described above, Figure 3-35 shows variations in the 30-minute in situ water quality record at each of the four sonde locations, with Figure 3-36 showing the smoothed data series $Y(t)$ for each location as constructed from the raw series $Y_{\text{raw}}(t)$ by a local smoothing process: At each value t, $Y(t) = f_t(t)$, where $f_t(\tau)$ is a function of the form:

$$f_t(\tau) = b_0 + b_1\tau + b_2\sin(2\pi\tau) + b_3\cos(2\pi\tau) + b_4\sin(4\pi\tau) + b_5\cos(4\pi\tau)$$

with the parameters fit by least-squares to the values of $Y_{\text{raw}}(t)$ in the forty-eight-hour window $t - 1 \leq \tau \leq t + 1$. Shows the observed Central Sonde data as predicted from the South Sonde surface in situ water quality data. Overall, the variations in observed vs. predicted data are small and the data transform was applied to normalize the spot measurements to local noon for each of the survey dates plotted in Appendix D (Figures D-45 though D-53). The resulting patterns show largely uniform water quality conditions in the SCRE.

Near surface water temperatures during habitat surveys were slightly cooler near the McGrath State beach berm and shallow water locations of the lower SCR during all seasons except fall 2015 which exhibited high SCR inflow temperatures (Figures E-45 though E-47). The VWRF outfall channel exhibited generally lower temperatures and conductivity from winter into early summer with some evidence of warmer surface temperatures in summer and early fall. DO conditions were generally uniform as well, except in the outfall channel as measured at the North Sonde, where DO was generally lower. Although a SCRE-wide DO depletion event was evident during the Spring 2015 surveys (Figure D-48), DO was near or above saturation during other surveys (Figures D-48 through D-50). Lastly, specific conductivity was also generally uniform with generally lower conductivity nearest the VWRF outfall channel (Figures E-51 through E-53).

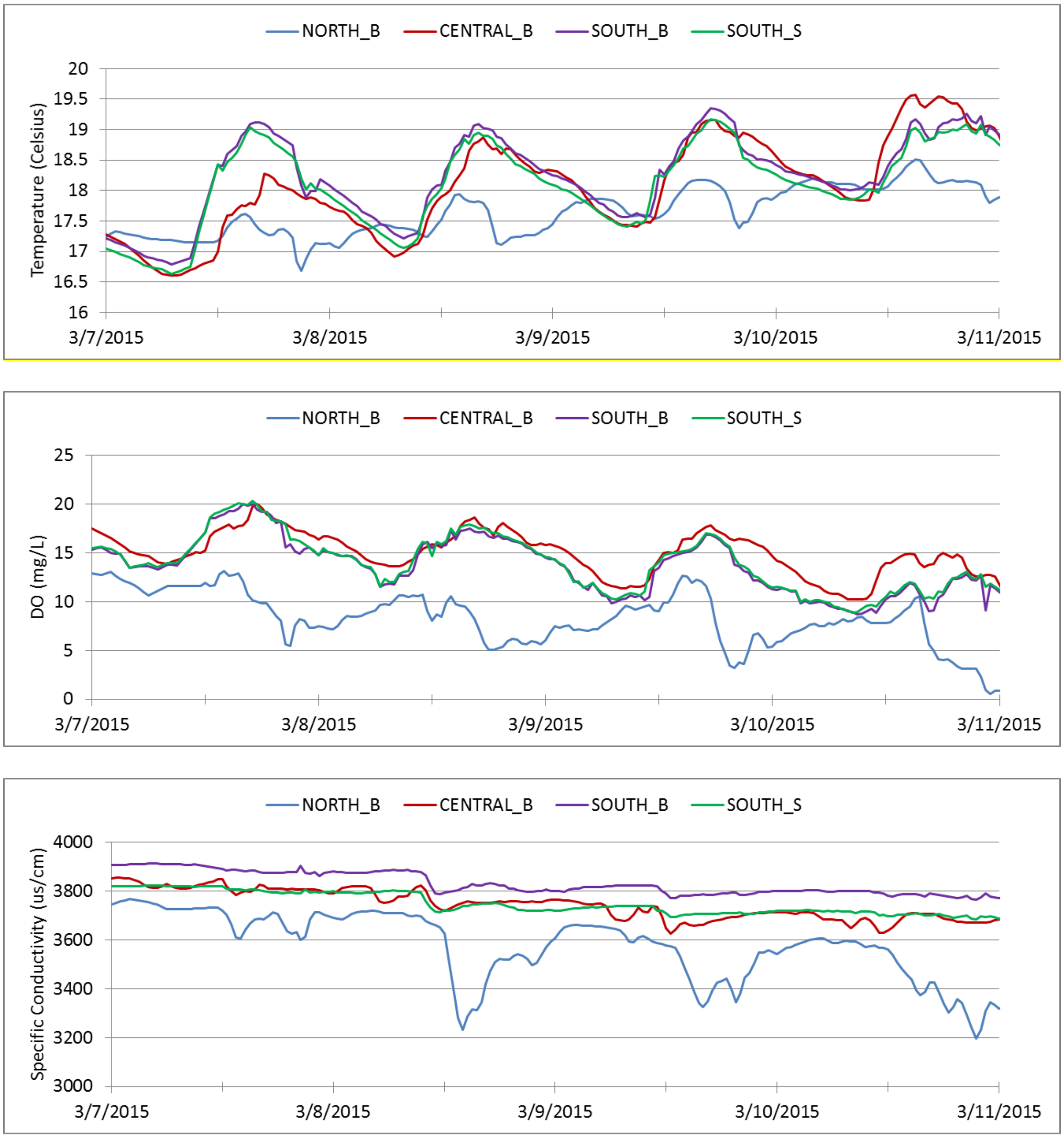


Figure 3-35. Observed diurnal variation in temperature, DO, and specific conductivity at the at the North, Central, and South Sonde locations from March 7 to March 11, 2015, with date labels showing the start of each day.

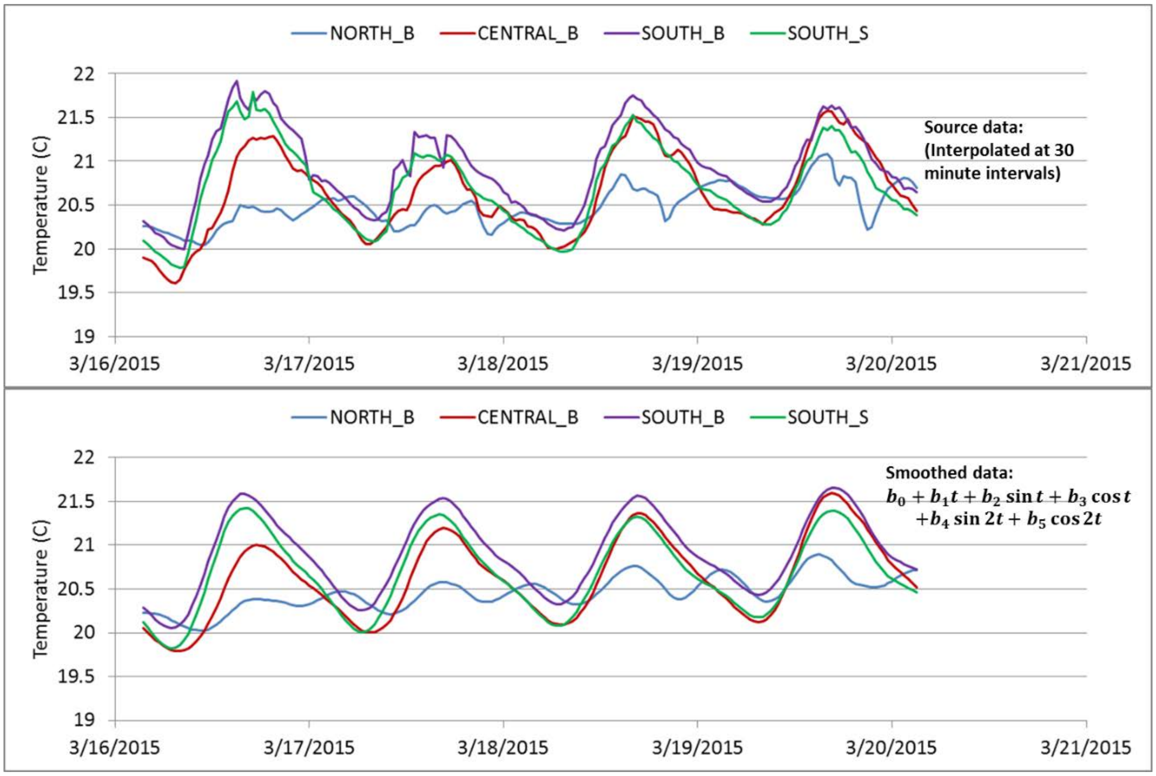


Figure 3-36. Example data smoothing of Temperature data collected from the North, Central, and South Sonde locations from March 16 to March 20, 2015, with date labels showing the start of each day.

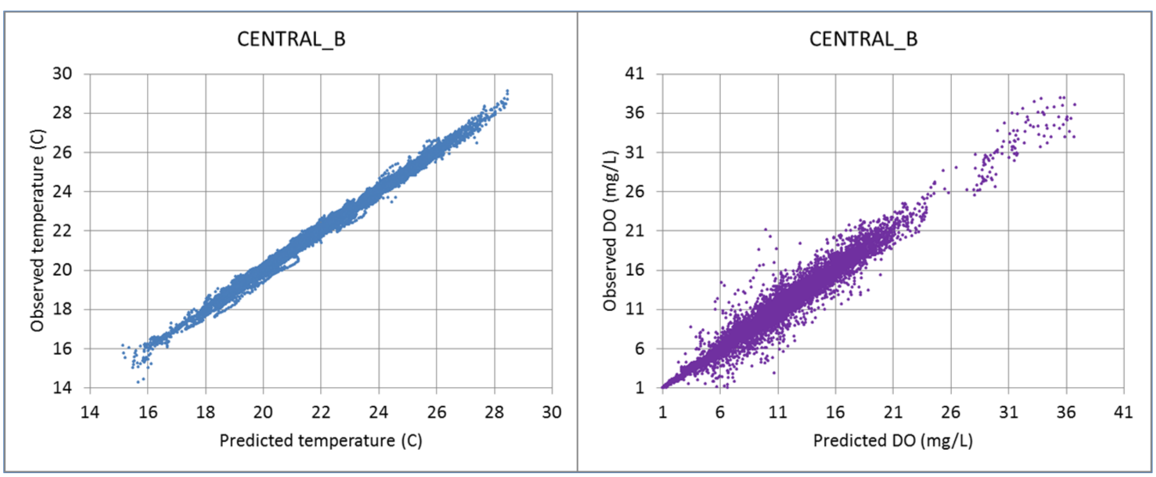


Figure 3-37. Observed vs Predicted Diurnal Temperature and DO at the Central Sonde location from March 16 to March 20, 2015.

3.4.1.5 Analytical water quality

Analytical water quality sampling collected in surface waters during the Phase 3 Study and as required by NPDES (No. CA0053651) were analyzed for nutrients (ammonia [NH_3], total Kjeldhal nitrogen [TKN], nitrate [NO_3], nitrite [NO_2], phosphate [PO_4], and total phosphorus[TP]), chlorophyll-a, and metals (total and dissolved zinc [Zn], total and dissolved copper [Cu]).

Nutrients

As found in the Phase 1 study, elevated nutrient levels and related Chlorophyll-a concentrations were found in the SCRE, reflecting inputs from the VWRF, local groundwater sources, runoff, riverine sources, and ocean exchanges. Table 3-11 through Table 3-14 present summary statistics of nutrient and Chlorophyll-a data under open and closed-mouth conditions for winter, spring, summer, and fall. If reported values were below a minimum detection limit (MDL), then the MDL was used for subsequent calculation of averages or other summary statistics.

Total nitrogen[11] within the SCRE during closed-mouth conditions is greatest during the fall and concentrations are generally similar between the spring, summer, and winter seasons (Table 3-11 through Table 3-14). Although ammonia was low throughout the SCRE, nitrate concentrations within the VWRF facilities exhibited higher nitrate (3 –16 mg/L) concentrations than SCRE sites. Nitrate concentrations within the outfall channel (0.4–15 mg/L) were generally similar to other SCRE sites (0.4–13 mg/L). Generally, inorganic nitrogen (as measured by NH_3 and NO_3) levels are higher and organic nitrogen (as measured by TKN) levels are lower under open mouth in comparison to closed mouth conditions in the SCRE. Although Chlorophyll-a measurements do not show a consistent pattern, the shifting proportions of inorganic vs organic nitrogen under open and closed mouth conditions suggest decreased uptake of inorganic nitrogen into algal biomass under open mouth conditions.

Total phosphorus levels were generally similar within the SCRE throughout all seasons, although average concentrations were slightly greater in the outfall channel (Table 3-11 through Table 3-14). The highest total phosphorus seasonal averages were greatest during the winter open-mouth conditions at the south estuary (Site E-1) and outfall channel (NPDES R-004). Ortho-phosphate collection was limited to Phase 3 (2015–2016). Concentrations of ortho-phosphate were greater with the outfall channel compared to the upstream sites (Table 3-11 through Table 3-14) and were similar between seasons.

High Chlorophyll-*a* concentrations within the SCRE are an indication of high rates of primary productivity of phytoplankton. Annual averages of Chlorophyll-a concentrations were consistently high during spring summer, and fall, and were lower at some sites during winter when SCRE water temperatures are cooler and daylight is shorter (Table 3-11 through Table 3-14). Chlorophyll-a concentrations were similar at most SCRE locations, with the exception of higher concentrations during open mouth condition in the upper estuary (Site E-2) and higher concentration during closed conditions in the estuarine/riverine site (NPDES Site R-005) (Figure 3-39).

Additional discussion on nutrient and Chlorophyll-a levels is provided in the trophic assessment in Section 3.4.1.6.

[11] Total nitrogen is the sum of NH_3, NO_3, NO_2, and organic nitrogen (TKN minus NH_3)

Table 3-11. Nutrients and chlorophyll-a in grab samples of the SCRE by mouth closure status during winter (2012–2016).

Site[1]	Mouth	Ammonia (mg-N/L)		TKN (mg-N/L)		Organic nitrogen[2] (mg-N/L)		Nitrate (mg-N/L)		Phosphate (mg-P/L)		Total phosphorus (mg-P/L)		Chlorophyll-a (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel															
ETS (M-001)	All	0.6	< 0.1–1	1.6	0.4–3.3	1	0–2.4	8	5.2-9.6	NA	NA	3.1	1.8–3.7	3	< 2–9
D-1 (M-001A)	All	< 0.1	< 0.1–0.2	2	1.6–2.4	1.9	1.5–2.3	6.9	5-10.4	2	1–3.1	2.9	1.9–3.8	101	23–220
North Sonde (R-004)	Open	0.3 [3]	NA	1.5 [3]	NA	1.2 [3]	NA	11 [3]	NA	NA	NA	3 [3]	NA	21 [3]	NA
	Closed	0.16	< 0.1–0.7	2	0.3–3	1.8	0.2–2.9	4.1	2.3–10	NA	NA	1.8	0.9–2.25	NA	NA
Lower estuary															
South Sonde (R-002)	Open	0.3 [3]	NA	1.8 [3]	NA	1.5 [3]	NA	1 [3]	NA	NA	NA	1 [3]	NA	< 2 [3]	NA
	Closed	0.21	< 0.1–0.7	2.8	1.1–4	2.6	0.9–3.9	2.9	< 0.4–8.4	NA	NA	1.4	0.5–2.5	NA	NA
E-1 (R-003)	Open	< 0.1 [3]	NA	1.4 [3]	NA	1.3 [3]	NA	11 [3]	NA	NA	NA	2.9 [3]	NA	23 [3]	NA
	Closed	0.15	< 0.1–0.7	2.3	0.5–4.4	2.1	0.4–4.3	2.6	0.9–4.8	0.55	0.3–1.0	1.4	0.3–2.7	138	1–360
Upper estuary and riverine															
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.33	< 0.1–0.8	2.8	0.2–4.3	2.5	0.1–4.2	1.4	0.5–2.52	0.53	< 0.2–1.1	1	0.6–1.5	197	1–400
(R-005)	Open	0.3 [3]	NA	1.6 [3]	NA	1.3 [3]	NA	2.1 [3]	NA	NA	NA	0.6 [3]	NA	< 2 [3]	NA
	Closed	0.25	< 0.1–1.0	2.2	0.3–4.3	1.9	0.1–3.8	2.3	< 0.4–5.7	0.43	< 0.2–0.9	1	0.4–1.9	154	< 2–260
R1 (NA) [4]	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016
1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.
2 Organic nitrogen concentrations were calculated by subtracting the ammonia concentrations from TKN.
3 Because value based upon a single reading, no range is reported

Table 3-12. Nutrients and chlorophyll-a in grab samples of the SCRE by mouth closure status during spring (2012–2016).

Site[1]	Mouth	Ammonia (mg-N/L)		TKN (mg-N/L)		Organic nitrogen[2] (mg-N/L)		Nitrate (mg-N/L)		Phosphate (mg-P/L)		Total phosphorus (mg-P/L)		Chlorophyll-a (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel															
ETS (M-001)	All	0.6	0.2–0.9	1.4	0.3–2.2	0.9	0.2–2	7.8	5.4-11.7	NA	NA	3.1	1.9–3.9	3.2	< 2–17
D-1 (M-001A)	All	0.2	< 0.1–0.4	1.8	0.6–3.3	1.6	0.5–3.2	6.2	3.3-10.4	1.4	0.6–2.1	2.6	1.2–3.6	130	11–380
North Sonde (R-004)	Open	0.4	0.3–0.5	1.8	1.1–2.6	1.4	0.8–2.1	5	2.4–9.4	NA	NA	2.5	1.9–3.4	137	4–310
	Closed	0.19	< 0.1–0.6	2.5	1.3–5.3	2.3	1.1–5.2	2.5	< 0.4–5.6	NA	NA	2.4	1.2–3.2	133	7–500
Lower estuary															
South Sonde (R-002)	Open	0.83	0.3–1.5	1.4	0.8–2.2	0.6	0.5–0.7	2.4	< 0.4–5.1	NA	NA	1.3	0.4–1.9	118	< 2–290
	Closed	0.27	< 0.1–0.6	3.3	1.5–4.7	3	1.4–4.4	0.9	< 0.4–2.8	NA	NA	1.2	0.6–1.9	115	< 2–380
E-1 (R-003)	Open	0.4	< 0.1–0.7	1	0.4–1.5	0.6	0.3–1.1	3.3	1.5–4.9	1.1 [3]	NA	1.3	0.7–2	175	3–430
	Closed	0.25	< 0.1–0.7	2.7	0.7–5.5	2.4	0.4–5.4	1.1	< 0.4–3.8	0.5	< 0.2–0.9	1.4	0.7–2.3	127	3–520
Upper estuary and riverine															
E-2 (NA)	Open	0.4 [3]	NA	1.4 [3]	NA	1 [3]	NA	< 0.4 [3]	NA	< 0.2 [3]	NA	1 [3]	NA	390	130–650
	Closed	0.14	< 0.1–0.2	1.5	0.7–2	1.4	0.6–1.8	0.5	< 0.4–1.1	0.3	< 0.2–0.7	0.8	0.3–1.3	133	< 2–500
(R-005)	Open	0.55	0.2–1	1.9	1.1–3.6	1.3	0.6–2.6	3	2.4–3.4	< 0.2 [3]	NA	0.5	0.2–1.2	68	< 2–270
	Closed	0.19	< 0.1–0.4	2.9	0.8–5.5	2.7	0.4–5.4	1.2	< 0.4–2.7	0.7	< 0.2–2.4	1	0.2–3.2	357	< 2–4,000
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.3	< 0.1–0.5	0.9	0.3–1.6	0.7	0.2–1.1	4.5	3.2–5.7	< 0.2[3]	NA	1.1	0.6–1.6	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

2 Organic nitrogen concentrations were calculated by subtracting the ammonia concentrations from TKN.

3 Because value based upon a single reading, no range is reported

Table 3-13. Nutrients and chlorophyll-a in grab samples of the SCRE by mouth closure status during summer (2012–2016).

Site[1]	Mouth	Ammonia (mg-N/L)		TKN (mg-N/L)		Organic nitrogen[2] (mg-N/L)		Nitrate (mg-N/L)		Phosphate (mg-P/L)		Total phosphorus (mg-P/L)		Chlorophyll-a (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel															
ETS (M-001)	All	0.7	0.2–1.2	1.6	1.1–2.1	0.9	0.2–1.3	8.3	5.8–9.7	NA	NA	2.6	1.2–4.2	4.3	< 2–12
D-1 (M-001A)	All	0.2	< 0.1–0.3	2.5	1.1–3.7	2.4	1–3.5	6.2	4.7–7.8	1.4	1–1.7	2.1	1.4–3.2	90	19–210
North Sonde (R-004)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.2	<0.1–0.3	3.1	1.4–6.4	3	1.2–6.1	3.2	< 0.4–7.4	NA	NA	2	0.1–3.9	243	12–990
Lower estuary															
South Sonde (R-002)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.4	< 0.1–2.9	3.8	1.1–8.2	3.4	1–8.1	0.5	< 0.4–0.9	NA	NA	0.9	0.2–2	261	10–920
E-1 (R-003)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.2	< 0.1–0.6	3.6	0.5–7.6	3.4	0.4–7.3	1.4	< 0.4–6.1	0.6	< 0.2–1.1	1.1	0.1–2	230	3–570
Upper estuary and riverine															
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	< 0.1	< 0.1–0.3	4.4	1.4–8.4	4.3	1.3–8.1	< 0.4	NA[3]	< 0.2	NA[3]	0.7	0.5–1.3	303	140–740
R-005 (R-005)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.2	< 0.1–0.5	3.9	1.4–9.2	3.7	1.1–8.7	0.5	< 0.4–0.9	< 0.2	NA[3]	0.8	0.2–1.5	295	24–870
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	3 [3]	NA	5.3 [3]	NA	2.3 [3]	NA	3.8 [3]	NA	<0.2 [3]	NA	1.6 [3]	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

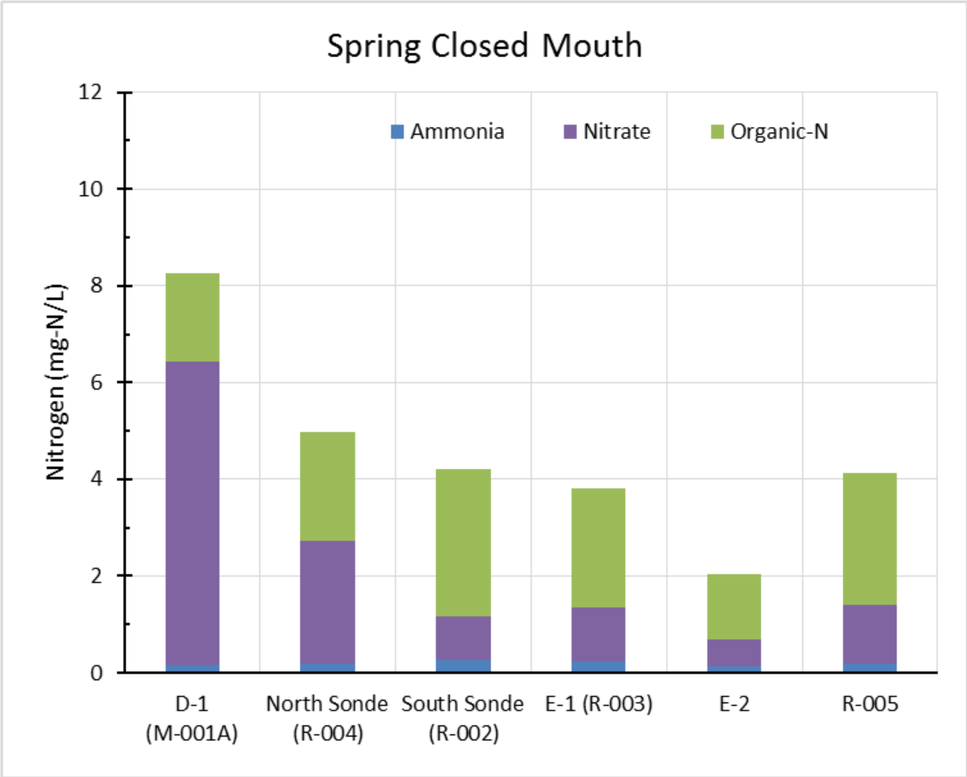
2 Organic nitrogen concentrations were calculated by subtracting the ammonia concentrations from TKN.

3 Because value based upon a single reading, no range is reported

Table 3-14. Nutrients and chlorophyll-a in grab samples of the SCRE by mouth closure status during fall (2012–2016).

Site[1]	Mouth	Ammonia (mg-N/L)		TKN (mg-N/L)		Organic nitrogen (mg-N/L)[2]		Nitrate (mg-N/L)		Phosphate (mg-P/L)		Total phosphorus (mg-P/L)		Chlorophyll-a (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel															
ETS (M-001)	All	0.6	< 0.1–0.9	1.7	0.8–5.2	1.1	0.1–4.6	8.4	5.6–10.2	NA	NA	2.4	1.3–3.9	3.5	< 2–8
D-1 (M-001A)	All	0.2	< 0.1–0.4	2.1	0.2–3.2	1.9	0–3.1	9.3	4.7–15.6	2	1.3–1.9	2	0.8–3.4	110	23–190
North Sonde (R-004)	Open	0	NA	0	NA	NA	NA	0	NA	NA	NA	0	NA	NA	NA
	Closed	0.13	< 0.1–0.2	3.17	0.9–5.3	3	0.7–5.2	4.8	< 0.4–15	NA	NA	1.5	0.7–2.5	156	17–310
Lower estuary															
South Sonde (R-002)	Open	0	NA	0	NA	NA	NA	0	NA	NA	NA	0	NA	NA	NA
	Closed	0.13	< 0.1–0.4	3.46	0.2–7.8	3.3	0–7.7	2.1	< 0.4–13	NA	NA	1.1	0.4–2.4	308	33–1,100
E-1 (R-003)	Open	0	NA	0	NA	NA	NA	NA	NA	NA	NA	0	NA	NA	NA
	Closed	0.11	< 0.1–0.2	3.8	1–7.8	3.7	0.8–7.7	3.1	< 0.4–15	0.7	< 0.2–1.5	1.1	0.4–1.9	236	14–540
Upper estuary and riverine															
E-2 (NA)	Open	< 0.1 [3]	NA	0.1 [3]	NA	NA	NA	3.2 [3]	NA	< 0.2 [3]	NA	0.7 [3]	NA	NA	NA
	Closed	0.16	< 0.1–0.4	2.64	1.4–3.5	2.5	1–3.4	3.9	< 0.4–12	0.5	< 0.2–1.1	0.8	0.5–1.4	232	34–310
R-005 (R-005)	Open	0	NA	0	NA	NA	NA	0	NA	NA	NA	0	NA	NA	NA
	Closed	0.16	< 0.1–0.4	3.22	0.8–4.6	3.1	0.6–4.5	1.3	< 0.4–5	< 0.2	< 0.2–0.52	0.7	0.2–1.2	170	10–350
R1 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	0.4 [3]	NA	0.8 [3]	NA	0.4 [3]	NA	4.5 [3]	NA	< 0.2 [3]	NA	0.12 [3]	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016
[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.
[2] Organic nitrogen concentrations were calculated by subtracting the ammonia concentrations from TKN.
[3] Because value based upon a single reading, no range is reported



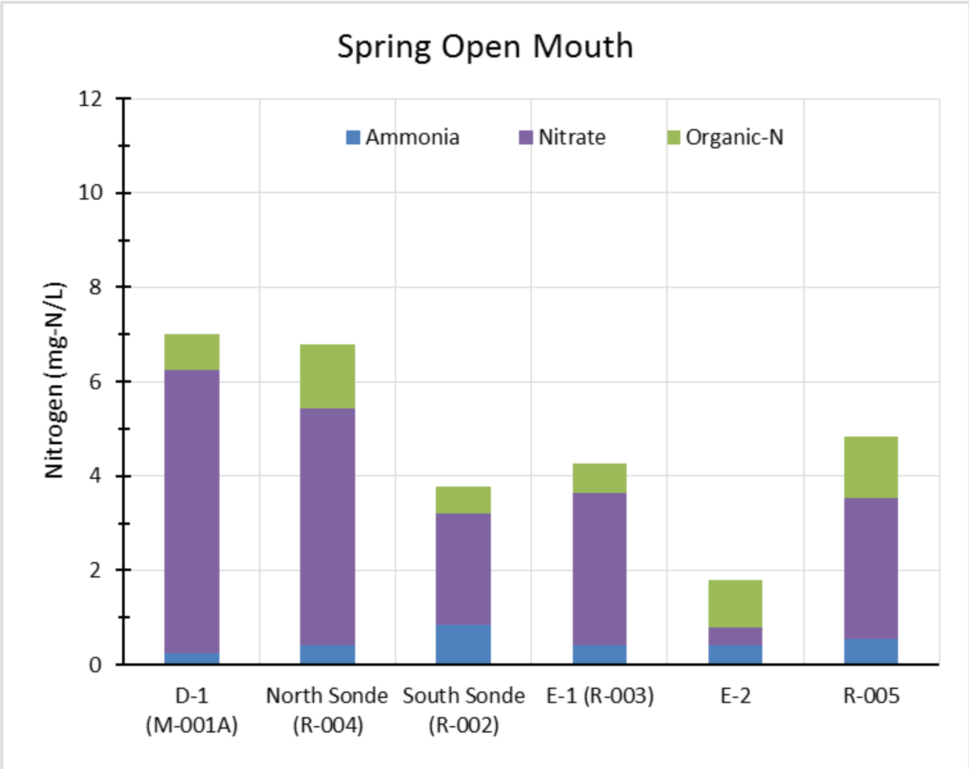


Figure 3-38. Nitrogen in grab samples collected during spring open (top) and closed (bottom) mouth conditions (2012–2015).

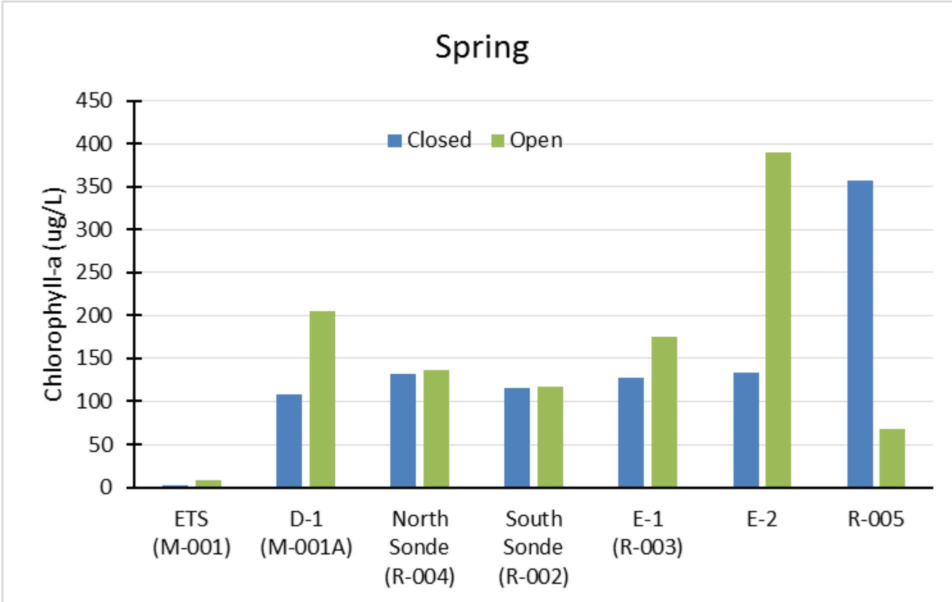


Figure 3-39. Chlorophyll-a (bottom) in grab samples collected during spring open and closed-mouth conditions (2012–2015).

Trace metals

Elevated concentrations of copper and zinc can have both acute and chronic effects on aquatic life. In high concentrations, both trace metals can be toxic to fish (Chapman and Stevens 1978). Chronic exposure to non-lethal concentrations of copper can impair olfaction in fish and disrupt homing of anadromous salmonids (Hecht et al 2007). Trace metal analysis results and summary statistics for total and dissolved zinc [Zn] and total and dissolved copper under open-mouth and closed-mouth conditions for all seasons are presented in Table 3-15 through Table 3-18. Due to sparse data collected during open-mouth conditions, comparisons between open and closed conditions are not discussed. Generally, total zinc and dissolved zinc are slightly greater at the VWRF ETS, Pond Outlet (D-1), and Outfall channel (NPDES R-004) compared to SCRE locations. However, the highest total zinc concentrations were measured at the upper estuarine/riverine site, R-005, (total zinc = 199 ug/L). Total copper is generally similar within the VWRF facilities, outfall channel, and SCRE locations, although, dissolved copper was slightly greater within the upper SCRE and estuarine/ riverine site, R-005, during in winter months when winter runoff is higher from the watershed.

Table 3-15. Metals in grab samples of the SCRE by mouth closure status during winter (2012–2016).

Site[1]	Mouth	Total zinc (ug/L)		Dissolved zinc (ug/L)		Total copper (ug/L)		Dissolved copper (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel									
ETS (M-001)	All	34.1	25.6–41.3	28.5 [2]	NA	2.9	< 2–5.1	2.1 [2]	NA
D-1 (M-001A)	All	29.8	11.2–42.6	19.5 [2]	NA	2.2	< 2–3.5	2.3 [2]	NA
North Sonde (R-004)	Open	38 [2]	NA	NA	NA	< 2 [2]	NA	NA	NA
	Closed	16	< 2.0–28	NA	NA	2.3	< 2–5.6	NA	NA
Lower estuary									
South Sonde (R-002)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA
E-1 (R-003)	Open	29 [2]	NA	NA	NA	< 2 [2]	NA	NA	NA
	Closed	15	5.6–22	18.7 [2]	NA	2.7	< 2–5.3	4.2 [2]	NA
Upper estuary and riverine									
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	23	17–29	19.7 [2]	NA	5.0	4.7–5.2	5.1 [2]	NA
(R-005) [3]	Open	6.2 [2]	NA	NA	NA	< 2	< 2 [2]	NA	NA
	Closed	14	< 2.0–33	NA	14-14	2.6	< 2–5.5	5	4.7–5.3
R1 (NA) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

2 Because value based upon a single reading, no range is reported

3 Riverine metals sample collected at NPDES site R-005 upstream of Harbor Blvd.

Table 3-16. Metals in grab samples of the SCRE by mouth closure status during spring (2012–2016).

Site [1]	Mouth	Total zinc (ug/L)		Dissolved zinc (ug/L)		Total copper (ug/L)		Dissolved copper (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel									
ETS (M-001)	All	32.9	23.5–42.9	37.1	35.3–38.9	2.5	< 2–8	3	3–3.1
D-1 (M-001A)	All	27.4	18.9–37	30.1	23.1–37	2.2	< 2–3.3	2.9	2.8–3.1
North Sonde (R-004)	Open	27	18–38	NA	NA	3.3	< 2–6.0	NA	NA
	Closed	19	6.9–52	NA	NA	2.2	< 2–4.9	NA	NA
Lower estuary									
South Sonde (R-002)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA
E-1 (R-003)	Open	8.7	3.5–13	NA	NA	< 2[2]	NA	NA	NA
	Closed	18	5.8–57	20	17–23	2.4	< 2–4.6	2.8	2.4–3.3
Upper estuary and riverine									
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	21	19–24	18	17–19	4.1	2.3–6.0	3.9	2.4–5.5
(R-005) [3]	Open	8.3	3.5–16	NA	NA	3.2	< 2–5.5	NA	NA
	Closed	28	2.7–199	18	15–20	3.3	< 2–10	4.2	2.6–5.9
R1 (NA) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016
1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.
2 Because value based upon a single reading, no range is reported.
3 Riverine metals sample collected at NPDES site R-005 upstream of Harbor Blvd.

Table 3-17. Metals in grab samples of the SCRE by mouth closure status during summer (2012–2016).

Site[1]	Mouth	Total zinc (ug/L)		Dissolved zinc (ug/L)		Total copper (ug/L)		Dissolved copper (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel									
ETS (M-001)	All	29.8	12.8–46	32.8	28.6–36.9	2.2	< 2–3.8	2.8	2.5–3
D-1 (M-001A)	All	25.5	17.9–35.6	31	27.1–34.9	2.2	< 2–3.4	2.5	2.2–2.8
North Sonde (R-004)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	21	7.0–30	NA	NA	< 2[2]	NA	NA	NA
Lower estuary									
South Sonde (R-002)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA
E-1 (R-003)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	13	2.4–26	19	19–20	2.1	< 2–2.9	2.3	2.1–2.5
Upper estuary and riverine									
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	11	9.0–14	15	9.5–20	3	2.2–3.8	2.7	2.2–3.3
(R-005) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	13	6.1–58	15	9.8–20	2.5	< 2–7.4	2.9	2.6–3.3
R1 (NA) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016

1 Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

2 Because value based upon a single reading, no range is reported

3 Riverine metals sample collected at NPDES site R-005 upstream of Harbor Blvd.

Table 3-18. Metals in grab samples of the SCRE by mouth closure status during fall (2012–2016).

Site[1]	Mouth	Total zinc (ug/L)		Dissolved zinc (ug/L)		Total copper (ug/L)		Dissolved copper (ug/L)	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range
VWRF, wildlife ponds and outfall channel									
ETS (M-001)	All	31.2	11.1–46.4	47	30.3–63.6	2.5	< 2–8.9	1.9	1.7–2
D-1 (M-001A)	All	32.3	20.6–51.2	37.8	33.1–42.5	2.6	< 2–7.6	2.3	1.8–2.9
North Sonde (R-004)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	19	7.6–32	NA	NA	< 2	< 2–< 2	NA	NA
Lower estuary									
South Sonde (R-002)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA
E-1 (R-003)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	16	4.8–45	29	24–34	2.2	< 2–3.2	2.6	2.3–3.0
Upper estuary and riverine									
E-2 (NA)	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	21	20–23	16	10–22	2.9	2.3–3.4	2.7	2.2–3.1
(R-005) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	16	< 2–41	26	14–38	2.2	< 2–4.7	2.8	2.4–3.2
R1 (NA) [3]	Open	NA	NA	NA	NA	NA	NA	NA	NA
	Closed	NA	NA	NA	NA	NA	NA	NA	NA

Data sources = 2015–2016 Phase 3 data included in Appendix B, Ventura 2012, 2013, 2014, 2015, 2016
[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.
[2] Because value based upon a single reading, no range is reported
[3] Riverine metals sample collected at NPDES site R-005 upstream of Harbor Blvd.

3.4.1.6 Changes in nutrient levels and trophic status of the SCRE

Since VWRF facility improvements were completed in late 2011, nitrate concentrations have been reduced in VWRF effluent and with the SCRE. Nitrate concentrations at the VWRF ETS and Pond outlets (Site D-1) have generally remained below 10 mg-N/L, and of the 287 SCRE water samples analyzed for nitrate between 2012–2016 less than 5% were greater than 10 mg-N/L (Appendix B). This a significant improvement to historical water quality where greater than 50% of water samples had a concentration of NO_3 + NO_2 greater than 10 mg-N/L (Stillwater Sciences 2011). To examine the effects of recent VWRF facility upgrades as well as other potential changes in nutrient limitations affecting algal blooms within the SCRE, we present calculations of a Trophic State Index (TSI) as well as an assessment of nutrients limiting algal productivity.

Trophic state index calculations

Although the term "trophic index" is commonly applied to lakes, we use a functional comparison of several water quality metrics applied 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. The subset of established threshold values used for evaluation are provided in Table 3-19. The means of open water site data for Chlorophyll-a, total inorganic nitrogen (TIN)(sum of inorganic nitrogen including NH_4^+, NO_3^-, and NO_2), phosphate and DO collected between 2012–2016 were used as references for the NEEA metrics (Appendix B). Table 3-20 presents the NEEA metrics with established threshold values using the range of daily averages at open water SCRE locations. Despite reductions in nitrate from the VWRF following facility upgrades, the SCRE continues to exhibit enriched (i.e., eutrophic) conditions found during the Phase 1 study (Stillwater Sciences 2011). Although, the status assessment indicates low potential for eutrophication in some conditions the maximum DO was super-saturated and exceeded 10 mg/L, indicating high rates of oxygen production by primary producers. This is supported by high levels of Chlorophyll-a within the SCRE and therefore can be assumed to have high potential for eutrophication indicating that ample nutrients and favorable conditions are available for excessive algal growth.

Table 3-19. 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).

Table 3-20. Trophic status assessment based upon observations of algal conditions, nutrients, and DO at open water sites in the SCRE (2012–2016)

Season	SCRE mouth	Phase 3 (2012–2016) Chlorophyll-a (ug/L)	TIN (mg-N/L)	PO_4[1] (mg-P/L)	DO (mg/L)
Winter	Open	12[2]	6.4[2]	NA	1.2–32.5
		Moderate potential	High potential	High potential	High potential
	Closed	1–343	0.9–7.9	0.2–1.6	4.4–13
		Hypereutrophic	High potential	High potential	Moderate potential
Spring	Open	3–273	3.2–6.0	0.5–1.3	3.3–27
		Hypereutrophic	High potential	High potential	Moderate potential
	Closed	4–721	0.6–5.1	0.2–1.4	3.2–12
		Hypereutrophic	High potential	High potential	Moderate potential
Summer	Open	NA	NA	NA	2.9–15
		NA	NA	NA	Moderate potential
	Closed	14–413	0.5–5.8	0.2–1.4	1.9–12
		Hypereutrophic	High potential	High potential	High potential
Fall	Open	NA	7.3[2]	0.8[2]	4.9–26
		NA	High potential	High potential	Low potential
	Closed	34–341	2.2–12	0.5–12	9.3–11
		Hypereutrophic	High potential	High potential	Low potential

1 Phosphate data presented is from 2012–2016 only.
2 Based on open water site averages from a single sampling event.

Nutrient limitation

The ratio of nutrients can be used to indicate the nutrient balance in estuarine systems and assist within indicating an over- or under-abundance (i.e., nutrient limitation) of one nutrient or another. The ratio of TIN to phosphate at a balanced state is often considered to be a 16:1 (molar) ratio for dissolved inorganic nitrogen and phosphorus (N:P) (Redfield 1958). The 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 mass ratio estimates above as a guide, Table 3-21 presents TIN:PO_4 ratios for the SCRE during 2015–2016 as calculated from data collected during the Phase 3 Study (Appendix B). The N:P ratios from 2015–2016 suggests that the SCRE is slightly nitrogen limited during closed-mouth conditions and that the riverine sites are phosphorus limited. These results show a shift in nutrient balance compared to previous results where the SCRE exhibited strong phosphorus limitation (Stillwater Sciences 2011). The shift to a nitrogen limited system is likely attributed to the reduction of nitrate in the VWRF effluent. Although, the nitrate was reduced the quantities of nutrients available for algal growth are still above saturation conditions for algal growth (0.2 mg/L-N and 0.02–0.08 mg/L-P) and therefore a large surplus of nutrients continues to be available to suspended as well as benthic algae. Although, ambient nitrate concentrations have reduced the nitrogen inputs via one pathway nitrate may continue to be supplied from multiple sources including VWRF effluent, groundwater, and upstream sources. The Phase 3 Study found elevated concentrations of nitrate in upstream groundwater wells (Figure 3-40) and upstream reaches of the SCR remain on the 2016 draft CWA 303(d) list for nitrate (RWQCB 2017) suggesting that upstream sources may continue to contribute to observed levels of nitrates in the SCRE.

Table 3-21. Average mass ratio of nitrogen to phosphorus (TIN:PO_4) in the SCRE by season and mouth closure status (2015–2016).

Site [1]	Winter			Spring			Summer			Fall		
	Open	Closed	Avg.	Open	Closed	Avg.	Open	Closed	Avg.	Open	Closed	Avg.
VWRF and wildlife ponds												
ETS (M-001)	NA	NA	3.4	NA -	NA -	5.5	NA	NA	4.6	NA	NA	5.4
D-1 (M-0001A)	NA	NA	3.8	NA -	NA -	5.0	NA	NA	4.5	NA	NA	6.2
Lower estuary												
E-1 (R-003)	NA	3.8	3.8	4.8	2.6	2.9	NA	3.4	3.4	9.5	4.4	5.8
Upper estuary and riverine												
E-2 (NA)	NA	3.5	3.5	4.0	2.3	2.8	NA	2.8	2.8	16.7	6.9	8.3
(R-005)	NA	5.8	5.8	17.5	5.9	7.6	NA	3.6	3.6	15.0	8.1	9.3
R1 (NA)	NA	29	29	23	NA	23	NA	34	34	NA	33	33

[1] Phase 3 sampling sites from Table 2-2 with NPDES site names shown in parentheses.

3.4.1.7 Toxicity testing results

Initial toxicity screening testing was conducted to determine the selection of the most sensitive species for the monitoring. The green alga (*Selenastrum capricornutum* [now called *Pseudokirchneriella subcapitata*]) growth was selected, per the NPDES Permit (No. CA0053651), requirements. Two additional species were selected as indicator species to better represent the special-status fish species that inhabit the SCRE—steelhead and tidewater goby. Because tidal exchanges with high salinity ocean waters are expected to strongly influence observed toxicity results, the species were selected depending upon SCRE mouth berm closure status. During closed-mouth conditions when freshwater conditions dominated the SCRE, rainbow trout (*O. mykiss*) juveniles[12] and the amphipod *Hyalella Azteca* were tested. During open-mouth conditions and depending upon the observed salinity in receiving water samples the marine species tested was the sea urchin (*Strongylocentrotus purpuratus*). When receiving water samples collected during open-mouth conditions exhibited a range of salinities, all samples in excess of 2 ppt were adjusted to full seawater strength (approx. 35 ppt) for these tests. Additional details on the species selected for the quarterly toxicity testing and toxicity results are presented in included in Appendix E**.**

Toxicity results during the Phase 3 quarterly testing from 2015–2016 indicated no significant effects on test organisms of water collected at the effluent transfer station water (ETS), lower SCRE (Site E-1), upper SCRE (Site E-2), and the riverine/estuarine site (R-005) (Appendix E). Additional toxicity results and discussion will be provided as part of the Nutrient and Toxicity Study as described in Stillwater Sciences 2015.

3.4.2 Groundwater quality

Analytical water quality samples were collected quarterly at GW-1 through GW-15 during the Phase 3 Study between 2015 to 2016. Annual averages of nitrogen species are presented in Figure 3-40 and summary statistics of grab sample analysis results are presented in Table 3-22. Nitrate was the dominant nitrogen species within most groundwater wells, with the highest concentrations at wells at the TNC, Bailard landfill, and north bank monitoring locations. The groundwater wells located near the VWRF ponds had low nitrate concentrations and a higher proportion of ammonia in the water, apart from GW-12. Orthophosphate (PO_4^{3-}) concentrations were generally low within the groundwater wells, with higher concentrations observed in the wells near the VWRF ponds.

[12] Juvenile test fish are generally 15–30 days old and approximately 25 mm fork length

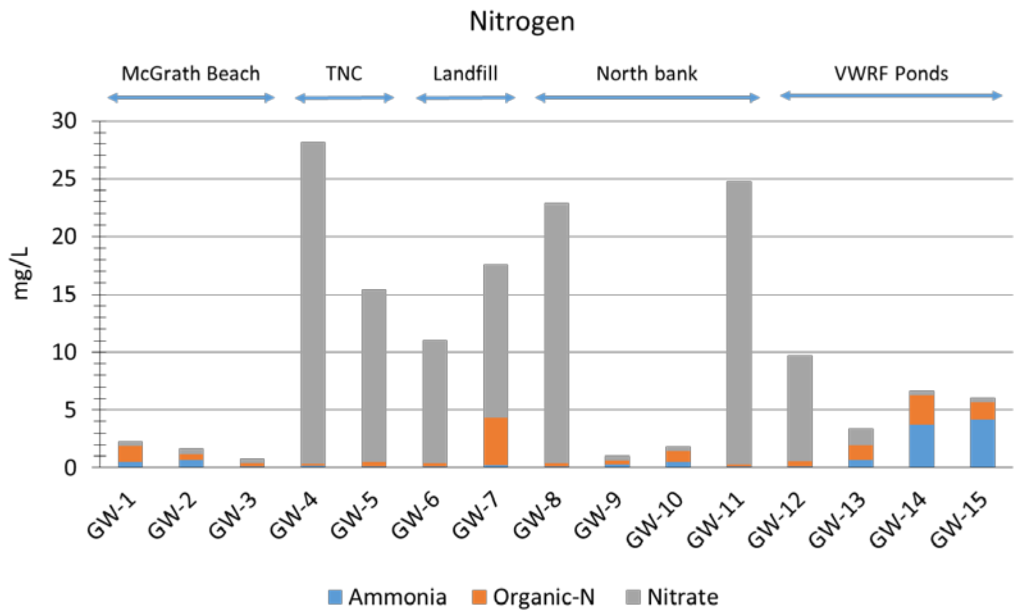


Figure 3-40. Annual mean nutrients of groundwater nitrogen collected during the Phase 3 Study (2015–2016).

Table 3-22. Nutrient concentrations in grab samples from groundwater wells (2015–2016).

Location	Site	Ammonia (mg-N/L)		Total kjeldahl nitrogen (mg/L)		Organic-N (mg-N/L)		Nitrate (mg-N/L)		Phosphate (mg-P/L)		Total phosphorus (mg-P/L)	
		Mean	**Range**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**	**Mean**	**Range**
McGrath beach	GW-1	0.5	0.1–0.9	1.9	1.03–3.8	1.4	0.93–2.9	0.4	0.4–0.4	0.2	0.15–0.4	1.6	0.3–3.6
	GW-2	0.7	0.1–1.3	1.2	0.4–1.7	0.5	0.03–1.0	0.5	0.4–1.52	0.2	0.15–0.4	0.4	0.1–0.7
	GW-3	0.1	0.1–0.2	0.4	0.1–0.7	0.3	0–0.6	0.4	0.4–0.45	0.2	0.15–0.4	0.2	0.1–0.3
TNC	GW-4	0.2	0.1–0.6	0.4	0.1–0.8	0.2	0.01–0.7	27.8	23–32	0.2	0.15–0.4	0.1	0.1–0.4
	GW-5	0.1	0.1–0.2	0.5	0.1–1.3	0.4	0–1.2	14.9	5.2–21	0.2	0.16–0.4	0.3	0.1–0.4
Bailard landfill	GW-6	0.1	0.02–0.2	0.4	0.1–1.1	0.3	0–0.9	10.6	0.4–29	0.2	0.15–0.4	0.7	0.2–1.4
	GW-7	0.2	0.1–0.4	4.4	1.1–6.6	4.1	1.1–6.4	13.3	11–18	0.2	0.2–0.2	0.4	0.3–0.4
North bank	GW-8	0.1	0.1–0.1	0.4	0.1–1.2	0.3	0.01–1.1	22.5	18–29	0.2	0.15–0.4	0.4	0.1–1.4
	GW-9	0.3	0.1–0.5	0.6	0.2–1.4	0.3	0.01–1	<0.4	NA	0.2	0.15–0.4	0.9	0.1–5.1
	GW-10	0.5	0.1–0.7	1.4	0.5–3.0	0.9	0.05–2.7	<0.4	0.4–0.4	0.2	0.15–0.4	0.4	0.1–0.7
	GW-11	0.1	0.1–0.1	0.3	0.1–0.7	0.2	0–0.6	24.5	20–29	0.2	0.15–0.4	0.6	0.02–4.1
VWRF ponds	GW-12	0.1	0.1–0.1	0.6	0.1–1.5	0.5	0–1.4	9.2	5–17	1.5	0.2–2.2	2.5	0.7–5.4
	GW-13	0.7	0.4–1.0	1.8	0.1–4.0	1.3	0.3–2.9	1.4	0.4–4.6	3.6	0.4–4.6	4.0	0.8–5.5
	GW-14	3.7	1.3–6.5	6.2	4.5–7.3	2.5	0.8–4	<0.4	NA	2.3	0.4–3.0	3.0	1.0–3.8
	GW-15	4.2	1.7–15.2	5.7	2.3–18.6	1.5	0.6–3.4	<0.4	NA	2.8	0.4–3.3	4.6	1.2–6.0

Data sources = 2015–2016 Phase 3 data included in Appendix B.

3.4.3 Constituents of emerging concern (CECs)

Constituents of Emerging Concern (CECs) encompass many unregulated chemicals that occur in air, water, sediment and biota, including pharmaceuticals, flame retardants, newly registered pesticides as well as other newly developed products such as nanomaterials. Generally, many of these chemicals have likely been present in the environment in the past, but at concentrations that were not detectable by commonly used analytical methods. Because recent advances in qualitative and quantitative analytical chemistry have indicated that treated wastewater effluent and storm water runoff represent major sources of CECs to California's inland waterways and coastal aquatic systems, several data collection initiatives have been established to estimate the potential hazards of CECs.

Per requirements of the VWRF NPDES permit, the City has collected CEC data for several locations including the ETS (NPDES M-001), Wildlife Polishing Pond discharge to the SCRE (D-1 or NPDES M-001A), SCRE receiving water locations (E-1 and E-2), and the upstream location (R-005). In accordance with the Workplan, the City has conducted annual Contaminants of Emerging Concern (CEC) sampling at three sampling events; August 12, 2015, November 12, 2015, August 10, 2016 and has compiled results of analyses of over 100 CECs (Appendix B). While not all CECs were measured at each event most CECs were measured in at least two events. The CECs measured in only one event include desulfinylfipronil, fipronil sulfide, and fipronil sulfone. It is noteworthy that the concentrations of many CECs were below the analytical detection limits.

3.4.3.1 Screening Approach

The potential risk to the health of aquatic species from CECs in the VWRF discharge and receiving water was assessed using an approach that relies on benchmarks for the predicted no-effect concentration (PNEC), and includes the following:

- Use of occurrence data to identify the CECs that should be evaluated for potential risks to aquatic health
- Identification of corresponding PNEC values for the selected CECs
- Calculation of the ratio of measured concentrations to the PNECs, and risk based evaluation.

CECs were selected for further assessment based on the combined analytical results of the three sampling events. CECs were selected for further assessment of aquatic health risk using the following criteria:

- A detected concentration at the Wildlife Pond Effluent (Site D-1 or NPDES M-001A), and
- If the maximum concentration at D-1 was greater than the minimum concentration at E-1 or E-2 (i.e., if the VWRF Wildlife Pond discharge concentration is greater than the SCRE concentration)
- Or if the maximum concentration at D-1 was greater than the minimum concentration at R-005 (i.e., if the VWRF Wildlife Pond discharge concentration is greater than the upstream concentration entering the SCRE)

Based on the criteria above, 36 CECs were selected for further analysis.

3.4.3.2 Predicted no-effect concentrations of identified CECs

As mentioned previously, the analysis of potential aquatic health effects is based on PNEC values. PNECs are the value below which exposure to a substance is not expected to cause adverse effects. PNECs are based on toxicological dose-response data for a range of organisms and includes a safety factor that accounts for the uncertainty of the available data, the variability in species response, and unknown long-term effects (Rauch-Williams et al. 2016). PNEC values are several orders of magnitude lower than the dose where toxicological effects are observed.

The approach used for this study follows the approach developed for a recent Water Research Foundation Report (Rauch-Williams et al. 2016). As part of the research report, an extensive and peer reviewed literature search for aquatic PNEC values was conducted. Recognizing that survival, growth, and reproductive endpoints are the most commonly reported toxicological data in the literature the researchers selected the lowest aquatic PNEC found in the literature for the purpose of using the most conservative response and thereby minimizing risk.

Among the selected 36 CECs with concentrations at the Pond discharge in excess of SCRE concentrations, there were 5 that were also included Rauch-Williams et al. (2016), including triclosan, TCEP, TDCPP, fipronil, and carbamzapine. For the remaining aquatic PNECs an additional literature search was conducted with assistance provided by some of the researchers involved in the previous Water Research Foundation study. Aquatic PNEC values were found for an additional 17 CECs. Table 3-23 presents aquatic PNEC values for 22 of the 36 selected CECs.

Table 3-23. Predicted no-effect concentration (PNEC) values for selected CECs.

CEC	Aquatic PNEC (ng/L)	Source
4-Nonylphenol-d4	Not Available	-
Atenolol	1,800	Trussel (2012)
Azithromycin	260	Zhou et al. (2016)
Caffeine	87,000	https://echa.europa.eu/registration-dossier/-/registered-dossier/10085/6/1
Carbamazepine	250	As reported in Rauch-Williams et al. (2016)
Cotinine	Not Available	-
DEET	7,700	Trussell (2012)
Desulfinylfipronil	Not Available	-
Diazepam	180	Ascough et al (2000)
Dichloran	Not Available	-
Fipronil	11	As reported in Rauch-Williams et al. (2016)
Fipronil sulfide	110,000	EURA Report (2009)
Fipronil sulfone	25,000	EURA Report (2009)
Fluoxetine	1,400	Trussell (2012)
Galaxolide (HHCB)	7,000	SCCWRP
Gemfibrozil	100,000	Luo (2014)
Ibuprofen	5,000	Luo (2014)
Iopromide	Not Available	-

CEC	Aquatic PNEC (ng/L)	Source
Meprobamate	Not Available	-
Methadone	Not Available	-
Naproxen	37,000	Luo (2014)
Nonylphenol	1,700	Trussell (2012)
Nonylphenol monoethoxylate	Not Available	-
Perfluorooctane sulfonate (PFOS)	1,200	Trussell (2012)
Perfluorooctanoic acid (PFOA)	280,000	Japan MOE (http://www.env.go.jp/en/chemi/chemicals/profile_erac/profile9/pf1-04.pdf; http://www.env.go.jp/en/chemi/chemicals/profile_erac/profile9/pf1-12.pdf; https://www.env.go.jp/en/chemi/chemicals/profile_erac/profile4/pf1-10.pdf)
Perylene-d12	Not Available	-
Phenytoin (Dilantin)	Not Available	-
Primidone	Not Available	-
Sulfamethoxazole	590	Straub (2016)
Sumithrin (Phenothrin)	Not Available	-
TCEP	65,000	As reported in Rauch-Williams et al. (2016)
TCPP	Not Available	-
TDCPP	10,000	As reported in Rauch-Williams et al. (2016)
Triclosan	69	As reported in Rauch-Williams et al. (2016)
Trimethoprim	5,100	Straub (2013)
Triphenyl phosphate	Not Available	-

3.4.3.3 Risk based assessment of identified CECs

For each CEC with an associated PNEC from the literature, the ratio of the maximum environmental concentration (MEC) to the PNEC was calculated. The maximum concentrations across the three sampling events were compiled for each sampling location and used to calculate the MEC/PNEC ratios, as shown in Table 3-24. In some cases, the D-1 and/or ETS concentrations were above the detection limits, but concentrations at R-1, E-1, and/or E-2 sites were below detection limits. In these cases, "ND" is shown in Table 3-24.

Table 3-24. Comparison of maximum environmental concentrations at Phase 3 monitoring sites to identified PNECs.

CEC	Aquatic PNEC (ng/L)	MEC/Aquatic PNEC Ratio				
		ETS	D1 (M001A)	E1 (R3)	E2	R1 (R-005)
4-Nonylphenol-d4	Not Available	-	-	-	-	-
Atenolol	1,800	4.3E-02	3.1E-02	4.4E-03	2.4E-03	3.1E-02
Azithromycin	260	1.2E+00	3.3E-01	ND	ND	ND
Caffeine	87,000	2.5E-04	5.2E-04	6.4E-04	6.2E-04	4.9E-04
Carbamazepine	250	1.1E+00	1.0E+00	5.6E-01	5.2E-01	5.6E-01
Cotinine	Not Available	-	-	-	-	-
DEET	7,700	3.9E-02	1.4E-02	1.4E-02	1.3E-02	1.3E-02
Desulfinylfipronil	Not Available	-	-	-	-	-
Diazepam	180	9.4E-02	3.1E-02	3.1E-02	1.8E-02	1.4E-02
Dichloran	Not Available	-	-	-	-	-
Fipronil	11	1.1E+01	5.2E+00	ND	ND	ND
Fipronil sulfide	110,000	1.2E-04	1.2E-04	ND	ND	ND
Fipronil sulfone	25,000	2.2E-03	1.7E-03	ND	ND	ND
Fluoxetine	1,400	3.7E-02	1.0E-02	4.5E-03	1.8E-03	1.6E-03
Galaxolide (HHCB)	7,000	5.7E-01	1.3E-01	5.4E-02	1.4E-02	1.1E-02
Gemfibrozil	100000	4.7E-05	6.4E-05	3.3E-05	4.4E-05	4.0E-05
Ibuprofen	5,000	6.4E-04	1.2E-03	3.2E-03	6.8E-04	4.8E-04
Iopromide	Not Available	-	-	-	-	-
Meprobamate	Not Available	-	-	-	-	-
Methadone	Not Available	-	-	-	-	-
Naproxen	37000	4.1E-04	4.1E-04	5.4E-05	ND	ND
Nonylphenol	1700	2.6E-01	7.6E-01	4.4E-01	5.4E-01	2.4E-01
Nonylphenol monoethoxylate	Not Available	-	-	-	-	-
Perfluorooctane sulfonate (PFOS)	1200	2.5E-02	5.7E-03	1.0E-02	1.3E-02	1.1E-02
Perfluorooctanoic acid (PFOA)	280000	9.3E-05	7.5E-05	8.6E-05	8.2E-05	7.9E-05
Perylene-d12	Not Available	-	-	-	-	-
Phenytoin (Dilantin)	Not Available	-	-	-	-	-
Primidone	Not Available	-	-	-	-	-
Sulfamethoxazole	590	1.9E+00	1.9E+00	1.2E+00	6.4E-01	6.3E-01
Sumithrin (Phenothrin)	Not Available	-	-	-	-	-
tris(2-chloroehtyl) phosphate (TCEP)	65,000	3.1E-03	3.1E-03	3.7E-03	3.1E-03	3.2E-03
TCPP	Not Available	-	-	-	-	-
tris(1,3-dichloro-2-propyl) phosphate (TDCPP)	10,000	4.3E-02	2.5E-02	2.4E-02	1.4E-02	1.2E-02
Triclosan	69	2.9E-02	9.1E-02	3.0E-02	9.7E-02	9.3E-02
Trimethoprim	5,100	3.7E-03	4.3E-03	2.2E-03	1.7E-03	1.4E-03
Triphenyl phosphate	Not Available	-	-	-	-	-

The shaded cells in Table 3-24 indicate a MEC/aquatic PNEC ratio that is greater than or equal to 1. A MEC/aquatic PNEC ratio that exceeds 1.0 does not necessarily represent a definitive concentration where aquatic health effects will be observed, but increasingly higher ratios over

1.0 are estimated to represent increasingly greater risks to aquatic health. Table 3-25 provides guidance on understanding risk associated with MEC/aquatic PNEC ratios.

Table 3-25. Risk interpretation of calculated MEC/aquatic PNEC ratios.

MEC/aquatic PNEC Ratio	Interpretation
<0.1	Insignificant environmental risk
0.1 to 1	Low environmental risk
1-10	Moderate environmental risk
>10	High environmental risk

Source: Adapted from Karlsson (2007) with reference to European Commission (2003)

Most of the MEC/aquatic PNEC ratios for the CECs in Table 3-24 are below 0.1, indicating an insignificant risk of adverse health effects on the aquatic species in the discharge channel and SCRE. The MEC/aquatic PNEC ratios exceeded 1 for only four CECs, as presented below.

1. **Azithromycin:** The MEC/aquatic PNEC ratio for the antibiotic azithromycin found in the ETS sampling location is 1.2. The ratio does not exceed 1.0 in the Wildlife Pond discharge (D-1), SCRE sites (E1 and E2) or the upstream site (R1). Because aquatic toxicity endpoints for invertebrate and fish species are several orders of magnitude higher than for reproductive endpoints for microbial and algal receptors (e.g., Selanastrum), these results suggest only a moderate risk of azithromycin related algae found in the Wildlife Ponds. However, while ratio for the ETS sample is a moderate risk level, less than detection level concentrations were observed in SCRE. Furthermore, while a moderate risk is associated with ratios greater than 1, a ratio of 1.2 is on the low end of the moderate risk level characterization.
2. **Carbamazepine:** The MEC/aquatic PNEC ratio for the pharmaceutical carbamazepine is 1.1 at the ETS site, and is exactly 1.0 at D-1 (Table 3-24) Although carbamazepine has been shown to affect reproduction of aquatic invertebrates such as Cladcoeran *Daphnia magna* because the MEC/aquatic PNEC ratios do not exceed 1.0 at the Wildlife Pond discharge, at the upstream riverine site (R-005) or any of the SCRE (E-1 and E-2) sites suggests a low to moderate risk for carbamazepine related adverse aquatic health effects for aquatic species. The ratios for the SCRE sites suggest a low environmental risk from carbamazepine.
3. **Fipronil**: The MEC/aquatic PNEC ratios for the insecticide fipronil are 1.1 and 5.2 at the ETS and the Wildlife Pond discharge (D-1), respectively (Table 3-24). Although fipronil has been shown to be toxic to aquatic invertebrates and some fish species, because fipronil concentrations in the SCRE and upstream locations were below detection levels, the ratio of 5.2 at the Wildlife Pond discharge suggests a low risk of fipronil related adverse aquatic health effects for aquatic species in the SCRE. However, while ratio for the ETS and discharge sites suggest a moderate risk level, less than detection level concentrations of fipronil were observed in SCRE.
4. **Sulfamethoxazole:** The MEC/aquatic PNEC ratios for the antibiotic sulfamethoxazole are 1.9 for at the ETS and discharge channel (D-1) sites, and 1.2 for one of the SCRE sites (E-1) (Table 3-24). The ratios were below 1 for the other SCRE site and upstream location (R-1). Because aquatic toxicity endpoints for many invertebrate and fish species are several orders of magnitude higher than for microbial and algal receptors (e.g., Cyanobacteria), the ratio of 1.9 in the Wildlife Pond discharge suggests only a moderate risk of sulfamethoxazole related adverse aquatic health effects for the aquatic species in the

Ponds. As the ratio for the SCRE is 1.2, the results suggest the SCRE is on the very low end of the "moderate environmental risk" categorization for sulfamethoxazole related adverse aquatic health effects for these aquatic species in the SCRE.

Based upon the results to date, the potential risks of the four identified CECs with VWRF or Wildlife Pond discharge concentrations in excess of those found in the SCRE is low for aquatic species within the SCRE. It is important to note that CEC related toxicity research is ongoing, and as more data are compiled and published, the body of research will expand. The literature should be revisited periodically to determine if new research has identified aquatic PNECs values for the 14 CECs currently not identified, and to determine if any of the of the previously selected aquatic PNEC values should be modified.

3.5 Vegetation and Wetlands

The current and historical conditions of vegetation (habitat types), including wetlands, are presented and compared in this section, and the potential habitat for special-status plant species in the project area is discussed. Because of the influence of short duration, high intensity flood events, the present-day Santa Clara River channel fluctuates between meandering and braided river forms, as defined by the gradient, discharge, and bed material grain size (Stillwater Sciences 2007b). These factors result in a mosaic of riparian vegetation that shifts in extent, structure, and composition in response to deposition, scour, and inundation by large flood flows. To enable comparisons of the changes in vegetation over time, previously mapped vegetation alliances (Stillwater Sciences 2011) have been grouped into broader habitat types. Habitat types in the SCRE include developed/disturbed, foredune, ocean, open beach, open water, riparian vegetation, and both freshwater and salt marsh wetlands (Table 3-26).

3.5.1 Current conditions

Using the previously mapped vegetation alliances (Stillwater Sciences 2011), habitat types were field verified during 2015 field surveys and further updated by conducting aerial imagery interpretation of two datasets—the coarser National Agriculture Imagery Program (NAIP) imagery depicts the lagoon at full stage in 2016 (NAIP 2016) and imagery from unmanned aircraft system (i.e., drone) flights conducted during September and December 2016 field efforts provides higher resolution data to increase accuracy. Together these data were used to update the position and extent of open water as well as boundaries of previously mapped habitat types (Figure 3-41 and Table 3-26). Descriptions of each habitat type, including the associated vegetation alliances, are based primarily on field data collected as part of the 2009 vegetation mapping effort (Stillwater Sciences 2011) and are provided in the sections below. Vegetation alliance names used in this section conform to the classification system in *The Manual of California Vegetation, Second Edition* (Sawyer et al. 2009); additional plant species naming conventions follow the taxonomy of *The Second Edition of the Jepson Manual* (Baldwin et al. 2012). Any vegetation alliances that are rare natural communities (i.e., natural community of special concern [S1–S3 on CDFW's *List of California Terrestrial Natural Communities*; CDFG 2010]) or environmentally sensitive habitat areas (ESHAs) defined in the City of Oxnard Local Coastal Plan (LCP; City of Oxnard 1982) are also noted.

3.5.1.1 Developed/Disturbed

Lands associated with McGrath State Beach campground, related facilities, portions of the VWRF, and access roads are mapped as developed/disturbed. Much of the developed/disturbed area surrounding the campground is disturbed wetlands (WRA, Inc. 2014a). The developed/disturbed area within the VWRF is a mix of developed trails and sandy areas colonized by the non-native invasive sea fig (*Carpobrotus chilensis*) and freeway iceplant (*Carpobrotus edulis*).

3.5.1.2 Foredune

Located in a strip along the coastal strand on both sides of the SCRE, this habitat type is comprised of predominantly native vegetation dominated by dune mat (*Abronia* spp. – *Ambrosia chamissonis* Herbaceous Alliance, a rare natural community [S3; CDFG 2010]). The foredune also includes large patches of the non-native invasive ice plant mats (*Carpobrotus edulis* or Other Ice Plants Herbaceous Semi-Natural Alliance). In the project area, it is found immediately inland from the open beach. Dunes are included as an ESHA in the LCP (City of Oxnard 1982).

3.5.1.3 Mudflat

Because the mouth of the SCRE was closed at the time of the 2016 mapping, mudflat habitats were initially mapped as open water. Habitat that is mapped as open water, but within interidal elevations (above MLW) under open mouth conditions is mapped as mudflat habitat. Mudflats provide foraging habitats for shorebirds when exposed and foraging for SCRE fish species when submerged.

3.5.1.4 Ocean

Ocean is included in acreage mapping to capture the effect of shoreline fluctuation over time with river flow and breach dynamics. It is distinct from estuary waters, which are included in the open water habitat type.

3.5.1.5 Open beach

Open beach is characterized as marine-associated open sand uncolonized by vegetation. In the project area, it is found immediately adjacent to the ocean and is generally bounded to the east by either foredune or estuarine open water.

3.5.1.6 Open water

The mapping effort considered open water to be any non-ocean water surface that lacks emergent or established vegetation. In the project area, this included the extent of the estuary, ponds, and river water. The mouth of the SCRE was closed at the time of the 2016 mapping.

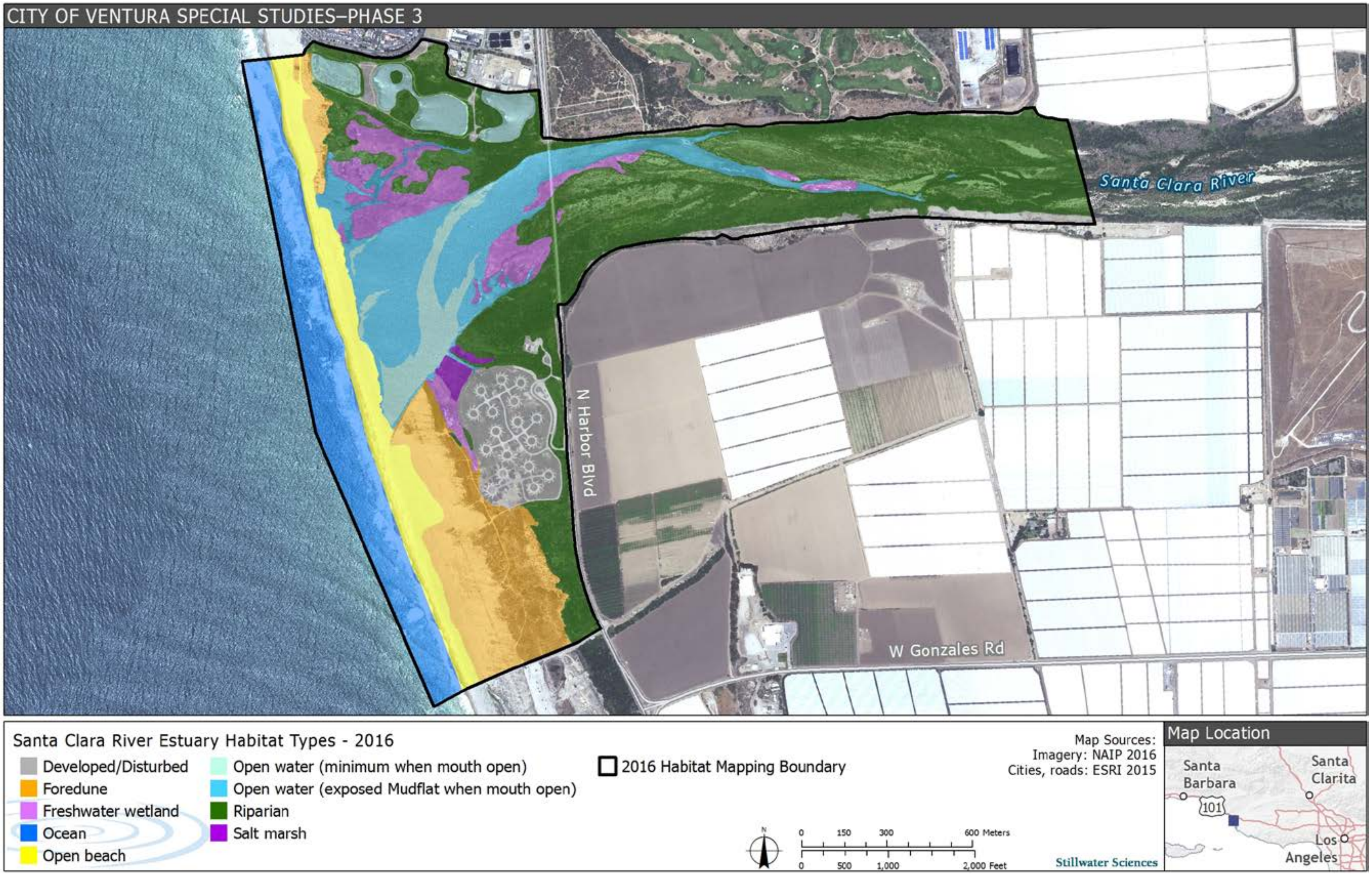


Figure 3-41. Existing habitat types in the Santa Clara River Estuary.

3.5.1.7 Riparian vegetation

This habitat type is dominated primarily by arroyo willow thickets (*Salix lasiolepis* Woodland and Shrubland Alliances), with smaller representation of shining willow groves (*Salix lucida* Woodland Alliance, a rare natural community [S3.2; CDFG 2010]) and mulefat thickets (*Baccharis salicifolia* Shrubland Alliance), and trace amounts of Riverwash Herbaceous. It also includes large, dense patches of high cover of the non-native invasive giant reed breaks (*Arundo donax* Herbaceous Alliance) in the riparian corridor. In the project area, it is generally found inland from the foredunes and in varying successional stages within a corridor between the bounds of the levees on the north and south sides of the river. Riparian habitat is included as an ESHA in the LCP (City of Oxnard 1982).

3.5.1.8 Wetland

This habitat type includes both freshwater wetland and small inclusions of salt marsh. The freshwater wetland is comprised of multiple alliances, largely represented by California bulrush marsh (*Schoenoplectus californicus* Herbaceous Alliance; a locally Rare plant species [Magney 2017]), with trace amounts of cattail marshes (*Typha* [*angustifolia*, *domingensis*, *latifolia*] Herbaceous Alliance), and false waterpepper-white sweetclover patches (*Persicaria hydropiperoides-Melilotus albus* Provisional Alliance). Freshwater wetland also includes some sporadic patches of giant reed mixed within other wetland communities, although generally giant reed was included within the riparian vegetation habitat type. The salt marsh is predominantly Pacific silverweed marshes (*Argentina egedii* Herbaceous Alliance [now *Potentilla anserina* subsp. *pacifica*], a rare natural community [S2; CDFG 2010] and locally Rare plant species [Magney 2017]), salt grass flats (*Distichlis spicata* Herbaceous Alliance), and creeping rye grass turfs (*Leymus triticoides* Herbaceous Alliance [now *Elymus triticoides*], a rare natural community [S3; CDFG 2010]). In the project area, freshwater wetland is generally found near the margins of estuary waters around the perimeter of the SCRE and adjacent to riparian vegetation. The freshwater wetland appears to die back in response to the draining of the SCRE during winter, more saline conditions when the sandbar is breached, and natural season die back. The salt marsh component occurs on the south edge of the SCRE, adjacent to McGrath State Beach, and is likely maintained by the northeast trending levee north of the campground (Swanson et al. 1990). Wetlands are included as an ESHA in the LCP (City of Oxnard 1982).

3.5.2 Historical conditions

The SCRE is representative of what was likely the rarest (in terms of area) of seven historical coastal estuarine archetypes in Southern California: steep, large river mouth estuaries (Grossinger et al. 2011). The very small estuaries at the mouths of the Ventura and Santa Clara rivers are the only examples of this archetype in Southern California identified by Grossinger and others (2011), who provide the following description:

> *"The mouths of the Santa Clara River and Ventura River were notable as very small estuaries with little tidal marsh. These systems at high energy stream mouths were dominated by willow-cottonwood swamps that transitioned into relatively small amounts of estuarine habitat types (in apparent contrast to the estuaries of other large South Coast rivers)."*

Of the non-riparian habitat types historically present, open water was most common (approximately 55–60%), with the remainder approximately evenly split between vegetated wetlands and salt flats (Figure 33 in Grossinger et al. 2011; the relative spatial distribution of these habitats is shown in Figure 6.7 in Beller et al. 2011). Below, we discuss more recent changes in vegetation types of the SCRE based on air photo interpretation as well as review of available survey data. The 2016 analysis includes an estimate of the portion of open water area that would have been mudflat under open mouth conditions; this estimation was not completed for earlier analysis years (i.e., 1977, 2002, and 2009) because the associated imagery for each of those years depicts closed mouth conditions.

3.5.2.1 Recent historical conditions and changes

Overall, the composition and extent of upland and tidal vegetation mapped in 2016 is very similar to that which was mapped in 1989 (Swanson et al. 1990), 2002 (ESA 2003), 2005 (Stillwater Sciences and URS Corporation 2007), and 2009 (Stillwater Sciences 2011). In addition to these data sources, California Department of Pesticide Regulation performed vegetation transect surveys in the McGrath State Beach and McGrath Lake areas in 2007/2008. Combining these resources with available aerial imagery under closed mouth conditions in 1977 (Teledyne Geotronics 1977), 2002 (Airphoto USA 2002), and 2009 (NAIP 2009), Stillwater Sciences also mapped the historical extent of habitat types (Figure 3-42 through Figure 3-44). Together, these mapping efforts provide a comparison of riparian and wetland vegetation conditions around the SCRE, and a description of the inter-annual variation that has occurred (Figure 3-45 and Table 3-26). It should be noted that because the SCRE mouth was closed at the time the photographs were taken, the extent of tidally exposed mudflats could not be estimated.

Table 3-26. Summary of acres of habitat types in the SCRE by year.

Habitat type		1977	2002	2009	2016
Ocean		95.2	46.9	30.7	75.7
Open beach		52.1	49.2	46.3	48.8
Foredune		60.8	95.3	87.6	77.4
Open water[1]		123.9	102.4	127.8	114.1
Tidally Exposed Mudflat					74.6
Wetlands	Freshwater	10.2	29.2	43.0	40.1
	Salt marsh	23.5	8.0	4.2	3.6
Riparian		202.2	258.2	246.3	228.3
Developed/Disturbed		75.2	53.8	55.5	55.1

[1] The estimate for "Open Water" includes the area of the SCRE that remains open water under open-mouth conditions when the SCRE reaches its minimum stage of 4.5 ft NAVD88 and the constant open water area from the VWRF ponds.

The primary, substantive changes in recent historical vegetation composition and 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 January–February 2005 floods that resulted in large areas of bare riverwash. The vast majority of this area was mapped as Riverwash Herbaceous Super-Alliance in 2006, defined by Stillwater Sciences and URS Corporation (2007) as dominated by white sweetclover (*Melilotus albus*), generally ranging from 10–35% cover, as well as some areas with up to 10% cover of seedlings of willows (*Salix exigua*, *S. laevigata*, *S. lucida*, *S. lasiandra*, and *S. lasiolepis*), mulefat (*Baccharis salicifolia*), and occasionally Fremont cottonwood (*Populus fremontii* subsp. *fremontii*). In addition, low levels (up to 10% cover) of the non-native giant reed

(*Arundo donax*) were often present. By 2009, the herbaceous vegetation had 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).

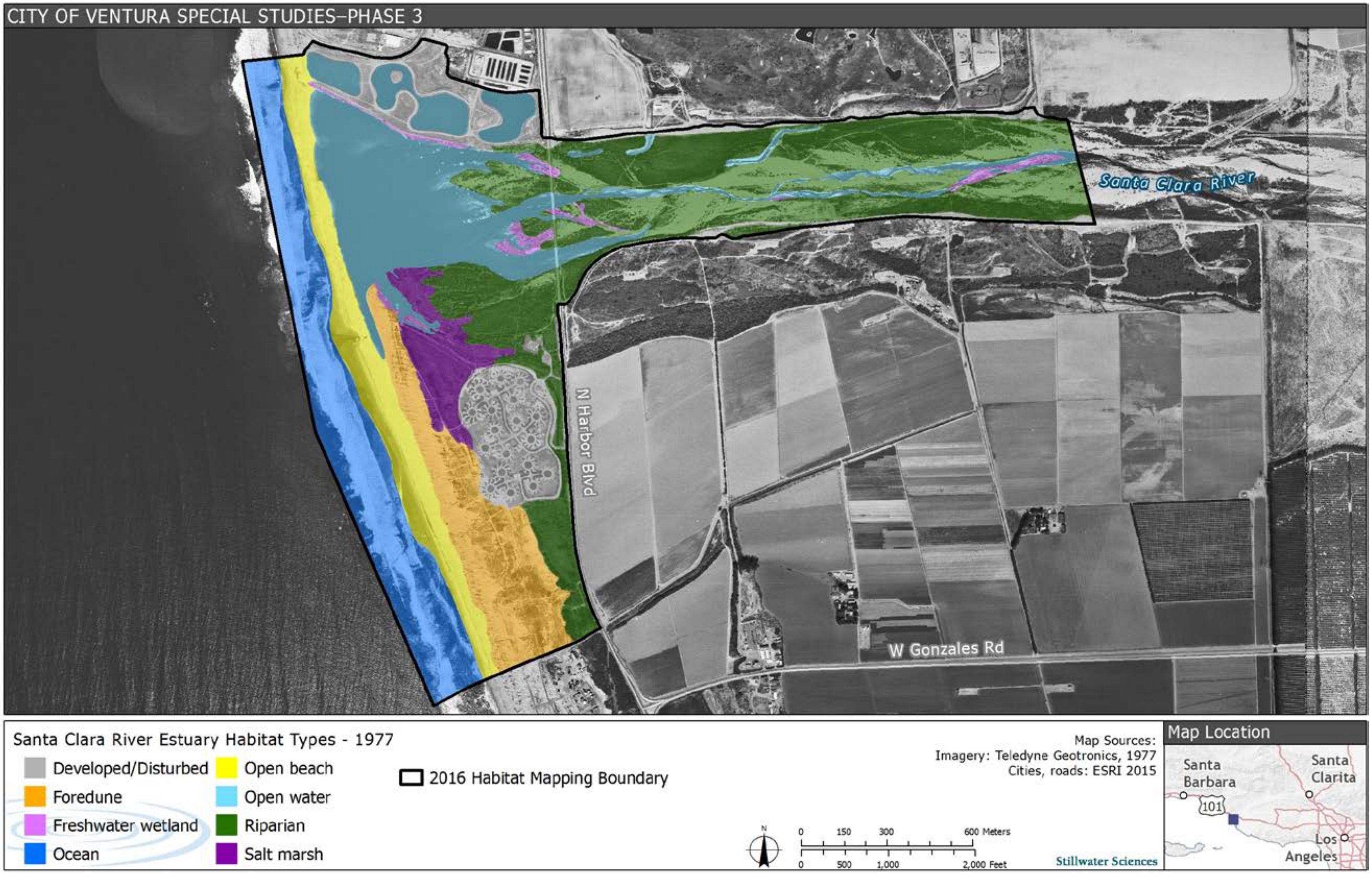


Figure 3-42. Historical (1977) habitat types in the Santa Clara River Estuary.

Figure 3-43. Historical (2002) habitat types in the Santa Clara River Estuary.

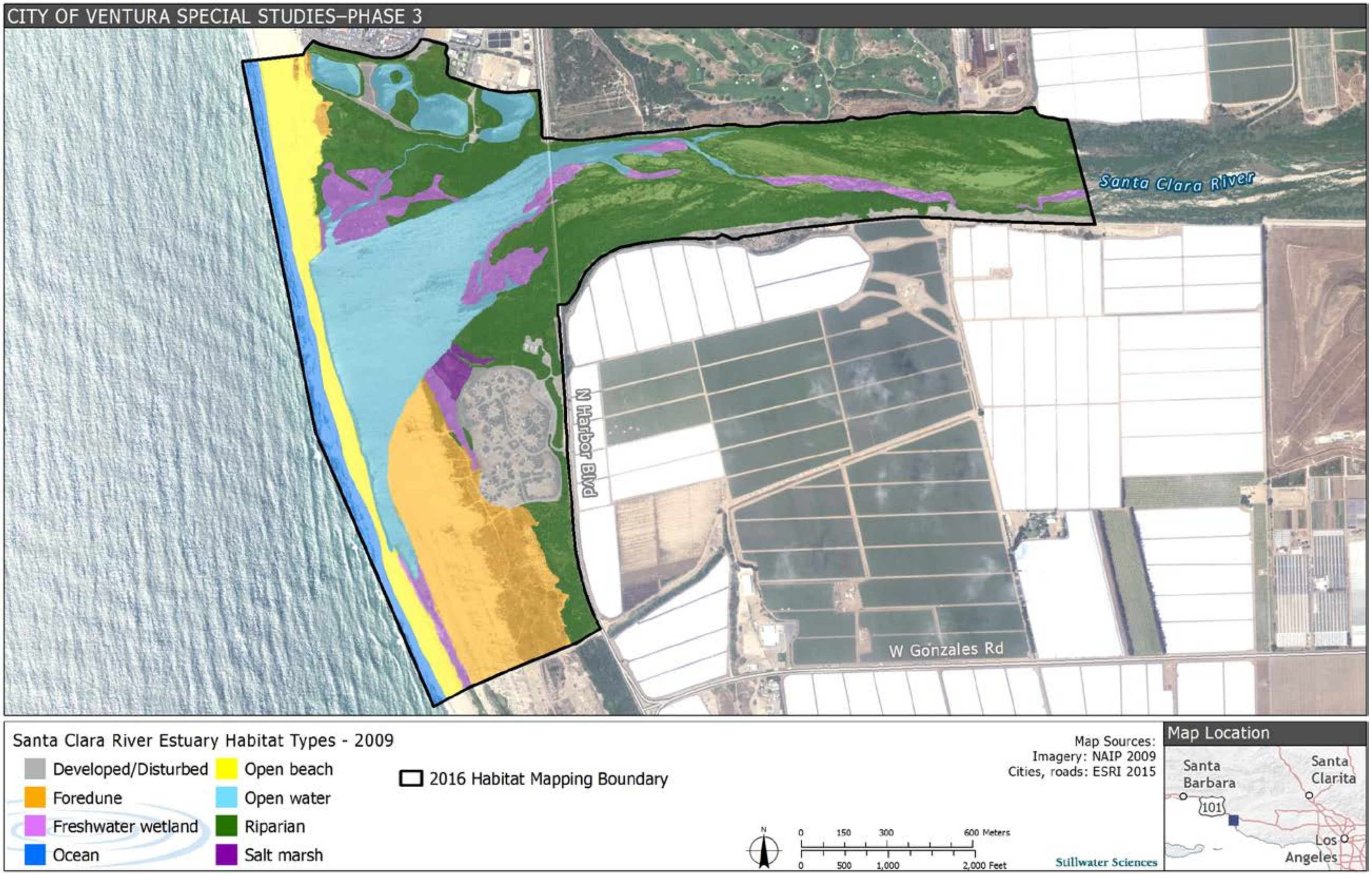


Figure 3-44. Historical (2009) habitat types in the Santa Clara River Estuary.

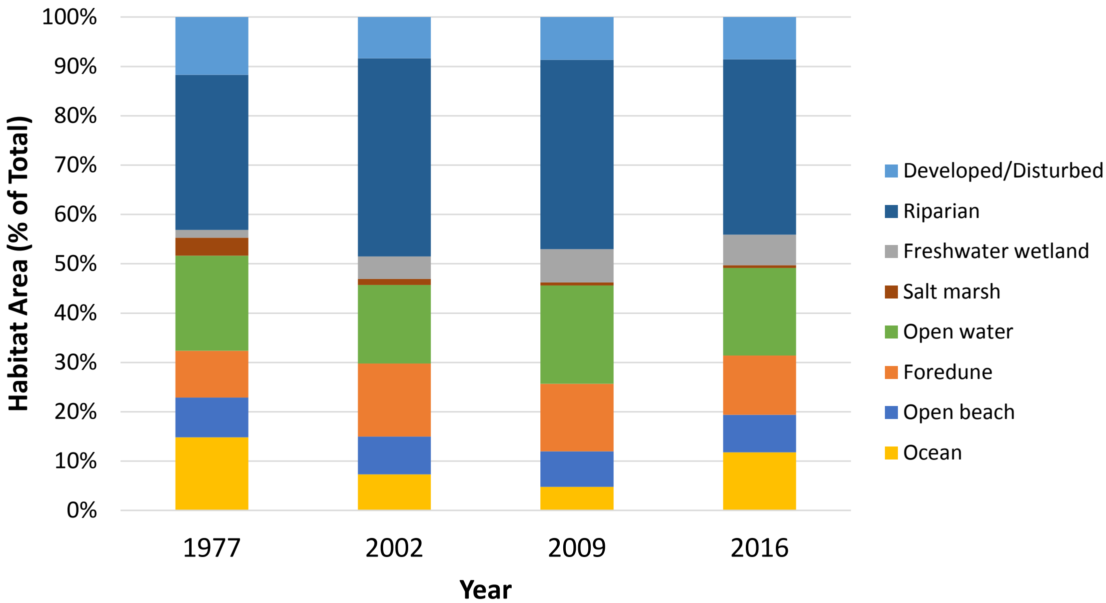


Figure 3-45. Composition of habitat types in the SCRE by year (1977–2016).

3.5.2.2 Previous historical conditions

Historical conditions in and around the SCRE dating back to 1855 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). 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 (Figure 3-46).

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 one mile behind the sand dunes and patches of vegetated wetland and salt flat/playa occurred. 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, it does map salt flat/playa. Swanson et al. (1990) found that vegetation and topographic patterns were suggestive of salt marsh and Nautilus Environmental (2005) identified approximately 16 acres 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.

Aerial photography from 1927 and 1938 supports this interpretation of the 1855 map (Stillwater Sciences 2011). 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. Multiple, braided channels were present, both within the wide, sandy mainstem channel and on the floodplains. A large tract of riparian vegetation, ½-mile wide in some places, occurred on the northern floodplain. Stillwater Sciences (2011) provides a thorough timeline of land use changes and developments after the 1855 map, as well as the impacts such actions have had on the Santa Clara riparian, estuarine, and marine vegetation. In summary, the major changes include a loss of riparian shrublands and woodlands, changes in freshwater wetland distribution, and invasion by non-native plant species, each of which are described in further detail below.

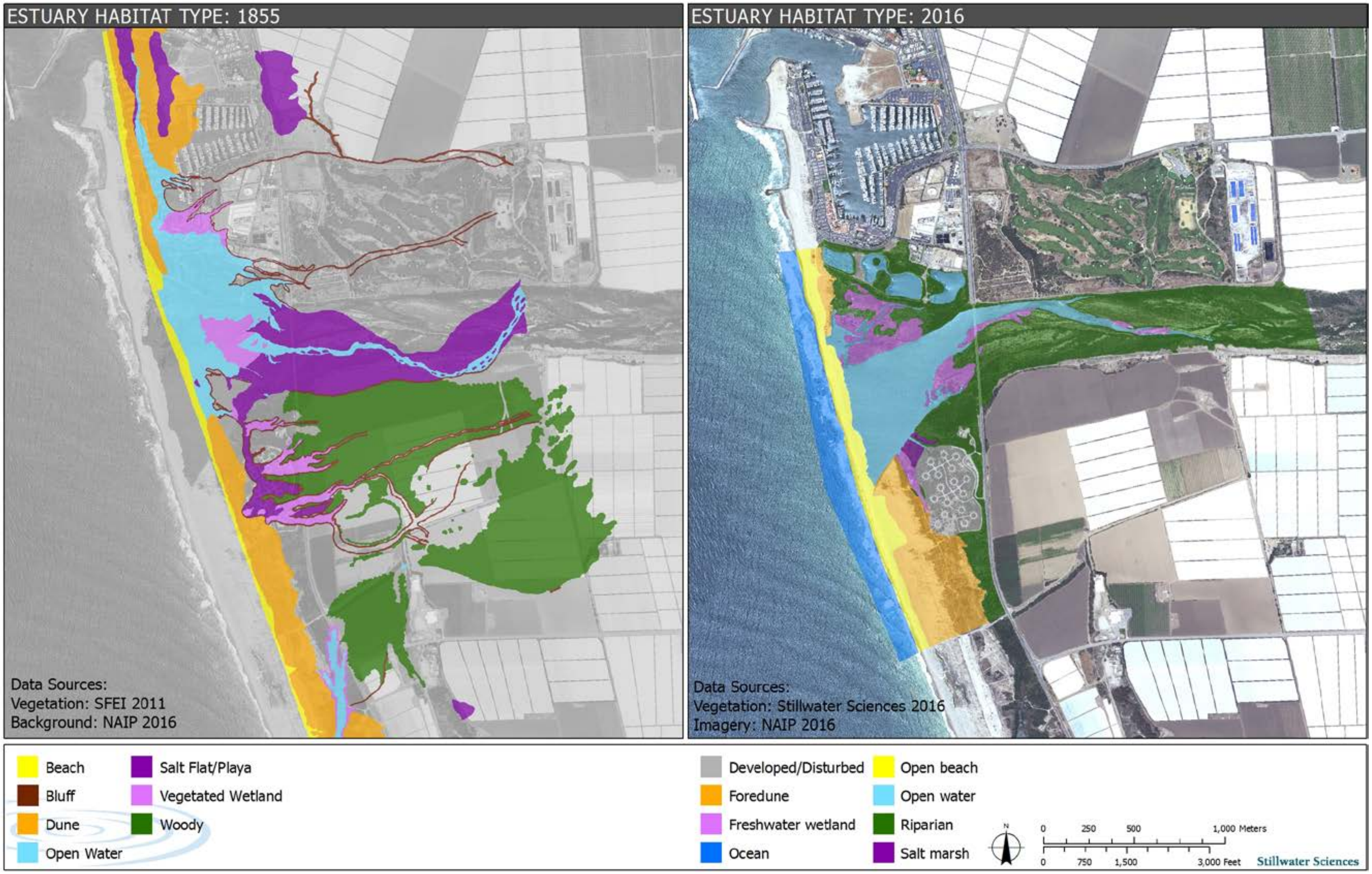


Figure 3-46. SCRE and adjacent floodplain vegetation in 1855 (Grossinger et al. 2011) and 2016 (Stillwater Sciences 2016).

3.5.2.3 Major changes to vegetation

Loss of riparian shrublands and woodlands

Under natural conditions, dynamic patterns of flooding, scour, and sediment deposition contributed to the formation and maintenance of riparian shrublands and woodlands in the SCRE area (Stillwater Sciences and URS Corporation 2007). 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. Simons, Li & Associates (1983) report that the Santa Clara River floodplain was, historically, as much as two miles wide in its lowermost reaches. The riparian area likely supported dense, multi-storied stands of broadleaf trees, including cottonwood (*Populus* spp.), western sycamore (*Platanus racemosa*), and willows (*Salix* spp.), that rely on shallow groundwater. Based on a review of numerous historical resources, Grossinger et al. (2011) concluded 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 seven miles of the river, and Nautilus Environmental (2005) calculated a 91% reduction in willow riparian woodlands around the SCRE. 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 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. 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 support 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.

Changes in freshwater wetland distribution

Comparison of the 1855 and 2016 vegetation maps suggests that the total acreage of freshwater wetland vegetation in the SCRE area has not changed dramatically, but that the way it is distributed in the landscape has. In 1855, wetland vegetation (and potentially salt marsh, per Nautilus Environmental 2005), was found in the backwater areas, side-channels, and sloughs of the lower Santa Clara River. Based upon analysis of historical aerial photographs, the vegetation in these areas had changed by 1927 with wetland vegetation and a small amount of salt-marsh now occurring as a ring around the perimeter of the SCRE (Beller et al. 2011, Stillwater Sciences 2011).

Invasion by non-native plant species

Giant reed and iceplant are both aggressive, non-native invasive plants that have become widespread in and around the SCRE.

Iceplant

Over 50 acres of iceplant were mapped in the foredunes both north and south of the SCRE, and it was documented as 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 it 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, an 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). In addition, iceplant traps more sand than native dune species and generally stabilizes dune communities unnaturally (DiTomaso and Healy 2007).

Giant reed

Over 16 acres of giant reed (*Arundo donax*) were mapped in the SCRE, primarily along the northern perimeter, and it is a dominant or co-occurring species in much of the riparian shrublands and woodlands in the area (Stillwater Sciences 2009). Giant reed is a highly aggressive member of the grass family that invades riparian zones by establishing dense, monospecific clonal stands generally propagating and establishing from rhizomes that wash downstream from eroded banks. It has a higher tolerance for inundation relative to bulrush and other native marsh species, and thus can establish further into open water areas. Dense stands typically develop, which often displace or discourage native vegetation, diminish wildlife habitat, and increase flooding and siltation in natural areas (DiTomaso 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 decrease surface and groundwater availability in semiarid regions due to a high evapotranspiration rate, which is estimated to be approximately three times higher than that of native riparian species (DiTomaso 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).

3.5.3 Special-status plants

Information on the presence and potential presence of special-status plants in the project vicinity and study area is primarily from the biological resources assessment performed in September 2014 (WRA, Inc. 2014b and cbec, Inc. et al. 2015) with additional information obtained from plant species observations recorded during the 2009–2010 vegetation mapping effort (Stillwater Sciences 2011). Neither of these studies included a comprehensive floristic survey; the biological resources evaluation was an evaluation of the potential for the site to support special-status plant species and the vegetation mapping effort was targeted at updating the aquatic and terrestrial vegetation assemblages within the project area. Thus, the most current information is based on incidental sightings recorded during a season when many taxa are unidentifiable (e.g., past blooming and/or possibly senescent). As a result, the information cannot be considered as complete results that would be obtained thorough protocol-level botanical survey.

For the purposes of this report, special-status plant species are defined as those listed, proposed, or under review as rare, threatened, or endangered by the federal or state government and/or on the California Department of Fish and Wildlife's (CDFW) *Special Vascular Plants, Bryophytes, and Lichens List* (CDFW 2016); plants with a California Rare Plant Rank (CRPR) of 1, 2, 3, or 4; and plants designated as rare or uncommon by the local California Native Plant Society (CNPS) Channel Islands Chapter (Magney 2017).

Using the previously documented information on special-status plant species in the Project Area, 31 special-status plant species have either been observed within the project area or are considered to have a moderate or high potential to occur (Table 2.5–2). The biological resources evaluation (WRA, Inc. 2014b) documented records of 26 special-status plant species in the project vicinity (i.e., database queries included the U.S. Geological Survey [USGS] 7.5-minute quadrangle map that the Project lies within [Oxnard] and two of the five adjacent quadrangles [Ventura and Point Mugu]). Of the 26 species, 19 special-status plant species were determined to either occur or have a moderate to high potential for occurrence within the boundaries of the project area. However, these results may underrepresent the number of potential special-status plant species in the project vicinity, as only three of the relevant six USGS quadrats were queried for known occurrences in the project vicinity and no protocol-level surveys were conducted (WRA, Inc. 2014b). Four of the 19 special-status plant species were incidentally observed within the study area during the September 2014 site assessment (WRA, Inc. 2014b). In addition, 15 special-status plant species were observed incidentally as part of the Stillwater (2011) vegetation mapping, three of which were included in the WRA (2014b) biological resources evaluation.

Habitat associations for the 31 special-status plant species with the potential to occur in the project area are provided in Table 3-27. These species are listed in Table 3-26 by habitat type in the following sections.

3.5.3.1 Developed/Disturbed

Twenty of the special-status plant species listed in Table 3-26 have the potential to occur within areas mapped as developed/disturbed within the project area (Table 3-27). Nine of those species have been incidentally observed within the project area; however, location data for those observations is generally unavailable as the species were not documented as part of protocol-level botanical surveys. Generally, there are two different types of developed/disturbed lands with the potential to support special-status plant species – disturbed sandy areas near foredunes and within the VWRF may support coastal dune species (e.g., *Abronia maritima*, *Astragalus pycnostachyus* var. *lanosissimus*, *Dudleya caespitosa*, etc.), and disturbed wetlands found within the campground may support wetland species (e.g., *Argentina egedii* [current name *Potentilla anserina* subsp. *pacifica*], *Atriplex lentiformis*, etc.).

In the project area, sand dune sedge (*Carex pansa*) was found on a sandy berm south of the campground, between the backdune and a grove of myoporum (*Myoporum laetum*) (WRA, Inc. 2014b and cbec, Inc. et al. 2015). Fragrant flatsedge (*Cyperus odoratus*) was found on the northern side of the campground in a disturbed wetland (WRA, Inc. 2014b and cbec, Inc. et al. 2015).

3.5.3.2 Foredune

Eighteen of the special-status plant species listed in Table 3-26 have the potential to occur within areas mapped as foredune within the project area (Table 3-27). Eight of those species have been incidentally observed within the project area; however, location data for those observations is generally unavailable as the species were not documented as part of protocol-level botanical surveys.

In the project area, red sand-verbena (*Abronia maritima*) was found at the southwestern edge of the site in limited patches on the foredunes (WRA, Inc. 2014b and cbec, Inc. et al. 2015). Sea lettuce (*Dudleya caespitosa*) was found in disturbed backdunes at the southern end of the study area (WRA, Inc. 2014b and cbec, Inc. et al. 2015).

3.5.3.3 Riparian vegetation

Seven of the special-status plant species listed in Table 3-26 have the potential to occur within areas mapped as riparian within the project area (Table 3-27). Two of those species have been incidentally observed within the project area; however, location data for those observations is generally unavailable as the species were not documented as part of protocol-level botanical surveys.

3.5.3.4 Wetland

Eighteen of the special-status plant species listed in Table 3-26 have the potential to occur within areas mapped as wetland within the project area – twelve of which may be found in either freshwater wetland or salt marsh, three of which are limited to freshwater wetlands or other non-wetland habitats, and three of which are limited to salt marsh wetlands or other non-wetland habitats (Table 3-27). Eleven of those species have been incidentally observed within the project area; however, location data for those observations is generally unavailable as the species were not documented as part of protocol-level botanical surveys.

In the project area, extensive amounts of the freshwater wetland were mapped as California bulrush marsh (*Schoenoplectus californicus* Herbaceous Alliance, Stillwater Sciences 2011). Much of the salt marsh, mapped just north of the campground, was mapped as Pacific silverweed marshes (*Argentina egedii* Herbaceous Alliance [now *Potentilla anserina* subsp. *pacifica*]; Stillwater Sciences 2011).

Table 3-27. Special-status plant species previously documented in the Project vicinity with the potential to occur in the project area[1].

Scientific name	Common name	Status[2] Federal/ State/ CRPR/ Local	Family	Blooming period	Habitat associations	Potential in the Project Area[3]
Abronia maritima	red sand-verbena	-/-/4.2/-	Nyctaginaceae	Feb-Nov	Coastal dunes	Documented in project area (Stillwater Sciences 2011 and WRA 2014b)[4]
Amsinckia spectabilis var. *spectabilis*	seaside fiddleneck	-/-/-/U	Boraginaceae	Apr-Aug	Coastal dunes, sandy bluffs	High potential to occur
Aphanisma blitoides	aphanisma	-/-/1B.2/R	Chenopodiaceae	Feb-Jun	Sandy or gravelly areas of: coastal bluff scrub, coastal dunes, and coastal scrub	High potential to occur
Argentina egedii (current name *Potentilla anserina* subsp. *pacifica*)	Pacific silverweed	-/-/-/R	Rosaceae	Mar-Oct	Coastal wetlands, often brackish	Documented in project area (Stillwater Sciences 2011)[4]
Astragalus pycnostachyus var. *lanosissimus*	Ventura marsh milkvetch	FE/CE/1B.1/R	Fabaceae	Jun-Oct	Coastal dunes, coastal scrub, and edges of coastal salt or brackish marshes and swamps	High potential to occur (WRA 2014b) and documented in project area (Stillwater Sciences 2011)[4]
Atriplex coulteri	Coulter's saltbush	-/-/1B.2/R	Chenopodiaceae	Mar-Oct	Alkaline or clay areas of: coastal bluff scrub, coastal dunes, coastal scrub, valley and foothill grassland	Moderate potential to occur
Atriplex lentiformis	big saltbush	-/-/-/W Limits	Chenopodiaceae	Jul-Oct	Alkaline or saline washes, dry lakes, scrub	Documented in project area (Stillwater Sciences 2011)[4]
Atriplex pacifica	south coast saltscale	-/-/1B.2/R	Chenopodiaceae	Mar-Oct	Coastal bluff scrub, coastal dunes, coastal scrub, playas	Moderate potential to occur
Atriplex serenana var. *davidsonii*	Davidson's saltscale	-/-/1B.2/R	Chenopodiaceae	Apr-Oct	Alkaline areas of: coastal bluff scrub, coastal scrub	Moderate potential to occur
Calystegia soldanella	seashore false bindweed	-/-/-/U	Convolvulaceae	Apr-Aug	Sandy seashores, coastal strand	Documented in project area (Stillwater Sciences 2011)[4]

Scientific name	Common name	Status[2] Federal/ State/ CRPR/ Local	Family	Blooming period	Habitat associations	Potential in the Project Area[3]
Camissonia cheiranthifolia (current name *Camissoniopsis cheiranthifolia*)	beach suncup	-/-/-/R	Onagraceae	Apr-Aug	Sandy slopes, flats, coastal dunes	Documented in project area (Stillwater Sciences 2011)[4]
Carex pansa	sand dune sedge	-/-/-/R	Cyperaceae	Apr-Jul	Sandy coastal areas	Documented in project area
Chaenactis glabriuscula var. *orcuttiana*	Orcutt's pincushion	-/-/1B.1/R	Asteraceae	Jan-Aug	Sandy coastal bluff scrub, coastal dunes	High potential to occur
Chloropyron maritimum subsp. *maritimum*	salt marsh bird's-beak	FE/CE/1B.2/R	Orobanchaceae	May-Oct	Coastal dunes, coastal salt marshes and swamps	Moderate potential to occur
Cyperus odoratus	fragrant flatsedge	-/-/-/R	Cyperaceae	Jul-Oct	Wet disturbed soils	Documented in project area
Dudleya caespitosa	sea lettuce	-/-/-/R	Crassulaceae	Apr-Aug	Rocky or sandy coastal areas	Documented in project area
Ericameria ericoides	California goldenbush	-/-/-/U	Asteraceae	Sep-Nov	Dunes, inland sandy soils	Documented in project area (Stillwater Sciences 2011)[4]
Heterotheca sessiliflora subsp. *sessiliflora*	beach goldenaster	-/-/1B.1/R	Asteraceae	Mar-Dec	Coastal chaparral, coastal dunes, coastal scrub	High potential to occur
Jaumea carnosa	marsh jaumea	-/-/-/U	Asteraceae	Apr-Dec	Coastal salt marshes, bases of sea cliffs	Documented in project area (Stillwater Sciences 2011)[4]
Juncus acutus subsp. *leopoldii*	southwestern spiny rush	-/-/4.2/U	Juncaceae	(Mar) May-Jun	Mesic coastal dunes, alkaline seeps, coastal salt marshes and swamps	Moderate potential to occur and documented in project area (Stillwater Sciences 2011)[4]
Juncus torreyi	Torrey's rush	-/-/-/R	Juncaceae	Jun-Sep	Meadows, moist woodland	Documented in project area (Stillwater Sciences 2011)[4]

Scientific name	Common name	Status[2] Federal/ State/ CRPR/ Local	Family	Blooming period	Habitat associations	Potential in the Project Area[3]
Lasthenia glabrata subsp. *coulteri*	Coulter's goldfields	-/-/1B.1/R	Asteraceae	Feb-Jun	Coastal salt marshes and swamps, playas, vernal pools	Moderate potential to occur
Malacothrix similis	Mexican malacothrix	-/-/2A/R	Asteraceae	Apr-May	Coastal dunes	High potential to occur
Phacelia ramosissima var. *austrolitoralis*	south coast branching phacelia	-/-/3.2/R	Hydrophyllaceae	Mar-Aug	Sandy or sometimes rocky areas of: chaparral, coastal dunes, coastal scrub, coastal salt marshes and swamps	High potential to occur
Pluchea odorata	saltmarsh fleabane	-/-/-/U	Asteraceae	Jun-Nov	Moist, often saline valley bottoms	Documented in project area (Stillwater Sciences 2011)[4]
Schoenoplectus acutus var. *occidentalis*	common tule	-/-/-/U	Cyperaceae	Summer	Marshes, shores, fens, shallow lakes, often emergent	Documented in project area (Stillwater Sciences 2011)[4]
Schoenoplectus californicus	southern bulrush	-/-/-/U	Cyperaceae	Spring-summer	Brackish to fresh marshes, shores	Documented in project area (Stillwater Sciences 2011)[4]
Suaeda esteroa	estuary seablite	-/-/1B.2/R	Chenopodiaceae	May-Oct (Jan)	Coastal salt marshes and swamps	High potential to occur
Suaeda taxifolia	woolly seablite	-/-/4.2/-	Chenopodiaceae	Jan-Dec	Coastal bluff scrub, coastal dunes, margins of coastal salt marshes and swamps	High potential to occur
Stephanomeria virgata subsp. *pleurocarpa*	wand wirelettuce	-/-/-/R	Asteraceae	Jun-Nov	Chaparral openings, grassland	Documented in project area (Stillwater Sciences 2011)[4]

Scientific name	Common name	Status[2] Federal/ State/ CRPR/ Local	Family	Blooming period	Habitat associations	Potential in the Project Area[3]
Typha latifolia	broadleaved cattail	-/-/-/U	Typhaceae	Jun-Jul	Unpolluted to nutrient-rich freshwater (brackish) marshes	Documented in project area (Stillwater Sciences 2011)[4]

[1] Unless otherwise noted, the list of potential special-status plant species was obtained from WRA, Inc. 2014b; there may be additional potential special-status species either documented (if protocol-level surveys are conducted) or with a moderate or high potential to occur (if a full USGS quadrangle search is conducted [i.e., including the project site quadrangle and all surrounding quadrangles]).

[2] Status:

	Federal		State
FE	Federally Endangered	CE	California Endangered
–	No federal status	–	No state status

California Rare Plant Rank (CRPR)

List 1B Plants rare, threatened, or endangered in California and elsewhere
List 2A Plants presumed extirpated in California, but common elsewhere
List 3 Plants about which more information is needed – a review list
List 4 Plants of limited distribution -a watch list

CRPR Threat Ranks:

0.1 Seriously threatened in California (high degree/immediacy of threat)
0.2 Moderately threatened in California (moderate degree/immediacy of threat)

Local (Channel Islands CNPS Chapter)

R Rare (5 or fewer occurrences in Ventura County)
U Uncommon (6-10 occurrences in Ventura County)
Limits Are at the limits of their distribution range (southernmost, northernmost, westernmost extent)

[3] Unless otherwise noted, this is as reported by WRA, Inc. (2014b). Note that those species documented at the site were documented incidentally, no protocol-level botanical surveys were conducted.

[4] This species was documented incidentally (Stillwater Sciences 2011); no protocol-level botanical surveys were conducted.

Table 3-28. Special-status plant species potential habitat associations in the project area[1,2].

Potential in the Project Area[3]	Scientific name	Common name	Developed/ Disturbed	Foredune	Riparian vegetation	Wetland	
						Freshwater	Salt marsh
Documented in project area	*Abronia maritima*[3,4]	red sand-verbena	X	X			
	Argentina egedii (current name *Potentilla anserina* subsp. *pacifica*)[4]	Pacific silverweed	X			X	X
	Astragalus pycnostachyus var. *lanosissimus*[4]	Ventura marsh milkvetch	X	X			X
	Atriplex lentiformis[4]	big saltbush	X				X
	Calystegia soldanella[4]	seashore false bindweed		X			
	Camissonia cheiranthifolia (current name *Camissoniopsis cheiranthifolia*)[4]	beach suncup		X			
	Carex pansa	sand dune sedge	X	X			
	Cyperus odoratus	fragrant flatsedge	X		X	X	X
	Dudleya caespitosa	sea lettuce	X	X			
	Ericameria ericoides[4]	California goldenbush	X	X			
	Jaumea carnosa[4]	marsh jaumea	X				X
	Juncus acutus subsp. *leopoldii*[4]	southwestern spiny rush		X		X	X
	Juncus torreyi[4]	Torrey's rush			X	X	
	Pluchea odorata[4]	saltmarsh fleabane				X	X
	Schoenoplectus acutus var. *occidentalis*[4]	common tule				X	
	Schoenoplectus californicus[4]	southern bulrush				X	X
	Stephanomeria virgata subsp. *pleurocarpa*[4]	wand wirelettuce	X				
	Typha latifolia[4]	broadleaved cattail				X	

Potential in the Project Area[3]	Scientific name	Common name	Developed/ Disturbed	Foredune	Riparian vegetation	Wetland	
						Freshwater	Salt marsh
High potential to occur	*Amsinckia spectabilis* var. *spectabilis*	seaside fiddleneck	X	X			
	Aphanisma blitoides	aphanisma	X	X			
	Chaenactis glabriuscula var. *orcuttiana*	Orcutt's pincushion	X	X			
High potential to occur	*Heterotheca sessiliflora* subsp. *sessiliflora*	beach goldenaster	X	X	X	X	X
	Malacothrix similis	Mexican malacothrix	X	X			
	Phacelia ramosissima var. *austrolitoralis*	south coast branching phacelia	X	X	X	X	X
	Suaeda esteroa	estuary seablite				X	X
	Suaeda taxifolia	woolly seablight	X	X		X	X
Moderate potential to occur	*Atriplex coulteri*	Coulter's saltbush	X	X	X	X	X
	Atriplex pacifica	south coast saltscale	X	X	X		
	Atriplex serenana var. *davidsonii*	Davidson's saltscale	X		X		
	Chloropyron maritimum subsp. *maritimum*	salt marsh bird's-beak		X		X	X
	Lasthenia glabrata subsp. *coulteri*	Coulter's goldfields				X	X

1 List of potential special-status plant species was obtained from WRA, Inc. 2014b; there may be additional potential special-status species either documented (if protocol-level surveys are conducted) or with a moderate or high potential to occur (if a full USGS quadrangle search is conducted [i.e., including the project site quadrangle and all surrounding quadrangles]).
2 Habitat associations (Baldwin et al. 2012 and CNPS 2017) are grouped according to the habitat types presented in Section 3.5.1.
3 Unless otherwise noted, this is as reported by WRA, Inc. (2014b). Note that those species documented at the site were documented incidentally, no protocol-level botanical surveys were conducted.
4 This species was documented incidentally (Stillwater Sciences 2011); no protocol-level botanical surveys were conducted.

3.6 Aquatic Habitat and Species

Aquatic habitat types present in the SCRE include open water, freshwater wetlands and a minimal amount of salt marsh (Figure 3-41). Depending upon river flows and mouth closure status, the largest aquatic habitat type is open water characterized by shallow sand-bedded flats with scattered emergent vegetation (Nautilus Environmental 2005, Stillwater Sciences 2011). Larger patches of emergent vegetation are found around the margins of the estuary, especially along the southern bank and near the outfall channel on the north side of the lagoon. Submerged aquatic vegetation was abundant around the perimeter of the estuary during periods of high water; otherwise submerged aquatic vegetation or other instream cover was found to be scarce within the SCRE during recent surveys. Substrate found in the SCRE is predominantly sand and silt although some patches of coarser substrate are found in bar complexes in the lower reaches of the Santa Clara River (Section 3.2.2).

The following sections review the available data and scientific literature related to the aquatic community of the SCRE and key factors influencing aquatic habitat conditions. Particular attention is paid to the focal fish species for this study, tidewater goby and southern California steelhead. Habitat for focal species serves as the primary basis for evaluation of habitat conditions for rare species (federally or state listed species) (see Section 5.5.7). Information on other aquatic species, including benthic macroinvertebrates and other native and introduced fishes is important for understanding the overall ecological condition of the SCRE and factors affecting native aquatic species, and is considered for aquatic habitat-related beneficial uses in the comparison of VWRF discharge scenarios in Section 5. The beneficial use assessment relies on this information for qualitative assessments of habitat conditions, and the relative degree to which alternative VWRF management scenarios affect the propagation of native species.

3.6.1 Benthic macroinvertebrates (BMI)

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). The variable nature of these drivers in the SCRE results in a seasonally and annually varying BMI community structure dominated by taxa tolerant of dynamic conditions. Previous BMI sampling in the SCRE showed inter- and intra-annual variability in community structure likely in response to changes in water levels, salinity, substrate, and disturbance associated with annual and seasonal changes in river flows and breaching of the SCRE mouth (ABC Laboratories 2004–2016, Entrix 2002b).

Monitoring since 1997 has indicated that the BMI community in the SCRE 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 2004–2016). In particular, species of midge larvae (Diptera: *Chironomidae*), oligochaete worms, copepods, ostracods, and gammarid amphipods tended to dominate during the 1997–2015 monitoring period, although the taxonomic composition of the dominant species varied seasonally and annually. One anomalous result, in which the BMI community drastically shifted from previous observations, occurred in May 2005 following severe storms that scoured much of the SCRE. By 2006, the BMI community returned to a composition consistent with pre-2005 observations, indicating that disturbance caused by large storm events temporarily changes BMI community structure, but that the community quickly recovers to stable-state structure.

Comparison of the 2002 SCRE BMI community to those in other similarly-sized estuaries within coastal southern California indicated that the SCRE BMI community composition varied considerably from other estuaries evaluated (Entrix 2002b). While the BMI communities in other estuaries overlapped with each other, the SCRE community showed very little overlap with any of the reviewed estuaries with the exception of San Dieguito Lagoon. In comparison to the other estuaries evaluated, the SCRE had greater numbers of oligochaetes, tubificid worms, and insect larvae (predominantly midges: *Chironomidae*), but fewer polychaetes and decapod and isopod crustaceans. Many of the other estuaries evaluated predominantly experience hypersaline or marine conditions (salinity >30 ppt), and many show more variable conditions than the SCRE. Additionally, the other estuaries evaluated have been dredged at the mouth of estuary to encourage more frequent tidal exchange. The SCRE is generally dominated by taxa that prefer freshwater but are tolerant of variable and brackish conditions; conspicuously absent from the SCRE are some BMI taxa that prefer higher salinities that were common in other estuaries. This is likely due to the prevalence of freshwater conditions in much of the estuary during prolonged closed mouth periods and low frequency of tidal exchange.

BMI sampling for the Phase 3 Study, which was conducted by ABC Laboratories quarterly during 2015, again showed a predominance of tolerant taxa, with oligochaetes, ostracods, and chironomid larvae comprising over 95% of total abundance. It should be noted that the abundance and diversity values resulting from 2015 sampling represent benthic infauna, as epibenthic taxa are unlikely to be sufficiently characterized by the methods used. As described in Section 2.3.1, in order to comply with requirements for the protection of tidewater goby by USFWS sweep nets and kick nets were not used during 2015 BMI sampling; only a grab sample was taken using a 0.02 m^2 petite ponar. Species richness and total abundance were both highest in the summer (June) and spring (March) and lowest in the winter (January) and fall (September) (Table 3-29). The 2015 BMI sampling showed a notable paucity of both taxa richness and total abundance of macroinvertebrates, particularly during the winter and fall (Table 3-29). The Shannon Diversity Index (H) is a metric used to evaluate the diversity of a community based on both the number of taxa represented, as well as how equally abundance is represented across taxa. In the 2015 BMI surveys, mean H was equal to 0.48 (range 0–1.01). By comparison, from 2003–2014, mean H was equal to 1.01 (range 0–1.97) (ABC Laboratories 2004–2016). Figure 3-47 shows seasonal abundance, taxa richness, and H values for one sampling station (R-004[13]) from 2003–2015.

In the context of prior years and other estuaries, the low abundance observed during 2015 BMI sampling was not unprecedented. In the SCRE, abundance was similarly low in the fall of 2013, summer of 2011, and fall of 2009 (Figure 3-47). Furthermore, in their summary of BMI studies in seven southern California estuaries, Entrix 2002 report low taxa richness in Malibu Lagoon, Batiquitos Lagoon, and San Dieguito Lagoon, suggesting that low diversity and abundance of benthic macroinvertebrates may not be an uncommon phenomenon in southern California estuaries. Variability of BMI abundance and diversity likely depends on the complex interactions of a number of biotic and physical variables, including substrate, resource availability, physical water quality, disturbance, and consumptive effects. A number of mechanisms have been proposed to explain low abundance or diversity in other estuaries. For example, low diversity observed in Malibu Lagoon was attributed to a sewage spill that occurred the year before BMI sampling. In Batiquitos lagoon, low diversity was suggested to result from the highly variable salinity regime, which ranges from 0–100 ppt. Low diversity in San Dieguito lagoon was also suggested to result from variable salinity conditions. While the salinity range in the SCRE is narrower than those of Batiquitos or San Dieguito Lagoons, the predominance of freshwater punctuated by periods of brackish and marine salinities likely selects for tolerant taxa. However, while this may explain low taxa richness, it does not explain low abundance of BMI.

One plausible explanation for the low abundance of BMI during winter and fall is that seasonal variations in predator abundance resulted in seasonally varying degrees top-down control of BMI abundance. This hypothesis is consistent with previous research that has shown that the density and taxa richness of soft-sediment invertebrate communities in lagoons increased following experimental predator exclusion (Peterson 1979). However, the highest number of fish captured in any of the 2015 seining surveys occurred during the summer, corresponding to the highest BMI abundance (see Section 3.6.2). Furthermore, fish abundance and BMI abundance were both lowest in the winter (though this apparent coupling does not hold in fall, when BMI abundance was low but fish abundance was high). This indicates that top-down control by predation is not a

[13] Station R-004 was called B3 prior to the issuance of the 2008 NPDES Permit

primary driver of BMI abundance in the SCRE. Rather, the co-occurrence of peak fish abundance and peak BMI abundance during the summer suggests that summer conditions were more favorable to secondary and tertiary production in the SCRE, relative to winter conditions.

Analysis of water quality data collected during BMI sampling events showed no univariate relationship between water quality parameters (DO, temperature, salinity, pH) and abundance or taxa richness across sampling dates. However, information reviewed in Sutula et al. (2012) indicates that low DO may impact some BMI species that serve as common food resources in the SCRE. While DO tolerance is highly variable across species, it is likely that reducing the frequency with which DO concentrations fall below 4 mg/L would likely increase the abundance and diversity of food resources for fishes in the SCRE. Conditions reducing the frequency and severity of algal blooms, including large reductions in nutrient loading, may increase BMI abundance and diversity in the SCRE.

During 2015 BMI sampling, substrate ranged from silt/clay to coarse sand. Substrate type did not explain variability in BMI abundance. The SCRE has a relatively low degree of habitat complexity, which may in part explain the low diversity of benthic invertebrates in the SCRE. Tests conducted in fulfillment of NPDES permit requirements indicated that no sediment contaminants accumulated in excess of designated thresholds and no toxicity was observed (ABC Laboratories 2016). As stated above, BMI abundance and diversity in the SCRE are likely controlled by the complex interactions of a number of variables. The BMI community in the SCRE may benefit from periodic scouring of fine sediments that usually occurs during high winter flows. The persistence of drought conditions and the lack of scouring of SCRE sediments may have contributed to the low abundance of BMI observed during 2015 surveys. While conditions that increase the frequency of tidal exchange are likely to propagate the dominance of tolerant taxa by creating a more variable salinity regime, the scouring of fine sediments may increase total BMI abundance by increasing the quality of benthic substrates.

Table 3-29. Number of taxa, abundance, and Shannon diversity results for 2015 BMI sampling.

Season	Number of taxa				Abundance				Shannon diversity index			
	E-1	E-2	R-1	Outfall	E-1	E-2	R-1	Outfall	E-1	E-2	R-1	Outfall
Winter	1	0	2	0	6	0	3	0	0	--	0.63	--
Spring	2	2	3	1	10	136	577	3	0.61	0.3	0.49	0
Summer	7	3	4	3	2,356	70	1,992	16	0.97	0.15	0.46	0.74
Fall	0	0	3	2	0	0	6	7	--	--	1.01	0.41

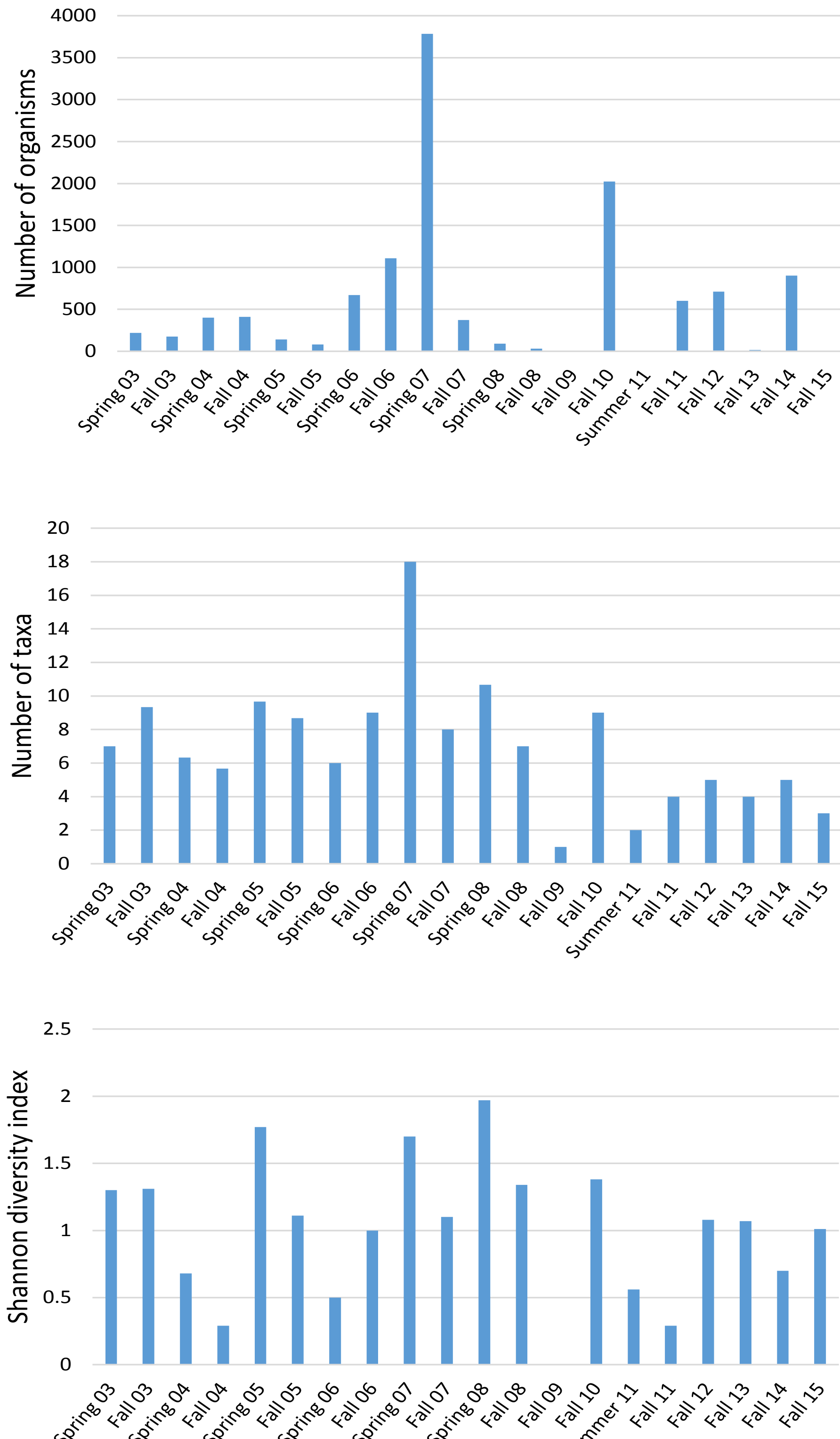


Figure 3-47. Seasonal abundance, taxa richness, and Shannon diversity index for station R-004[13,14].

[14] From data collected in fulfillment of NPDES Permit requirements (ABC Laboratories 2004–2016)

3.6.2 SCRE fish species

Data collected as part of the City's *Resident Species Study* (Entrix 2002b), recent monitoring, and other sources have documented twenty fish species in the SCRE, including eleven native and nine non-native species (Table 3-30 and Table 3-32). The fish assemblage present in the SCRE varies seasonally and annually, largely depending on environmental conditions that regulate habitat suitability and access to the SCRE from upstream and coastal ocean sources. This variety of fishes includes freshwater species, euryhaline species (species adapted to variable salinity conditions found in estuarine habitats), and stenohaline species (species that move into the estuary opportunistically as part of their life cycle) (Entrix 2004, Nautilus Environmental 2009, USFWS 1999) (see Table 3-32). 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*) (Table 3-30). This variety of species supports a number of designated beneficial uses of the SCRE, including COMM, EST, MAR, MIGR, RARE, and SPWN.

Table 3-30. Fish species found in the SCRE (1997–2016), relative abundance, status, and applicable beneficial uses.

Species[1]	Origin	Abundance[1]	Listed species status	Applicable beneficial uses[2]					
				COMM	EST	MAR	MIGR	RARE[3]	SPWN
arrow goby (*Clevelandia ios)*	native	uncommon	—		X	X			X
arroyo chub (*Gila orcutti*)	introduced[4]	abundant	CA Species of Special Concern		X			X	
bay pipefish (*Syngnathus leptorhynchus*)	native	uncommon	—		X	X			
California killifish (*Fundulus parvipinnis*)	native	uncommon	—	X	X	X (rarely)			X
common carp (*Cyprinus carpio*)	introduced	common	—	X					
fathead minnow (*Pimephales promelas*)	introduced	common	—						
goldfish (*Carassius auratus*)	introduced	uncommon	—						
green sunfish (*Lepomis cyanellus*)	introduced	common	—	X					
Mississippi silverside (*Menidia audens*)	introduced	common	—		X	X			X
Pacific lamprey (*Lampetra tridentata*)	native	migratory (rearing unknown)	—	X	X	X	X		
prickly sculpin (*Cottus asper*)	native	variable [5]	—		X	X	X		X
Santa Ana sucker (*Catostomus santaanae*)	introduced[4]	uncommon	Federally Threatened					X	
southern steelhead DPS (*Oncorhynchus mykiss*)	native	migratory (rearing unknown)	Federally Endangered		X	X	X	X	

Species[1]	Origin	Abundance[1]	Listed species status	Applicable beneficial uses[2]					
				COMM	EST	MAR	MIGR	RARE[3]	SPWN
staghorn sculpin (*Leptocottus armatus*)	native	common	—	X	X	X			
striped mullet (*Mugil cephalus*) [3]	native	common (marine visitor)	—	X	X	X			
tidewater goby (*Eucyclogobius newberryi)*	native	variable[6]	Federally Endangered		X	X (rarely)		X	X
topsmelt (*Atherinops affinis*)	native	common (marine visitor)	—	X	X	X			X
threespine stickleback (Gasterosteus aculeatus williamsoni)	native	variable [5]	—[7]		X	X (rarely)		X	X
western mosquitofish (*Gambusia affinis*)	introduced	common	—						
yellowfin goby (*Acanthogobius flavimanus*) [8]	introduced	uncommon	—	X	X	X			X

1 Sources: Entrix, Inc. 2004, 2014; Stillwater 2010; Stillwater seine surveys 2015-2016
2 Sources for designation of species to beneficial uses: wildlife.ca.gov/Fishing/Ocean/Fish-ID/Sportfish; Sutula et al. (2012); Allen et al. (2006); With the exception of COMM, AHP analysis of beneficial uses listed here emphasized native species
3 Only state or federally listed species were considered to support the RARE beneficial use, which deviates from designations in Sutula et al. (2012)
4 Otherwise native, but considered introduced in the Santa Clara River drainage (Swift et al. 1993)
5 Prickly sculpin and threespine stickleback were documented in high abundance during previous surveys (Entrix 2008, Cardno/Entrix 2010); however, only one prickly sculpin was captured and no three-spine stickleback were captured during sampling in 2015 and 2016.
6 Tidewater goby were one of the most abundant fishes during some surveys (e.g., 2008, 2010) but were very low in abundance 2013-2016. Several hundred tidewater goby were observed during a flow survey in July 2017.
7 The threespine stickleback in the SCRE and lower reaches of the SCR are likely to be partially or fully armored. There is a very low probability that the unarmored form, which are federally listed as endangered, will occupy the SCRE.
8 Last observed in 1997.

Eleven fish species were documented during 2015 and 2016 seine surveys (Table 3-31). Fish were most abundant in the summer and fall with low number observed during the winter and spring. The most abundant fish species captured in 2015–2016 were Mississippi silverside (*Menidia audens*), western mosquitofish (*Gambusia affinis*), and green sunfish (*Lepomis cyanellus*) (Table 3-31). From 1997–2016, the most abundant fishes captured during seine surveys included 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*). It should be noted that while beach seining is a widely accepted method for estimating fish abundance and taxa richness, there are some associated sampling biases. Goldstein (1978) noted that seine surveys may bias toward smaller species, indicating that larger fishes such as common carp, green sunfish, or steelhead may be more abundant in the SCRE than reported herein. Steele et al. (2006) found that seining underestimated abundance of demersal fishes, but not pelagic fishes, which suggests that demersal fishes such as tidewater goby, suckers, and sculpin may also be more abundant in the SCRE than reported herein. Additionally, seining is less effective in areas that are deep or have complex habitat structure including aquatic or flooded riparian vegetation. Despite these potential biases, consistent methodology across sampling dates enables comparison of relative abundance and assemblage composition across seasons and years.

Five native species use the estuary for migratory or opportunistic purposes, including two anadromous fishes (steelhead and Pacific lamprey), one catadromous species (prickly sculpin) and two marine species, (topsmelt and striped mullet). Several stranded *O. mykiss* were identified following an unauthorized third-party breach event in September 2010 (Cardno/Entrix 2010).

Table 3-31. Fish counts by species and sampling dates for seine surveys conducted 2015–2016.

Species	Fish (and amphibian) count by sampling date					
	Jan-15	**Mar-15**	**Jun-15**	**Sep-15**	**Sep-16**	**Total**
African clawed frog	0	0	79	0	0	79
arroyo chub	1	0	0	0	0	1
bay pipefish	0	0	0	0	1	1
common carp	0	0	18	0	1	19
crayfish	1	1	3	1	0	6
fathead minnow	0	0	2	5	0	7
goldfish	0	0	0	0	2	2
green sunfish	0	0	69	45	0	114
Mississippi silverside	51	1	3,500	2,955	231	6,738
prickly sculpin	1	0	0	0	0	1
striped mullet	29	0	0	0	0	29
tidewater goby	0	0	0	10	12	22
tree frog	0	0	2	0	0	2
western mosquitofish	63	91	85	462	1,022	1,723
Total	**146**	**93**	**3,758**	**3,478**	**1,269**	**8,744**

3.6.3 Focal fish species

The sections below describe the life history traits and habitat requirements for the two focal fish species selected for the Phase 3 Study: southern California steelhead and tidewater goby. In order to identify the physical, chemical, and biological characteristics of the SCRE that are important for species behavior and long-term success, we reviewed the scientific literature and available data on these species, with emphasis on site-specific information where possible. As both of these species are federally listed as endangered under the Endangered Species Act, we also reviewed critical habitat designations to evaluate the extent to which Primary Constituent Elements[15] (PCEs) of habitat are realized in the SCRE (USFWS 2013), and to determine whether these factors might be affected by VWRF discharge.

3.6.3.1 Steelhead

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 bordering Santa Barbara and San Luis Obispo counties to the U.S-Mexico border (NMFS 2006). This DPS is listed as endangered under the federal Endangered Species Act (NMFS 2006). In addition, the National Marine Fisheries Service (NMFS) designated the SCRE as critical habitat for the Southern

[15] Now referred to as physical and biological features (PBFs) by USFWS and NMFS; however, we have retained the PCE terminology since this was the convention at the time of Critical Habitat designation

California steelhead under the ESA (70 Fed. Reg. 52,488 (Sept. 2, 2005) [designating, among other areas, the SCRE as critical habitat for steelhead]). 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, the major tributaries west of the Piru Creek confluence have been suggested to be potentially suitable for steelhead spawning (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 prep).

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).

To the north, within Scott Creek (Santa Cruz County, CA), competition for space within upstream freshwater habitats has given rise 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 than 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 size 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 communities 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; Section 3.6.1).

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 smolts, 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).

Habitat requirements in the Santa Clara River Estuary

This section reviews the available scientific literature to develop habitat suitability criteria for steelhead in the SCRE. As part of the critical habitat designation under the Endangered Species Act, PCEs have been established for southern California steelhead (Table 3-32). The extent to which these PCEs are realized in the SCRE, as well as their relationship with VWRF discharge were considered when determining habitat suitability criteria for the comparison of VWRF discharge scenarios and the beneficial use assessment presented in Section 5.

There are too few observations of steelhead in the SCRE to relate to suitability and there are no generally accepted habitat criteria for steelhead in estuaries. Little data on steelhead upmigration and spawning timing exists for the Santa Clara River, although both spawning timing and distribution within the basin is related to the timing, frequency, and duration of mouth berm opening and winter flow conditions. Adult steelhead occurrence in the SCRE is limited to periods when the estuary is open, and they are expected to use the SCRE as a migration corridor to the upper watershed as soon as water depth in the river allows. Accordingly, for the purposes of quantifying adult habitat herein, habitat requirements for adult steelhead will focus on migration opportunities, rather than rearing conditions. Similarly, because spawning and early rearing occurs in habitat well upstream of the SCRE, PCE 1 in Table 3-32 is not considered in this assessment. Timing of smolt outmigration depends on adequate flow conditions to connect the estuary to the ocean. Smolts may rear in the lagoon prior to outmigration, as lagoons have been identified as valuable rearing habitat for juvenile steelhead in other systems based on the abundance of high quality food (e.g. Hayes et al. 2008, Boughton et al. 2017). Hayes et al. (2011) identified a "down-up-down" life history pathway in Scott Creek in which juvenile steelhead rear in lagoon habitats for some period of time, retreat to upstream habitats in the fall to continue rearing, and then enter the lagoon again as smolts prior to outmigration. However, this life history pathway is considered infeasible in the SCRE given the lack of connectivity between the SCRE and upstream habitats in fall due to low flows (see Section 3.3.3). Previous research in the SCRE in 2008 has suggested that juvenile steelhead tend to rear in upstream locations and spend only limited time in the lagoon (< 3 days) prior to outmigration (Kelley 2008). However, several stranded *O. mykiss* in the size range of 10–12 inches were identified following an unauthorized third-party breaching event in September 2010 (Cardno/Entrix 2010), indicating that sub-adult *O. mykiss* may use the lagoon for more extended periods of rearing under some conditions. Accordingly, analysis of habitat for sub-adult *O. mykiss* will focus on the suitability of rearing habitat.

Table 3-32. Primary Constituent Elements of Critical Habitat for southern California steelhead.

PCE Number	PCE Description	Source
1	Water quantity and quality supporting spawning, incubation and larval development	70FR52488 – 52627, (c)(1)
2	Freshwater rearing sites	70FR52488 – 52627, (c)(2)
2(i)	Water quality and floodplain connectivity that support juvenile growth and mobility	70FR52488 – 52627, (c)(2)(i)
2(ii)	Water quality and forage supporting juvenile development	70FR52488 – 52627, (c)(2)(ii)
2(iii)	Natural cover	70FR52488 – 52627, (c)(2)(iii)
3	Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover	70FR52488 – 52627, (c)(3)
4	Estuarine areas free of obstruction and excessive predation	70FR52488 – 52627, (c)(4)
4(i)	Water quality, water quantity, and salinity conditions supporting physiological salinity transitions	70FR52488 – 52627, (c)(4)(i)
4(ii)	Natural cover	70FR52488 – 52627, (c)(4)(ii)
4(iii)	Juvenile and adult forage	70FR52488 – 52627, (c)(4)(iii)

Rearing Sub-Adults

Microhabitat selection for juvenile steelhead is largely dictated by predator avoidance (Moyle 2002). 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). Observations of steelhead rearing in lagoons have indicated a similar preference for deeper habitat (e.g., Smith 1990, Anderson et al. 2007). Because adult steelhead as well as resident *O. mykiss* spawn at upstream locations within the tributary watershed of the Santa Clara River, fry are assumed to rear in upstream locations, with only smolt-sized individuals (i.e., sub-adults) using the lagoon for rearing prior to emigration. Although non-native piscivorous fishes have been observed in high abundance in the SCRE, they are not considered a predation threat to rearing steelhead because sub-adult steelhead are expected to be of comparable or greater size relative to these predators (e.g., sculpins, green sunfish). Piscivorous birds are the predominant predatory risk to steelhead in the SCRE. Moyle (2002) notes that piscivorous birds present a predation risk to steelhead in shallow water or near the water surface. PCE 4(iii) calls for natural cover for rearing steelhead, while PCE 4 calls for estuarine areas free from obstruction and excessive predation. Obstruction is primarily relevant to upmigrating adults and outmigrating smolts and is not considered here. However, the risk of predation and the availability of cover are closely related. Because features that typically provide cover from predation in riverine habitats (e.g., large boulders, large woody debris, undercut banks) are not prevalent in the SCRE (Kelley 2008), depth is expected to be the primary source of cover from avian predation for rearing sub-adult steelhead. Gawlik (2002) found experimentally that avian predation on fish decreased with increasing water depth. There are no good data available on minimum water depths used by steelhead in estuaries, but a minimum depth of 0.5 m is generally considered suitable for avoiding avian predation—no maximum water depth is believed to apply in this habitat (Daniels et al. 2010). Based on the above, for the purposes of quantifying rearing habitat for juvenile steelhead as a function of estuary stage, depths greater than 0.5 m were considered suitable habitat for sub-adult steelhead in the SCRE, though some predation may still occur at these depths.

PCE 2(i) calls for water quality and floodplain connectivity that support juvenile growth and mobility. While floodplain habitat is largely absent in the SCRE, water quality conditions supporting growth are an important consideration. Similarly, PCE 2(ii) calls for water quality and forage supporting juvenile development. Water quality conditions for PCEs 2(i) and 2(ii) will be assessed on the basis of water temperature, dissolved oxygen, and salinity.

Several studies regarding the relationship between water temperature and growth in steelhead have reported varying optimal and unsuitable temperatures depending on the location or population of fish studied. In their laboratory study of two Northern California strains of rainbow trout (Eagle Lake and Mt. Shasta), Myrick and Cech (2000) found that fish acclimated to 25°C exhibited a critical thermal maximum (CTM) of 31.5–32.0°C, depending on strain. Fish acclimated to 19°C exhibited CTM of 29.3–29.6°C. This is consistent with a literature review conducted as part of a larger study on southern California steelhead, which found that steelhead can persist in areas where short-term temperatures do not exceed 29–30°C (Boughton et al. 2015). Based on this review, Boughton et al. (2015) considered habitat for southern California steelhead to be thermally suitable if daily maximum temperatures were below 29°C and daily average temperatures were below 25°C, with the acknowledgement that physiological stress increases with increasing degree hours above 21°C. Sublethal bioenergetic effects of increased water temperatures may limit growth of steelhead (Stillwater and URS Corp 2007), and consequently survival in the ocean environment. In their study of two northern California strains of *O. mykiss*, Myrick and Cech (2000) noted that growth of both strains slowed from 19–25°C, which approaches their upper incipient lethal temperature of 26°C. However, a study of *O. mykiss* in the Scott Creek estuary along the central California coast found the highest growth rates were observed at temperatures between 15–24°C (Hayes et al. 2008). Recent evidence has indicated that rainbow trout show signs of local adjustment to high temperatures: Verhille et al. (2016) found that rainbow trout in the lower Tuolumne River maintained 95% of their peak aerobic scope from 17.8–24.6°C. Spina (2007) found that juvenile O. mykiss rearing in southern California streams remain active at temperatures well exceeding temperature preference ranges reported in the literature. Based on the above information, daily average temperatures greater than 26ºC may be unsuitable for rearing sub-adult steelhead.

In lagoon habitats similar to the SCRE, dissolved oxygen concentrations greater than 5 mg/L are considered suitable for rearing steelhead (ISU 2008). Boughton et al. (2017) considered concentrations < 6 mg/L to constitute moderate impairment to steelhead in the Russian River estuary. Dissolved oxygen concentrations near saturation are generally required for growth, but steelhead can survive at dissolved oxygen concentrations as low as 1.5–2.0 mg/L at low temperatures (Moyle 2002). However, dissolved oxygen concentrations of less than 6 mg/L are considered unsuitable for the purposes of this study.

PCE 4(i) calls for water quality, water quantity, and salinity conditions supporting physiological salinity transitions. Although rearing steelhead can withstand high salinity for brief periods, growth may decrease with increasing salinity (Morgan et al. 2011). Smolting steelhead require low to moderate salinities while they undergo physiological transitions in preparation for outmigration to ocean salinities. Salinities less than 10 ppt (i.e., isotonic concentrations) are generally considered suitable for sub-adult steelhead rearing in estuaries (e.g., Boughton et al. 2017). As a predominantly freshwater lagoon during closed mouth summer conditions, the SCRE provides freshwater rearing habitat for juvenile steelhead (PCE 2). Accordingly, for the purposes of evaluating habitat availability in the SCRE, salinities greater than 10 ppt will be considered unsuitable for rearing steelhead, with the understanding that short periods of high salinities are unlikely to impact rearing sub-adult steelhead.

Several studies have measured copper olfactory toxicity during freshwater rearing of juvenile salmonids (Baldwin et al. 2003; Sandahl et al. 2007; McIntyre et al. 2008) with effects at concentrations as low as 2 ug/L in laboratory settings (Hecht et al 2007) suggesting that urban runoff and other sources of dissolved copper to rivers may impair predator avoidance during rearing as well as later homing ability of returning adults. Significant gaps in understanding of the effects of dissolved copper exist in estuaries, however, and two recent studies suggest that the threshold for copper olfactory toxicity increases rapidly with salinity to the point that 50 ug/L produced no olfactory toxicity in simulated brackish estuarine (10 ppt) or seawater (32 ppt) due to complexation reactions (Baldwin 2015). Because the presence of organic matter and dissolved organic carbon (DOC) is also known to reduce the toxicity of copper to the olfactory system (McIntyre et al. 2008) and background levels of both salinity in the SCR as well as DOC within the SCRE are relatively high, it can be expected that little if any olfactory toxicity would be observed at dissolved copper levels below 5 ug/L.

Juvenile forage (PCE 4iii) is considered an important factor for steelhead rearing in the SCRE. However, given data limitations regarding invertebrate abundance in the SCRE (see Section 3.6.1) and the difficulty in modeling invertebrate community response to changing conditions in the SCRE, forage was not explicitly considered as an assessment criterion for the comparison of VWRF discharge scenarios. Instead, metrics of habitat area and water quality were considered to broadly represent habitat conditions for steelhead in the SCRE.

Based on the information reviewed above, a curve of habitat area as a function of stage for sub-adult *O. mykiss* rearing in the SCRE (Figure 3-48) was developed using the following suitability criteria:

- Depth: > 1.6 ft (0.5 m)

Based on habitat requirements reviewed above, the resultant habitat area will be evaluated qualitatively or quantitatively in the comparison of VWRF discharge scenarios using following water quality suitability criteria:

- Temperature: < 25ºC
- Salinity: < 10 ppt
- Dissolved oxygen: > 6 mg/L
- Metals: < 5 ug/L Dissolved Copper

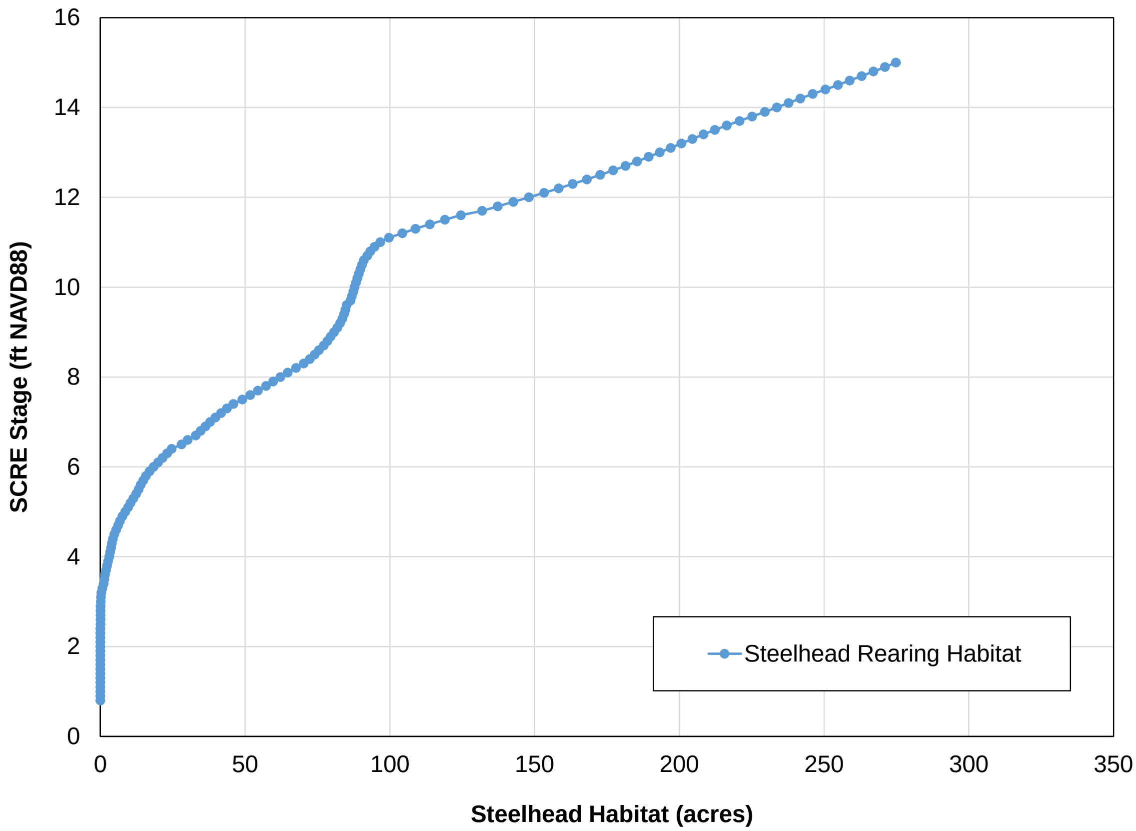


Figure 3-48. Extent of suitable steelhead habitat as a function of SCRE stage under current conditions.

Upmigrant Adults

Upmigration opportunities for adult steelhead depend on open mouth conditions and sufficient water depth in the SCRE for upstream passage (PCEs 3 and 4). For this analysis, we have assumed that habitat criteria for upmigrant adults is applicable for smolts during outmigration from the SCRE. The minimum depth for upstream passage is reported by the California Department of Water Resources as 7 inches (0.18 m) (DWR and USBR 2000), which is generally met in the SCRE during or shortly after open mouth conditions. Accordingly, passage opportunities for adult steelhead will be assessed based on the frequency of open mouth conditions during the steelhead migration window (November–May). The dynamic nature of the estuary mouth makes it difficult to estimate depth when the mouth is open because flows scour away at the loose sand deposited along the berm; however, during each breach there is typically a period where high flows draining the estuary scour a channel that provides depths greater than 7 inches (0.18 m) (personal observations by Stillwater Staff during Phase 1 and Phase 3 monitoring). For this analysis, we have assumed (a) that a migratory corridor with depths of at least 7 inches is available for steelhead migration during open mouth conditions and (b) that adult steelhead hold in the lagoon for only negligible periods of time prior to upmigration. In addition to open mouth and depth criteria discussed above for upmigrant adult steelhead, certain water quality parameters can present barriers to migration, including high temperatures, low levels of dissolved oxygen, high levels of turbidity, as well as certain trace metals such as dissolved copper.

While it is widely recognized in the literature that high water temperatures can prevent upmigration of adult anadromous salmonids, no data are available to quantitatively determine suitable temperatures for upmigration of adult southern California steelhead. Previous work on salmonids from regions other than southern California has suggested that migrating juveniles and adults have similar temperature preferences (Bell 1991). Based on this information, we have assumed that the temperature ranges deemed suitable for rearing sub-adult steelhead (see above) are also suitable for adult upmigration and smolt outmigration.

Low dissolved oxygen and high turbidity have also been suggested as mechanisms interfering with adult upmigration. Bjornn and Reiser (1991) suggested that dissolved oxygen concentrations greater than 5 mg/L are necessary for adult upmigration. Lastly, although Cordone and Kelley (1961) suggested that very high levels of turbidity may delay upmigration of anadromous salmonids, because turbidity levels are largely determined by rainfall runoff events with little relationship to VWRF operations, turbidity will not be considered in analysis of migration opportunities for steelhead in the SCRE. Based on the information presented for rearing sub-adult steelhead above regarding the copper interfering with olfactory function, we have assumed that copper concentrations in excess of 5 ug/L can also interfere with homing functions of upmigrant adults and outmigrant smolts.

Fish migration opportunities will be assessed in Section 5.3.4. Passage opportunities for upmigrant adult steelhead and outmigrant smolts will be quantified based on the following criteria:

- Frequency of open mouth conditions during the migration window (November–May)

The resultant estimates of passage opportunities will be evaluated qualitatively or quantitatively in the comparison of VWRF discharge scenarios using the following water quality criteria:

- Temperature: < 25ºC
- Dissolved oxygen: > 6 mg/L
- Metals: < 5 ug/L Dissolved Copper

3.6.3.2 Tidewater Goby

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). Populations in very small estuaries (<5 acres) are at risk of extirpation, while populations in intermediate-sized estuaries (5–125 acres) tend to be the most stable (USFWS 2005). With some exceptions (e.g. Lake Tolowa/Lake Earl, 4800 acres), stable tidewater goby populations are not common in very large estuaries (USWFS 2005). The population in the SCRE is federally listed as endangered under the Endangered Species Act (62 FR 43937). California has also designated the tidewater goby as a species of special concern.[16] The U.S. Fish and Wildlife Service (USFWS) has designated the SCRE as critical habitat for the tidewater goby under the ESA. (73 Fed. Reg. 5920 (Jan 31, 2008), and also published a Recovery Plan for the endangered fish,[17] which is a plan required by Section 4(f) of the ESA that delineates reasonable actions that are believed to be required to recover and/or provide future protections for a listed species.

Threats to tidewater goby include lack of freshwater due to diversions, pollution, siltation, and competition and predation by invasive 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 tidewater goby that are relevant to the Santa Clara River watershed include agricultural discharges, sewage treatment effluent, water diversions, and exotic species (USFWS 2005).

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 goby 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). They mainly occur in the lower portion of the watershed; however, in the Santa Ynez River, Santa Barbara County, tidewater goby have been found as far as 8 km (5 mi) upstream of tidal lagoon areas, sometimes in sections of stream impounded by beavers (*Castor canadensis*) (SCR Project Steering Committee 1996)

Tidewater goby feed mainly on small animals, usually mysid shrimp (*Mysidopsis bahia*), ostracods, gammarid amphipods, and aquatic insects, particularly chironomid midge (Diptera: *Chironomidae*) larvae (Swift et al. 1989; Swenson 1995; Moyle 2002; Swenson and McCray 1996). Juvenile tidewater goby are generally day feeders, although adults mainly feed at night (Swenson and McCray 1996). Tidewater goby 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). Swenson and McCray (1996) suggested that tidewater goby in lagoon habitats showed selectivity for ostracods, amphipods, and chironomid larvae—which are

[16] Cal. Dept. of Fish and Game, *Species of Special Concern in California* 79, 235 (2d ed. 1995). Species of special concern are those with low, scattered, or highly localized populations and require active management to prevent them from becoming threatened or endangered. (*Id*. at p. 3.)
[17] Fish and Wildlife Service, Pacific Region, *Recovery Plan for the Tidewater Goby (Eucyclogobius Newberry)* (Recovery Plan) (issued Jan. 7, 2005) (*available at* http://www.fws.gov/pacific/ecoservices/endangered/recovery/documents/TidewaterGobyFinalRecoveryPlan.pdf).

typically among the most abundant benthic macroinvertebrates in the SCRE (Section 3.6.1)—but are capable of opportunistic feeding as well.

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 goby 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 inches) 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 goby tends to be significantly larger in marshes (43–45 mm [1.7–1.8 in] standard length) when compared to tidewater goby from lagoons or creek habitats (USFWS 2005, Swenson 1999). This may be because the more stable physical conditions of the marsh foster improved growth or a more consistent or abundant supply of prey (USFWS 2005, Swift et al. 1997).

Habitat requirements in the Santa Clara River Estuary

This section reviews the scientific literature related to tidewater goby habitat use and requirements, USFWS designation of critical habitat, and observations of tidewater goby in the SCRE to develop habitat suitability criteria for both rearing and spawning tidewater goby. As part of the critical habitat designation under the Endangered Species Act, PCEs have been established for tidewater goby (USFWS 2013) (Table 3-33). The extent to which these PCEs are realized in the SCRE, as well as their relationship with VWRF discharge were considered when determining habitat suitability criteria for the comparison of VWRF discharge scenarios and the beneficial use assessment presented in Section 5.

Table 3-33. Primary Constituent Elements of Critical Habitat for tidewater goby.

PCE Number	PCE Description	Source
1	Persistent, shallow (in the range of about 0.1 to 2 m), still-to-slow-moving, aquatic habitat most commonly ranging in salinity from 0.5 ppt to about 10 to 12 ppt	Fed Register 73, 21 (2)
2	Substrates (e.g., sand, silt, mud) suitable for the construction of burrows for reproduction	Fed Register 73, 21 (2)
3	Submerged and emergent aquatic vegetation that provides protection from predators	Fed Register 73, 21 (3)
4	Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity	Fed Register 73, 21 (4)

The suitable ranges for identified for some habitat parameters are relatively broad, representing the range of conditions known to occur in habitats occupied by tidewater goby. In some cases, the scientific literature has identified a narrower “preferred” range where they appear to be most abundant and persistent during specific life stages. Where applicable, this assessment uses this preferred range for habitat suitability criteria.

Suitability criteria were determined for two tidewater goby lifestages: rearing and spawning. While the pelagic larval lifestage may have somewhat different habitat requirements from benthic rearing individuals, there are not sufficient data to characterize these requirements, as the larval stage is not often observed (Swenson 1999). Larval tidewater goby are pelagic and are often found in deeper water than adults (Swenson 1999). Larva emerge at 4–5 mm SL and become benthic at 16–18 mm SL (Swift et al. 1989), with pelagic larval phase lasting 18-31 days (Spies et al. 2014). Maturity is reached at 24–27 mm SL (Moyle 2002). Benthic individuals of different size classes are generally captured in the same location (Swenson 1999), suggesting that post-larval juveniles and adults occupy similar habitats. Swift et al. (1989) described aggregations of tidewater gobies as lacking any observable size segregation. Some capture data has indicated that juveniles may have somewhat narrower preference ranges than adults (Tetra Tech 2000), but on balance there is insufficient evidence to suggest that habitat suitable for adults is not suitable for juveniles. Based on the paucity of data regarding larval and juvenile tidewater goby habitat requirements, we have not attempted to estimate larval habitat suitability as a function of estuary stage and have assumed that post-larval juveniles have the same habitat requirements as adults.

Rearing Juveniles and Adults

Tidewater goby appear to prefer shallow water (< 1 m) near emergent vegetation, possibly to avoid predation by wading birds and piscivorous fish (Moyle 2002), although they are frequently observed in areas lacking vegetative cover. In 2015 and 2016 fish surveys of the SCRE, tidewater goby were most frequently captured in unvegetated sand-bedded areas near the beach berm, though they have been observed in high frequency near vegetated areas in prior years. Access to vegetative cover is included as a PCE of tidewater goby habitat (Table 3-33). Given that submerged vegetation and other structural elements of cover are sparse in the SCRE, depth was selected as the primary physical habitat assessment criterion for predator avoidance functions. Reported minimum depths may be associated with foraging depth thresholds for wading birds such as herons; in general, avian predation efficiency decreases with increasing depths greater than 0.2 m (Gawlik 2002). The literature indicates that juvenile and adult tidewater goby prefer depths ranging from 0.2 to 1.0 m, based primarily on sampling with beach seines. However, tidewater goby have been captured in Big Lagoon at depths of up to 4.6 m using a small-frame trawl towed by a small boat (C. Chamberlain, pers. comm., 2006). The USFWS (2013) includes water depths from 0.1 to 2.0 m as a PCE for tidewater goby (Table 3-33). During seine surveys of the SCRE in 2015 and 2016, tidewater goby were observed at sites with maximum depths ranging from 0.5 to 1 m. Based on the above and for the purposes of quantifying tidewater goby rearing habitat as a function of estuary stage, we define suitable tidewater goby rearing depths as 0.1 to 2.0 m.

Tidewater goby are poor swimmers and are intolerant of swiftly moving water (Swenson 1999, Moyle 2002). Moyle (2002) noted that populations of tidewater goby often plummet following breaching of lagoon berms and suggested that mouth berm breaching controls tidewater goby populations. Tidewater goby require stable lagoon or off-channel habitats, and can persist in habitats that wash out under high flows as long as a velocity refuge is present (Moyle 2002, Lafferty et al. 1999, Chamberlain 2006). Still-to-slowly moving areas of water are a PCE of tidewater goby critical habitat (Table 3-33). Analysis of tidewater goby capture data does not indicate that open mouth conditions reduce abundance (Figure 3-49). This apparent decoupling of tidewater goby abundance (as represented by capture data) from mouth breach dynamics indicates that there is sufficient flow refuge during open mouth conditions in the SCRE. Some sheltered areas of the SCRE, including the outfall channel, may provide low-flow refuge during flood events. The SCRE is a naturally flashy system, with long periods of closed-mouth, stable conditions in summer punctuated by periods of high flow and open-mouth conditions, typically during winter (Section 3.3.5). The USFWS (2013) includes stable, closed-mouth conditions late spring through fall, and the relative stability in salinity and water levels as a PCE of tidewater goby habitat (PCE 4; Table 3-33).

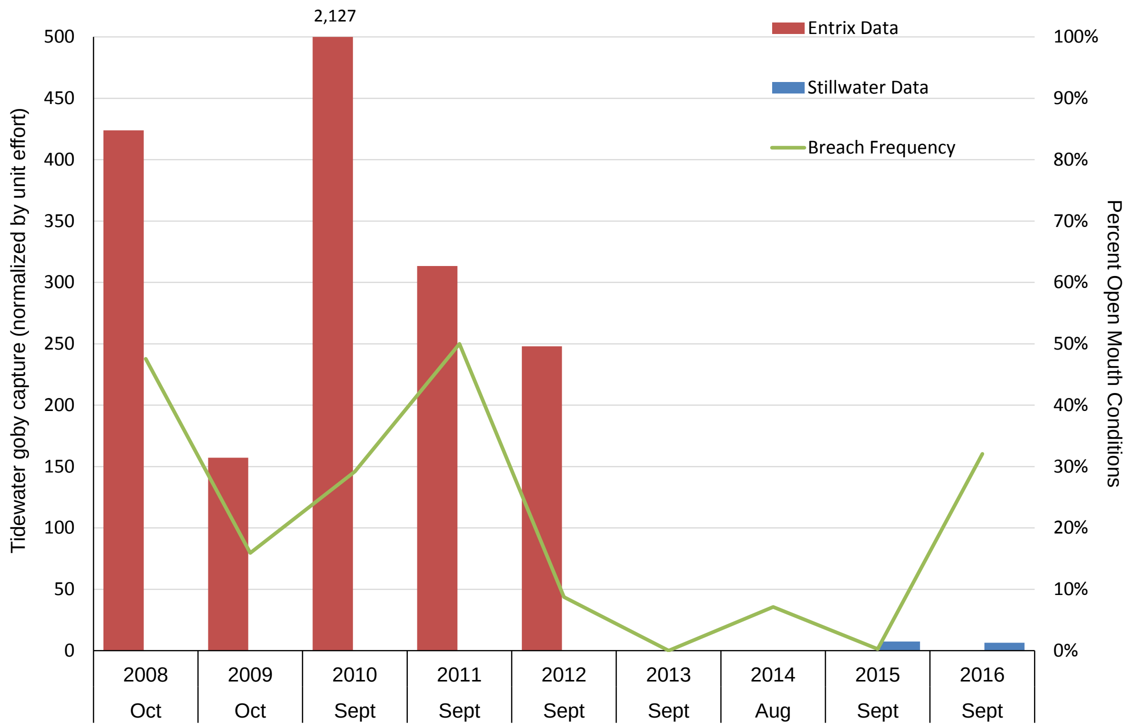


Figure 3-49. Number of tidewater goby captured (normalized by unit effort) (2008–2016) and percent open mouth conditions.

Tidewater goby appear to be tolerant of a wide range of temperature, dissolved oxygen, and salinity conditions. They have been observed in DO levels ranging from <1–19 mg/L, though a preferred range of 4–19 mg/L has been reported (Irwin and Soltz 1984 as cited in Chamberlain 2006, Chamberlain 2006, Tetra Tech Inc. 2000); temperatures ranging from 8–25°C (Swift et al. 1989, Swenson 1995, Swenson 1999); and salinities that range from 0–41 ppt, and (Swift et al. 1989, Swenson 1997). The USFWS (2013) did not designate temperature as a primary constituent element for tidewater goby critical habitat; however, salinities ≤ 12 ppt are identified as a primary constituent element of critical habitat.

Based on the information reviewed above, a curve of habitat area as a function of stage for tidewater goby rearing in the SCRE (Figure 3-50) was developed using the following suitability criteria:

- Depth: 0.3–6.5 ft (0.1–2 m)

Based on habitat requirements reviewed above, the resultant habitat area will be evaluated qualitatively using following water quality and habitat suitability criteria:

- Stable closed-mouth conditions during late spring through fall (June 1 through October 31)
- Temperature: Daily average temperatures < 25ºC, as measured near the substrate surface
- Salinity: ≤ 12 ppt, with the understanding that this represents the preferred range, and that tidewater goby are tolerant of much higher salinities
- Dissolved oxygen: 4–19 mg/L on a continuous basis, although they are tolerant of periodic low DO

Adult Spawning

No data were found indicating that preferred spawning depths varied from preferred rearing depths (see above). Accordingly, habitat for tidewater goby spawning is considered suitable at depths of 0.1 to 2.0 m for the purposes of quantifying spawning habitat as a function of estuary stage.

The USFWS (2013) includes sand, silt, and mud substrates suitable for constructing burrows as a PCE of tidewater goby critical habitat (PCE 2; Table 3-33). However, during 2015 and 2016 surveys in the SCRE, tidewater goby were only observed in areas with sandy substrate, except one individual that was captured at a site with muddy substrate. A number of studies have shown that tidewater goby tend to spawn in sand or coarse sand (Swift et al. 1989; Capelli 1997), and that they prefer sand to mud when both are available (Swenson, unpublished data, as cited in Swenson 1997). For the purposes of quantifying spawning habitat as a function of estuary stage, suitable substrate will be defined as sand or coarse sand, with the understanding that this represents preferred habitat and is thus a conservative measure of total suitable habitat.

Tidewater goby have been documented breeding in a wide range of temperatures (9–25°C in the field, 17–22°C in captivity) and salinities (2–27 ppt), though they seem to prefer salinities of 8–15 ppt (USFWS 2005, Swenson 1999). This temperature range is often met in the SCRE, though some extended periods of warmer temperatures were observed during recent monitoring (Section 3.4.1.2). Salinity in the SCRE is highly variable: extended periods of closed mouth conditions lead to salinities characteristic of freshwater environments, while open mouth conditions or surf over-topping the mouth berm increase salinity, resulting in brackish conditions. Small changes in salinity probably do not prevent successful hatching of the eggs (Entrix 2004)

A curve of habitat area as a function of stage for tidewater goby spawning in the SCRE (Figure 3-50) was developed using the following suitability criteria:

- Depth: 0.3–6.5 ft (0.1–2.0 m)
- Substrate: sand or coarse sand

Based on habitat requirements reviewed above, the resultant habitat area will be evaluated qualitatively or quantitatively using the following water quality and habitat suitability criteria:

- Stable closed-mouth conditions during late spring through fall (June 1 through October 31)
- Temperature: 9–25°C
- Salinity: 8–15 ppt
- Dissolved oxygen: 4–19 mg/L on a continuous basis

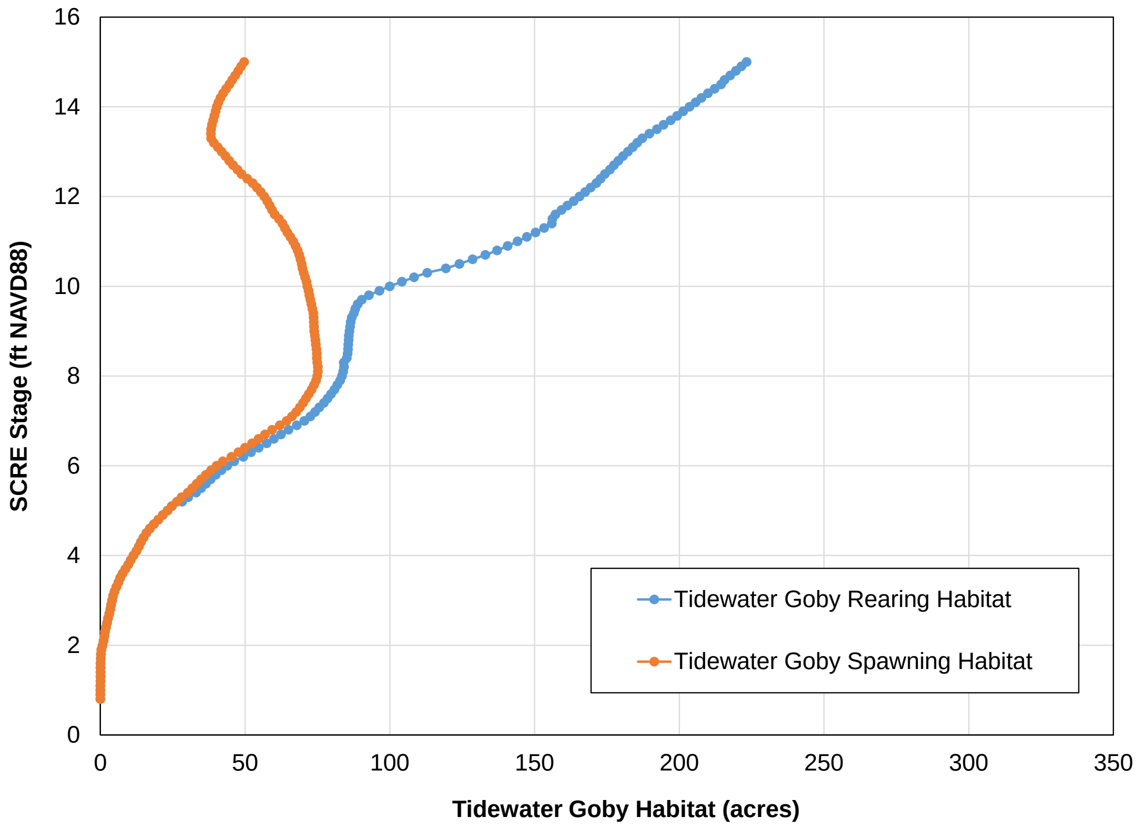


Figure 3-50. Extent of suitable tidewater goby spawning and rearing habitat as of function of SCRE stage under current conditions.

3.6.4 SCRE fish assemblage dynamics

The fish species composition in the SCRE is variable seasonally and annually. In order to determine the primary drivers of this variability and make inferences regarding inter-species interactions, scientific literature and available data were used qualitatively evaluate fish assemblage dynamics. Primary dimensions defining niche space for estuarine fish species were identified as water quality parameters (namely temperature, salinity, and dissolved oxygen tolerance), food resources and feeding characteristics, and physical habitat associations. Table 3-32 summarizes the niches of each of the resident fish species in the SCRE, which were characterized based on published observations or experimental data. The following sections provide a brief overview of how conditions in the SCRE affect the composition, distribution, and abundance of its fishes.

Table 3-34. Primary niche dimensions for native and introduced fishes in the SCRE.

Species	Feeding habit[1]	Food resources	Temperature range (°C)	Salinity range (ppt)	Dissolved oxygen requirements	References
Native fish species						
arrow goby (*Clevelandia ios)*	benthopelagic	diatoms, green algae, tintinnids, eggs and young of shrimp	4.0–30.5 (16–24 preferred)	0.9–33.9 (11–32 preferred)	Unknown	Hart (1973) as cited in Froese and Pauly (2017); Hieb (2000)
bay pipefish (*Syngnathus leptorhynchus*)	demersal	small crustaceans	Unknown	fresh to seawater (1–20 preferred)	Unknown	Hart (1973) as cited in Froese and Pauly (2017); Allen et al. (2006)
California killifish (*Fundulus parvipinnis*)	benthopelagic	wide range of benthic and pelagic invertebrates: amphipods, ostracods, insect larvae, fish eggs	11–25 preferred	0–128	Tolerant of changes in DO	Moyle (2002); Feldmeth &Waggoner (1972)
Pacific lamprey (*Lampetra tridentata*)	demersal	ammocoetes: filter feed on organic detritus and algae, though expected to rear upstream of estuary adults: predatory stage occurs in ocean	< 22 preferred	< 30 [2]	Thought intolerant of low DO, though lower DO limit unknown	Reid (2015); Richards and Beamish (1981)
prickly sculpin (*Cottus asper*)	demersal	benthic invertebrates, gammarid amphipods, insect larvae, molluscs, isopods, amphipods, larval and small fish and frogs	< 30	fresh to seawater	No data, but water quality thought to limit distribution in some areas	Moyle (2002); Swift (1989)
southern steelhead DPS (*Oncorhynchus mykiss*)	benthopelagic	gammarid amphipods, isopods, corixid beetles, mysid shrimp, chironomids, polychaetes, snails, small fish	< 26	< 10	> 5 mg/L	see Section 3.6.3.1
staghorn sculpin (*Leptocottus armatus*)	demersal	benthic amphipods, mysid shrimp, gammarid amphipods, nereid worms, insect larvae, small fish	< 26.5	0–67 (0 – 34 preferred, though adults are not commonly found in freshwater)	Tolerant of low DO	Moyle (2002); Morris (1960); Speers-Roesch et al. (2013); Swift (1989)

Species	Feeding habit[1]	Food resources	Temperature range (ºC)	Salinity range (ppt)	Dissolved oxygen requirements	References
striped mullet (*Mugil cephalus*)	benthopelagic	juveniles: zooplankton adults: benthic invertebrates, microalgae, detritus	> 14	0–75	> 4 mg/L	Moyle (2002); Thomson (1963); Collins (1985); Sutula et al. (2012); Sylvester et al. (1975)
tidewater goby (*Eucyclogobius newberryi)*	demersal	ostracods, gammaris amphipods, mysid shrimp, insect larvae (e.g., chironomids)	< 25	0–41 (< 12 preferred)	4–19 mg/L, though they are tolerant of brief periods of lower DO	see Section 3.6.3.2
topsmelt (*Atherinops affinis*)	pelagic	juveniles: algae, kelpfly larvae adults: zooplankton	5–29	0–80 (prefer 0–34, adults rarely found in freshwater)	2–18 mg/L	Moyle (2002); Baxter et al. (1999)
threespine stickleback (*Gasterosteus aculeatus williamsoni*)	benthopelagic	aquatic insects (e.g., chironomid larvae), crustaceans (e.g., ostracods), earthworms, fish eggs including conspecifics	< 34 (3–19)	fresh to seawater	> 2 mg/L	Moyle (2002); Allen & Wootton (1982); Feldmeth & Baskin (1976)
Introduced Fish Species						
arroyo chub[3] (*Gila orcutti*)	benthopelagic	predominantly algae, but also small insects and crustaceans	10–30+ (10–24 preferred)	freshwater	Tolerant of low DO	Greenfield and Deckert (1973); Moyle (2002); Castleberry and Cech (1986)
common carp (*Cyprinus carpio*)	benthopelagic (juveniles more pelagic, adults more benthic)	insect larvae, crustaceans, molluscs, annelids, algae, organic detritus	4–36 (~24 preferred)	< 16 (freshwater preferred)	Tolerant of low DO	Moyle (2002)
fathead minnow (*Pimephales promelas*)	demersal	diatoms, filamentous algae, organic detritus, small invertebrates	< 33 (22–23 preferred)	Tolerance unknown (freshwater preferred)	Tolerant of periodic hypoxia, but chronic exposure to < 4 mg/L deleterious	Keast (1966) as cited in Moyle (2002); Moyle (2002); Brungs 1971

Species	Feeding habit[1]	Food resources	Temperature range (ºC)	Salinity range (ppt)	Dissolved oxygen requirements	References
goldfish (*Carassius auratus*)	benthopelagic	algae, zooplankton, benthic invertebrates, macrophytes, organic detritus	0–41 (27–37 preferred)	0–14 (1–3 preferred)	Tolerant of low DO	Moyle (2002); Kucuk (2013)
green sunfish (*Lepomis cyanellus*)	benthopelagic	juveniles: fish larvae, zooplankton, insect larvae adults: small fishes, large aquatic insects, terrestrial insects, crayfish	< 38 (26–30 preferred)	< 2	Tolerant of low DO	Moyle (2002)
Mississippi silverside (*Menidia audens*)	pelagic	zooplankton (particularly copepods), terrestrial insects, larval fish, larval insects	8–34 (20–25 preferred)	< 33 (10 – 15 preferred, but also thrive in freshwater)	> 4.3 mg/L [4]	Strongin et al. (2011); Cohen & Bollens (2008); Moyle (2002); Swift et al. (2014); Hubbs et al. (1971); Sutula et al. (2012)
Santa Ana sucker[3] (*Catostomus santaanae*)	demersal	mostly algae and organic detritus, occasionally insect larvae	15–28 (< 22 preferred)	freshwater	Unknown	Moyle (2002); Swift (2001); Saiki (2000) as cited in Haglund et al. (2003)
Shimofuri goby[5] (*Tridentiger bifasciatus*)	demersal	hydroids, barnacles, tube-dwelling amphipods, copepods, isopods, polychaetes, ostracods; larval fish	< 37 (10–25 preferred)	< 21 (<17 preferred)	Unknown	Matern (2001); Howard and Booth (2016); R. Swenson as cited in Howard and Booth (2016)
western mosquitofish (*Gambusia affinis*)	benthopelagic	algae, aquatic insects, terrestrial insects, zooplankton, varied aquatic invertebrates, fish eggs and larvae including conspecifics	0.5–42 (25–30 preferred)	0–58 (< 25 preferred)	Highly tolerant of hypoxia; > 0.2 mg/L	Moyle (2002); Swift (1989)

Species	Feeding habit[1]	Food resources	Temperature range (ºC)	Salinity range (ppt)	Dissolved oxygen requirements	References
yellowfin goby (*Acanthogobius flavimanus*) 6	demersal	copepods, mysid shrimp, gammarid amphipods, crabs, small fish	15–32	0–40 (5–40 for breeding)	3–14 mg/L	Moyle (2002)

[1] Feeding orientations based listing on FishBase.org (Froese & Pauly 2017)
[2] For the life history Phases expected in the estuary (adults and metamorphs at Phase 6 or later (Richards and Beamish 1981)
[3] Santa Ana sucker and arroyo chub are ottherwise native to the region, but considered introduced in the Santa Clara River drainage (Swift et al. 1993)
[4] Based on proxy species *Menidia menidia* (Sutula et al. 2012)
[5] Has not been observed in the SCRE, but is present in upstream locations and is predicted to invade the SCRE (Howard and Booth 2016)
[6] Last observed in 1997

3.6.4.1 Water quality conditions

In general, species -specific physiological properties such as aerobic scope and passive tolerance determine a fish's environmental niche dimensions (i.e., the range of suitable abiotic conditions) (Pörtner et al. 2010). These abiotic conditions (salinity, temperature, DO) are widely recognized as driving factors of fish distribution and assemblage composition in estuaries (e.g., Matern et al. 2002), and are expected to act as the primary filter on the fish species occupying the SCRE. Variability in these parameters varies the composition of species for which the estuary provides suitable habitat at any given time. When changing conditions become unsuitable for a given species, those individuals will retreat to more suitable water if possible, typically in upstream or coastal ocean habitats, depending on their requirements. Those locations can be thought of as refuge for long-term estuary residents during periods of adverse conditions, or as source pools for short-term or opportunistic estuary occupants. If abiotic conditions are unsuitable in these alternate locations, or these locations are inaccessible (due to mouth closure or lack of in-stream flow) these individuals might perish. At the same time, access permitting, other species may enter the SCRE if abiotic conditions become more suitable than their current location, or if conditions are equally suitable, but occupation of the SCRE confers some other benefit (e.g., access to food resources, reduced risk of predation).

Salinity appears to be a primary driver (i.e. environmental filter) of fish assemblage composition in the SCRE, which is consistent general trends in other estuaries (e.g., Martino and Able 2003; Barletta et al. 2005). Of the fish species found in the SCRE, most prefer lower salinities (i.e., fresh to brackish water), but can tolerate higher salinities for shorter periods of time (Table 3-32). Most of the introduced species in the SCRE thrive in freshwater, consistent with prior work suggesting that water transfers are a primary mechanism of fish introductions in southern California (Moyle and Williams 1990) though several of the species can tolerate moderate levels of salinity and two (yellowfin goby and western mosquitofish) can tolerate hypersaline conditions (Table 3-32). On average, native species found in the SCRE can thrive in a wider range of salinities and have higher maximum salinity tolerance relative to non-native species (juvenile steelhead are a notable exception to this rule) (Table 3-32). Based on this information, prolonged periods of elevated salinity might cause long-term shifts in the SCRE fish assemblage with the potential to benefit some native fishes. It should be noted that extended periods of elevated salinity would likely be detrimental to rearing juvenile steelhead, however, which highlights the need for weighing competing species demands. Short-term periods of elevated salinity might confer some benefits to native fishes as well (e.g., increase survival during a critical lifestage).

Analysis of recent fish survey data (2008–2016) indicates that while native fish were typically more abundant than non-native fish from 2008-2012, a shift in the fish assemblage composition toward non-native species beginning in 2013 (Figure 3-51). This shift coincided with low breach frequency conditions (<10% open mouth conditions annually) that persisted from 2012–2015 (Figure 3-26 and Figure 3-51). Mouth breaching controls salinity in the SCRE by allowing for tidal exchange (and wave overwash shortly after mouth closure) in a system otherwise dominated by freshwater inputs (Section 3.3.5). Extended closed mouth periods increase the predominance of freshwater conditions and may shift assemblage dynamics in favor of non-native species. The percent of fish captured that were native positively correlates with the percent of open mouth conditions (R^2=0.45), suggesting that mouth status affects the relative abundance of native vs. non-native fishes in the SCRE. For example, non-native green sunfish, which likely prey on native fishes including tidewater goby, are a true freshwater fish (< 2 ppt) and might be depressed by the introduction of saline water to the SCRE. However, green sunfish were one of the most abundant fishes captured during 2015 seine surveys in the SCRE, likely as a result of persistent drought conditions and low variability in salinity (Table 3-31). Shimofuri goby, a non-native species that has been identified in the upper Santa Clara River watershed, but has not yet been observed in the SCRE, has a lower salinity tolerance relative to its potential native competitors in the SCRE, including tidewater goby (Howard and Booth 2016). Despite the prevailing freshwater conditions in the estuary in recent years, low in-stream flow (another symptom of persistent drought) may have created a barrier preventing Shimofuri goby invasion of the SCRE.

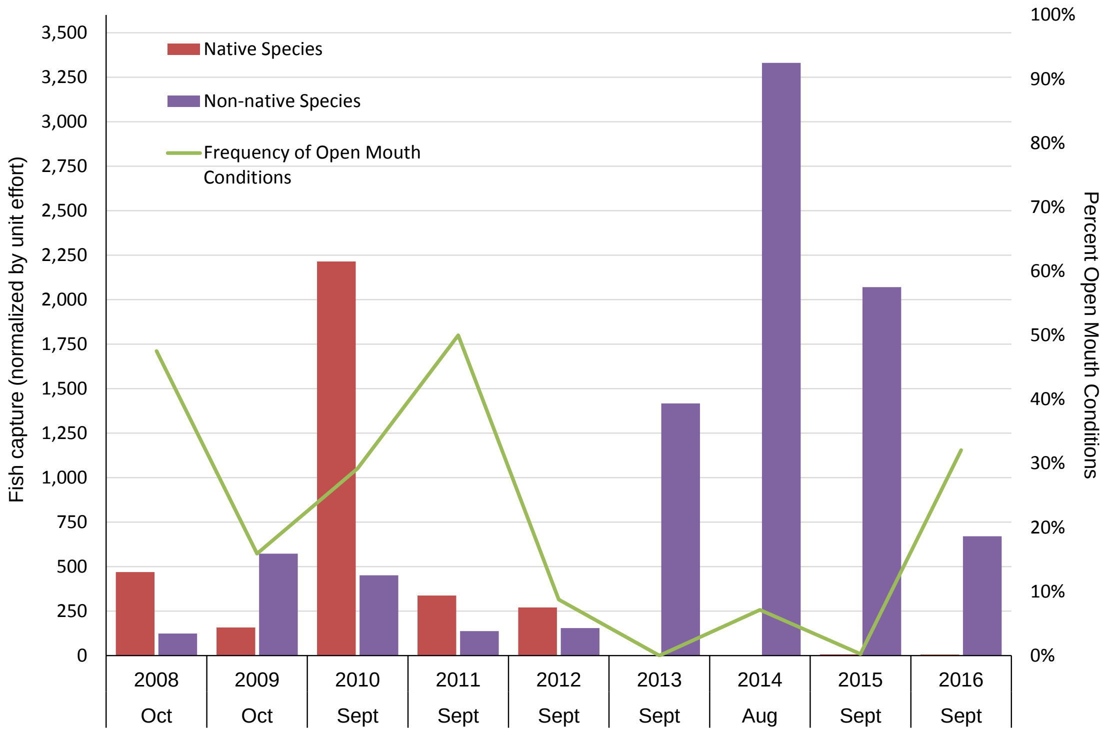


Figure 3-51. Number of native and non-native fish captured (normalized by unit effort) (2008–2016) and percent of days with open mouth.

In general, the introduced species in the SCRE have higher upper thermal tolerances and higher preferred temperature ranges (Table 3-32). Many of introduced species are also highly tolerant of hypoxic or low DO conditions (Table 3-32). While many native species in the SCRE can tolerate high temperatures, most prefer temperatures well below 25ºC. A number of introduced species, on the other hand, are adapted to warm water and are nearer to their metabolic optima at temperatures greater than 25ºC. Many native species in the SCRE are tolerant of periodic low DO conditions, and many have adaptations allowing them to survive severe but short-lived DO crashes. Some native species, however, including Pacific lamprey and steelhead, are relatively intolerant of low DO conditions. As transient residents of the SCRE, these species may avoid the lagoon during low DO periods, which can interfere with migration timing (for both species) and growth opportunities for rearing steelhead. Many of the introduced fishes in the SCRE are adapted to eutrophic environments with prolonged or chronic low DO or hypoxic conditions. For example, western mosquitofish and fathead minnow, which are considered pioneer species based on their ability to occupy water uninhabitable by most other species, are likely to thrive during extended periods of depressed DO, elevated temperatures, or in the presence of environmental pollutants (Moyle 2002). Common carp, goldfish, and green sunfish are also known to thrive in eutrophic waters with frequent and extended periods of low DO. The abundance of non-native fishes in the SCRE and their wide range of environmental tolerances are likely to result in suppression of native fish production by competition and predation; depressed DO and elevated temperature are likely to exacerbate this effect.

3.6.4.2 Food resources and trophic interactions

When abiotic conditions in the SCRE are suitable for multiple species, a species' ability to exploit available food resources likely determines successful establishment. The fish species pool in the SCRE is comprised primarily of invertivorous or omnivorous generalists, both native and non-native, most of which have highly flexible diets (Table 3-32). A number of these generalists are also known to eat larval fish when available, while a smaller number eat juvenile fishes or adults of smaller-bodied fishes, creating a potential for consumptive interactions between native and non-native fishes, and potential top-down control on native fish populations (Table 3-32). There is considerable overlap in both abiotic and trophic niche dimensions of a number of SCRE fishes. This overlap can potentially result in competition between species, and may result in suppression of some populations by reducing access to food resources or competitive exclusion from optimal habitat. Based on the predominance of invertivorous generalists in the SCRE, total availability of invertebrate food resources is likely to limit fish production. Also note, however, that while invertebrates comprise the primary food items for many of the fishes of the SCRE, a number of fishes are omnivorous and will consume algae or detritus especially when invertebrate production is low. It is difficult to predict the relative abilities of fishes to exploit food resources based on the available data, as they likely depend on the complex interactions of a number of factors. However, relative physiological performance under given environmental conditions (temperature, DO, salinity) is expected to be a primary determining factor of competition dynamics.

In addition to BMI abundance, BMI community composition may have some effects on fish assemblage dynamics, though this is not expected to be a primary driver, given the generalist feeding strategies of most fishes in the SCRE. Despite these generalist strategies, some degree of resource partitioning (i.e., local specialization) might be expected. While no data are available to evaluate resource partitioning in the SCRE, some of the generalist species have been known to specialize in other systems, presumably to avoid competition or exploit an abundant food resource (e.g., Swenson and McCray 1996). This is only likely to impact fish assemblage dynamics when both diversity and abundance of food resources are low. For example, BMI surveys in the SCRE in 2015 found benthic invertebrate abundance to be very low in the fall and winter (Section 3.6.1), suggesting food limitation that may have resulted in a highly competitive environment. Additionally, those surveys found a low diversity of benthic taxa, which could have increased competition between species that might otherwise avoid competition by partitioning resources. Tidewater goby may be particularly susceptible to population suppression by competition, which may in part explain the low abundance of tidewater goby during 2013–2016 (Figure 3-49). Chase et al. (2016) found that under experimentally reduced ration conditions (75% and 50% satiation), tidewater goby that were held with threespine stickleback and/or rainwater killifish decreased in weight and showed physiological signs of stress. Under 50% ration conditions, tidewater goby held with threespine stickleback and/or killifish also showed reduced survival relative to controls over the 28-day study period. During 2015 BMI surveys, ostracods, oligochaetes, and chironomid larve comprised over 95% of observed BMI, with very low total abundance. While there are no data on the availability of epibenthic or pelagic prey during this period, this paucity of benthic food abundance and diversity might have left tidewater goby competing with a number of species, both native and non-native for limited food resources. However, during 2015 fish surveys, Stillwater staff observed high abundance of cladoceran and other zooplankton, which are high value food resources for pelagic fish. Because of data limitations, it is difficult to make inferences about food availability in the SCRE.

Little research has evaluated the effects of inter-species interactions on steelhead in southern California estuaries and lagoons. Emmett (1997) suggested that the feeding habits of juvenile salmonids in Northwest estuaries indicated a high diet overlap between different species, but the data did not conclusively indicate that such diet overlap has resulted in competition. Competition is less likely to suppress rearing steelhead production than tidewater goby, as sub-adult steelhead primarily feed on pelagic invertebrates, though they also feed opportunistically on active benthic invertebrates (Moyle 2002). There are fewer pelagic-oriented invertivorous fishes in the SCRE relative to demersal fishes (Table 3-32). Additionally, sub-adult steelhead are strong swimmers, effective predators, and have been known to aggressively defend feeding territory; they are thus less likely to be displaced from preferred feeding habitat or outcompeted for available food resources (Moyle 2002). Boughton et al. (2017) identified gammarid amphipods and isopods as primary prey items for steelhead rearing in the Russian River estuary, while corixid beetles and chironomids constituted secondary prey items. Gammarid amphipods have been abundant in the SCRE in the past and chironomids are consistently abundant. While chironomids were considered secondary in terms of abundance in gut contents, Boughton et al. (2017) note that they are energetically rich relative to some of the primary prey items and can confer a growth advantage. As steelhead size increases, piscivory becomes increasingly important for the steelhead diet (Moyle 2002). Larger steelhead may compete with larger green sunfish for food resources.

In addition to suppression by competition, production of some fishes in the SCRE may be controlled by predation. Some fishes of the SCRE are known to eat fish eggs, larvae, juveniles, or adults as part of diet (Table 3-32). While no data are available to quantify or directly assess the effects of predation on tidewater goby populations in the SCRE, the presence of several species that are thought to be predators of various life stages of tidewater goby suggests that predation may exert control over the tidewater goby population in the SCRE. While displacement during open mouth conditions, not predation, is typically thought to be the primary mechanism regulating tidewater goby population size, analysis of data from seining surveys (2008–2016) indicates that the tidewater goby population size is decoupled from the frequency of open mouth conditions in the SCRE (Figure 3-49). This apparent decoupling suggests that tidewater goby have sufficient refuge from high flows during open mouth conditions, and that other mechanisms (e.g., predation, competitive suppression) exert stronger control over tidewater goby population size in the SCRE. The high abundance of potential tidewater goby predators, including several non-native species, indicates a high potential for predatory control on the tidewater goby population in the SCRE. USFWS (2013) considers absence of abundant invasive species a primary constituent element of critical habitat for tidewater goby. Swenson (1997) suggested that green sunfish prey on adult tidewater goby, and predation by green sunfish is thought to have caused at least one extirpation of tidewater goby (D. Holland, unpublished; as cited in Lafferty and Page 1997). During 2015 and 2016 fish sampling efforts in the SCRE, green sunfish were the third most abundant fish species captured (Table 3-31). Swift et al. (1989) suggested that staghorn sculpin, prickly sculpin, and steelhead/rainbow trout (all are native to the SCRE) are likely predators of adult tidewater goby and reported one instance in which the stomach contents of one steelhead contained 6–10 goby. Swift et al. (1989) also suggested that some smaller omnivorous fishes known to occupy lagoon habitats, including mosquitofish, may prey heavily on larval or juvenile tidewater goby. During 2015–2016 fish sampling efforts in the SCRE, mosquitofish were the second most abundant fish species captured. Mississippi silversides are also known to prey on larval fish, and have been suggested as a threat to tidewater goby populations (Swift et al. 2014). Mississippi silverside were by far the most abundant fish in 2015 and 2016 seine surveys (Table 3-31), indicating a substantial potential predation threat for larval tidewater goby. African clawed frogs are also abundant in the SCRE; 79 were captured during seining surveys in 2015 and 2016. African clawed frogs are voracious predators and have been documented feeding on tidewater goby in the SCRE (Lafferty and Page 1997). The SCRE also provides foraging habitat for several piscivorous avian species (see Section 3.6.4.1) that are likely to prey on tidewater goby. Sub-adult steelhead may be vulnerable to predation by larger avian species such as kingfishers, mergansers, and herons, particularly during periods when depth cover is not available (Moyle 2002).

3.6.4.3 Physical habitat associations

The SCRE is primarily composed of sand, mud, and silt bottomed flats with low habitat complexity. Emergent or submerged vegetation is not abundant, nor are structural features such as large boulders or woody debris, which provide cover for some fishes. A number of fishes found in the SCRE are typically associated with emergent vegetation or other cover features in other systems. Elements providing cover or shelter are considered primary constituent elements of critical habitat for both tidewater goby and steelhead (USFWS 2013, NOAA 2005). Habitat complexity is presumed to reduce predation risk for both tidewater goby and steelhead, and is thought to provide flow refuge during high flow, open mouth conditions. Flooding of the riparian habitat surrounding the McGrath State Beach campground allows transient access to vegetative cover, and the outfall channel on the north side of the lagoon may serve as cover for fishes and flow refuge during storm flows. Because of the lack of large boulders and woody debris in the SCRE, deeper water habitat is likely to serve as cover from avian predation. The limited habitat complexity of the SCRE may increase the frequency of inter-species interactions in the SCRE relative to more complex systems. However, as described in Section 3.6.4.2, the apparent decoupling of tidewater goby population fluctuations and annual frequency of open mouth conditions indicates that sufficient refuge from high flows is available in the SCRE.

There is considerable overlap in substrate associations between native and non-native fishes in the SCRE that, combined with low habitat complexity, may have negative impacts on native fishes including tidewater goby. For example, common carp root around silty bottom sediments to stir up benthic invertebrates, which they then pluck from the water column (Moyle 2002). Some native fishes in the SCRE are known to spawn in similar habitat. This disturbance of the sediment may disrupt spawning or early development of some species. While sand is the preferred substrate for tidewater goby spawning (Section 3.6.3.2), they have also been known to construct burrows in mud or silt bottoms in which carp feed.

3.7 Wildlife Habitat and Species

The habitats within and adjacent to the SCRE 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 and a number of bird species also use habitats provided by the VWRF Wildlife/Polishing Ponds. The following sections review the available data and scientific literature related to the wildlife community of the SCRE and key factors influencing wildlife habitat conditions. Particular attention is paid to the focal avian species for this study, California least tern and western snowy plover. Habitat for focal species serves as the primary basis for evaluation of wildlife habitat-related beneficial uses (see Section 5). However, information on other wildlife species—including special-status 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 CDFW—is presented to provide context for understanding the overall ecological condition of the SCRE and factors affecting native wildlife species. The beneficial use assessment relies on this information for qualitative assessments of habitat conditions, and the relative degree to which alternative VWRF management scenarios affect the propagation of native species.

3.7.1 Wildlife species by habitat type

3.7.1.1 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 were selected as the focal species for this study and are discussed in more detail in Section 3.7.2. 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).

3.7.1.2 Mudflat

Mudflats are defined as “expanses of mud contiguous with a water body, often covered and exposed by tides” (CDFG 1988). In the SCRE mouth breaching drains the estuary, converting open water habitat to exposed or shallow mudflat habitat. These areas can be rich in benthic infauna such as burrowing crustaceans and oligochaetes, epibenthic fauna such snails, as well as stranded nekton or pelagic invertebrates, and serve as valuable foraging habitat for the shorebirds and wading birds described in the previous section, including western snowy plover. Raccoons and opossums may opportunistically forage in these habitats as well when they are fully exposed.

3.7.1.3 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*), Baja California treefrog (*Pseudacris hypochondriaca hypochondriaca*), 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 feeding 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, 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.), and California meadow vole (*Microtus californicus*). Bat species may forage for insect prey over wetlands.

3.7.1.4 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*), striped skunk (*Mephitis mephitis*), bobcat (*Lynx rufus*), raccoon (*Procyon lotor*), Virginia opossum (*Didelphis virginiana*), and brush rabbit (*Silvilagus bachmani*) (ESA 2003).

3.7.1.5 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/Polishing 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). Open water in the SCRE and Wildlife/Polishing Ponds also serve as foraging habitat for California least tern (see Section 3.7.2).

3.7.2 Focal avian 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 30 years (R. Smith, Ventura Audubon Society, pers. comm., 2010). Annual monitoring reports for western snowy plover and California least tern since 2007 have been provided by California State Park's Channel Coast District and CDFW. The sections below describe the life history traits and habitat requirements for these species. In order to identify the habitat characteristics of the SCRE that are important for species behavior and long-term success, we reviewed the scientific literature and available data on these species, with emphasis on site-specific information where possible. As both of these species are federally listed under the Endangered Species Act, we also reviewed critical habitat designations to evaluate the extent to which Primary Constituent Elements[18] (PCEs) of habitat are realized in the SCRE, and to determine whether these factors might be affected by VWRF discharge.

3.7.2.1 Western snowy plover

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 (58 Fed. Reg.

[18] Now referred to as physical and biological features (PBFs) by USFWS and NMFS; however, we have retained the PCE terminology since this was the convention at the time of Critical Habitat designation

12864 (Mar. 5, 1993) [listing the Pacific Coast population as threatened throughout its range]), while interior populations are a state species of special concern. Further, USFWS has designated the SCRE as critical habitat for wester snowy plover under the federal Endangered Species Act. (64 Fed Reg. 68,508 (Dec. 7, 1999); 77 Fed. Reg. 36,727 (June 19, 2012).)

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.

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).

Recent surveys

California State Park's 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.

Annual western snowy plover visual nest surveys are typically conducted weekly by a USFWS 10(a)(1)(A) recovery permit holder, who walks meandering transects throughout the study area in search of nests or sign (More 2008, Smith 2009, CA State Parks 2013–15). Nesting surveys are typically initiated in March and finished after nesting is complete, typically around August. Efforts are 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 are conducted the rest of the year by CCD staff or Ventura Audubon Society volunteers.

The number of nests at McGrath State Beach between 2003 and 2009 has ranged between 10 and 23 (Smith 2009); and the number of nests between 2013 and 2015 has ranged between 13 and 18 (CA State Parks 2013–15). Number and locations of western snowy plover nests observed between 2007 and 2009 and 2013 and 2015 are illustrated in Figure 3-52. 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 (Figure 3-52); and near McGrath Lake. Over years 2007–2009, nest locations have shifted from next to McGrath Lake, to along the outer beach from the lake to opposite the campground (Smith 2009). Some of the beach present during 2007–2009 has receded in current years.

Western snowy plover nesting activity in this area usually peaks in May or June. The primary cause of nest failure is often 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).

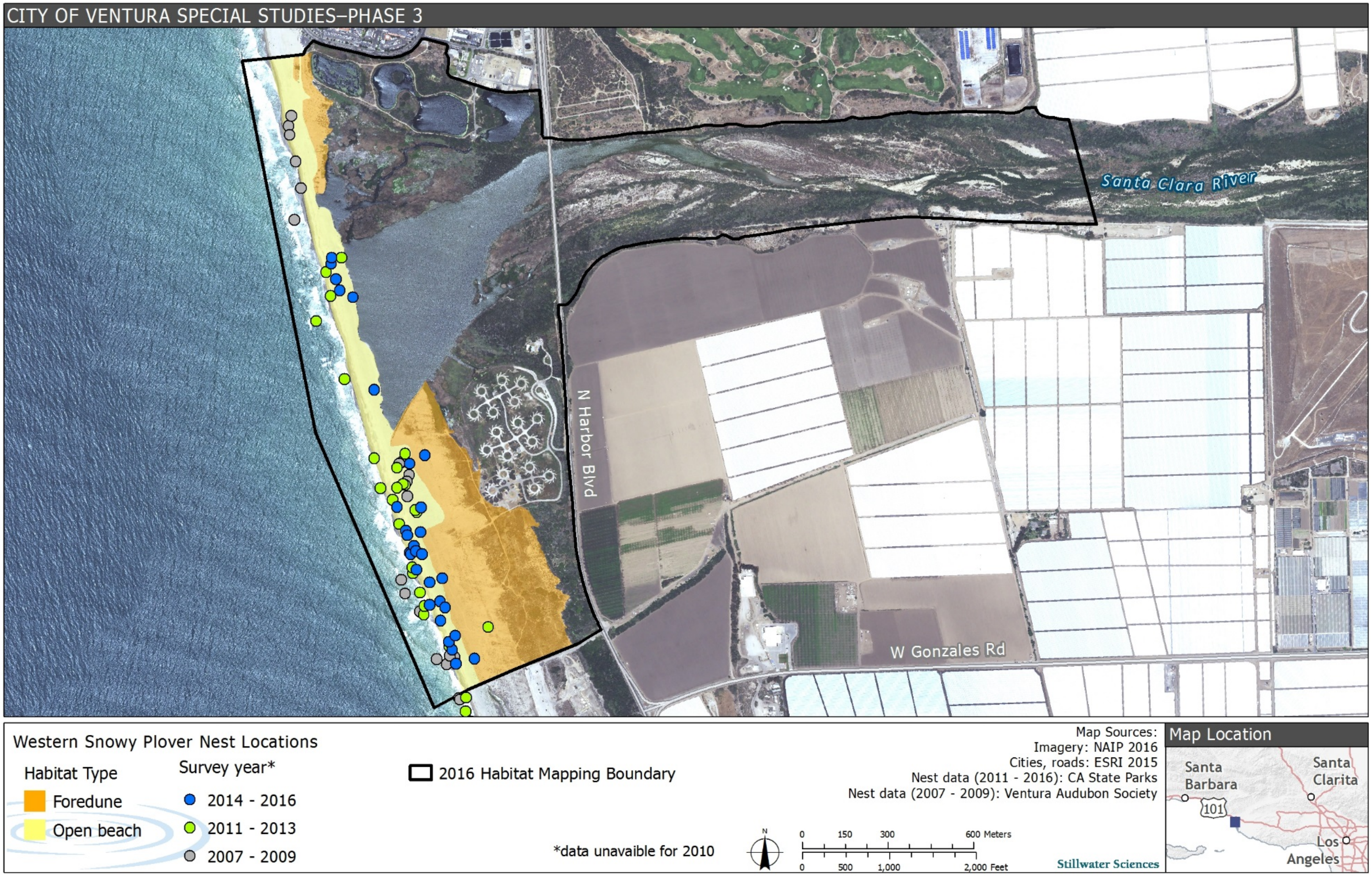


Figure 3-52. Habitats and western snowy plover nest locations observed near the SCRE during surveys in 2007–2009 and 2011–2016.

Habitat use within and around the SCRE

This section reviews the available scientific literature and habitat use information to develop habitat suitability criteria for western snowy plover in the SCRE. As part of the critical habitat designation under the Endangered Species Act, PCEs have been established for western snowy plover (Table 3-35). The extent to which these PCEs are realized in the SCRE, as well as their relationship with VWRF discharge were considered when determining habitat suitability criteria for the comparison of VWRF discharge scenarios and the beneficial use assessment presented in Section 5.

Table 3-35. Primary Constituent Elements of Critical Habitat for western snowy plover.

PCE Number	PCE Description	Source
1	Sandy beach or dune system areas that are below heavily vegetated or developed areas and above the daily high tides	77FR36727, pg 36747-36748
2	Shoreline habitat areas for feeding that support small invertebrates, with minimal vegetation, that are between the annual low tide or low-water flow and annual high tide or highwater flow, subject to inundation but not constantly under water	77FR36727, pg 36747-36748
3	Surf- or water-deposited organic debris, such as seaweed (including kelp and eelgrass) or driftwood located on open substrates that supports and attracts small invertebrates described in PCE 2 for food, and provides cover or shelter from predators and weather, and assists in avoidance of detection (crypsis) for nests, chicks, and incubating adults	77FR36727
4	Minimal disturbance from the presence of humans, pets, vehicles, or human-attracted predators, which provide relatively undisturbed areas for individual and population growth and for normal behavior	77FR36727

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 3.4.3.1, 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).

Habitat use data in the SCRE suggest that the open beach habitat between the estuary and Pacific Ocean provide high value nesting habitat (Figure 3-52). Sandy beach areas below vegetated areas are included as a PCE of critical habitat for western snowy plocer (PCE 1; Table 3-35). 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 3-52). Figure 3-53 shows physical habitat curves for 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 sand-water interface that supports or attracts invertebrates is a PCE of western snowy plover critical habitat (PCE 3; Table 3-35). While the habitat curve in Figure 3-53 shows potential foraging habitat for western snowy plover, the sand-water interface along the beach berm is the most likely foraging habitat for western snowy plover. Accordingly, the estimated length of the beach berm was used to compare foraging habitat for western snowy plover under alternative VWRF discharge scenarios in Section 5. Availability of nesting habitat is relatively more stable than availability of foraging habitat as stage increases, since foraging habitats include the areas of the SCRE that are inundated at higher stage, while this is not considered suitable nesting habitat based on the risk of flooding the nest with increased stage. Because nesting habitat is largely invariable with stage, nesting habitat was not included in the comparison of discharge scenarios (Section 5.5.6.2). The stage-habitat curve shows an inverse relationship with the habitat curve modeled for the two focal fish species, reflecting tradeoffs between fish and shorebird 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 reducing water levels would temporarily increase the amounts of dry sandy nesting habitat as well as mudflats used for foraging, vegetation—such as willows—would eventually adapt to the new water levels and inundation periods and replace such habitat (see Section 5.3.1). Minimal disturbance from humans is a PCE of western snowy plover critical habitat (PCE 4; Table 3-35). However, this was not explicitly incorporated into habitat analyses in the comparison of VWRF discharge scenarios since it is considered largely unaffected by VWRF discharge.

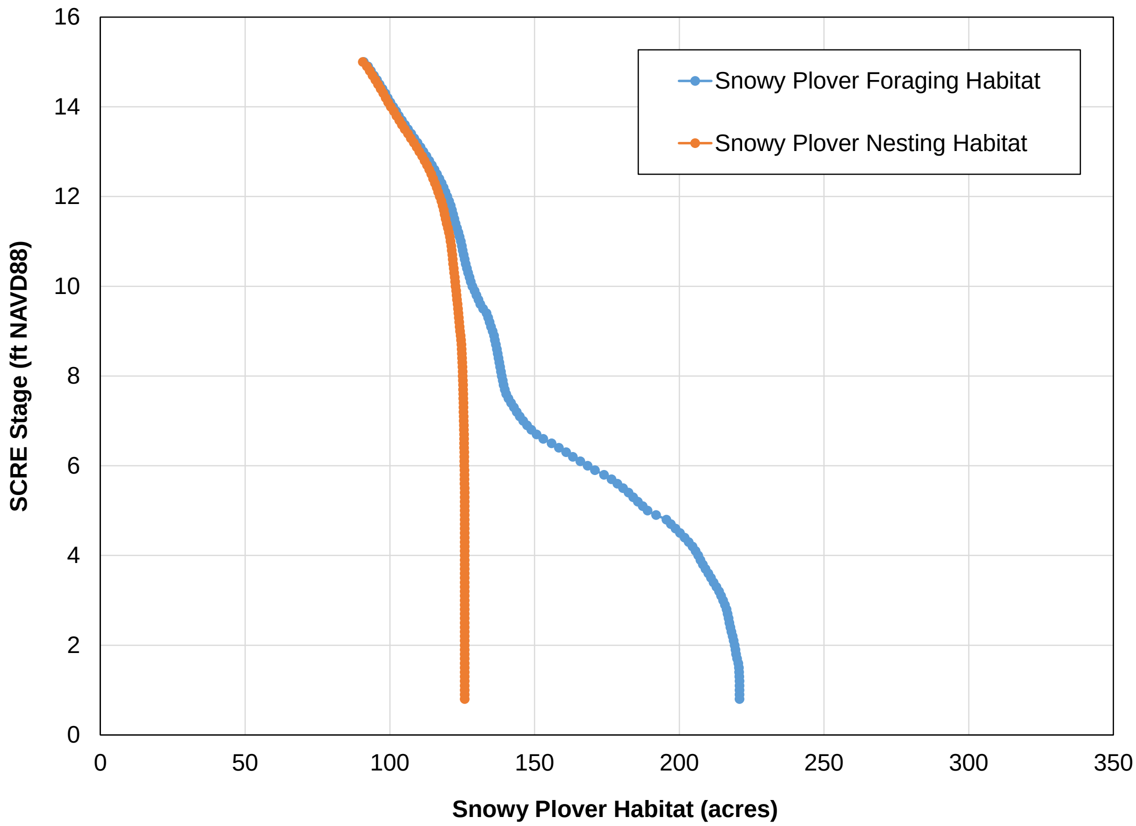


Figure 3-53. Extent of suitable snowy plover foraging and nesting habitat as of function of SCRE stage under current conditions.

3.7.2.2 California least tern

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, and the only subspecies of least tern found in California. The California least Tern is federally listed (35 Fed. Reg. 8,491 (June 2, 1970) and state listed (June 27, 1971) as endangered throughout its range. The California least tern is also designated for protection as a California Fully Protected Species.

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).

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. A lack of sufficient foraging resources is thought to be a significant factor limiting the growth of the California least tern population (CA State Parks 2016).

Recent surveys

Annual California least tern nest surveys are conducted once weekly by a USFWS 10(a)(1)(A) recovery permit holder, who walks meandering transects throughout the study area in search of nests or sign (Smith 2008, 2009, CDFG 2011–2012, CDFW 2013–2016, CA State Parks 2016). 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, CA State Parks 2016). 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 during surveys conducted between 2010 and 2015 ranged between 4 and 72, with the highest number of nests and fledglings in 2015 (Table 3-33). The number of nests at McGrath State Beach between 2000 and 2009 ranged between 3 and 97, with a general upward trend in the number of nests compared with earlier surveys (Smith 2008, 2009). Number and locations of California least tern nests observed between 2007 and 2009, for which spatial data were available, are illustrated in Figure 3-54.

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 3-54). 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).

Table 3-36. Estimated number of breeding pairs, nests, and fledglings at McGrath State Beach, 2010–2015.

Survey year	Estimated number of breeding pairs	Number of nests observed	Estimated number of fledglings
2010	36	36	14
2011	26	26	20
2012	38–39	42	8
2013	37	37	0
2014	4	4	2
2015	45–69	72	27

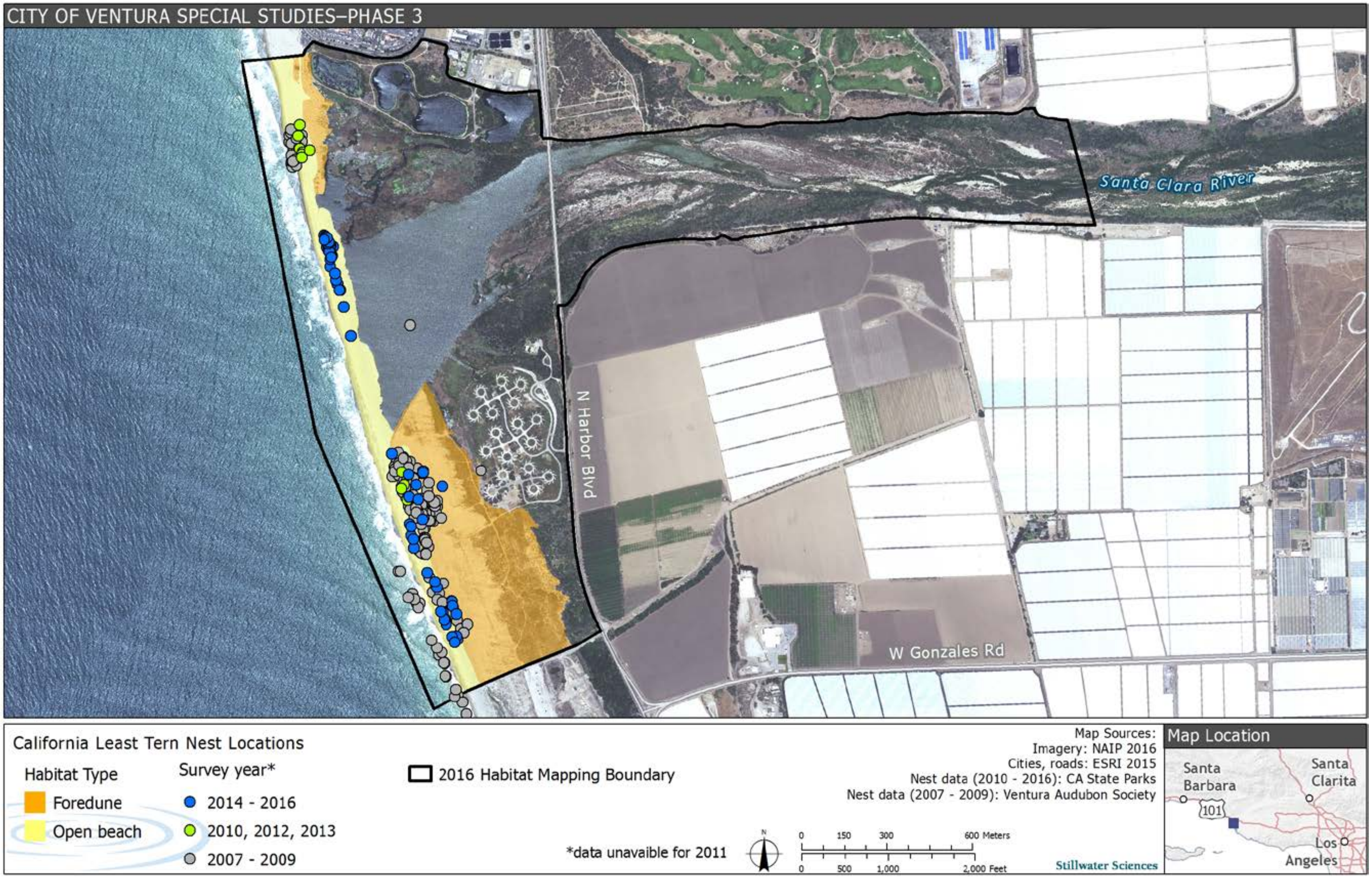


Figure 3-54. Habitats and California least tern nest locations observed near the SCRE during surveys in 2007–2016.

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.

Habitat use within and around the SCRE

This section reviews the available scientific literature and habitat use information to develop habitat suitability criteria for California least tern in the SCRE. Although California least tern are listed as an endangered species under the Endangered Species Act, critical habitat and associated PCEs have not been established. However, the general PCE template that USFWS uses as the basis for establishing species-specific PCEs can be used to relate habitat conditions in the SCRE to California least tern requirements (Table 3-37).

Table 3-37. Primary Constituent Elements of Critical Habitat for California least tern.

PCE Number	Features	SCRE Conditions
1	Space for individual and population growth and for normal behavior	Open water habitat for foraging
2	Food, water, air, light, minerals, or other nutritional or physiological requirements	Small pelagic fish occupying open water habitat for foraging
3	Cover or shelter	None identified in SCRE
4	Sites for breeding, reproduction, or rearing (or development) of offspring	Open beach and foredune habitat for nesting
5	Habitats that are protected from disturbance or are representative of the historical, geographical, and ecological distributions of a species	Foraging and nesting habitat as described above; risk of nest flooding associated with altered hydrology

The area surrounding the SCRE provides only summer nesting and rearing habitat for California least terns (i.e., the species does not winter here). 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 3.4.3.1, 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 Wildlife/Polishing 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 3-54). Figure 3-55 shows potential 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 the same as the relationship between nesting habitat and stage for western snowy plover, described above. 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 slightly less nesting 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. Because nesting habitat is largely invariable with stage, nesting habitat was not included in the comparison of discharge scenarios (Section 5.5.6.2).

The relationship between potential California least tern foraging habitat and SCRE stage shows that as stage increases, foraging habitat increases with the corresponding increase in inundated area. The relationship between California least tern foraging area and SCRE stage also illustrates that a dry estuary would dramatically decrease foraging area, with only the VWRF Wildlife/Polishing Ponds (19.1 acres) remaining within the estuary boundaries considered in the development of the curves (Figure 3-55). While Figure 3-55 shows potential foraging habitat as inundated area, the comparison of discharge scenarios (Section 5.5.6.2) uses open water habitat area, which does not include shallow inundated areas. It is important to note that the lagoon and VWRF Polishing Ponds are not the sole source of least tern foraging habitat in the area. Least terns also forage in McGrath Lake (which is within the SCRE subwatershed), as well as in 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 is a conservation priority for this species. However, nesting habitat area in the SCRE is largely unaffected by SCRE stage (Figure 3-55). Some risk of nest flooding can occur when sand bedded areas of the lagoon are dry during open mouth conditions but become inundated following mouth closure and estuary filling. Accordingly, risk of nest flooding was assessed in the comparison of VWRF discharge scenarios (Section 5).

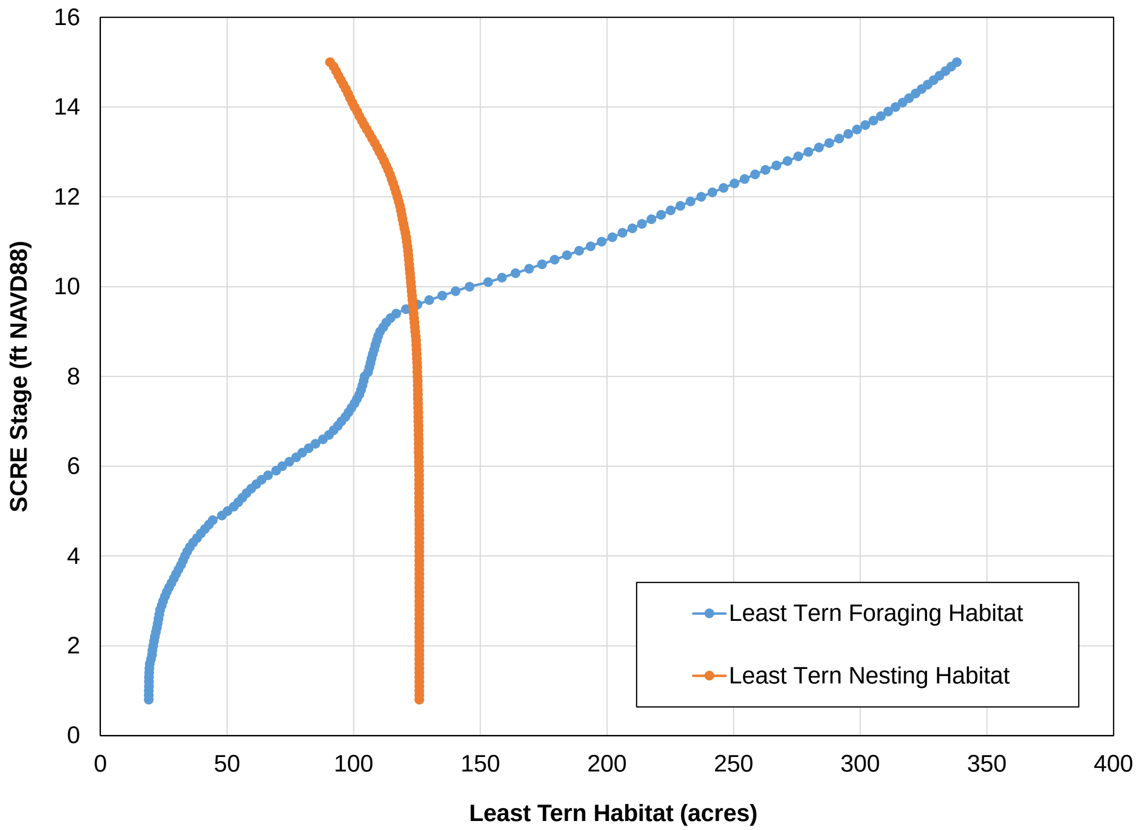


Figure 3-55. Extent of suitable least tern foraging and nesting habitat as of function of SCRE stage under current conditions.

4 WATER BALANCE EVALUATION

The Phase 3 Study water balance evaluation updates the previous water balance evaluation by incorporating new hydraulic and hydrologic data collected at previously established and new monitoring locations to better constrain modeled hydrologic variables. In response to identified data gaps from the Phases 1 and 2 Studies and stakeholder concerns, this update is intended to provide more accurate, reliable, and current estimates of the effects of various water inputs to and outputs from the SCRE on ecosystem function, with the ultimate goal of evaluating the effects of VWRF management measures on beneficial uses of the SCRE. As stated in the approved Workplan for the Phase 3 Studies, the updated water balance evaluation includes:

- Continued measurements of surface water inflows to the SCRE, including VWRF discharge
- Improved estimates of groundwater inflows and outflows between the SCRE and the underlying Semi-perched aquifer
- Improved estimate of water balance unmeasured flow by incorporating known, but unmeasured hydrologic processes into unmeasured flow equation
- Improved estimates of berm breach/closure dynamics
- Updated lagoon morphology
- Estimates of McGrath Lake inflow[19]

The water balance evaluation includes data collected specifically for the Phase 3 Study as well as data collected concurrently by others, and is based on a combination of measured and interpolated information and literature-derived empirical values. The Phase 3 Study monitoring period began for all water-balance components in January 2015 and continued through early December 2016, representing a nearly 2-year period. Presented below is the SCRE water balance for the current conditions, which roughly corresponds to water year 2015–2016. The water balance specifically spans January 13, 2015 through December 6, 2016. The results of the evaluation of VWRF management measures on beneficial uses of the SCRE are presented in Section 5).

4.1 Methods

The Phase 3 water balance evaluation relied largely on the same methods as the previous studies (Stillwater Sciences 2011). A water balance is a method for accounting for the inflows, outflows, and change in water stored in a specific reservoir over a given period (Hornberger et al. 1998). The change in water volume stored over a certain time can be expressed by the equation:

$$\frac{dV}{dt} = I - O \qquad \text{(Eqn. 4.1)}$$

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

$$V(t+1) = V(t) \pm \frac{dV}{dt} \qquad \text{(Eqn. 4.2)}$$

The reservoir volume at any time is equal to the reservoir volume from the previous time step plus the amount of inflow minus the outflow during that time step. When the reservoir is a measured quantity, the volumetric difference between successive time steps represents the total water volume gained or lost. Combining this information with measured rates of water inflow and outflow between successive time steps may be used to constrain discrete rates of water inflow and outflow for which limited data are available. Compiling these data over longer periods (i.e., many

[19] The SCRE has been historically observed to extend a narrow arm southward at high water surface elevations and appeared to connect with the outfall channel of McGrath Lake (Stillwater Sciences 2016). No evidence of a surface water connection was observed during Phase 3 monitoring, so McGrath Lake surface inflows to the SCRE were assumed be negligible.

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 4-1), which include the following:

- Direct precipitation upon SCRE lagoon—volume of inflow estimated from local rainfall recordings by VCWPD in downtown Ventura
- VWRF treated effluent discharge via the outfall channel—volume of inflow estimated from measured discharge recorded by the City of Ventura
- Santa Clara River discharge—volume of inflow estimated from computed discharge recordings by VCWPD and UWCD upstream of the SCRE
- Subwatershed runoff—volume of inflow estimated from computed discharge from runoff received in the subwatershed between Victoria Ave. and the SCRE
- Groundwater inflow—volume of inflow estimated from computed subsurface flow from Semi-perched aquifer as a function of water-table elevation and SCRE stage
- Unmeasured inflow—volume of inflow estimated from otherwise unspecified flow sources within the vicinity of the SCRE (e.g., base flow, groundwater exchanges, wave overwash), computed as a function of time during closed-mouth periods and a constant base flow term
- Subsurface inflow through beach berm—volume of inflow during closed-mouth periods estimated from computed subsurface flow as a function of tidal elevation and SCRE stage
- Surface-water inflow through beach berm (Tidal inflow)—volume of inflow during open-mouth periods estimated from computed surface-water flow as a function of tidal elevation and SCRE stage

Water outflows from the SCRE including the following:

- Direct evaporation from the SCRE lagoon—volume of outflow estimated from local evaporation recordings by VCWPD near Saticoy
- Groundwater outflow—volume of outflow estimated from computed subsurface flow from Semi-perched aquifer as a function of water-table elevation and SCRE stage
- Subsurface outflow through beach berm (Closed lagoon) —volume of outflow during closed-mouth periods estimated from computed subsurface flow as a function tidal elevation and SCRE stage
- Surface-water outflow through beach berm (Draining lagoon)—volume of lagoon outflow during open-mouth periods estimated from computed surface-water flow as a function of tidal elevation and SCRE stage

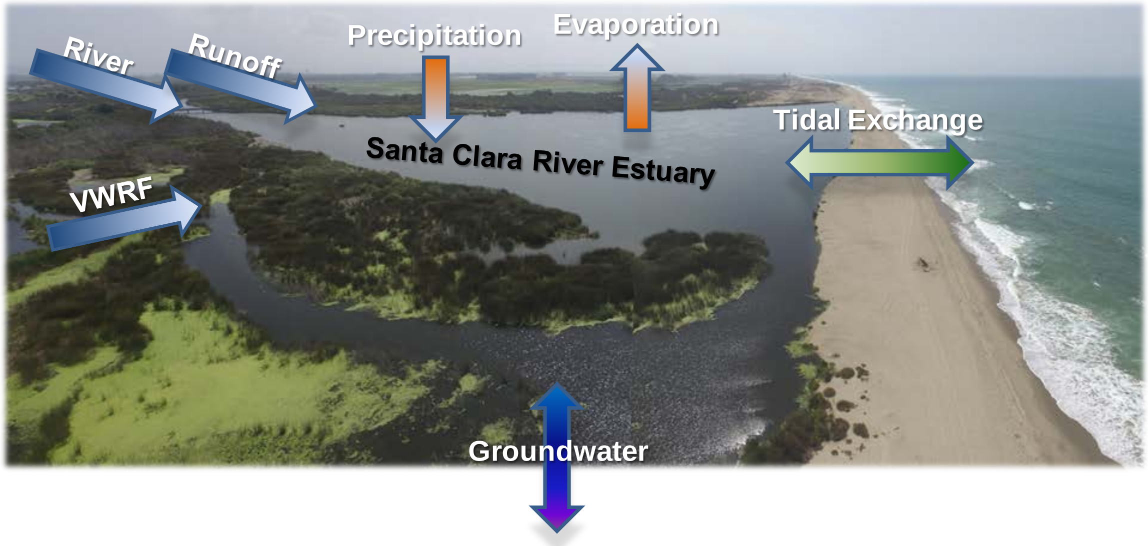


Figure 4-1. Conceptual model diagram of the dominant SCRE inflows and outflows, view looking south.

4.1.1 Lagoon and berm morphology

The SCRE water volumetric change over time and the contribution of individual inflow and outflow components to the SCRE water balance were determined combining SCRE volume measurements with calculated inflow and outflow rates. A key component of the water balance was the use of an updated elevational surface of the SCRE representing 2015–2016 conditions. As described above in Section 3.2.2., a contemporary DEM surface was generated with sufficient coverage and resolution for use in the water balance evaluation. The relationships between SCRE stage and inundated area and volume were presented in Figure 3-3.

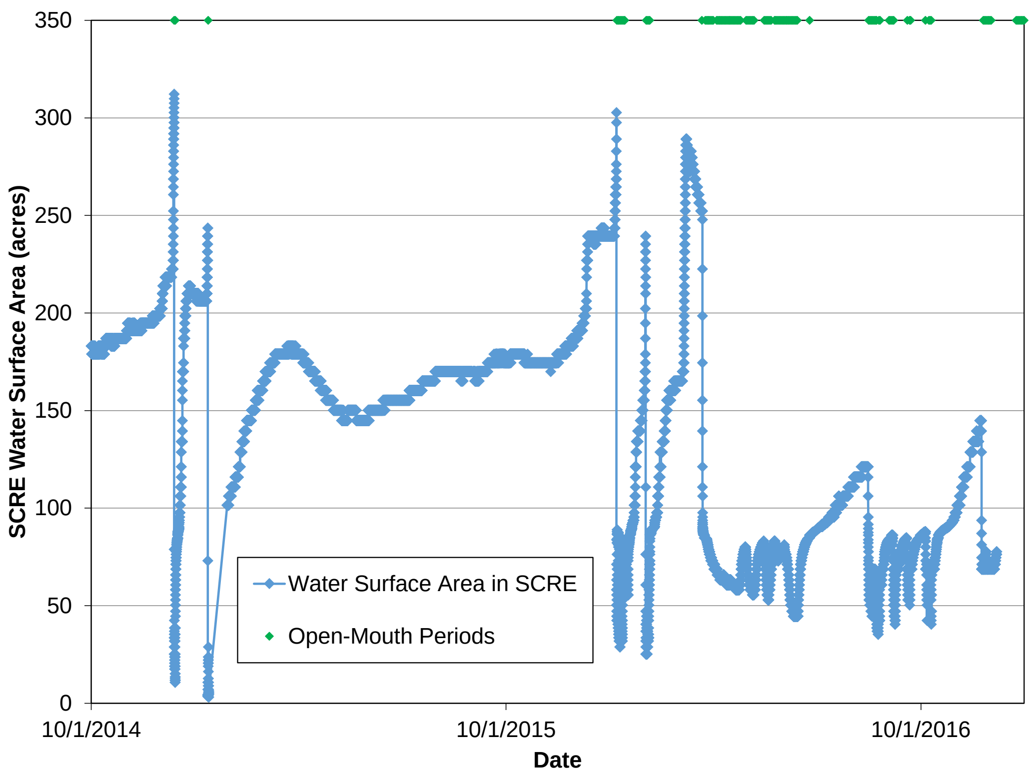
4.1.2 Monitored stage, area, and volume

As was determined during the Phase 1 Study, the quasi-equilibrium "full" stage of the SCRE is approximately 10 ft NAVD88. This elevation aligns just above the inflection point apparent in the regression curve of SCRE stage and water-surface area depicted above in Figure 3-3a. Thus, the SCRE's lagoon and associated open-water habitat type is considered fully inundated when the SCRE stage has reached 10 ft NAVD88.

The compiled SCRE stage recorded between October 1, 2014 and December 6, 2016, as introduced above in Section 3.3.5 SCRE Stage and Inundation Dynamics, was combined with the stage-area and stage-volume relationships derived from the contemporary DEM surface during open- and closed-mouth conditions (Figure 4-2). During this monitoring period, the SCRE inundated area and water volume ranged between approximately 3 and 310 acres (151,350 and 13,600,000 square feet) and 2 and 1,500 acre-feet (79,000 and 65,400,000 cubic feet), respectively. The stage associated with a "full" lagoon condition of approximately 10 ft NAVD88 corresponds to a water surface area of approximately 129 acres (5,600,000 square feet) and a water volume of 416 acre-feet (18,100,000 cubic feet). The SCRE remained at or above "full" status for approximately 62% of the time. More specifically, the SCRE remained full 90% of water year 2015 (October 2014–September 2015) and 41% of water year 2016 (October 2015–September 2016).

The stage fluctuated during the monitoring period in response to changes in inflows and outflows, and to natural and anthropogenic berm-breach events. During storm-induced breaching events, the associated SCRE stage ranged between 12.4 and 14.7 ft NAVD88 (see Table 3-4 above). Thus, for the water balance evaluation, a SCRE stage exceeding 11.2 ft and 14 ft NAVD88 were assumed to initiate berm breaching during low-flow and high-flow periods.

a)



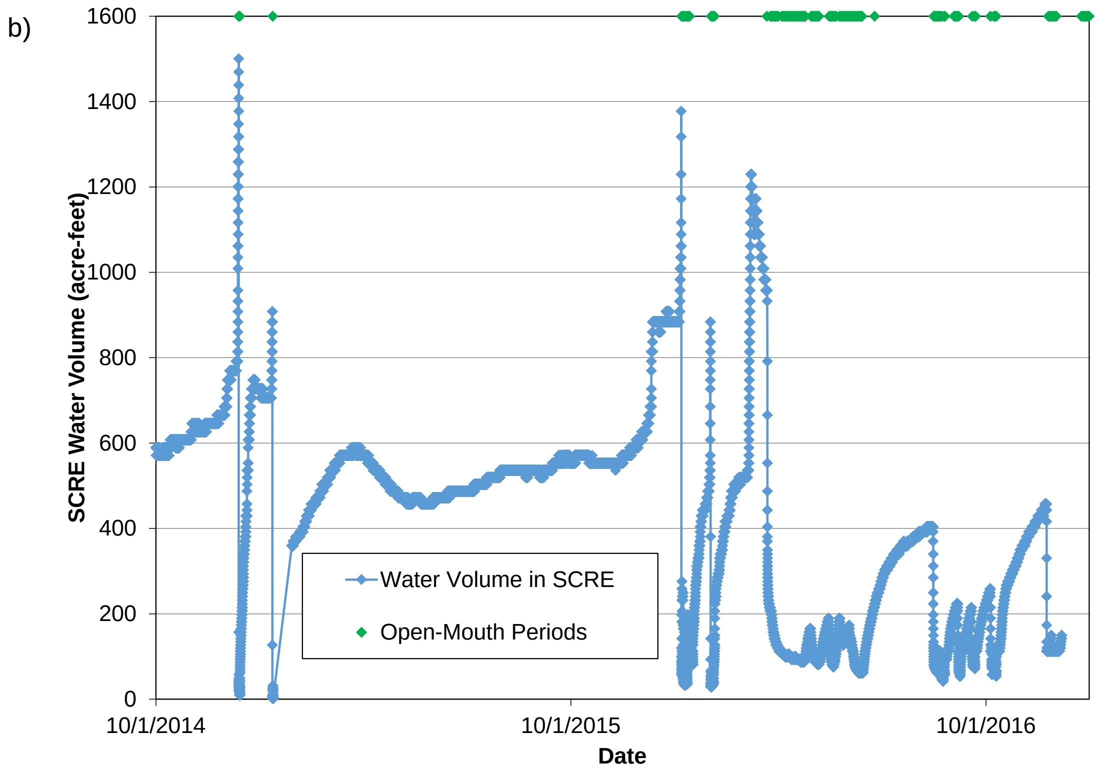


Figure 4-2. SCRE inundated area (a) and volume (b) during the Phase 3 Study period.

4.1.3 Precipitation

The volume of direct precipitation upon the water surface of the SCRE was derived from a combination of hourly precipitation recorded in downtown Ventura (VCWPD station 66E; see Figure 3-7) and the SCRE inundated area time series (see Figure 4-2a). Hourly precipitation estimates were converted to a 30-minute time step assuming a constant rate of precipitation within any given hour. The precipitation volume time series from January 2015 to December 2016 is shown in Figure 4-3. The maximum rate of precipitation input occurred on January 6, 2016 during a modest storm that initiated breaching of the beach berm the following day. Approximately 10 acre-feet of rain fell on the inundated portion of the SCRE over a 30-minute period on that day, which is equivalent to a constant inflow of 230 cfs (150 MGD).

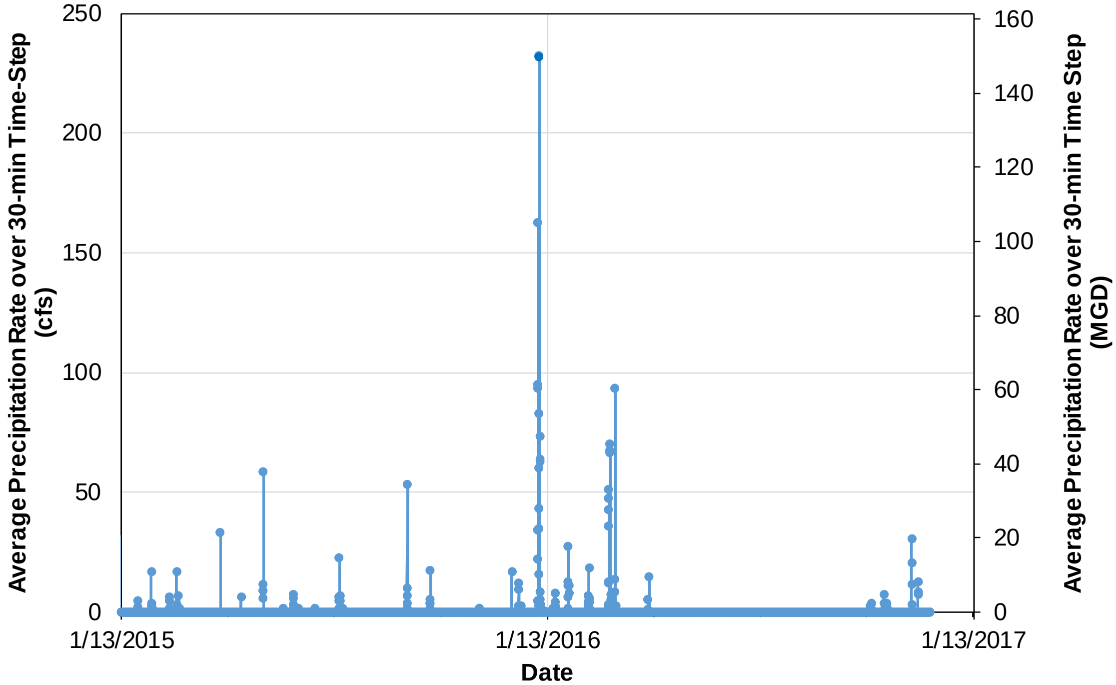


Figure 4-3. Precipitation volumetric inflow rate to the SCRE during the water balance period.

4.1.4 Evaporation

The volume of SCRE water lost to evaporation was derived from hourly pan evaporation data recorded at the El Rio-UWCD Spreading Grounds (VCWPD station 239) (see Figure 3-8) and the SCRE inundated area time series (see Figure 4-2a). Hourly evaporation estimates were converted to a 30-minute time step assuming a constant rate of evaporation within any given hour. The evaporation volume time series from January 2015 to December 2016 is shown in Figure 4-4. Greater rates of evaporation occurred during warm summer days when the SCRE was at or above its full condition. The maximum estimated evaporation rate for the water balance period was 0.17 acre-feet, which is equivalent to a constant outflow of 4 cfs (2.6 MGD).

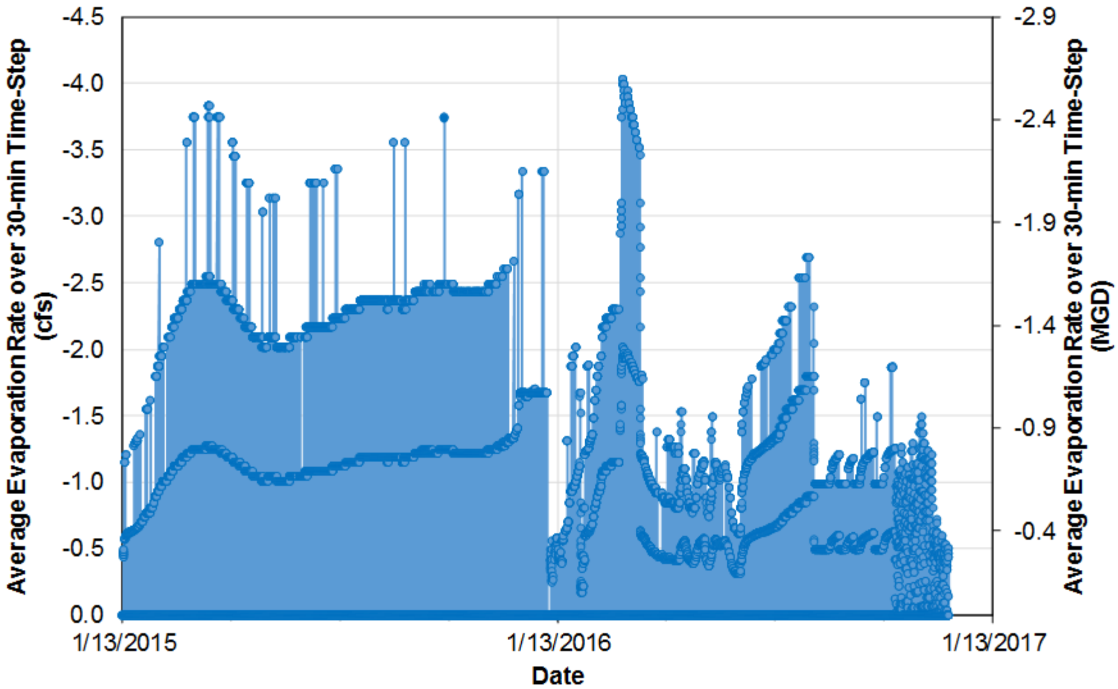


Figure 4-4. Evaporation volumetric outflow rate to the SCRE during the water balance period (note the variation in night vs. day).

4.1.5 VWRF discharge

The volume of VWRF effluent discharge to the SCRE was derived directly from the daily recordings made in the outfall channel at station D-1 (see Figure 3-11). The daily discharge estimates were converted to a 30-minute time step assuming a constant rate of inflow within any given day. The VWRF effluent volume time series from January 2015 to December 2016 is shown in Figure 4-5. The inflows ranged from approximately 6 cfs (4 MGD) to 12 cfs (8 MGD), with an average value of about 8 cfs (5 MGD). Discharge values were generally higher in the winter months and lower in the summer months.

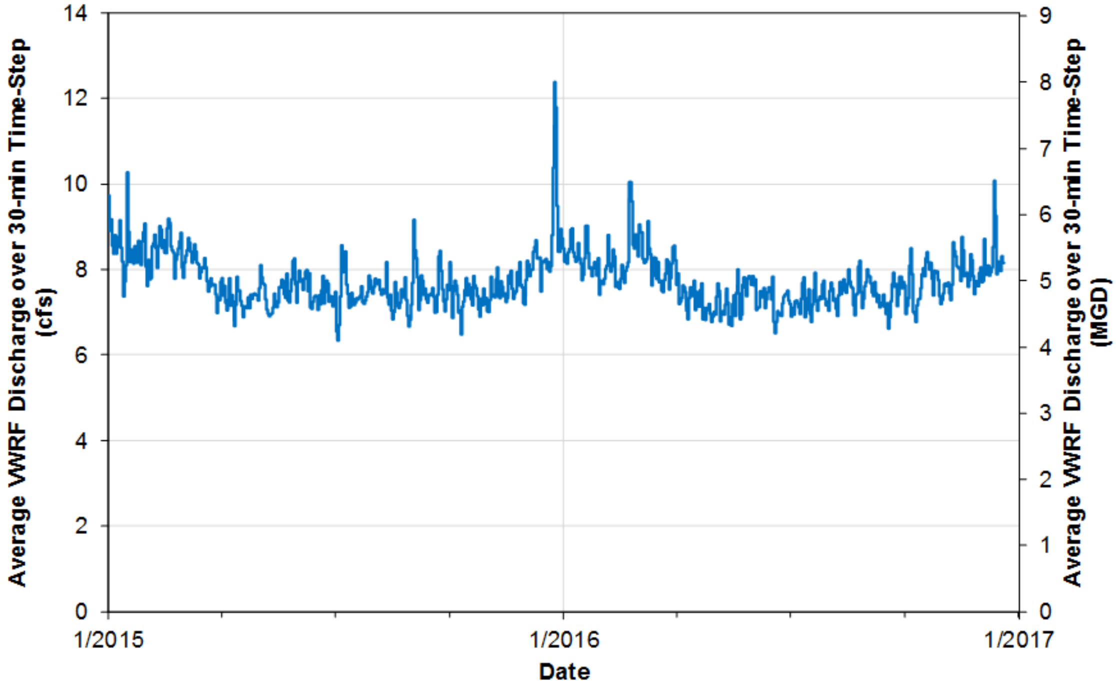


Figure 4-5. VWRF effluent volumetric inflow rate to the SCRE during the water balance period.

4.1.6 Santa Clara River discharge

The volume of Santa Clara River discharge flowing into the SCRE was determined using two information sources: daily mean discharge recordings made 1.5 miles upstream at the Victoria Ave. bridge (VCWPD station 723) during water year 2015 and recordings made 9 miles upstream immediately below Freeman Diversion (UWCD) during and since water year 2016 (see Figure 3-9 above). The use of data collected farther upstream was required due to the unavailability of data published by VCWPD at their Victoria Ave. station since September 2015. While flows measured immediately below Freeman Diversion are expected to decrease by the time they enter the SCRE due to percolation in the Oxnard Forebay reach, no adjustments were made to the flows measured immediately below Freeman Diversion because of uncertainty about the amount of percolation and the limited influence of flows on the SCRE water balance during water year 2016. The daily mean discharge estimates were converted to 30-minute time steps assuming a constant rate of inflow within any given day. The river discharge time series from January 2015 to December 2016 is shown in Figure 4-6. The inflows ranged from 0 cfs (i.e., no flow) to 560 cfs (360 MGD). Discharge values were consistently highest in the winter months and nearly absent during the other months.

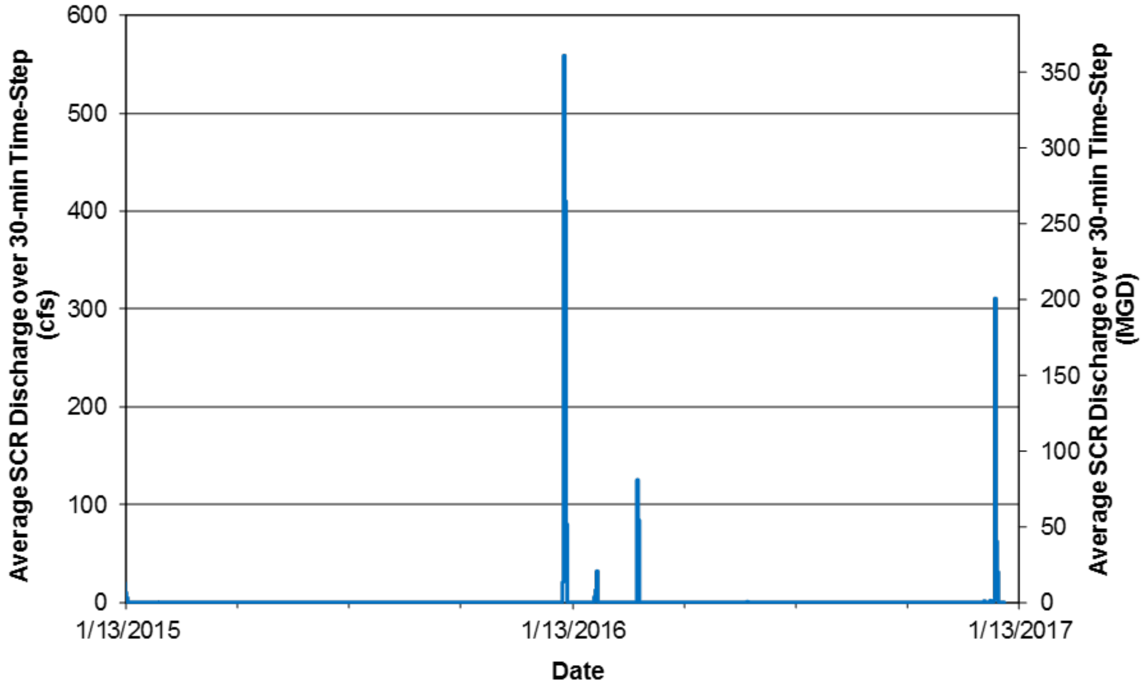


Figure 4-6. Santa Clara River volumetric inflow rate to the SCRE during the water balance period.

4.1.7 Subwatershed runoff

The SCRE receives additional surface-water inflow from the local drainage area not represented by the VCWPD gaging station at Victoria Ave, thus requiring estimation of that local runoff using the Rational Method. This method consists of a simple equation for calculating storm-induced discharge from relatively small drainage areas as a function of that area and precipitation characteristics in that area. The equation can be expressed as:

$$Q_{peak} = CIA \qquad \text{(Eqn. 4.3)}$$

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

The Phase 1 Study previously estimated values for the runoff coefficient and contributing drainage area using a GIS feature dataset that determined the total contributing area and the runoff characteristics of discrete parcels. Each unique parcel was assigned a runoff coefficient based on degree of development and impervious area extent (Dunne and Leopold 1978). The total contributing drainage area for both the north bank floodplain (2,517 acres) and south bank floodplain (874 acres) that lie downstream of Victoria Ave. is primarily composed of 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 computed for the total north bank (C=0.45) and south bank (C=0.44) floodplain areas. Rainfall intensity for storms occurring during the Phase 3 Study period was calculated using the hourly rainfall records for downtown Ventura (VCWPD station 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: (1) considers only "significant" storms where more than 1 inch was received during the water balance period; (2) no runoff would be received from contributing areas prior to and following the storm; (3) the time from peak discharge to zero discharge is twice the storm duration and is essentially the total hydrograph duration; (4) the rising and falling limbs of the storm hydrograph are linear from zero discharge to peak discharge; and (5) runoff from the north bank and south bank floodplain areas joins the lower Santa Clara River at one single confluence point. Table 4-1 presents the storm duration, intensity, and calculated peak discharge values for the three "significant" storms that occurred during the water balance period.

Table 4-1. Storm intensity and calculated peak runoff for storm events during the water balance period.

Storm event	Storm duration (hr)	Storm total (in)	Storm intensity (in/hr)	Hydrograph duration (hr)	Q_{peak} for north bank floodplain (cfs)	Q_{peak} for south bank floodplain (cfs)
January 5–6, 2016	10	1.7	0.1	20	98	34
January 6–8, 2016	22	2.4	0.1	44	63	22
March 5–6, 2016	16	1.2	0.04	32	42	14

The time series of subwatershed runoff inflow to the SCRE was created with 30-minute time steps between January 2015 and December 2016, as depicted in Figure 4-7. The inflows ranged from 0 cfs (i.e., no flow) to 130 cfs (85 MGD). Discharge values were consistently highest in the winter months and nearly absent during the other months.

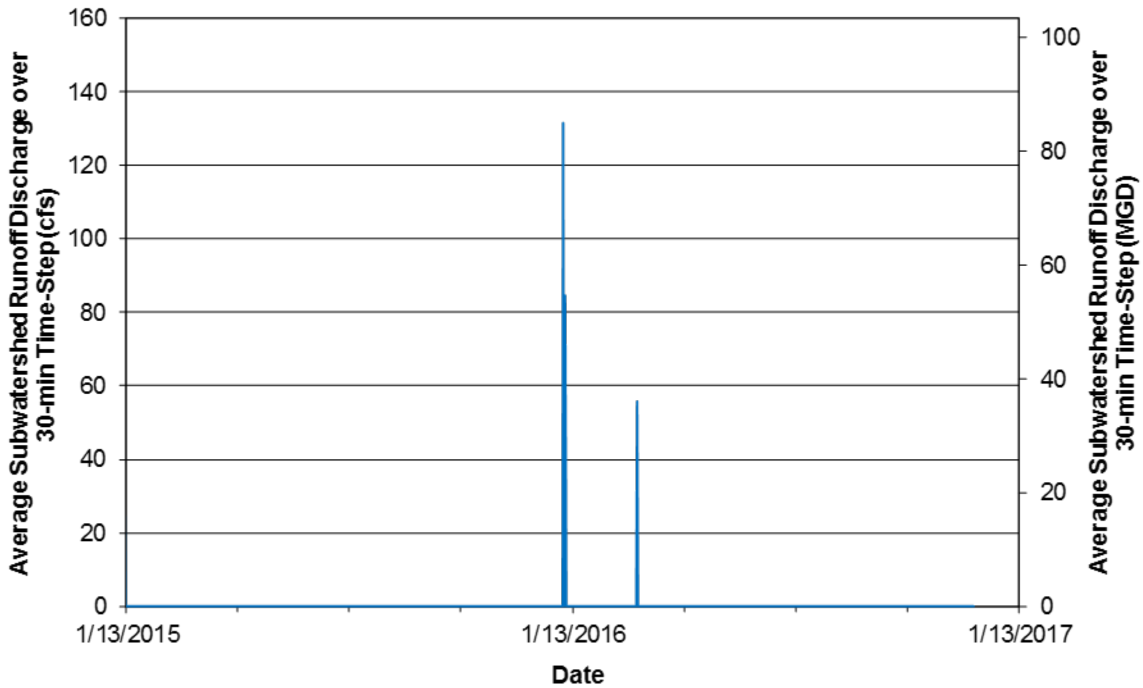


Figure 4-7. Subwatershed runoff volumetric inflow rate to the SCRE during the water balance period.

4.1.8 Groundwater flow

As discussed above in Section 3.3.4.2., the groundwater flow through the floodplain areas adjacent to the SCRE is directed towards the SCRE during open-mouth conditions and then directed away from the SCRE when the SCRE becomes full following mouth closure. Estimates of groundwater flow volumes were made for five distinct areas surrounding the SCRE:

- “Beach Berm”—beach area separating the SCRE from the ocean and having a stage-dependent seepage-face length along the western boundary of the SCRE
- “South Bank Floodplain”—floodplain area immediately south consisting of the McGrath State Beach campground and having a 3,000-ft long seepage face along its boundary with the SCRE between the beach and Harbor Blvd.
- “North Bank Floodplain–Ponds”—floodplain area immediately north consisting of the VWRF Wildlife/Polishing Ponds and having a 3,200-ft long seepage face along its boundary with the SCRE between the beach and Harbor Blvd.
- “North Bank Floodplain-West”—floodplain area north of the SCRE and lower river consisting of the Olivas Links golf course and agricultural lands, and having a 6,000-ft long seepage face along its boundary with the SCRE and lower river between Harbor Blvd. and half the distance to Victoria Ave.
- “North Bank Floodplain-East”—floodplain area north of the lower river consisting of agricultural lands and having a 6,000-ft long seepage face along its boundary with the river channel between Victoria Ave. and half the distance to Harbor Blvd.

Similar to the Phase 1 Study methods, groundwater flow volumes were estimated for the Phase 3 Study using the Darcy's Law equation and cross-sectional area of the seepage face. Darcy's Law combines hydraulic conductivity (i.e., the capacity of sediment to transmit water) and hydraulic gradient (i.e., potentiometric slope) to calculate flow velocity, and is expressed as:

$$V = -K\left(\frac{dh}{dl}\right) \quad \text{(Eqn. 4.4)}$$

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 gradients were calculated at 30-minute time steps following the methods described above in Sections 3.3.4.2 Groundwater-Flow Dynamics and 3.3.4.3 Beach Berm Seepage. Hydraulic conductivity (K) estimates were derived from the particle size data derived from samples taken at various depths at the three south bank floodplain monitoring wells (GW-1, GW-2, GW-3) and lithologic descriptions recorded for all monitoring wells, and the Hazen equation for determining the hydraulic conductivity (K) of sandy soils, which is expressed as:

$$K = C\left(D_{10}\right)^2 \quad \text{(Eqn. 4.5)}$$

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. Computed hydraulic conductivity stratigraphic layers present within the five groundwater-flow source areas are presented in Table 4-2.

Table 4-2. Estimated hydraulic conductivity of stratigraphic layers in groundwater-flow sources to the SCRE.

Stratigraphic Layers	Hydraulic Conductivity (*K*) of Groundwater Flow Sources				
	Beach Berm (ft/day)	**South Bank Floodplain (ft/day)**	**North Bank Floodplain – Ponds (ft/day)**	**North Bank Floodplain – West (ft/day)**	**North Bank Floodplain – East (ft/day)**
Surficial Layer	N/A (undifferentiated from Upper Layer)		1 (Pond lining)	N/A (undifferentiated from Upper Layer)	
Upper Layer	150	140	30	10	10
Lower Layer	30	30	10	30	10

The groundwater flow velocities (K) were combined with area of the seepage face to determine groundwater flow volume at 30-minute time steps. The seepage-face area, or flow area, was derived from the length and depth of the seepage face. The lengths were discussed above. The depth of the flow field utilized the estimated base elevation of the Semi-perched aquifer, which is assumed to be approximately -60 ft NAVD88 (Turner 1975). The top of the flow field was estimated from the measured water table elevation recorded in the monitoring wells.

The groundwater flow estimates were computed at 30-minute time steps as a product of flow velocity and seepage-face area. The groundwater flow time series for the five source areas from January 2015 to December 2016 are shown in Figure 4-8 through Figure 4-12. Describing the following components:

- **SCRE Mouth Berm Outflow:** Subsurface flow through the beach berm were almost exclusively directed westward toward the ocean, thus representing outflow of water from the SCRE (Figure 4-8). The estimated outflow values ranged up to -16 cfs which coincided with the maximum SCRE stage and “full” extent on March 8, 2016.
- **South Bank Floodplain (GW-1 through GW-3):** Groundwater flow through the south bank floodplain varied in direct response to SCRE stage, with estimated values ranging between approximately -5 cfs (outflow) and 5 cfs (inflow) during closed-mouth and open-mouth periods, respectively (Figure 4-9).
- **South Bank Floodplain, Upstream (GW-6 and GW-7):** Because groundwater gradients in the riverine portion of south bank floodplain upstream nearest Bailard landfill were consistently to the south and no ponded SCRE water was available for outflow, no outflow estimates were developed for this source.
- **North Bank Floodplain—Ponds (GW-12 through GW-15):** Groundwater inflows from the VWRF Wildlife/Polishing Ponds were estimated to range between 0.7 and 1.6 cfs (Figure 4-10).
- **North Bank Floodplain—West (GW-8 through GW-11):** Groundwater flows through the western half of the north bank floodplain were primarily directed away from the SCRE and lower river due to the prolonged duration of closed-mouth, “full” conditions during the monitoring period, particularly during 2015. The inflows and outflows through this area ranged between 0.2 and -0.3 cfs, respectively (Figure 4-11).
- **North Bank Floodplain—East (GW-4 and GW-5):** The groundwater flow from the eastern half of the north bank floodplain was estimated to consist exclusively of inflows, ranging between 0.07 and 0.1 cfs (Figure 4-12), thus constituting the smallest fraction of water inflow to the SCRE.

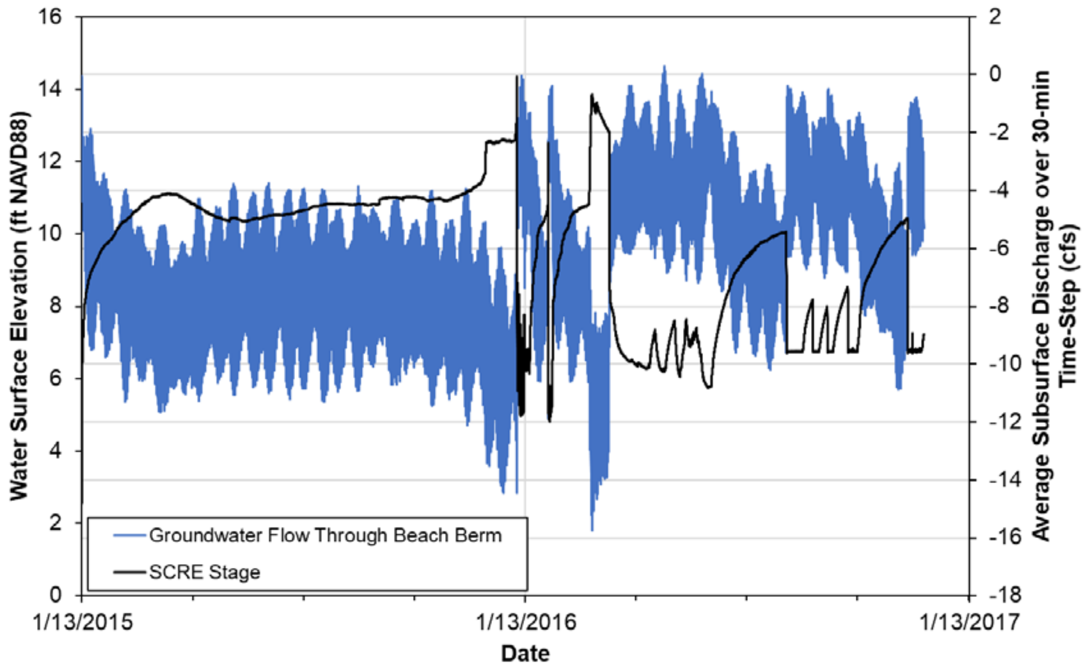


Figure 4-8. Subsurface volumetric inflow (positive values) and outflow (negative values) rates through the beach berm during the water balance period.

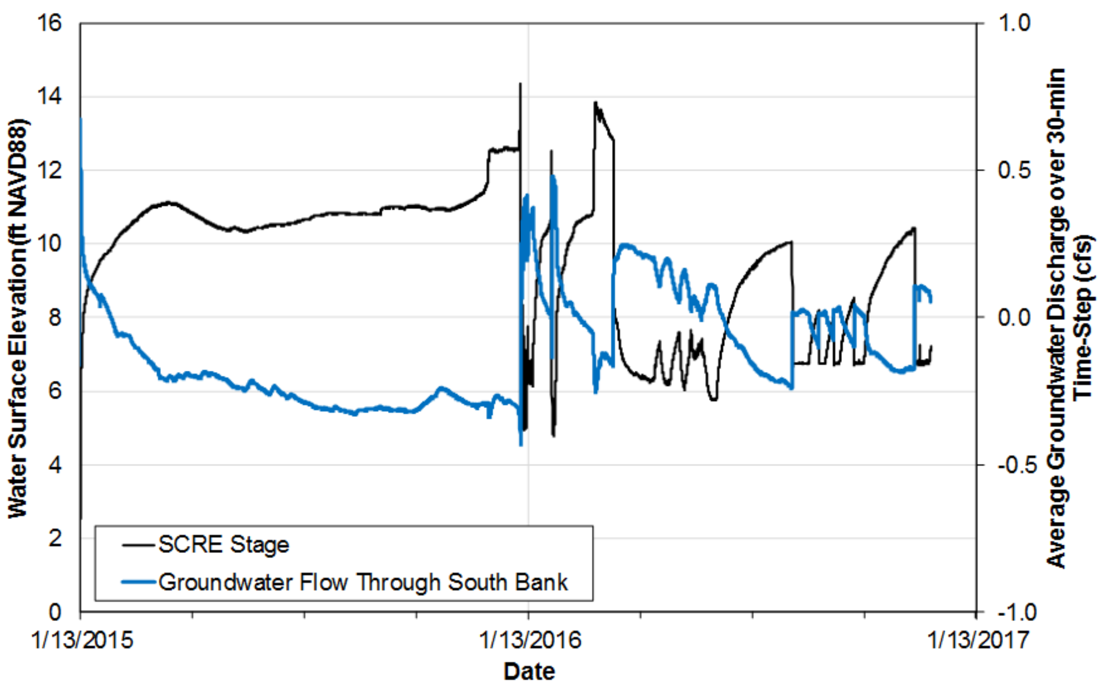


Figure 4-9. Groundwater volumetric inflow (positive values) and outflow (negative values) rates through the south bank floodplain during the water balance period.

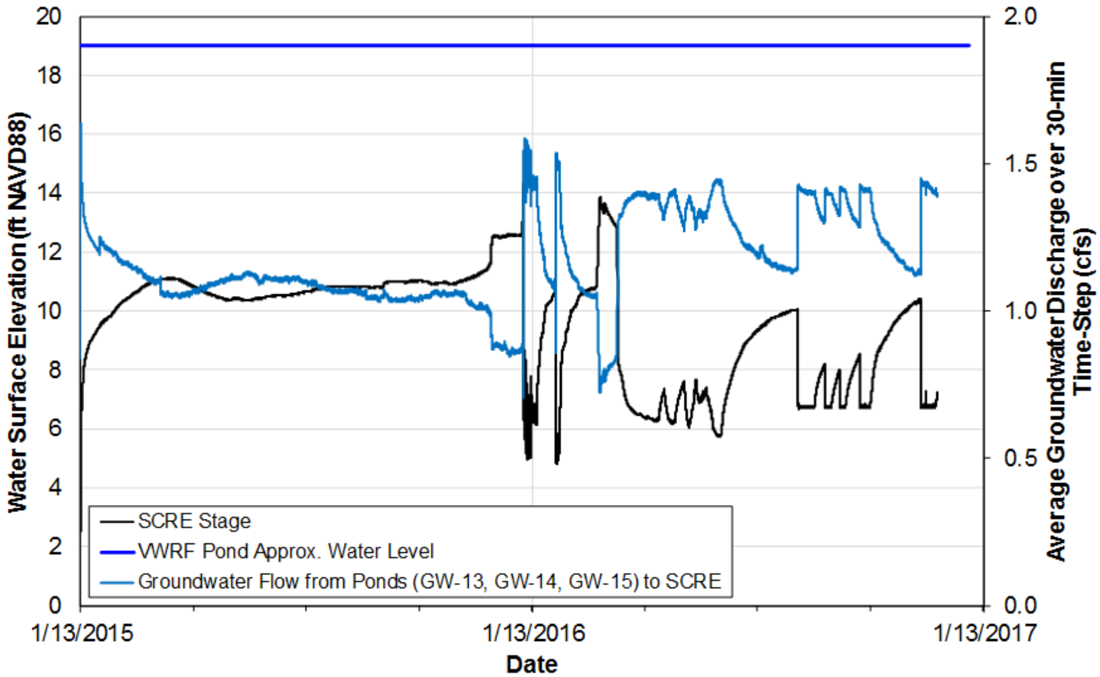


Figure 4-10. Groundwater volumetric inflow (positive values) rates from the VWRF Wildlife/Polishing Ponds through the north bank floodplain-ponds area during the water balance period.

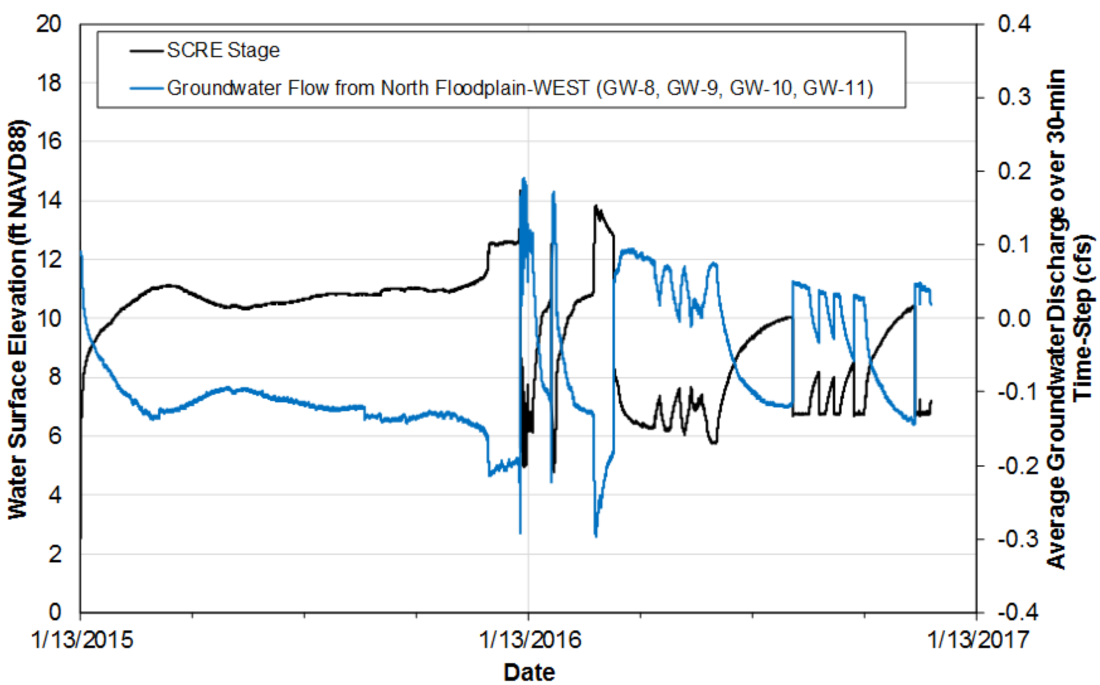


Figure 4-11. Groundwater volumetric inflow (positive values) and outflow (negative values) rates through the north bank floodplain-west area during the water balance period.

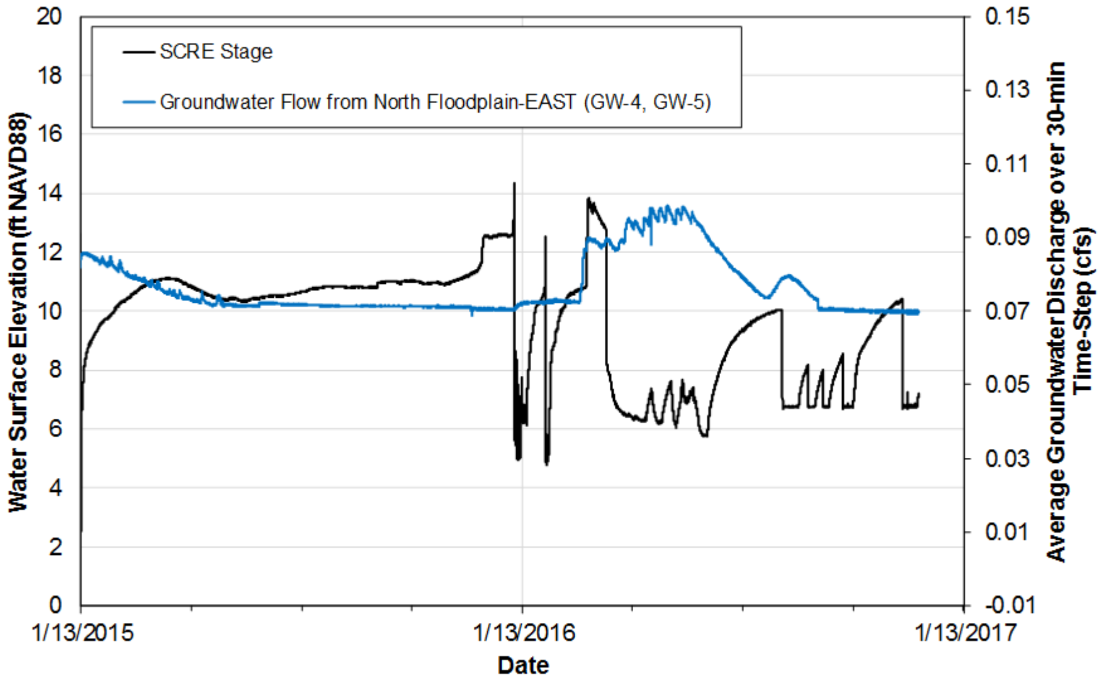


Figure 4-12. Groundwater volumetric inflow (positive values) and outflow (negative values) rates through the north bank floodplain-east area during the water balance period.

4.1.9 Unmeasured flow

Following a similar approach initially employed during the Phase 1 Study, a certain fraction of the inflows to the SCRE that remained unmonitored during the present study required estimation to complete the SCRE water balance. Unmeasured flow is assumed to be primarily a combination of base flow, wave overwash, and bank storage groundwater. A portion of the unmeasured flow is assumed to be base flow because during both Phase 1 and Phase 3 monitoring, streamflow was observed to be perennial in the Santa Clara River channel downstream of the Victoria Ave. bridge. In Phase 1, the amount of flow that enters the Santa Clara River channel downstream of the Victoria Ave. bridge and upstream of the Harbor Blvd. bridge was estimated to be 2.1 cfs from a comparison of surface flow measurements made at the two bridges during November before the first rain of the season. During July 2017, flow in the channel was measured between 0.08 cfs and 0.36 cfs between the Victoria Ave. bridge and the upstream extent of the SCRE with flow increasing in the downstream direction. While the groundwater component of the water balance accounts for groundwater contributions from the floodplains adjacent to the channel, the part of the unmeasured flow accounts for subsurface flow entering through the Santa Clara River channel between the Victoria Ave. bridge and the upstream extent of the SCRE.

Another assumed source of unmeasured flow is attributed to the wave overwash that constructs and overtops the beach berm at high tide during closed-mouth periods. Wave overwash into estuaries occurs during periods when the wave height at the foot of the berm blocking the estuary exceeds the berm height. Models of wave overwash (see Laudier et al., 2011 for a comparison of various models) predict the flux of water into the estuary as a nonlinear function of the wave height at the base of the berm, scaled by the wave height relative to height of the berm, times the length of the berm being overtopped. Because the flow from overwash is non-linearly dependent on the height difference between the wave and the berm, overwash is larger as seasonal high tides and large waves due to storms increase wave height or when the berm is lower, particularly as the berm is rebuilding after a breach.

In the SCRE, wave overwash is expected to strongly depend on the elevation of the berm. Berm elevations vary based on both the wave energy delivering sediment and the migration and plugging of the SCRE outlet. Rapid changes to the local berm height are likely to occur as sand deposits in the outlet during plugging leading to variations in overtopping along the length of the berm with lower elevation portions more susceptible to overtopping than higher elevation portions. The time required for the outlet to seal depends on the discharge from the river and the delivery of sand to the breach by waves (a function of wave energy) (Behrens et al., 2013). Predictions of wave overwash also non-linearly depend on the berm height at the time of large waves, which can evolve quickly when a breach is sealing. Repeated observations in the Russian River estuary found that it takes several days for the mouth breach to fill (Behrens et al., 2013). During the sealing of a breach, wave overwash is anticipated to be a significant contributor to the water balance in the SCRE, but it is not possible to explicitly model the contribution of overwash because the spatial and temporal variations in berm height, especially during the initial sealing of a breach, is unknown. As such, wave overwash is included in the unmeasured flow component of the water balance.

Another portion of the unmeasured flow component is assumed to be from bank storage groundwater that flows into the SCRE when the water surface elevation decreases result in groundwater stored in the variably inundated margins of the SCRE to drain into the estuary. The spatially distributed nature and unavailability of data to specifically quantify bank storage groundwater results in it being grouped into the unmeasured flow component of the water balance. While the bank storage groundwater would begin draining once a breach lowers the estuary water surface, the amount of flow attributed to these bank storage groundwater sources is anticipated to be most significant once a breach closes then decreasing over time as the SCRE refills.

The conditions most likely to lead to large overwash contributions (a recently plugged mouth outlet), would also lead to large bank storage groundwater contributions. While temporally detailed measurements of salinity coupled with stream discharge measurements have been used to determine wave overwash into an estuary (see Laudier et al., 2011), there is not sufficient data in the SCRE to differentiate the proportions of overwash and bank storage groundwater.

A general equation to represent the unmeasured flow in the water balance was developed from base flow measurements in the Santa Clara River channel, unmeasured flow estimates during closed- and open-mouth periods, and expectations of wave overwash and bank storage groundwater patterns. Unmeasured flow attributed to base flow was assumed to be a constant since there was sufficient data to identify a base flow existed, but not enough data to determine any temporal variations in base flow. After a berm breach seals, the unmeasured flow in the water balance exponentially decreases suggesting the other portion of unmeasured flow can be represented by an exponential function. Both the wave overwash and bank storage groundwater processes would contribute the most flow immediately after a breach seals, then would decrease over time as the berm height builds up and SCRE water elevation increases. There is insufficient data to quantify the proportion of the exponentially decreasing unmeasured flow to either wave overwash or bank storage separately, so both are grouped together into a single exponential term in the general equation for unmeasured flow. The general equation for unmeasured flow, $Q_{unmeasured}$ (cfs), is:

$$Q_{unmeasured} = Q_{base\ flow} + Q_{og}\ e^{(-k\ t_{close})} \quad \text{(Eqn. 4.6)}$$

where $Q_{base\ flow}$ (cfs) is the component of the unmeasured flow from base flow, Q_{og} (cfs) is an empirically determined maximum flow from the combined wave overwash and groundwater portion of the unmeasured flow, k (1/day) is an empirical constant representing the rate the overwash and groundwater contribution to the unmeasured flow decreases, and t_{close} (day) is the time from when a breach closes.

The $Q_{base\ flow}$ was assumed to be a constant value of 0.36 cfs during the entire period of the water balance based on the measurement of flow in July 2017. The maximum flow from the combined wave overwash and groundwater portion of the unmeasured flow, Q_{og}, and the rate of decrease in the overwash and groundwater contribution, k, were determined by optimizing the water balance Nash-Sutcliffe Efficiency (NSE), percent bias, root mean square error-observations standard deviation ratio (RSR), and visual fit between the measured and computed SCRE volume. The Q_{og} varied seasonally with higher Q_{og} during Fall and Winter than in Spring and Summer periods. No breaching occurred during Spring 2015 through Fall 2015 so a specific Q_{og} could be determined during these periods. The estimated Q_{og} from other periods was used to calculate an average maximum flow from wave overwash and groundwater, Q_{og_avg}, that was then assigned to the unmeasured flow equations from Spring 2015 through Fall 2015. The rate of decrease in the overwash and groundwater contribution was estimated as 0.2 day^{-1} indicating the majority of the overwash and groundwater contribution occurs within the first 5 days of a berm breach sealing which is consistent with observations of the time it takes for mouth closure in the Russian River (Behrens et al., 2013). Table 4-3 summarizes the seasonal unmeasured flow equations and the average unmeasured flow discharge rates during the water balance period. As would be expected, the maximum unmeasured inflows occurred during periods following mouth breaching and during the initial re-filling of the SCRE (Figure 4-13).

Table 4-3. Unmeasured flow relationships calculated average discharge rates during the water balance period.

Season: period duration	% of period duration with closed-mouth conditions	Unmeasured flow relationship over 30-minute time-step (cfs)	Average discharge over 30-minute time-step (cfs)
Winter 2015: 1/13/2015–3/31/2015	100% (Filling Period)	$Q_{unmeasured}$ = 0.36+ 13,000*exp(-0.2*t_{close})	0.82
Spring 2015: 4/1/2015–6/30/2015	100% ("Full" No Breach Period)	$Q_{unmeasured}$ = 0.36+ 24,200*exp(-0.2*t_{close})	0.36
Summer 2015: 7/1/2015–9/30/2015	100% ("Full" No Breach Period)	$Q_{unmeasured}$ = 0.36+ 24,200*exp(-0.2*t_{close})	0.36
Fall 2015: 10/1/2015–12/31/2015	100% ("Full" No Breach Period)	$Q_{unmeasured}$ = 0.36+ 24,200*exp(-0.2*t_{close})	0.36
Winter 2016: 1/1/2016–3/31/2016	75% (Storm and Man-made Breaching and Re-filling Period)	$Q_{unmeasured}$ = 0.36+ 65,000*exp(-0.2*t_{close})	4.29

Season: period duration	% of period duration with closed-mouth conditions	Unmeasured flow relationship over 30-minute time-step (cfs)	Average discharge over 30-minute time-step (cfs)
Spring 20165: 4/1/2016–6/30/2016	31% (Man-made Breaching and Re-filling Period)	$Q_{unmeasured} = 0.36 + 13{,}000 \cdot \exp(-0.2 \cdot t_{close})$	1.87
Summer 2016: 7/1/2016–9/30/2016	54% (Man-made Breaching and Re-filling Period)	$Q_{unmeasured} = 0.36 + 5{,}000 \cdot \exp(-0.2 \cdot t_{close})$	0.77
Fall 2016: 10/1/2015–12/6/2016	54% (Storm and Man-made Breaching and Re-filling Period)	$Q_{unmeasured} = 0.36 + 26{,}000 \cdot \exp(-0.2 \cdot t_{close})$	1.88

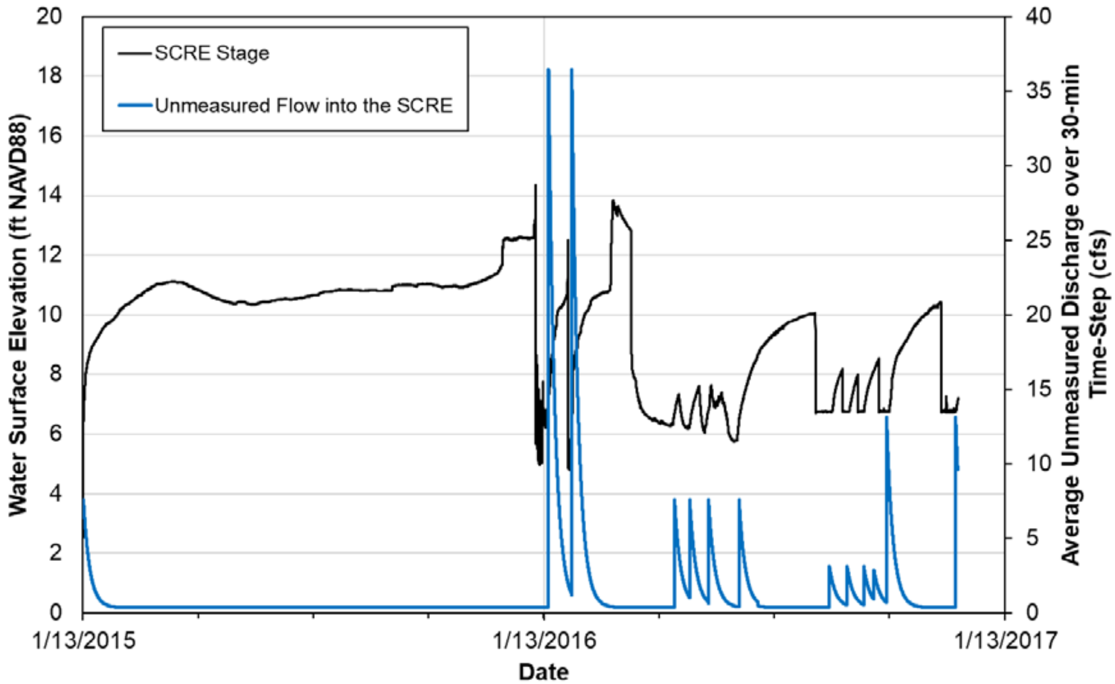


Figure 4-13. Unmeasured flow estimates during the water balance period.

4.1.10 Surface-water flow across mouth berm

The surface-water flow volume in and out of the SCRE during open-mouth periods was calculated at 30-minute time steps based on the difference between the measured water volume (derived from the SCRE stage-volume relationship) and the sum of all other water balance components described above. Therefore, when the sum of all components was less than the measured SCRE volume, there was surface flow from the ocean into the SCRE and when the sum of components was greater than the measured SCRE volume, there was surface flow out of the SCRE. That is, flow was directed into the SCRE during those moments when the tidal elevation exceeded the SCRE stage, while flow was directed out of the SCRE when tidal elevation was less than the SCRE stage (Figure 4-14).

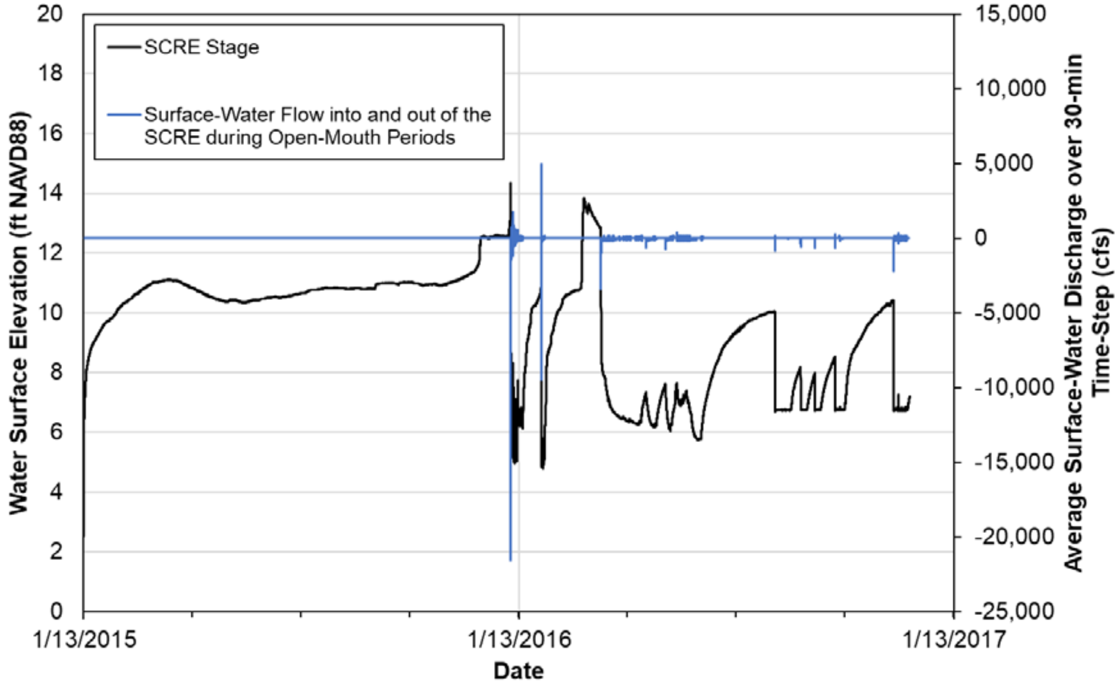


Figure 4-14. Surface-water volumetric inflow (positive values) and outflow (negative values) rates during open-mouth conditions of the water balance period.

4.2 SCRE Water Balance for WY 2015-2016

4.2.1 Calibration of calculated SCRE water balance

The model calibration process utilized empirical relationships described in Section 4.1 (including SCRE stage-dependent groundwater flow and unmeasured flow) as well as measured data (including river discharge, effluent discharge from the VWRF, SCRE stage, and local tidal elevation) in order to compare modeled and measured estuary stage and volume. Because groundwater flows were not constrained by measured data, water balance calibration was accomplished by selection of reasonable values of hydraulic conductivity in various soil mixtures (i.e., relative amounts of clay, silt, and sand) around the SCRE perimeter (Table 4-2), and then the unmeasured flow amount was estimated for the sum of all water balance components to fit to the observed stage versus volume relationship during SCRE filling periods. Adjustments were iteratively made to the empirical unmeasured flow Q_{og} to optimize the overall water balance model NSE, percent bias, RSR, and visual fit. The optimum value for the NSE is 1, percent bias is 0, and RSR is 0 with acceptable values varying depending on the time-scale of the model and the parameter being modeled for percent bias. Moriasi et al. (2007) classified model performance "very good" (the highest performance category) when NSE was greater than 0.75, percent bias was +/- 10% for streamflow, and RSR was less than 0.5. The final model performance statistics for the entire water balance period were 0.87 for the NSE, -3.4% for the percent bias, and 0.36 for the RSR. A secondary calibration step involved adjustments of the assumed breaching elevation thresholds to match observed stage data during natural breaching events. The comparison of measured SCRE volume as derived from the stage-volume relationship and the summed volume components is shown in Figure 4-15.

Comparison of the computed water-balance volume time-series to the measured volume time-series reveals a close relationship during periods of SCRE filling following breach events throughout the evaluation period, which lends confidence to the use of the water-balance components and the equations developed to estimate the unmeasured flow component. Compiling the total inflow and outflow volumes from each component illustrates the relative contributions (Figure 4-16). The consistently largest *inflow* source during the relatively dry water years of the water balance period was the VWRF effluent, while smaller inflow sources (listed in descending order) were river discharge, flow through the open mouth (i.e., tidal inflow), unmeasured flow, groundwater flow from the VWRF ponds, subwatershed runoff, precipitation, and groundwater flow from the eastern half of the north bank floodplain. Approximately 73% of the unmeasured flow occurs during the refilling of the estuary with the remaining approximately 27% of the unmeasured flow attributed to baseflow contributions. The consistently largest *outflow* source was flow through the open mouth (i.e., rapid emptying of lagoon at onset of a breach event), followed in descending order by groundwater flow (seepage) through the beach berm, evaporation, groundwater flow through the south bank, and groundwater flow through the western half of the north bank floodplain.

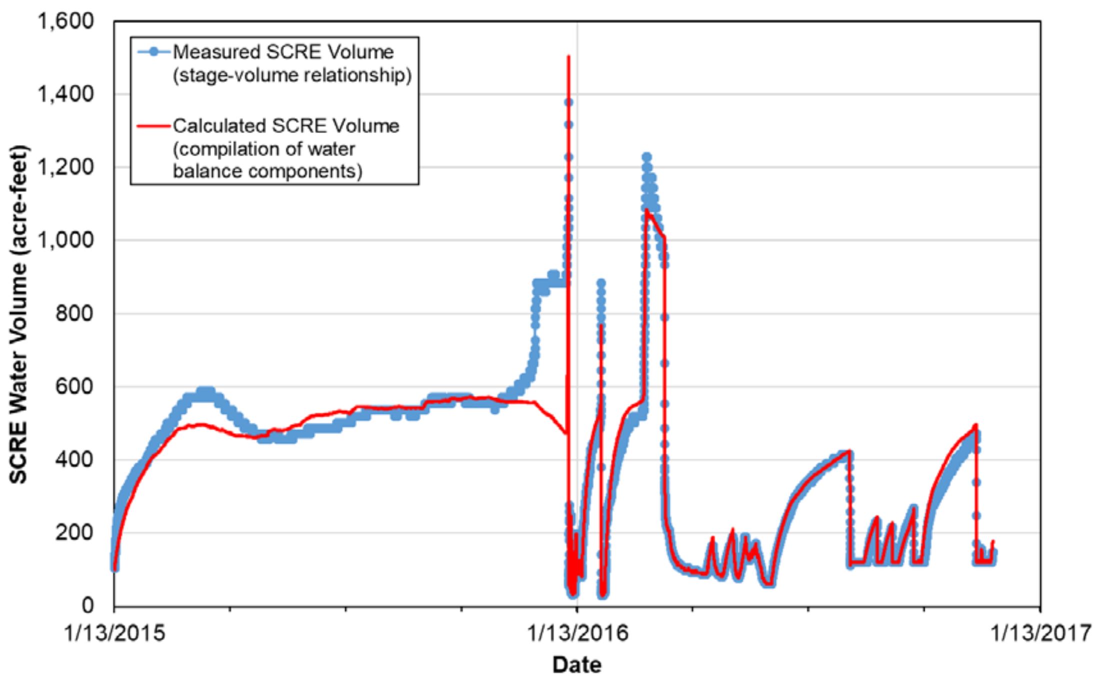


Figure 4-15. Comparison of measured SCRE inundated volume and calculated volume derived from compiling inflow and outflow rates during the water balance period.

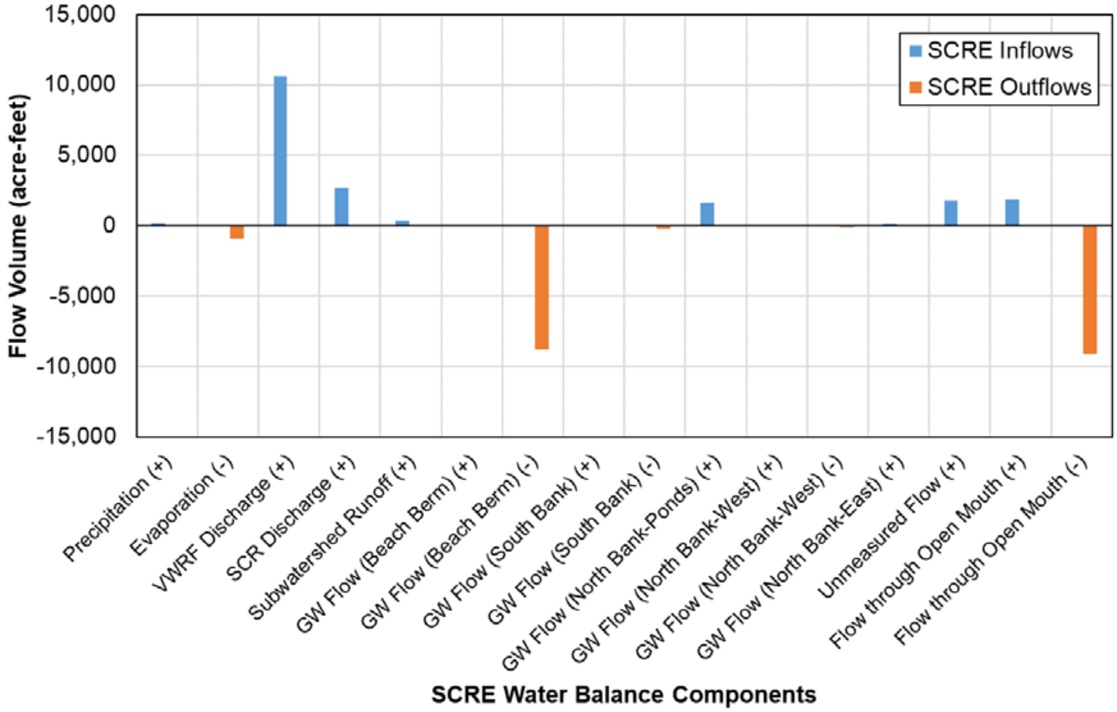


Figure 4-16. Total SCRE inflow (positive) and outflow (negative) volume by component during the water balance period.

There are two distinct periods when the measured and computed volumes do not satisfactorily match. The magnitude of SCRE volume computed from the water-balance components appears underpredicted between mid-February 2015 and the end of April 2015, and then underpredicted from late November 2015 until January 5, 2016 when the berm breaches. During the first "underpredicted" period, the measured volume increases more rapidly than the water-balance volume until late March 2015 after which the measured volume decreases until it becomes similar to the water-balance volume again. While the overall trend of increase and decrease is similar in both the measured and computed volume during this period, the magnitude of the volume variations is greater in the measured volume. Thus, it appears that the actual measured SCRE volume during the first "underpredicted" period was influenced by a process not considered by the water balance evaluation between about mid-February 2015 and the end of April 2015.

Examination of aerial photography taken during this period (Figure 4-17) reveals a large sand deposit along the ocean-side of the northern end of the SCRE berm. The material was placed in this location as part of permitted maintenance dredging activities for the Ventura Harbor, resulting in a deposit along 2,000 feet of the SCRE beach berm and projecting seaward approximately 400 feet. The deposit is present in imagery taken in February 2015 and is assumed to have eroded by wave action between mid-February 2015 and late-April 2015. The coincident timing of this deposit with the unaccounted rise and fall of in SCRE water volume suggests that the deposit may have temporarily increased the length of the seepage pathway through the mouth berm. An increase in the berm width would impeded SCRE outflows, decrease outflow through the berm, and account for the more rapid increase in SCRE volume during the early part of 2015. As the deposit was gradually eroded, the effective seepage path shortened and the estimated SCRE volume approached the measured volume (Figure 4-15).

The second distinct period when the measured and computed volumes do not adequately match is referred to herein as the second "underpredicted" period, which occurred between mid-November 2015 and January 5, 2016. Here the measured volume exhibited an exponential increase while the calculated volume continued at a steady (flat) level until December 12, 2015 when it began to steadily decline. Between November 15 and December 12, 2015 there was no recorded rainfall, river flow, or increase in VWRF discharge, yet the SCRE experienced significant filling. No aerial photography was identified to corroborate similar maintenance dredging deposit during this period as in early 2015. An alternative explanation of the stage increase is related to wave overwash of the SCRE closed mouth berm during exceptionally high tides during this period. While the unmeasured flow was designed to account for wave overwash during sealing of the mouth berm, it did not account for any closed mouth berm wave overwash since detailed measurements of berm height were unavailable. Measured high-tide levels reached up to about 7 ft NAVD88 during late November and early December, with the tidal elevation on November 26, 2015 exceeding 7.2 ft NAVD88—the highest level, or "extreme tide," recorded during the entire Phase 3 Study period. For reference, the estimated mean higher high tide during the Phase 3 Study period is 5.4 ft NAVD88.

While water quality sondes were not deployed during the time in question, continuous measurements of conductivity in the weeks leading up to November 23, 2015 show specific conductivity measurements < 2,500 uS/cm, while measurements following the redeployment of the sondes on December 17, 2015 show conductivity >10,000 uS/cm that persisted until a January 5, 2016 storm event associated with a complete draining of the SCRE (Figure 4-15). It is therefore assumed that inflow to the SCRE from wave overwash at high tide occurred between late-November and mid-December 2015 adding a significant contribution to the measured volume between November 25 and December 12, 2015. The difference between the measured and calculated SCRE volumes continued until a storm occurred on January 5, 2016 resulting in immediate filling of the SCRE and breaching of the beach berm the following day. The measured and calculated SCRE volumes closely match during the remainder of this breach event, and thereafter.

Figure 4-17. Aerial image of Ventura Harbor maintenance dredging deposits used for beach replenishment during the "overpredicted" period in the water balance.

As a final calibration step, the water-balance model was then run to adjust the elevation thresholds for berm breaching and closure. This process focused on three periods between January 2015 and December 2016 dictated by the dominant berm-breaching mechanism: (1) natural breaching period between January 13, 2015 and March 6, 2016 which lacked known man-made influences on berm breaching; (2) man-made breaching period “part A” between June 14 and August 23, 2016 which consisted of re-filling of the lagoon following a series of man-made breaches initiated on March 21, 2016, and breaching of the berm on 8/15/2016; and (3) man-made breaching period “part B” between October 11 and December 6, 2016 which consisted of re-filling of the lagoon following another series of man-made breaches initiated in early September 2016, and breaching of the berm on November 23, 2016. Figure 4-18 presents the results of the water-balance model runs for these three periods. The following conditions were applied and thresholds were determined during these model runs:

- The model considers hydrologic-component inputs and outputs at 0.5-hr time steps over the model period.
- The minimum measured SCRE stage when the berm is open varies during the monitoring period between 2.5 and 6.5 ft NAVD88. The average SCRE water surface elevation when the berm was open during water year 2015 was 4.5 ft NAVD88, but progressively increased in water year 2016. The model runs between 2015 and February 2016 begin closed-mouth conditions with a SCRE stage of 4.5 ft NAVD88. Model runs from February to June 2016 begin closed-mouth conditions with a SCRE stage of 4.8 ft NAVD88, while runs from June through December 2016 begin closed-mouth conditions with a SCRE stage of 6.0 ft NAVD88.
- For the SCRE mouth to open after being closed, the modeled berm status must be open for at least 24 hours.
- For the SCRE mouth to close after being open, the modeled berm status must be closed for at least 24 hours.
- Breaching stages under low and high inflow conditions were determined from SCRE stage and berm-breaching observations collected during the study period during quiescent and storm conditions (see Table 3-4) and on the results of the calibration-model runs. Quiescent (or non-storm) conditions are assumed in absence of rainfall in the watershed as indicated when the computed local runoff from the adjacent subwatershed is less than 5 cfs, whereas storm conditions are assumed with the local runoff is greater than 5 cfs. Thus, the low- and high-flow SCRE stages associated with berm breaching from January 2015 through January 2016 was determined to be 11.2 and 14 ft NAVD88, respectively. The low- and high-flow SCRE stages associated with berm breaching from February through December 2016 was estimated as 10.6 and 14 ft NAVD88, respectively except during man-made breaching events which tended to occur at a low-flow SCRE stage 10 ft NAVD88.
- The tidal range and water inflows thresholds that promote SCRE-mouth closure were derived from observed conditions (see Table 3-4) and on the results of the calibration-model runs. The winter/spring tidal range is < 5.0 ft NAVD88 and net inflow is < 65 cfs, and the summer/fall tidal range is < 4.0 ft NAVD88 and net inflow is < 30 cfs. The winter/spring period extends from December 1st to April 30th, and the summer/fall period extends from May 1st to November 30th.

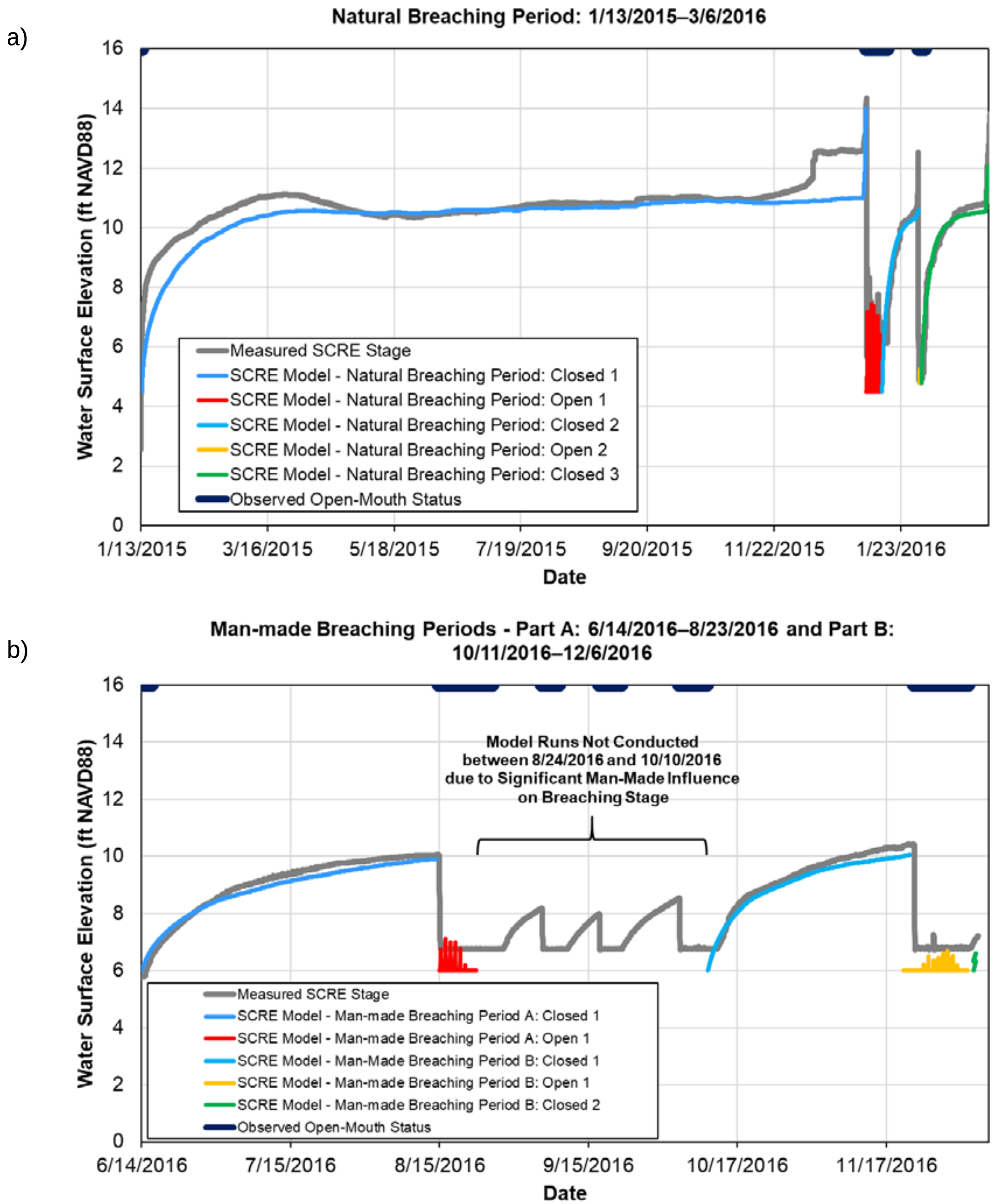


Figure 4-18. Results from the berm breaching and closure calibration runs of the water balance model during three breaching periods: natural breaching period (a) and man-made breaching periods (b).

4.2.2 Potential uncertainty and error in the water balance

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 the compiled elevational surface intended to represent topographic/bathymetric conditions during the water balance period. Using this surface to develop the stage-volume relationship used in the water balance evaluation likely induces error in the volume estimates due to potential instrument and measurement error (on the order of hundredths to tenths of a foot), and errors due to actual changes in the bed morphology from sediment accumulation and scour occurring during the study period. 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 groundwater monitoring wells, 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. The length of the seepage flow path through the berm is assumed to be a constant, but water levels change in the SCRE may change the flow path length and the amount of the seepage through the mouth berm. 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. Additionally, the study relied on river flow data provided by VCWPD for water year 2015 as recorded at Victoria Ave. bridge, but had to rely on riverflow data recorded much farther upstream near Freeman Diversion due to unavailability of data recorded by VCWPD for water year 2016. The Santa Clara River discharge is based upon a stage versus 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.

Under equilibrium conditions, uncertainty in groundwater seepage into the SCRE as well as berm seepage out of the SCRE under equilibrium conditions is one of the largest sources of uncertainty in the water balance. Although assumptions regarding the effective depth of the seepage face, the length of the seepage flow path through the mouth berm, and the hydraulic conductivity of the berm based upon soil type and grain size (Section 4.1.8) could not be readily explored, assuming all other flow contributions to the water balance are accounted for enables the water balance closure to serve as a constraint on alternative assumptions. That is, although these terms were not used for fine tuning the water balance calibration (Section 4.2.1) large decreases in the assumed depth of the seepage face, the length of the flow path through the mouth berm, or reductions in conductivity would result in departure from the modeled versus observed SCRE stage shown in Figure 4-15.

Lastly, because of the dynamic nature of the SCRE mouth berm morphology as well as river flows, the “unmeasured flow” estimates were needed to close the water balance especially during periods immediately following mouth berm closure to achieve model calibration (Figure 4-13). To implement the unmeasured flow component into an MS Excel spreadsheet tool, equations were developed (Table 4-3) that are intended to represent the combined influences of base flow, wave overwash during the initial tidal cycle(s) following mouth berm closure, and soil porewater drainage from the shallow aquifer surrounding the SCRE (bank storage groundwater). This unmeasured term has little influence at higher stages under equilibrium closed mouth conditions and apart from the changes in berm seepage related to dredging deposits along McGrath State Beach during winter 2015 and 2016 and closed berm wave overwash in late 2015, the water balance is well calibrated during both filling and full conditions (Figure 4-15). While the estimate of base flow in the unmeasured flow term was based on individual measurements of base flow in the Santa Clara River channel, the seasonal variations in base flow are unknown and potentially introduce uncertainty to the model. Although wave overwash influences would be expected to be a minor flow contribution to the water balance as berm stability increases in the days and weeks following mouth closure, insufficient berm data makes wave overwash a source of uncertainty. The empirically determined constants Q_{og}, maximum flow from the combined wave overwash and groundwater, and *k*, the rate of decrease in the overwash and groundwater contribution in the unmeasured flow term also are a source of uncertainty in the calibrated water balance model. Variations in the contribution of base flow, wave overwash, and bank storage groundwater may occur under as the water table of the shallow aquifer approaches equilibrium resulting in lower estimates of equilibrium stage under reduced VWRF flow conditions.

In summary, the calibrated water balance is sensitive to assumptions in berm geometry, short term changes in berm seepage due to dredge deposits, and longer term changes in berm length and height which are not readily represented. The water balance is calibrated for the current conditions in the SCRE so significant changes in the SCRE, especially morphology or groundwater elevations, may require a recalibration of the water balance model. The unmeasured flow contributions contain empirical constants that could not be validated by comparisons of the relative flow contributions between SCRE filling periods and periods with sustained low VWRF discharge. Subject to these qualifications the calibrated water balance is considered suitable for use in examining the relative changes in SCRE volume, open water habitat amounts, as well as changes in water quality contributions associated with modifying VWRF discharge rates.

4.3 Estuary Mixing Model

As a means of understanding the relative contributions of nutrients from the local watershed (VWRF and upstream sources) under current or alternative future conditions, we have developed a simplified nutrient balance for the SCRE. A mass balance of algal nutrients such as nitrogen and phosphorus (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

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 depth and consistent onshore winds. Although there are several 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:

$$\frac{dC_E V_E}{dt} = Q_R C_R + Q_{VWRF} C_{VWRF} + \sum Q_G C_G + Q_O C_O + Q_{UM} C_{UM} - \left(Q_R + Q_{VWRF} + \sum Q_G + Q_O + Q_{UM}\right) C_E - Rxn \quad \text{(Eqn. 4.7)}$$

Here the volumetric flows (Q) and mass concentrations (C) refer to the lower Santa Clara River (R), VWRF, shallow groundwater sources (G) from areas along the north and south banks, Estuary sources (E), tidal exchanges from the ocean (O), and unmeasured flow (UM). 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 January 2015 through December 2016 (Section 4.2) shows that the volume of the SCRE is non-steady for large portions of the 2015–2016 monitoring period, we cannot use the steady state 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:

$$C_E \approx \frac{\sum_{i=1}^{n} Q_i C_i - Rxn}{\sum_{i=1}^{n} Q_i + \frac{dV_E}{dt}} \quad \text{(Eqn. 4.8)}$$

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 open water Estuary sites were used to construct mass loading and outflow estimates for salts (using Conductivity as a proxy), total nitrogen (TN = TKN + NO_3), and phosphate (PO_4).

The amount of nutrient removal due to reactions is unknown due to data limitations, but it is reasonable to assume that some combination of algal uptake and microbial processes such as denitrification are occurring in the estuary, especially during summertime conditions that would promote these reactions. After calculating the nutrient concentration assuming no reactions, we estimated the nutrient concentration with reactions assuming zero-order (areal) removal and fitting a single average zero-order removal rate for all times using the observed nutrient concentrations.

Electrical conductivity was modeled at a 30-minute time-step using the calibrated water balance for flows and changes in SCRE volumes, estimates of average Santa Clara River, local runoff, VWRF conductivity, assumptions of groundwater and unmeasured flow conductivity, and sonde conductivity measurements to evaluate model predictions. The Santa Clara River and local runoff conductivity varied widely in measurements made during Phase 3 sampling, but the average conductivity was approximately 2948 uS/cm outside of high flow periods. VWRF conductivity had a much smaller range and was estimated to have an average conductivity of 2564 uS/cm from Phase 3 sampling measurements. Groundwater and the base flow component of the unmeasured flow was assumed to have a conductivity of 1757 uS/cm based on typical groundwater conductivities (Ventura Water 2016). The combined unmeasured flow resulting from wave overwash and the release of bank storage groundwater following breach closure was assumed to be dominated by the wave overwash component and assigned a conductivity of 50,000 uS/cm (Cox et al. 1967). Total nitrogen and phosphate were initially modeled assuming no algal uptake and removal using the analytical sampling results over the discrete time periods between sampling events. Similarly, the total nitrogen and phosphate concentrations including potential algal uptake and removal were modeled using the analytical sampling results over the discrete time periods between sampling events and fitting the zero-order removal rate for the algal uptake and removal reaction term from the measured total nitrogen or phosphate concentrations.

For the simplest application of the mixing model to salinity, Table 4-4 shows that predicted conductivity was within measured conductivity by -2 to 4% during extended closed mouth conditions that occurred during 2015 (Figure 4-19a). The modeling approach tended to overestimate conductivity in the periods shortly after a mouth breach closure so larger prediction errors of 63–268% were apparent during 2016 when there were more frequent breaching events (Figure 4-19b). Tidal exchanges with the ocean would be expected following breaches on January 12, 2015 prior to the March 2015 sampling event, and on January 7th as well as February 2nd of 2016 prior to the March 2016 event (Table 3-4). As many as nine breaches were apparent in the stage record from late March through late November 2016 that would be associated with significant salt water inputs to the SCRE. Model overprediction during frequent breaching periods is likely due to incomplete mixing conditions immediately following a breach and temporary density salinity stratification in the estuary. For example, modeled conductivity typically fell between the measured surface and bottom conductivity during the sampling events. On December 6, 2016 when the model estimated the depth averaged conductivity was approximately 16.0 mS/cm, the average measured surface conductivity was 4.6 mS/cm, but the measured conductivity increased to 12.5 mS/cm in the middle depth sampled and 21.3 mS/cm at the bottom depth sampled. Additionally, a higher than measured conductivity after a mouth breach closure was expected from the model since it was assumed the proportion of unmeasured flow from wave overwash and bank storage groundwater was all wave overwash.

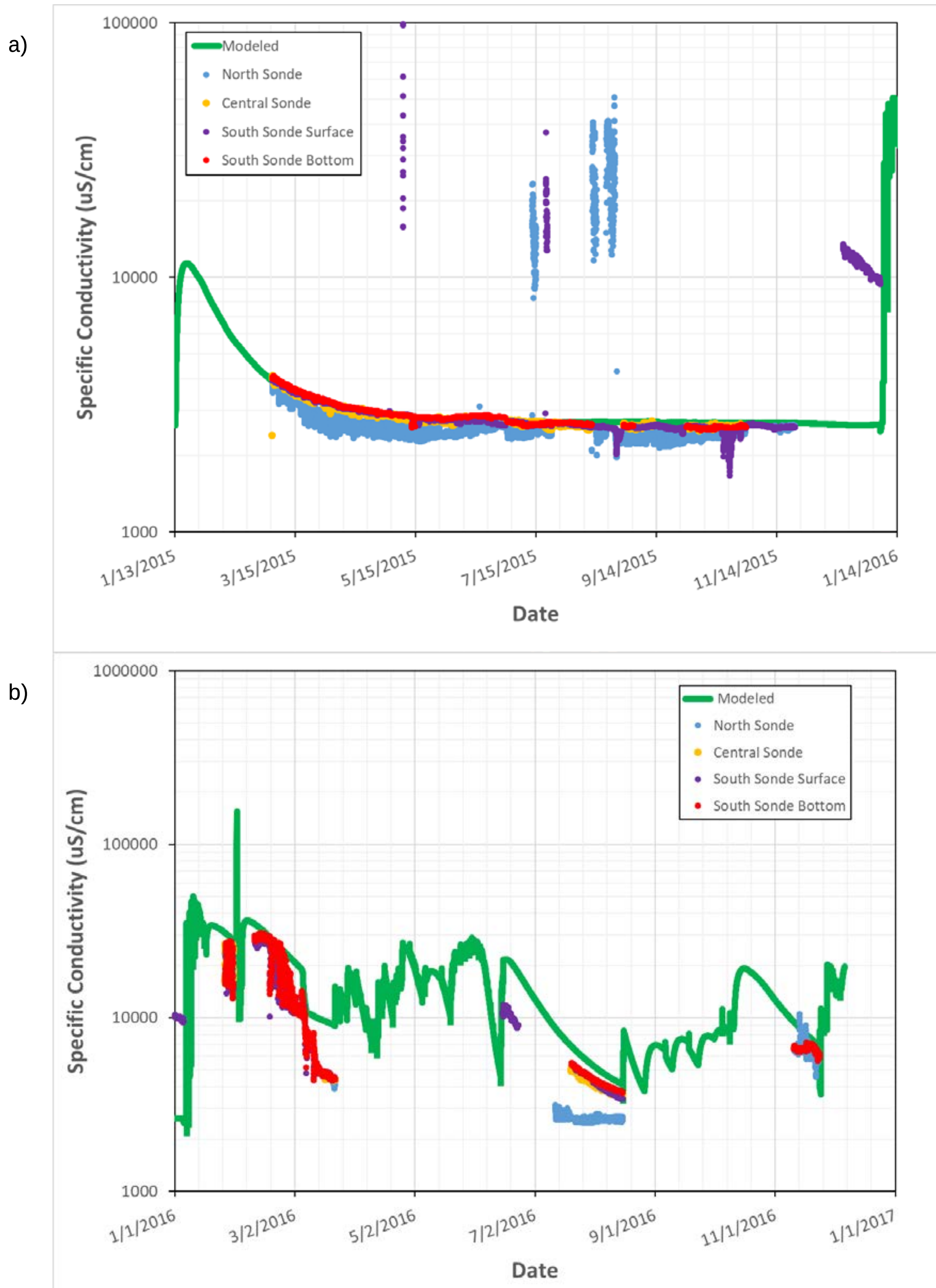


Figure 4-19. Estimated conductivity from the mixing model for in (a) 2015 (b) 2016.

Table 4-4. Predicted and observed conductivity and nutrient concentrations for 2015–2016 sampling period.

Sampling date	Estuary (measured)			Estuary (predicted)					Prediction error				
	Cond (mS/cm)	TN (mg-N/L)	PO_4 (mg-P/L)	Cond (mS/cm)	TN (mg-N/L) at assumed algae uptake/denitrification rates [1]		PO_4 (mg-P/L) at assumed algae uptake rates [2]		Cond (%)	TN (%) at assumed algae uptake/ denitrification rates [1]		PO_4 (%)at assumed algae uptake rates [2]	
					0 mg-N /m²-d	171 mg-N /m²-d	0 mg-P /m²-d	40 mg-P /m²-d		0 mg-N /m²-d	171 mg-N /m²-d	0 mg-P /m²-d	40 mg-P /m²-d
3/18/2015	3.46	4.6	1.1	3.43	10.0	4.6	2.6	1.3	-1%	118%	0%	134%	17%
6/9/2015	2.80	2.6	0.7	2.74	7.7	3.0	2.5	1.4	-2%	201%	16%	255%	50%
9/1/2015	2.60	3.2	0.5	2.71	8.7	2.5	2.0	0.6	4%	172%	-27%	348%	21%
12/1/2015	2.60	4.7	0.4	2.66	8.3	2.0	1.7	0.2	2%	78%	-135%	328%	-74%
3/16/2016	4.97	1.7	0.5	9.70	11.4	2.3	2.4	0.3	95%	588%	29%	392%	-61%
6/1/2016	7.71	4.1	0.7	14.47	7.2	5.6	1.6	1.3	88%	75%	27%	150%	49%
9/14/2016	4.54	6.2	0.6	7.14	9.1	7.2	0.8	0.4	58%	47%	14%	51%	-43%
12/6/2016	4.56	16.3	1.3	15.99	10.0	9.1	1.0	0.7	251%	-39%	-80%	-26%	-75%

Extending the mixing model approach to nutrients (N, P) resulted in large overestimates of observed nutrient levels when no algal uptake and removal was assumed (i.e., the zero-order removal rate was zero), suggesting that biological uptake is important to the nutrient dynamics in the SCRE and it should be included in the assessment of future conditions. For TN, overestimates occurred during spring and summer conditions (Table 4-4), likely due to unaccounted uptake by benthic algae or denitrification occurring in the surficial sediments of the SCRE.

The inclusion of algal uptake and removal with a zero-order removal rate in the mixing model significantly reduced the difference between the modeled and observed nutrient concentrations (Table 4-4). An average zero-order removal rate for each nutrient (N, P) was fitted using the measured nutrient concentrations and the SCRE surface area estimates from the quarterly sampling events during 2015 and 2016. The fitting of a single average zero-order removal rate for each nutrient resulted in both overestimates and underestimates of nutrient concentrations since algal uptake and removal is not actually a constant throughout the year. Modeled TN concentrations including algal uptake and removal had better agreement with measured TN concentrations with as little as 0% difference, but the model tended to underestimated TN levels during winter when actual SCRE algal uptake and removal rates would be reduced. The fitted average zero-order removal rate that improved agreement between the modeled and observed TN levels was 171 mg-N/m^2-day which was within the 0 mg-N/m^2-day to 360 mg-N/m^2-day range of removal rates that have been observed in other systems (Seitzinger 1988, as cited by Horne and Goldman 1994). Overestimates in PO_4 levels in Table 4-4 would also be reduced through inclusion of removal mechanisms such as algal uptake and sedimentation. A fitted average zero-order removal rate for phosphate of 40 mg-P/m^2-day improved agreement between modeled and measured phosphate concentrations which was within the 0.2 mg-P/m^2-day to 60 mg-P/m^2-day range of observed phosphate removal rates in other estuarine systems (Faust and Correll 1976). The modeled phosphate level was within 17% of the measured phosphate levels during some sampling events, but, similar to modeled TN, modeled phosphate concentrations tended to be underpredicted during winter.

While the mixing model approach used here has limitations and uncertainties regarding nutrient uptake and removal, the modeling tool does allow for the assessment of conductivity and the relative magnitude of nutrient loadings to the SCRE from contributing sources. The dynamic nature of the SCRE may result in large departures of estimated concentrations from actual conditions due to several factors related to hydraulics (e.g., rapid changes in SCR flows, ocean exchanges), mixing (e.g., wind, salinity), and seasonal algal growth dynamics. Mixing assumptions are generally met for the SCRE following extended periods of berm closure and predicted conductivity matches observed data well (Table 4-4), but under more transient conditions such as after breaching and berm closure, the model overestimates conductivity until SCRE approaches equilibrium conditions again. The mixing model was not designed to represent transient conditions, so it also does not capture conductivity variations due to closed-mouth wave overwash conditions like those observed in 2015.

Although large errors between predicted and observed nutrient levels under equilibrium conditions are apparent with no removal, the inclusion of zero-order nutrient removal in Equation 4.8 improves agreement between measured and modeled nutrient levels. The remaining differences between the modeled with zero-order removal and measured nutrient concentrations are attributed to two main factors though other factors may also contribute to algal uptake and removal. A single average zero-order removal rate was used to model algal uptake and removal throughout 2015 and 2016, but the actual algal uptake and removal would be expected to have seasonal variations as conditions in the SCRE changed. Both overpredictions and underpredictions would occur from using an average zero-order removal rate since the average would overpredict uptake during winter when algae would uptake less nutrients and underpredict uptake during summer when algae would uptake more nutrients. The SCRE hydraulics and mixing conditions also may contribute to differences between the modeled and measured nutrients. Wave overwash in December 2015 that caused filling of the estuary along with frequent open-mouth conditions during much of 2016 would temporarily alter SCRE mixing conditions along with salinity concentrations that could impact algal growth and die-offs. 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.

4.4 Estuary Equilibrium Heat Balance Model

A simplified well-mixed equilibrium heat balance model for the SCRE was developed to assess the relative influence of VWRF discharges on the equilibrium SCRE water temperature. A well-mixed assumption is considered representative of typical conditions in the SCRE because water temperature data from the sondes (Section 3.4.1.3) after QA/QC (Section 2.2.3.2) along with vertical profile measurements (Section 3.4.1.4 and Appendix D) document that the SCRE is generally thermally well-mixed, especially during closed mouth conditions. While the dynamic nature of the SCRE morphology and intensive data requirements prohibit the development of a fully predictive water temperature model, a simplified model balancing the primary heat exchange processes under well-mixed equilibrium conditions can be used to understand the relative contributions and effects of VWRF discharges upon SCRE water temperature. Assuming well-mixed equilibrium conditions have been reached and heat transport from conduction is negligible compared to other heat transport processes, the net heat transport from advection and surface exchanges is equal to zero and is written as:

$$h_{\text{advection, net}} + h_{\text{surface, net}} = 0 \qquad \text{(Eqn. 4.9)}$$

where $h_{\text{advection, net}}$ (Joules [J]) is the net heat transport from all advection terms and $h_{\text{surface, net}}$ (J) is the net heat transport for all surface heat exchange processes. In this simplified model, the net heat transport from the advection of water is the sum of the heat from the inflows and outflows determined in the water balance model and is written as:

$$h_{\text{advection, net}} = \left(\sum_i q_i T_i - q_{\text{out}} T_{\text{estuary}}\right) K_1 \qquad \text{(Eqn. 4.10)}$$

where q_i (cfs) is the flow rates for the various inflows into the SCRE, T_i (ºC) is the corresponding water temperature for each inflow into the SCRE, q_{out} (cfs) is the sum of the flow rates for the various inflows into the SCRE since the model assumes the SCRE is in equilibrium, T_{SCRE} (ºC) is the average SCRE water temperature, and K_1 is a constant that accounts for the necessary unit-conversion factors, density of water, and heat capacity of water.

The simplified heat balance model considers the advective transport of heat from all the flow terms in the water balance model including precipitation, VWRF discharge, Santa Clara River discharge, local runoff, groundwater flows across the South Bank, North Bank (separated into Ponds, West, and East as described in Section 4.1.8), and Beach Berm, unmeasured flow calculated during the water balance, and the surface water flow across the mouth berm during open-mouth periods. Water temperature for each of the advective components of the heat balance model are determined from available temperature data or assumed from typical temperature conditions. The precipitation, local runoff, and Santa Clara River inflows are assumed to be the ambient air temperature as reported at NOAA station NTBC1 - Santa Barbara, CA because studies have documented precipitation and ambient air temperature are similar (Byers et al. 1949) and the majority of local runoff and Santa Clara River inflows to the estuary occur from precipitation events. The VWRF inflow water temperature is assumed to be equal to the water temperature recorded at the North Sonde. The various groundwater inflows from the South and North Bank are all assumed to be 19ºC based on analysis of the incidental water temperature measured in groundwater monitoring wells. The average groundwater temperature ranged from from 23.3ºC to 16.9ºC in monitoring wells with a regional average temperature of 19.3ºC. Flows from the Beach Berm and the ocean are assumed to be equal to the ocean surface temperature reported at SCRIPPS Station 46216 - Goleta Point, CA. When the model is calibrated to historical data, the average of the Central Bottom Sonde and South Bottom Sonde temperature records is used as a surrogate for the mean SCRE temperature.

In the simplified heat balance model, the net heat transport from the surface exchange ($h_{\text{surface, net}}$) is the sum of the heat from insolation (solar radiation), long wave radiation, and evaporation and is written as:

$$h_{\text{surface, net}} = h_{\text{insolation}} + h_{\text{long wave, in}} - h_{\text{long wave, out}} - h_{\text{evaporation}} \qquad \text{(Eqn. 4.11)}$$

where $h_{\text{insolation}}$ (J) is the heat transport from insolation, $h_{\text{long wave, in}}$ (J) is the heat transport from long wave radiation into the SCRE, $h_{\text{long wave, out}}$ (J) is the heat transport from long wave radiation out of the SCRE, and $h_{\text{evaporation}}$ (J) is the heat transport from evaporation. All other surface heat exchange processes are assumed to be negligible. The surface exchange heat transport components were calculated using the formulae from Theurer et al. (1984).

The heat transport from insolation is calculated:

$$h_{\text{insolation}} = \eta I A \qquad \text{(Eqn. 4.12)}$$

where η is the fraction of solar radiation absorbed by the water, I (W/m²) is the insolation reported at CIMIS Station 198 - Santa Paula, CA, and A (m^2) is the SCRE surface area determined from the SCRE stage in the water balance analysis. Solar insolation from CMIS Station 198 was used because it was the closest station to the SCRE with the most complete data record between 2015 and 2016. The fraction of solar radiation absorbed by water has been found to be have an average value of approximately 0.73 in coastal waters (TVA 1972) though it ranges from about 0.61 in clear water to over 0.8 in turbid water. In the simplified equilibrium heat balance model, the fraction of solar radiation absorbed by the water is a calibration parameter that is determined for a specified time period using the historical data in the SCRE

The heat transport from long wave radiation into the SCRE is calculated using:

$$h_{\text{longwave, in}} = (1 - r_l)\varepsilon_a \sigma (T_{\text{air}} + 273.16)^4 A \qquad \text{(Eqn. 4.13)}$$

where r_l is the fraction of radiation reflected at the surface, here taken to be 0.03, ε_a is the atmospheric emissivity, T_{air} (°C) is the air temperature as reported at NOAA station NTBC1 – Santa Barbara, CA, σ is Boltzman's constant equal to $5.672 \times 10^{-8}\ \text{W/m}^2/(°\text{K})^4$, and A (m^2) is the SCRE surface area. The atmospheric emissivity is approximated using Swinbank's method which is calculated:

$$\varepsilon_a = 9.062 \times 10^{-6}(T_{\text{air}} + 273.16)^2 \qquad \text{(Eqn. 4.14)}$$

where T_{air} (°C) is the air temperature as reported at NOAA station NTBC1 – Santa Barbara, CA.

The heat transport from long wave radiation out of the SCRE is calculated using:

$$h_{\text{longwave, out}} = \varepsilon_w \sigma (T_{\text{SCRE}} + 273.16)^4 A \qquad \text{(Eqn. 4.15)}$$

where ε_w is the emissivity of water equal to 0.9526, σ ($\text{W/m}^2/(\text{°K})^4$) is Boltzman's constant as discussed above, T_{SCRE} (°C) is the average SCRE water temperature, and A (m^2) is the SCRE surface area.

The heat transport from evaporation is calculated:

$$h_{\text{evaporation}} = E(2500 - 2.39T_{\text{SCRE}})AK_2 \qquad \text{(Eqn. 4.16)}$$

where E is the evaporation rate (m/s) determined in the water balance from from hourly pan evaporation data recorded at the El Rio-UWCD Spreading Grounds (VCWPD station 239) and the SCRE inundated area time series, T_{SCRE} (°C) is the average SCRE water temperature, A (m^2) is the SCRE surface area, and K_2 is a constant equal to 998.2 kg/m^3, the density of water at 20°C, in the units used.

The heat fluxes for the individual components of the heat balance model described above were calculated from measured data, water balance estimated flows, and the literature average value for the fraction of solar radiation absorbed by water. The calculated heat fluxes are compared in fall 2015 because SCRE conditions are assumed to be the closest to well-mixed equilibrium conditions during this time when the beach-berm has been closed for over eight months, there are minimal variations in measured inflows and outflows, and the observed and water balance modeled stage are in close agreement. During September through October 2015, the total heat flux in is approximately equal to the total heat flux out (Figure 4-20) indicating the heat balance captures the main heat fluxes under equilibrium conditions. The small difference between the heat flux in and out is expected since it is a simplified model that does not include all heat flux terms or dynamics of heat exchange. During September through October 2015, the VWRF discharge averaged 7.5 cfs so the calculated heat fluxes in Figure 4-20 also highlight the relative magnitude of heat fluxes from this VWRF discharge with the other heat fluxes into and out of the SCRE. The heat flux from convection is listed in Figure 4-20 in brackets to recognize its potential influence on the heat balance, but to indicate that it is not included in the heat balance.

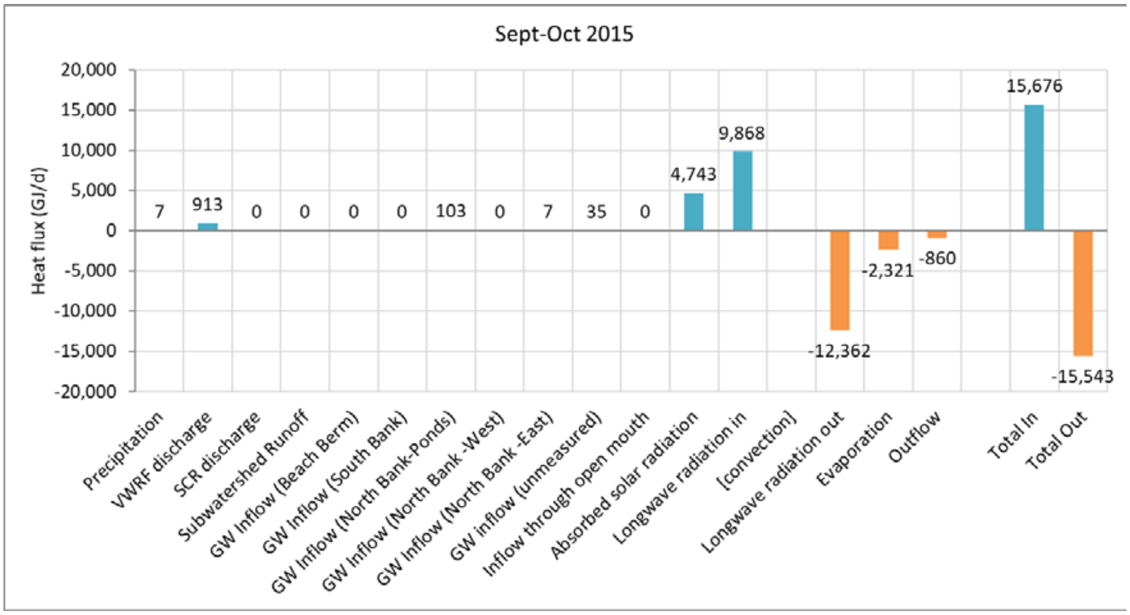


Figure 4-20. Observed magnitude of heat flux from the individual components of the simplified heat balance under non-equilibrium conditions

The simplified equilibrium heat balance model provides a tool to assess the relative influence of VWRF discharges on estuary water temperature despite the uncertainties and limitations of the model. As with all models, there are multiple uncertainties contained in the model including those from measurements of water or air temperature to those associated with assumptions of groundwater temperature or constants in the heat exchange equations. The model also has uncertainties introduced by the inputs from the water balance. The model is only designed to assess the relative influence of VWRF flows on estuary temperature at equilibrium since it is a simplified model not intended to represent the dynamic complexity of estuarine heat exchange processes in the SCRE. Calculations of the heat fluxes from the different components of the model show that the total heat fluxes in are approximately equal to the total heat fluxes out verifying that the model captures the main heat exchange processes in the estuary. While model estimated equilibrium estuary water temperatures may vary from observed estuary water temperatures, the modeling approach will still capture the relative influence of alternative VWRF discharge scenarios developed in later sections of this report on SCRE water temperature.

5 COMPARISON OF EXISTING AND ALTERNATIVE DISCHARGE SCENARIOS

For the purposes of meeting the conditions of the current NPDES Permit (R4-2013-0174) and the terms of the Consent Decree this Section compares existing and alternative VWRF discharge scenarios and the degree to which they support each of the 11 beneficial uses of the SCRE, using a combination of approaches. Additonally, potential realization of municipal water supply uses (MUN) associated with diversions of VWRF tertiary treated flow to potable reuse (via advanced treatment) are examined. The results of these comparisons will be used to make recommendations as outlined in Section 1.2 on the following:

1. **Enhancement Discharge Levels**. To determine the recommended Enhancement Discharge Levels, if any, analysis in this section is used to evaluate the effects of a "no-discharge" scenario in comparison to the effects of 10 other VWRF discharge scenarios on the basis of both quantitative and qualitative assessments to determine which of the discharge scenarios, are likely to more fully support the 11 beneficial uses of the SCRE in the Basin Plan, as well as the potential for realizing additional municipal water supply (MUN) uses within the SCRE subwatershed. This analysis complies with requirements of the NPDES Permit, and facilitates the RWQCB's determination of "enhancement" (as defined by EBE Policy) likely to be associated with different levels of discharge for purposes of the next renewal of the VWRF NPDES Permit.
2. **Maximum Ecologically Protective Diversion Volume**. To determine the MEPDV, analysis in this section is used to evaluates the degree to which a range of VWRF diversions—from 0% diversion (current discharge conditions) to 100% diversion (no discharge)—may either better support or potentially adversely impact existing ecological conditions and related beneficial uses of the SCRE, with particular weight and emphasis upon habitat use by native listed species and designated critical habitat. This analysis is anticipated by the NPDES Permit, complies with the Consent Decree, and may also facilitate environmental review under NEPA and CEQA of potentially significant environmental impacts to existing SCRE conditions that may be associated with diversions of VWRF tertiary treated flow to augment and increase reliability of local municipal (MUN) water supplies. In addition, this analysis facilitates review, consideration, and future permitting determinations by NMFS, USFWS and CDFW with respect to significant adverse impacts of diversions, if any, on listed species and associated critical habitats within the SCRE.
3. **Continued Discharge Levels**. To determine the recommended Continued Discharge Levels, if any, needed to protect the SCRE's existing ecological resources, this Report evaluates the threshold at which potential impacts to existing ecological resources of the SCRE and related beneficial uses are likely to result from reducing current VWRF discharges of tertiary treated effluent to (i.e., increasing diversions of effluent from) the SCRE, taking into account existing conditions and current discharge levels. This analysis complies with requirements of the NPDES Permit, and facilitates a determination by the RWQCB regarding an appropriate average annual flow that may be permitted or required under the next VWRF NPDES Permit cycle.

Essentially, determination of the MEPDV and the Continued Discharge Level are the converse of one another, and determining both the MEPDV and the Continued Discharge Level are informed by the detailed beneficial uses assessments conducted in the sections below to recommend Enhancement Discharge Levels. The same factors, considerations, and analysis appropriately determine the degree to which VWRF discharges support fuller realization of beneficial uses or "enhancement" as compared to the absence of discharge can also inform recommendations regarding the MEPDV and the Continued Discharge Levels, if any, necessary to protect existing native species and their habitats within the SCRE. In the sections below, a combination of the technical information and modeling tools developed in prior sections are integrated using a multi-criteria assessment framework based upon the Analytic Hierarchy Process (Saaty 1980) to determine the relative ecological benefit of a wide range of different VWRF discharge scenarios as they relate to realization and protection of designated beneficial uses of the SCRE, improvement of local water supply and MUN within the watershed, and protection of existing ecology, species and habitats of the watershed.

As an overview, Section 5.1 describes the assessment framework for comparing discharge scenarios by weighing likely effects of the different scenarios on potentially competing beneficial uses and specific attributes or factors affecting realization of each beneficial use. Section 5.2 introduces the VWRF discharge scenarios evaluated, ranging from 0% diversion (current discharge conditions) to 100% diversion (0 MGD discharge conditions). Sections 5.3 through 5.4 compare these VWRF discharge scenarios on the basis of comparing predicted changes in habitat and other conditions based on the information, methods, and models described in Sections 3 and 4 of this Report:

- variations in vegetation and habitat types
- variations in water quality conditions
- variations in breaching dynamics and salinity conditons following SCRE breaches

Based upon these predictions of changes in physical conditions and habitat types, Section 5.5 defines detailed metrics for quantifying the extent to which each VWRF discharge scenario supports each designated beneficial use of the SCRE as well as potential MUN uses within the SCRE subwatershed in isolation, and in comparison to one another. Discharge scenarios are scored for each of a number of factors for each beneficial use. Each of these factors is weighted according to their relative importance using the assessment framework described in Section 5.1 developed with input by the TRT and Resources Agencies. In Section 5.6, scores for each discharge scenario and its support for each of the individual beneficial uses are compiled, and compared across beneficial uses, which have also been weighted by relative importance of each beneficial use according to the assessment framework. The resulting scores for each discharge scenario represent the extent to which the scenarios support the designated SCRE beneficial uses as well as MUN uses within the SCRE subwatershed, taking into account consensus developed among the TRT, Resources Agencies, and other stakeholders regarding relative importance of beneficial uses, and the factors and attributes that likely vary with discharge. The results of this assessment are then considered along with qualitative analysis and professional judgement to provide recommendations regarding Enhancement Discharge Levels, if any, the Continued Discharge Level, if any, and the MEPDV.

5.1 Assessment Framework

Comparison of discharge scenarios inherently entails value-based judgements since there are potentially competing beneficial uses with potentially competing factors and attributes supporting the various beneficial uses. This complexity presents a considerable challenge in determining those VWRF discharge scenarios that more fully realize beneficial uses (i.e., provide “enhancement” as defined in the EBE Policy) of the SCRE, the recommended MEPDV, and appropriate Continued Discharge Level, if any, necessary to protect native listed species and critical habitat. To address this challenge, we adopted an analytical framework that uses a series of pairwise comparisons to assign weights to each of a hierarchical series of decision-making factors. This framework is based on the Analytic Hierarchy Process (AHP) (Saaty 1980, 2008). The AHP is a mathematical approach to multiple criteria decision making that has been applied to a range of applications in solving large and complex decision problems. To develop a rational and repeatable framework to compare and select recommended levels of discharge/diversion among various discharge alternatives, we have applied the AHP by the following four steps:

1. **Analytic Hierarchy** – Initially, factors considered in a decision such as comparison of the VWRF discharge alternatives are identified and organized as a hierarchy of interrelated decision elements. Typically, this is visualized as a tree containing the overall decision or goal at the top and lower levels (Tiers) of contributing factors. In the case of the SCRE, our

hierarchy consists of three tiers: Tier 1 factors are the ten designated SCRE beneficial uses analyzed (note that there are two beneficial uses not considered, MAR[20] and NAV[21]); Tier 2 factors are the physical, chemical, and biological attributes supporting the realization of that beneficial use; and Tier 3—which was only used for the RARE beneficial use—includes factors supporting the Tier 2 factors. Based upon informed input by the TRT and Resources Agencies, these attributes and ultimately the beneficial uses are linked to VWRF discharge scenarios using qualitative and quantitative information from the observed data, the water balance model, the estuary mixing model, and the heat balance model.

2. **Pairwise comparisons** – To determine which factors are more important in reaching a decision, a series of pairwise comparisons are made amongst all the decision elements. This is generally arranged as a series of questions as to whether each factor is more or less important in making a decision than other factors within the same tier, and secondarily by how much on a numeric (Saaty) scale from 1-9 (Table 5-1).

Table 5-1. Saaty scale of relative importance.

Intensity of importance	Definition of factor importance	Explanation
1	Equal Importance	Two factors contribute equally to comparison
3	Moderate Importance	Experience and judgement moderately favor one factor over another
5	Strong Importance	Experience and judgement strongly favor one factor over another
7	Very Strong Importance	A factor is strongly favored and its dominance demonstrated in practice
9	Absolute Importance	The evidence favoring one factor over another is of the highest possible order of affirmation
2, 4, 6, 8	Intermediate values between the two adjacent judgements	When compromise is needed

Source: Saaty (1980)

3. **Weighting** – The results of the pairwise comparisons are arranged in a square matrix $B_{M \times M}$. The comparison is made by identifying the impact of the factors on the left side of the matrix to the elements at the top of the matrix. A factor compared with itself is always assigned the value 1, so the main diagonal entries of the pair-wise comparison matrix are all 1. Below the main diagonal there are the inverse of the pairwise comparisons above the diagonal. With the matrix completed the relative normalized weight of each factor is calculated from the geometric mean of the m^{th} row, and by normalizing the geometric means of rows in the comparison matrix.

[20] Factors affecting MAR beneficial uses such as marine species that might use the SCRE are assessed under other beneficial uses, including EST, WILD, and RARE and are not assessed separately.
[21] Because the SCRE is not used extensively for navigational purposes, NAV beneficial use was not assessed.

$$\boldsymbol{B}_{M\times M}= \begin{array}{c} \boldsymbol{B}_1 \\ \boldsymbol{B}_2 \\ \boldsymbol{B}_3 \\ \vdots \\ \boldsymbol{B}_M \end{array} \begin{bmatrix} \mathbf{1} & \boldsymbol{b}_{12} & \boldsymbol{b}_{13} & \cdots & \cdots & \boldsymbol{b}_{1M} \\ \boldsymbol{b}_{21} & \mathbf{1} & \boldsymbol{b}_{23} & \cdots & \cdots & \boldsymbol{b}_{2M} \\ \boldsymbol{b}_{31} & \boldsymbol{b}_{32} & \mathbf{1} & \cdots & \cdots & \boldsymbol{b}_{3M} \\ \vdots & \vdots & \vdots & \vdots & \cdots & \vdots \\ \boldsymbol{b}_{M1} & \boldsymbol{b}_{M2} & \boldsymbol{b}_{M3} & \cdots & \cdots & \mathbf{1} \end{bmatrix}$$

4. **Scoring and Alternatives Decision** – With the relative weighting of assessment factors completed, metrics and criteria at terminal branches of the hierarchy are assigned scores based upon an agreed upon system, with the results of the weighting Steps 1–3 used to calculate the weighted summations in each tier and to arrive at a final score for the alternatives under comparison.

Pairwise comparisons for weighting beneficial uses of the SCRE as well as factors contributing to each beneficial use were initially performed in a workshop setting that included natural resource experts retained by the City, and the TRT convened by Heal the Bay and Wishtoyo. In this workshop process discussed in Appendix H, expert participants were asked to reach consensus regarding each pairwise comparison, as well as the resulting weighting of the beneficial uses, beneficial use factors, and attributes of factors considered. After presenting the draft results, additional workshops were held to obtain verbal and written feedback on the AHP, the included beneficial uses, factors, and scoring thresholds. The individual expert pairwise comparisons were used to update the AHP weighting and scoring framework, then the VWRF discharge scenarios scores were calculated for each beneficial use from the contributing factors and the weights associated with each of the beneficial uses and factors (See Section 5.5). To synthesize these results, the weighted scores calculated for each of the beneficial uses are combined into a normalized score and priority for each VWRF discharge scenario in Section 5.6. The VWRF discharge scenarios are compared based on the normalized score and priority, and considered in light of best professional judgment regarding protection of the SCRE's beneficial uses, existing ecological resources, including listed species and critical habitats, and potential for improvements to local water supply and MUN uses within the SCRE subwatershed.

5.2 VWRF Discharge Scenarios Evaluated

For the purposes of evaluating future VWRF scenarios, the closed-mouth, dry-weather conditions occurring during 2015/2016 were selected to represent the most critical condition for assessing discharge scenarios as well as current conditions. The 2015/2016 conditions were selected since the berm morphology has receded since 2005 (Figure 3-1) so the 2015/2016 would best characterize the current SCRE conditions. Additionally, the influence of VWRF discharges on the SCRE under the below normal hydrologic conditions in 2015 and 2016 would potentially be larger since the VWRF discharges would make up a larger percentage of flows into the SCRE than in wetter water year types.

The 2015/2016 Average Dry Weather Flow (ADWF) entering the SCRE was approximately 4.7 MGD at the Wildlife Pond outlet (Site M-001A). The 11 potential discharge scenarios, each representing a 10% reduction in discharge at 0.47 MGD, are shown in Table 5-2. By application of modeling tools developed in prior section of this report, changes in physical, water quality and habitat conditions in the SCRE are described in the following sections over a range in discharge from current conditions (4.7 MGD or 0% diversion) to 0 MGD discharge (i.e., 100% diversion).

Table 5-2. Phase 3 VWRF discharge scenarios.

Discharge scenario	% of current discharge (4.7 MGD) diverted	VWRF discharge to SCRE (MGD)	VWRF flow diverted to other uses (MGD)
1	0%	4.7	0
2	10%	4.2	0.5
3	20%	3.7	0.9
4	30%	3.3	1.4
5	40%	2.8	1.9
6	50%	2.3	2.3
7	60%	1.9	2.8
8	70%	1.4	3.3
9	80%	0.9	3.7
10	90%	0.5	4.2
11	100%	0	4.7

5.3 Variation in Equilibrium Closed Mouth Habitat Conditions by Discharge Scenario

For each discharge scenario (Table 5-2), the compiled water balance (Section 4.1) was used to provide an estimate of the relative area and extent of different vegetation and habitat types, and relative water quality changes that might be anticipated to develop under different discharge scenarios during dry weather, closed berm conditions when the influence of VWRF discharge on the SCRE would dominante. These estimates were produced using average seasonal SCRE inflows and outflows for the water balance components (Table 5-3) and adjusting the VWRF discharge inflows according to the discharge scenario assumptions.

While long-term (multi-year) averaging of external flow inputs to the water balance model is possible, as was done in the Phase 1 Study, changes and effects likely to result from each discharge scenario are quantitatively assessed for the most critical conditions (e.g., dry water year or summer) with the assumptions that the SCRE mouth berm remains closed (i.e., no breaching) and the SCRE water surface elevation is near or at equilibrium. These conditions and assumptions were analyzed because the model predicts that even moderate flows from the Santa Clara River result in rapid filling and breaching of the SCRE mouth berm, the discharge impacts on SCRE beneficial uses and ecological resources are most significant during representative closed mouth, dry weather conditions, and the model performance is limited under the dynamic filling or emptying of the SCRE present during open-mouth conditions.

Each discharge scenario was assessed for this SCRE condition, assuming similar lagoon morphology and baseline flows that are the same as those encountered during the relatively dry

2015–2016 monitoring period. While the SCRE experiences changes in lagoon morphology (Section 3.2.1) as well as vegetation and habitat types (Section 3.5.2) over time in response to non-VWRF discharge factors like storm events, the current assessment acknowledges the uncertainties associated with the prediction of future vegetation and habitat types and water quality conditions (Sections 5.6.3 and 5.6.4).

Figure 5-1 shows the calculated water surface elevation of the SCRE for each VWRF discharge scenario following a hypothetical closed-mouth, dry-weather filling period that begins on March 1st and continues through the summer period until the lagoon reaches a quasi-steady equilibrium water surface elevation. Unmeasured flow is a constant for all VWRF discharge scenarios since it is composed of a constant base flow and an initial wave overwash and groundwater drainage flow from areas surrounding the SCRE (bank storage groundwater) that are only a function of time from mouth closure so the combined base flow and wave overwash and groundwater drainage are the same under all scenarios. The initial rate of estuary filling is primarily driven by unmeasured flow attributed to both wave overwash and drainage of bank storage groundwater and the contribution of VWRF effluent. Variations in the magnitude of unmeasured flow would influence the initial rate of estuary filling, but would not alter the equilibrium estuary stage beyond the filling period. As the unmeasured flow portion of water filling the lagoon decreases over time to the constant flows shown at equilibrium in Table 5-3, the VWRF effluent becomes the primary contribution to total inflow volume that produces the lagoon water surface elevation at the closed-mouth, dry-weather equilibrium conditions under each scenario.

Table 5-3. Estimated average flows (MGD) to the SCRE Water Balance during closed-mouth, dry-weather equilibrium conditions for discharge scenarios 1 through 11.

Discharge Scenario	Surface water flow contributions					Groundwater flow contributions					
	Santa Clara River[1]	VWRF Discharge	Precipitation/ Runoff	Evaporation[2]	Berm Outflow	North Bank Floodplain—Ponds (GW-12 through GW-15)	North Bank Floodplain—West (GW-8 through GW-11)	North Bank Floodplain—East (GW-4 and GW-5)[3]	South Bank Floodplain (GW-1 through GW-3)	South Bank Floodplain, Upstream (GW-6 and GW-7)	Unmeasured flow
1	0.00	4.70	0.00	-0.53	-4.48	0.16	-0.10	0.05	0.00	0.00	0.23
2	0.00	4.20	0.00	-0.43	-4.14	0.17	-0.09	0.05	0.02	0.00	0.23
3	0.00	3.70	0.00	-0.36	-3.75	0.18	-0.07	0.05	0.03	0.00	0.23
4	0.00	3.30	0.00	-0.34	-3.41	0.18	-0.05	0.05	0.04	0.00	0.23
5	0.00	2.80	0.00	-0.33	-2.97	0.19	-0.03	0.05	0.06	0.00	0.23
6	0.00	2.30	0.00	-0.31	-2.54	0.20	-0.01	0.05	0.07	0.00	0.23
7	0.00	1.90	0.00	-0.28	-2.21	0.21	0.00	0.05	0.09	0.00	0.23
8	0.00	1.40	0.00	-0.23	-1.82	0.22	0.02	0.05	0.10	0.00	0.23

9	0.00	0.90	0.00	-0.18	-1.42	0.22	0.03	0.05	0.11	0.00	0.23
10	0.00	0.50	0.00	-0.14	-1.09	0.23	0.05	0.05	0.12	0.00	0.23
11	0.00	0.00	0.00	-0.09	-0.43	0.00	0.06	0.05	0.13	0.00	0.23

[1] Santa Clara River flow based on daily mean discharge recordings made 1.5 miles upstream at the Victoria Ave. bridge (VCWPD station 723) during water year 2015

[2] Evaporation decreases with decreasing VWRF discharge since the surface area of the SCRE decreases at lower VWRF discharges.

[3] A constant inflow was estimated for the groundwater flow from the North Bank Floodplain - East since measured data showed a consistent inflow that did not vary significant with SCRE stage (Section 4.1.8).

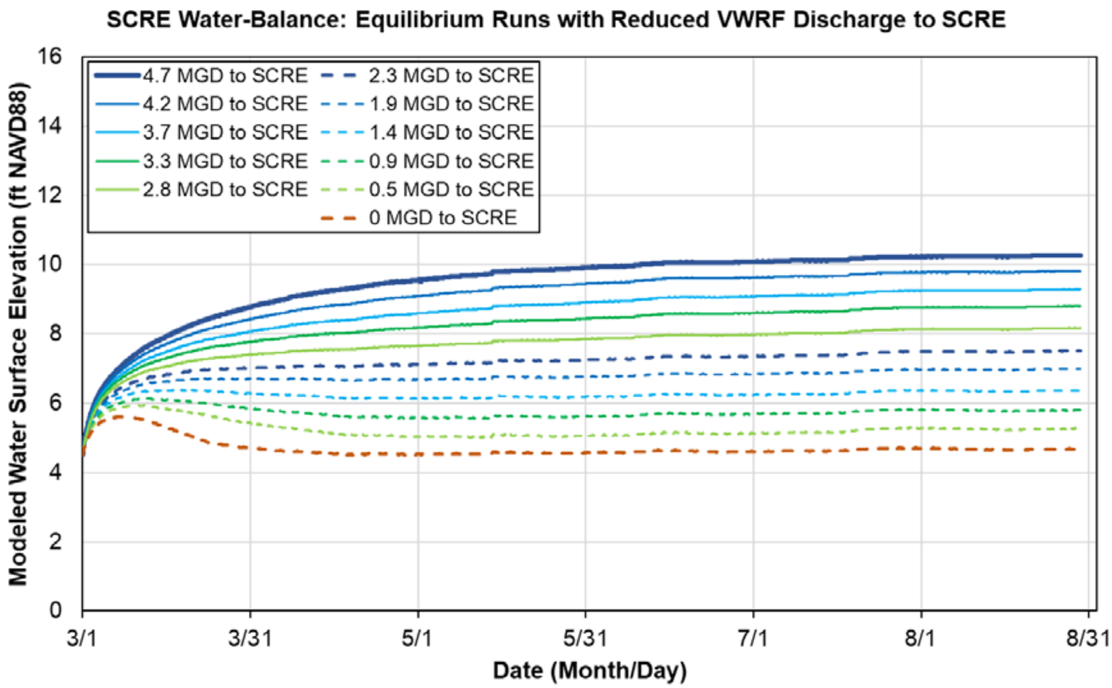


Figure 5-1. Modeled SCRE stage time series during idealized dry season conditions for discharge scenarios 1 through 11.

5.3.1 Changes in vegetation community and habitat types by discharge scenario

As described in the preceding sections (Section 3.2 and 3.3), the SCRE periodically experiences large pulses of water and sediment from the Santa Clara River, which have effects upon channel form, bed substrate, vegetation assemblages, and estuary extent over both 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 lagoon inundation resulting from VWRF discharges, wet weather increases in river inflows, as well as berm breaching and ocean water exchange. Although reductions in average SCRE water elevations and the amounts of both open water and inundated habitats in the SCRE are nearly certain under reduced VWRF discharge scenarios, changes in the relative distribution of vegetation communities and habitat types are likely to take many years to observe, and are also likely to be affected over the long-term by changes in sea level, flood frequency, rainfall, and the occurrence of flood scour events.

5.3.1.1 Succession modeling of future habitat types

Recognizing that historical changes in the SCRE mouth berm position and morphology are also a major factor driving changes in vegetation communities and habitat types, the overall extent and composition of vegetation and habitat types generally approach long-term averages that are controlled primarily by SCRE berm position and stage (Section 3.5.2). For the purposes of this assessment, we have estimated the relative amounts and distributions of future vegetation and habitat types during stable, closed-mouth, dry weather lagoon conditions for each VWRF discharge scenario on the basis of water depth relationships using a spatially-explicit GIS model within key regions extending laterally from the SCR thalweg as well as inland from the Pacific Ocean (Figure 5-2). The habitat successional rules used in the model are detailed in Table 5-4, and were determined based upon the landscape position within the study area (i.e., the study area was divided up into five regions: riverwash/lagoon, riparian, beach/foredune, campground, and VWRF ponds). The rules assign predicted habitat changes based upon the model-input habitat type and the relationship between each location and the modeled water surface elevation. Below we summarize the rationale for habitat succession within each of the five regions:

1. *Riverwash/Lagoon Region*

   The Riverwash/Lagoon region is characterized by areas experiencing greater frequency of scour and inundation which control the successional stage of riparian and marsh vegetation present along the active channel corridor. Scour areas within the active channel and recent floodplain deposits upstream of Harbor Blvd. provide surfaces for plants to establish. For the purposes of this assessment, the habitat successional rule assigns the riparian riverwash habitat type to all Riverwash/Lagoon regions based upon recent mapping in these regions during the period reviewed for this Study. Riparian riverwash habitat is primarily comprised of barren floodplain deposits with sparse cover of low stature pioneer species such as willows (*Salix* spp.), which have high seed output and rapid growth rates, and are well suited for rapid colonization. Accordingly, application of the habitat successional rules for this assessment, predicts that areas in the Riverwash/Lagoon region with ground surfaces above predicted water surface elevations (WSEs) for each discharge scenario will convert to (or remain) riparian riverwash habitat.

   Although small patches of freshwater wetland were mapped within the Riverwash/Lagoon region along the channel margin upstream of Harbor Blvd. in 2016 (Figure 3-41), these stands would continue to be vulnerable to regular scour during even moderate flood events. As such, the habitat succession rules assume that some freshwater wetland habitat remains when appropriate elevations (i.e., 0–3 ft below WSE) are modeled within pre-existing patches of freshwater wetland, but the rules do not predict that new areas of freshwater wetland would develop within the Riverwash/Lagoon Region at 0–3 ft below the modeled equilibrium WSE for each discharge scenario based upon water-depth realtionships for typical species as well as direct observations (cbec 2015) discussed under the *Riparian Region* below. Within areas that are predicted to be inundated under the VWRF discharge scenarios, open water habitats are expected to remain intact, although at a reduced extent in proportion to modeled reductions in the equilibrium lagoon stage.

2. Riparian Region

The Riparian region is characterized under existing conditions by higher elevation surfaces away from the active channel as well as the margins of SCRE freshwater wetland habitats. Growth of riparian vegetation in this region is controlled by plant colonization that promotes sediment trapping, which can then extend the bar surface to create new seedbeds for successive riparian vegetation recruitment events (Strahan 1984, Scott et al. 1996). For example, following recovery from the 2005 flood scour event, areas that had been previously mapped as Riverwash Herbaceous and Floodplain Wetland prior to the 2005 high flows shifted to arroyo willow thickets with smaller representations of shining willow groves and mulefat thickets, and trace amounts of Riverwash Herbaceous a few years later (Orr et al. 2011). The region also includes large, dense patches of high cover of the non-native invasive giant reed breaks (*Arundo donax* Herbaceous Alliance). For the purposes of this assessment, we have assumed that most areas in the Riparian region with ground surfaces above predicted WSEs associated with each VWRF discharge scenario will convert to (or remain) a mixed riparian habitat type, except foredune, salt marsh, and disturbed habitat will remain unchanged.

Within shallow water areas, freshwater wetlands typically show a zonation of plant species and vegetation types according to water depth (Mitsch and Gosselink 2000): cattail (*Typha latifolia*) stands typically develop in shallow water up to 1.5 ft deep (DiTomasso and Healy 2003), while bulrush or tule species, such as hardstem bulrush (*Schoenoplectus acutus*) can spread in somewhat deeper waters (3–5 ft deep) (Tilley 2012). Consistent with the literature sources above, and the observed elevations and habitat types documented in the SCRE for representative plant species (Table 3 in cbec et al. 2015) during the Study period, the habitat successional rules assign freshwater wetland habitats to areas of shallow water ranging from 0 to -3 ft NAVD88. Accordingly, freshwater wetlands are predicted to develop in areas of 0-3 ft below the modeled WSE for each VWRF discharge scenario in the Riparian region, with open water occurring in deeper areas (>3 ft in depth) (Table 5-4). In addition, under these successional rules new freshwater wetland habitat is predicted to develop within this same shallow water depth range (0–3 ft below modeled WSE for each discharge scenario) in areas that currently support open water habitat. Lastly, we have assumed that salt marsh habitats that are inundated in this same depth range based on the predicted WSEs for each VWRF discharge scenario would eventually convert to a freshwater wetland habitat type, while any foredune areas below the WSEs for a given VWRF discharge scenarios would convert directly to open water habitat.

3. Beach/Foredune Region

Within the Beach/Foredune region, habitat amounts are controlled by coastal processes with periodic sand/sediment inputs during periods of flood that result in large changes to the SCRE berm position. Because the occurrence of these events is largely stochastic, the successional rules assume only minor changes in the amounts of beach and foredune habitats resulting from reduced WSEs under alternative VWRF discharge scenarios; in general, beach/foredune areas with ground surfaces above the modeled WSE remain unchanged and areas with ground surfaces below the modeled WSE convert to open water habitat. Freshwater wetland is the exception as it converts to open beach above the modeled WSE but remains freshwater wetland below the modeled WSE.

4. Campground Region

Within the Campground region, access roads as well as trail areas and riparian areas are presently mapped as developed/disturbed habitats (Section 3.5) with low lying areas mapped as disturbed wetlands (WRA, Inc. 2014a). Based on this mapping and expectations that campground uses will continue, for the purposes of this assessment the successional rules assume that developed and riparian areas will be maintained as developed/disturbed habitats at elevations above predicted WSEs under alternative VWRF discharge scenario. In shallow water areas (0–3 ft depth), the rules assume that any freshwater wetland that develops in current developed/disturbed areas would be classified as disturbed wetlands in the future, while areas above predicted WSEs convert to developed/disturbed habitat.

5. VWRF Pond Region

The VWRF Wildlife/Water Quality Ponds currently support a mix of open water habitats, disturbed/developed areas as well as riparian vegetation surrounding the ponds (Section 3.5). For the purposes of this assessment, the successional rules assumed that these habitats would remain largely unchanged because of relative constant water levels in the ponds under existing or foreseeable VWRF operations. For Scenario 11 (100% diversions/and no discharge), we have assumed that the ponds would be eliminated or operated at reduced water levels. Under this scenatio, although open water habitat amounts in the ponds are shown to remain, existing riparian habitat surrounding the ponds would be converted to developed habitat.

Figure 5-2. Modeled SCRE study area regions for application of habitat evolution rules under alternative VWRF discharge scenarios

Table 5-4. GIS modeling assumptions of future habitat evolution in relationship to water surface elevation (WSE) under alternative VWRF discharge scenarios.

Input habitat type	Future habitat type
Riverwash/Lagoon region	
Open Water	Above WSE, converts to riparian riverwash habitat
	Below WSE, remains open water habitat
Freshwater Wetland	Above WSE, converts to riparian riverwash habitat
	0–3 ft below WSE, remains freshwater wetland habitat
	>3 ft below WSE, converts to open water
Riparian/Riparian Riverwash	Above WSE, remains riparian riverwash habitat
	Below WSE, converts to open water
Riparian region	
Foredune	Above WSE, remains foredune habitat
	Below WSE, converts to open water habitat
Open Water	Above WSE, converts to riparian habitat
	0–3 ft below WSE, converts to freshwater wetland habitat
	>3 ft below WSE, remains open water habitat
Freshwater Wetland	Above WSE, converts to riparian habitat
	0–3 ft below WSE, remains freshwater wetland habitat
	>3 ft below WSE, converts to open water habitat
Salt Marsh	Above WSE, remains salt marsh
	0–3 ft below WSE, converts to freshwater wetland habitat
	>3 ft below WSE, converts to open water habitat
Riparian	Above WSE, remains riparian habitat
	0–3 ft below WSE, converts to freshwater wetland habitat
	>3 ft below WSE, converts to open water habitat
Developed/Disturbed	No change in habitat type
Beach/Foredune region	
Ocean	No change in habitat type
Open Beach	Above WSE, remains open beach habitat
	Below WSE, converts to open water habitat
Foredune	Above WSE, remains foredune habitat
	Below WSE, converts to open water habitat
Open Water	Above WSE, converts to open beach habitat
	Below WSE, remains open water habitat
Freshwater Wetland	Above WSE, converts to open beach habitat
	Below WSE, remains freshwater wetland habitat

Input habitat type	Future habitat type
Campground region	
Disturbed Wetland	Above WSE, converts to developed/disturbed habitat
	Below WSE, remains disturbed wetland habitat
Riparian	Above WSE, remains riparian habitat
	Below WSE, converts to disturbed wetland habitat
Developed/Disturbed	Above WSE, remains developed/disturbed habitat
	Below WSE, converts to disturbed wetland habitat
VWRF Pond region	
Open Water	Remains open water habitat for Scenarios 1–11
Riparian	Remains riparian habitat for Scenarios 1–10; converts to developed (surrounding the ponded water) or open water (within ponded water) for Scenario 11
Developed/Disturbed	No change in habitat type

5.3.1.2 Modeling results by VWRF discharge scenario

Habitat evolution under future equilibrium WSEs in the SCRE under alternative VWRF discharge scenarios was modeled by application of the rules in Table 5-4 applied to the mapped and observed 2016 current habitat types (Section 3.5.1) in a step-wise order. For example, to evaluate vegetation community types under Scenario 1 (0% diversion; existing discharge), rules in Table 5-4 were applied to the 2016 current habitat types; Scenario 2 was evaluated by applying these rules to the Scenario 1 GIS results; Scenario 3 was evaluated based on the Scenario 2 GIS results, and so forth (Appendix F). For instance, within the Riparian region, an area mapped in the 2016 mapping as open water is modeled to transition to riparian if it is above the modeled WSE, it is modeled to transition to freshwater wetland if it is between 0–3 ft below the modeled WSE, and if it is greater than 3 ft below the modeled WSE, it remains open water habitat. An area that was mapped in the 2016 mapping as riparian habitat is modeled to remain riparian if it above the modeled WSE, transition to freshwater wetland if it is between 0–3 ft below the modeled WSE, or transition to open water if it is greater than 3 ft below the modeled WSE.

Results of all future habitat types for each discharge scenario are summarized in Table 5-5 and depicted in figures provided in Appendix F. Habitat types and associated vegetation are described in Section 3.5.1. Note that vegetation associated with each habitat type is comprised of both native and non-native plant species; quality of the habitat is not assessed by this method and may vary within a habitat type. Freshwater wetlands, for example, may include areas dominated by invasive *Arundo*, which results in low quality wetland habitat. Conditions enabling *Arundo* invasion are evaluated separately under the WET beneficial use (Section 5.5.10). Around the western, northern, and eastern margins of the campground, some of the developed/disturbed areas and salt marsh (as shown in 2016 mapping) are modeled to be disturbed wetlands and freshwater wetlands (as shown in Scenario 1), respectively. These discrepancies are attributed to several factors including the time scales of freshwater wetland development in responses to average water level changes, changes to salt marsh in response to higher water levels, and assumptions of a stable equilibrium water level in the future.

Overall, several general trends are seen in the predicted future habitat types at progressively larger VWRF diversion/lower discharge volumes. In general, because simulations were evaluated only under current lagoon morphology, no changes in ocean and foredune habitat were estimated and only small increases open beach habitat are predicted as the VWRF diversions increase/VWRF discharges decrease (i.e., going from Scenario 1 (0% diversion; continued existing discharge) to Scenario 11 (100% diversion; 0 MGD discharge). The extent of riparian/riverwash habitat, which is a vital component of the SCRE, is predicted to increase with increasing diversions/VWRF discharge reductions across Scenarios 1 to 11. The extent of freshwater wetland habitat is modeled to decrease from Scenario 1 through Scenario 11 as diversions are increased and discharges are reduced because the transition to riparian habitat is predicted to outpace the colonization of open water with new wetland habitat. Open water and tidally exposed mudflat is also modeled to decrease from Scenario 1 through Scenario 11 as diversions are increased and VWRF discharges are reduced as riparian and riparian riverwash areas increase. The disturbed wetlands in the campground area revert entirely to developed/disturbed habitat by Scenarios 3 and 4.

Because of the large changes in the SCRE mouth position with flood events (Section 3.2.1), most of the predicted changes in the habitat type areas are within the range of historical variability for the habitat types within the SCRE (Section 3.5.2; Table 3-26) – including developed/disturbed, foredune, ocean, and open beach habitats. It should be noted that historical observations reflect current or greater than current VWRF discharge levels, rather than the reduced VWRF discharges assessed in Scenarios 2 through 11, which are predicted to support greater amounts of riparian vs. open water and wetland habitat types. For example, the amounts of open water and tidally exposed mudflat varied historically from 102.4 to 188.7 acres; however, for VWRF discharge scenarios diverting 80% or more (Scenarios 9–11), predicted SCRE equilibrium WSE drops below 5.8 ft NAVD88 (Figure 5-1) and the amount of open water combined with mudflat drops below what was documented historically. The amount of wetlands varied historically from 33.7 to 47.2 acres; however, for VWRF discharge Scenarios 2–11 (10% to 100% diversion) the amount of wetland habitat drops below what was documented historically. Riparian combined with riparian riverwash is the only habitat type predicted to increase with increased diversions to levels larger than those documented in 1977, 2002, 2009, or 2016 (Table 3-26) when the greatest geographic extent was estimated to be 258 acres; discharge scenarios 3–11 all estimate greater amounts of riparian and riparian riverwash between 271–324 acres. It is possible that areas modeled as riparian riverwash could develop into wetland habitats over time, given sufficient gaps between scour events. These model results are a snapshot of predicted potential future habitat and vegetation types associated with the varying discharge scenarios, which are useful for purposes of comparison of the discharge scenarios, but the model results are not necessarily likely to replicate actual conditions since the frequency and magnitude of storm events have a significant influence on the actual habitat and vegetation types that may develop in the future.

Table 5-5. Estimated amounts of future habitat types at SCRE stages predicted by water balance modeling of discharge Scenarios 1 through 11.

Scenario No. (% reduction)	1 (0%)	2 (10%)	3 (20%)	4 (30%)	5 (40%)	6 (50%)	7 (60%)	8 (70%)	9 (80%)	10 (90%)	11 (100%)
Discharge to SCRE (MGD)	4.7	4.2	3.7	3.3	2.8	2.3	1.9	1.4	0.9	0.5	0
Modeled Equilibrium SCRE Stage (ft, NAVD88)	10.2	9.8	9.3	8.8	8.1	7.5	7	6.4	5.8	5.3	4.7
Modeled equilibrium SCRE area estimate (acres) by habitat type for each VWRF discharge scenario											
Ocean	76	76	76	76	76	76	76	76	76	76	76
Open Beach	47	47	48	49	50	51	51	52	53	53	53
Foredune	76	76	76	76	76	76	76	76	76	76	76
Open Water[1]	108	104	100	96	93	88	86	77	58	49	41
Tidally Exposed Mudflat[2]	66	65	60	56	53	48	47	39	20	11	3
Freshwater Wetland	45	29	14	14	14	16	15	13	8	7	2
Salt Marsh	2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6
Riparian	218	236	252	255	257	259	261	264	269	270	256
Riparian Riverwash	15	16	19	19	20	20	20	28	45	54	62
Developed/ Disturbed	49.6	52.9	55.3	55.4	55.4	55.4	55.4	55.4	55.4	55.4	73.7
Disturbed Wetland	5.8	2.5	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

1 The estimate for “Open Water” includes the area of the SCRE that remains open water under open-mouth conditions when the SCRE reaches its minimum stage of 4.5 ft NAVD88 and the constant open water area from the VWRF ponds.

2. Mudflat habitats exposed during open mouth condition reflect the differences in inundated area between the predicted equilibrium WSEL and mean lower low water levels (MLLW) modeled in the SCRE (4.5ft NAVD88).

5.3.2 Changes in water quality conditions by discharge scenario

Estimates of future physical and chemical water quality conditions under each of the discharge scenarios in Table 5-2 were based upon both quantitative and qualitative approaches. Nutrient loads associated with each of the VWRF discharge scenarios were estimated using the nutrient balance developed in Section 4.3 and applying nutrient concentrations from Phase 3 along with other available monitoring data to average flows of the water balance components (Table 5-3) during the modeled closed-mouth, dry weather equilibrium stage condition. The analysis below centers upon variations in the amounts of background salinity (as measured by specific conductivity) and nutrients available for algal uptake such as total inorganic nitrogen (i.e., TIN as the sum of NH_4^+ and NO_3^-) as well as orthophosphate (PO_4) for each of the 11 VWRF discharge scenarios being studied in connection with diverting tertiary treated VWRF flow from the SCRE to advanced purification and subsequent potable reuse as described in Carollo Engineers (2017). Table 5-6 shows the estimated average concentrations for each inflow component accounted for by the water balance model with the resulting estimates of predicted surface water quality within the SCRE during closed-mouth, dry-weather equilibrium conditions for each discharge scenario in Table 5-7. The following water quality concentrations were based upon surface water quality measurements during the 2015-2016 Phase 3 monitoring period (Appendix B) as well as other sources noted below.

- **Santa Clara River Inflows and Surface Water Runoff -** Estimated specific conductivity and nutrient levels arriving from riverine sources were estimated as the 2015-2016 average of sites R-1 and R-005 (Table 5-6).
- **VWRF Inflows** - Estimated specific conductivity and nutrient levels arriving from the VWRF were based upon compiled 2015-2016 measurements at the outlet of the Wildlife/Polishing Ponds or site D-1 (Table 5-6).
- **Unmeasured Flow: Overwash** – Estimated specific conductivity for the wave overwash portion of the unmeasured flow was based on the typical seawater conductivity of 50,000 uS/cm (Cox et al. 1967). Nutrient levels in wave overwash were assumed to be negligible since there was insufficient data to support any assumptions of nutrient levels in the nearshore environment.
- **Groundwater sources -** Estimated nutrient levels arriving from groundwater sources were based upon groundwater wells within portions of the Santa Clara River floodplain (Table 5-6). Because no salinity or specific conductivity data was collected for these sources, specific conductivity was based upon average values for groundwater wells in the Mound Basin from the 2016 consumer water quality report (Ventura Water 2016). The actual salinity or specific conductivity from groundwater sources may be higher with measurements of electrical conductivity in monitoring wells in the semi-perched aquifer along the northwestern edge of the SCRE from a 1991 study reporting conductivity between 2160 uS/cm and 5880 uS/cm (Hopkins 2018). The 2016 specific conductivity was used for groundwater sources rather than the 1991 specific conductivity since it was a more recent measurement and the difference between the specific conductivity estimates would not significantly alter resulting SCRE specific conductivity values due to low flows from groundwater sources.

Table 5-6. Estimated surface water and groundwater quality contributions to SCRE mixing model under alternative VWRF discharge scenarios.

Flow component	Average specific conductivity (2015–2016)		Average nutrients (2015–2016)		
	Data sources	**Sp. Cond. (uS/cm).**	**Data sources**	**TIN (mg/L)**	**PO_4 (mg/L)**
Surface water sources					
Santa Clara River	R1 and R-005	2,948	R1 and R-005	2.2	0.4
VWRF	D-1	2,564	D-1	7.2	1.6
Runoff	R1 and R-005	2,948	R1 and R-005	2.2	0.4
Unmeasured Flow [1]	Seawater (Cox et al 1967)	50,000	NA	NA	NA
Groundwater sources					
North Bank Floodplain-Ponds	Ventura Water (2016) groundwater.	1,757	GW 12-15	5.0	2.6
North Bank Floodplain-West			GW 8-11	11.8	0.2
North Bank Floodplain-East			GW 4-5	21.5	0.2
South Bank Floodplain - McGrath			GW 1-3	0.9	0.2
South Bank Floodplain, Upstream			GW 6-7	NA	NA
Unmeasured Flow [2]			R1 (2016)	5.4	0.2

1 Assumed unmeasured flow dominated by wave overwash following breached berm closure
2 Unmeasured base flows within the Santa Clara River channel not represented by other groundwater sources.

5.3.2.1 Variations in equilibrium stage salinity

Salinity in the SCRE is typically very low during closed-mouth, dry-weather conditions due to the predominance of VWRF discharge combined with other fresh water inflows. Both Sonde monitoring data (Figure 3-29 through Figure 3-32) as well as modeled equilibrium stage water quality conditions (Table 5-7) indicate the SCRE specific conductivity approaches the specific conductivity of the primary flow contributions to the SCRE and generally ranges from 2,800–2,100 uS/cm (or approximately 1.1 –1.4 ppt based upon salinity conversion calculations by Schemel (2001). Overall, the closed-mouth, dry-weather equilibrium specific conductivity decreases under high VWRF diversion/lower VWRF discharge scenarios because the groundwater inputs with a lower specific conductivity than the VWRF effluent become the dominant inflow in the absence of wet weather and VWRF discharge. The equilibrium stage specific conductivity approaches the specific conductivity of the groundwater inputs as the VWRF discharge decreases, so application of higher estimates of the groundwater conductivity like those reported in Hopkins (2018) would result in higher estimates of the SCRE equilibrium stage conductivity for higher VWRF diversion/lower VWRF discharge scenarios.

Table 5-7. Equilibrium SCRE conductivity/salinity for alternative discharge scenarios 1 to 11.

Scenario (% VWRF reduction)	Discharge to SCRE (MGD)	SCRE volume (acre-ft)	SCRE conductivity (mS/cm)	SCRE salinity (ppt)
1 (0%)	4.7	443	2.8	1.4
2 (10%)	4.2	392	2.7	1.4
3 (20%)	3.7	340	2.7	1.4
4 (30%)	3.3	294	2.7	1.4
5 (40%)	2.8	232	2.7	1.4
6 (50%)	2.3	181	2.7	1.4
7 (60%)	1.9	142	2.7	1.4
8 (70%)	1.4	99	2.6	1.3
9 (80%)	0.9	66	2.5	1.3
10 (90%)	0.5	45	2.3	1.2
11 (100%)	0.0	27	2.1	1.1

5.3.2.2 Nutrient uptake and removal within the SCRE

The nutrient balance model predicts the nutrient concentrations (TIN and PO_4) in the SCRE generally decrease with increases in VWRF diversions/decreases in VWRF discharge, though only the relative change in model predicted nutrient concentrations between scenarios should be evaluated due to uncertainties associated with the nutrient balance model, including uncertainties in the amount of nutrient uptake and removal processes in the SCRE. The SCRE equilibrium stage nutrient concentrations are first modeled assuming no uptake or removal. Predicted equilibrium stage concentrations under alternative VWRF discharge scenarios range from 7.4 mg-N/L for Scenario 11 (100% diversion/no discharge) to 8.3 mg-N/L for Scenario 1 (0% diversion/existing discharge) and 0.3 mg-P/L for Scenario 11 (100% diversion/no discharge) to 1.7 mg-P/L for Scenario 1 (0% diversion/existing discharge) assuming no removal (Table 5-8). The concentrations for TIN and phosphate are generally higher at the larger VWRF discharge rates (Scenarios 1–4), but exhibit low variability in Scenarios 1–8. The one exception to this pattern occurs in Scenario 11, the lowest VWRF discharge rate, when TIN peaks at 8.3 mg-N/L. This occurs because the model predicts that diversion of 100% of the discharge per Scenario 11 results in elimination of the low TIN concentration groundwater flow from the North Bank Floodplain – Ponds since the Wildlife/Polishing ponds would no longer be filling and contributing to the local groundwater flow. In the absence of VWRF discharge and North Bank Floodplain-Ponds inflows, the higher TIN concentration groundwater flow from the North Bank Floodplain upstream of Harbor Blvd. (NBF-East) becomes a much larger percentage of the total inflow into the SCRE resulting in an increase in the SCRE equilibrium stage TIN concentration. The increase in TIN assuming no uptake or removal under Scenario 11 may be due to an overestimate in the TIN coming from groundwater inputs from NBF-East. Such variations in the NBF-East TIN concentrations only alter the predicted TIN levels in Scenarios 7 – 11, and only significantly alter the predicted TIN levels in Scenario 11.

A comparison of the predicted nutrient concentrations with observations of lower nutrients in the SCRE during 2015 under closed-mouth, dry-weather conditions (Section 3.4.1) and 2016, along with similarly lower observed nutrient levels in other inflows to the SCRE (e.g., VWRF, Groundwater, and Santa Clara River inflows) indicate that some combination of algal uptake and microbial processes (such as denitrification) effectively reduce TIN and PO_4 levels during close-mouth, dry-weather conditions and potentially in other seasons and conditions. Application of the nutrient balance model to the 2015-2016 sampling period (Section 4.3) demonstrated that inclusion of algal uptake and removal would improve agreement between modeled and observed nutrient concentrations in the SCRE. While high rates of nitrogen mineralization by denitrification have been observed in lakes, marshes and estuaries (Boynton and Kemp 2008) and removal rates were fitted from observed nutrient levels for the 2015-2016 sampling period, we have used conservative rates of TIN removal by a zero-order (areal) removal process of 50 mg-N/m^2-d. For phosphorus, we have assumed rates of algal uptake on the order of 20 mg-P/m^2-d (Boelee et al. 2011). A minimum background nutrient concentration was assigned if the nutrient balance model predicted the nutrient concentration was less than the USEPA typical nutrient concentration for sites with a high amount of human influence in the California South Coast ecoregion representing the Santa Clara River watershed (Fetscher et al. 2014).

The nutrient balance model with the assumed rates of nutrient removal (50 mg-N/m^2-d and 20mg-P/m^2-d) predicts the nutrient concentrations (TIN and PO_4) in the SCRE decrease with increased diversions/decreases in the VWRF discharge. At these uptake rates, TIN has a constant peak of 6.4 mg-N/L in Scenarios 1–4, after which the TIN concentrations consistently declines for VWRF discharge Scenarios 5-11. Note that with application of removal rates, TIN decreases with decreasing VWRF discharges, even under Scenario 11. Phosphate concentrations have a constant peak of 1.1 mg-P/L in Scenarios 1–5, then decline gradually with decreasing discharges associated with Scenarios 6–10. In Scenario 11, the nutrient balance model with nutrient removal reached the minimum background nutrient concentration for phosphate indicating the zero-order removal process is no longer applicable. In addition to the previously discussed nutrient balance model limitations and uncertainties, phosphate concentrations for Scenario 11 may be anywhere between the 0.3 mg-P/L when estimated with no nutrient uptake, and the minimum background phosphate concentration of 0.1 mg-P/L when the model with nutrient uptake predicts less than the minimum background phosphate concentration. As discussed further in Section 5.3.2.3 below, because of the inherent uncertainties in predicting nutrient uptake and removal processes such as algal growth as well as denitrification, linkages between nutrient levels and DO conditions for aquatic species were assessed using simplified nutrient loading assumptions.

Table 5-8. Flow weighted equilibrium nutrient concentration estimates from SCRE mixing model during closed-mouth, dry-weather conditions for discharge scenarios 1 through 11.

Scenario (% VWRF discharge reduction)	Discharge to SCRE (MGD)	Specific Cond. (mS/cm)	TIN (mg-N/L) at assumed algae uptake/denitrification rates [1]		PO_4 (mg-P/L) at assumed algae uptake rates [2]	
			0 mg-N/m²-d	50 mg-N/m²-d	0 mg-P/m²-d	20 mg-P/m²-d
1 (0%)	4.7	2.8	8.0	6.4	1.7	1.1
2 (10%)	4.2	2.7	7.9	6.4	1.7	1.1
3 (20%)	3.7	2.7	7.8	6.4	1.7	1.1
4 (30%)	3.3	2.7	7.8	6.4	1.7	1.1
5 (40%)	2.8	2.7	7.8	6.2	1.7	1.1
6 (50%)	2.3	2.7	7.8	6.1	1.7	1.0
7 (60%)	1.9	2.7	7.8	5.9	1.7	0.9
8 (70%)	1.4	2.6	7.7	5.8	1.6	0.9
9 (80%)	0.9	2.5	7.6	5.8	1.5	0.8
10 (90%)	0.5	2.3	7.4	5.6	1.4	0.7
11 (100%)	0.0	2.1	8.3	5.1	0.3	0.1[3]

1 Adapted from Boynton and Kemp (2008) assuming either zero or low estimates of areal TIN removal by algal uptake as well as denitrification.
2 Adapted from Boelee et al (2011) assuming either zero or low estimates of phosphorus uptake by algal uptake.
3 Minimum background PO_4 concentration reached based on USEPA typical nutrient concentration for sites with a high amount of human influence in the California South Coast ecoregion (Fetscher et al. 2014).

5.3.2.3 Variations in dissolved oxygen with the SCRE

Semi-hourly sonde measurements of DO during the Phase 3 monitoring (2015–2016) show generally conditions unsuitable for aquatic life at the North Sonde deployment in the VWRF outfall channel, with DO < 5 mg/L in 69% of records (Table 3-10). DO was generally >5 mg/L in 92–97% of semi-hourly *in situ* measurements from 2015–2016 at the South and Central Sonde deployment locations in the SCRE, indicating DO levels under current conditions are generally suitable throughout most of the SCRE during most periods. However, periods of unsuitable DO do occur intermittently within the SCRE, such as occurred in spring 2015 (Section 3.4.1.3). Nevertheless, the SCRE is relatively shallow and re-aeration by wind-mixing is relatively high, so extensive areas within the SCRE exhibiting hypoxic conditions were not observed. Re-aeration across the air-water interface as well as photosynthetic DO production, offset by algal die-offs[22], can be affected by light availability, wind, and temperature, which can fluctuate both regularly and intermittently throughout a diel cycle. For example, wind-driven mixing and the aeration of the estuary surface layer especially in shallow lagoons and estuaries is a potentially significant influence on estuary DO concentrations (Scully 2010, Bruner de Miranda et al. 2017).

Because several biotic and abiotic factors discussed above combine to affect DO in the SCRE, quantitative (i.e., predictive) modeling of DO under alternative scenarios was not considered feasible based on information collected as part of the Phase 3 Study. Overall, relative to current

[22] The availability of organic matter from algal detritus increases the biological oxygen demand by heterotrophic microorganisms, thus depleting DO from the water column.

conditions, decreases in VWRF discharge are expected to decrease nutrient concentrations in the SCRE, and relative to the absence of discharge, increasing discharges are predicted to increase nutrient concentrations in the SCRE (Table 5-8). For example, because TIN load reductions have been associated with reductions in algal blooms other southern California estuaries and lagoons (Sutula et al. 2006), some reduction of the incidence of low DO conditions might be expected under increased diversion/reduced VWRF discharge scenarios. However, even with expected decreases in nutrient concentrations associated with increased diversion/reduced VWRF discharge, modeled nutrient concentrations remain well above typical saturation levels for growth (0.2 mg/L for inorganic nitrogen and from 0.02–0.08 mg/L for orthophosphate) (Horne and Goldman 1994) due to elevated nutrients in groundwater and other surface water inputs. Algal blooms under reduced VWRF discharge scenarios are likely to still occur, though the frequency or duration of blooms may be reduced.

5.3.2.4 Variations in water temperatures in the SCRE

Variations in the VWRF discharge under alternative discharge scenarios are not anticipated to significantly alter SCRE water temperature because the SCRE equilibrium heat balance modeling results indicate that summer water temperatures during closed-mouth, dry-weather conditions in the SCRE are primarily driven by heat exchange fluxes across the SCRE water surface from evaporation, solar radiation, and long wave radiation. While VWRF effluent discharge is the predominant flow source during the conditions modeled (Figure 4-16), VWRF discharge is not the dominant heat flux (Figure 4-20). Sensitivity testing of parameters in the heat balance model shows variations in the VWRF discharge under alternative discharge scenarios would potentially alter the SCRE equilibrium stage water temperature by approximately +/- 0.5°C. Variations in the SCRE equilibrium water temperature from sensitivity testing only provide an order of magnitude estimate though since the heat balance model assumes simplified equilibrium conditions as discussed in Section 4.4.

A qualitative evaluation of potential water temperature conditions suggests the SCRE water temperature would potentially increase slightly under reduced VWRF discharge scenarios. In a shallower lagoon under reduced VWRF discharge scenarios, the mean water temperature would be expected to respond more quickly to air temperatures, particularly in summer, relative to a deeper equilibrium stage lagoon. Additionally, the absorption of solar radiation by lagoon sediments would also be expected to increase with decreasing equilibrium lagoon depth, resulting in increasing transmission of heat from sediments to the water column and potential increases in mean water temperatures relative to current conditions. A shallower equilibrium water depth under reduced VWRF discharge scenarios may also alter the potential for localized thermal stratification in the estuary. Estuary water temperature profile measurements (Appendix D) document thermal stratification was relatively infrequent in the SCRE with most instances related to density stratification due to wave overwash or seawater exchanges during open-mouth conditions. This indicates the shallow water of the SCRE is usually characterized by a uniform or well-mixed vertical water column. Assuming no change in the other factors influencing localized thermal stratification, decreases in the water depth under alternative discharge scenarios would result in the water column reaching well-mixed conditions more rapidly and decrease the potential for localized thermal stratification. Cumulatively, the information above suggests that the increased discharges and equilibrium lagoon depths results in somewhat lower temperatures than under reduced discharge scenarios, but reductions in VWRF discharge are not predicted to significantly increase estuary water temperature.

5.4 Variations in SCRE Breaching Dynamics and Associated Habitat Conditions by Discharge Scenario

In order to examine the relative influences of VWRF discharges upon SCRE breaching dynamics, modeling was conducted with the calibrated water balance to examine changes in breaching frequency, duration of open mouth conditions, as well as modeling of salinity variations following berm closure after a hypothetical breaching event.

5.4.1 Relative open-mouth duration for representative water year types

The water balance model was set-up to simulate SCRE hydrologic/hydraulic conditions under three representative water years (i.e., wet, normal, and dry) for all eleven discharge scenarios (i.e., 0% through 100% reductions) to assess differences in the modeled timing of beach-berm breaching and closure during the given water years. Water-year types were determined for this study based on equal partitioning of ranked annual total runoff amounts from the lower Santa Clara River between 1928 and 2016 (n=89 years). Because of data gaps in the compiled record of river flow, water-year typing based on streamflow recorded in lower Sespe and Santa Paula creeks and rainfall recorded in downtown Ventura and near Santa Paula were used to supplement the water-year type determinations as applicable to the lower watershed and SCRE. Results of the water-year typing analysis are presented in Appendix F. The representative "wet," "normal," and "dry" water years selected for the modeling were 2011, 2009, and 2015, respectively.

The modeling approach treats the three selected water years as representative water-year types (and expected to occur in the near future) using hydrologic inputs measured during those years, including streamflow, precipitation, evaporation, subwatershed runoff, tidal elevation, and tidal range, while applying a constant inflow of VWRF over the modeling duration based on the discharge-reduction scenarios. The modeling begins each water year with a closed-mouth and empty lagoon condition at elevation 4.5 ft (NAVD88) to aid the subsequent comparisons of results from the VWRF scenarios. A total of thirty-three individual model runs were performed: eleven discharge scenarios (0%, 10%, 20%, 30%, 40%, 50%, 60%, 70%, 80%, 90%, and 100%) *x* 3 representative water-year types.

The results of the model runs for the 0%, 50%, and 100% discharge scenarios for each water year type are displayed in Figure 5-3 through Figure 5-5 where they are grouped by water-year type. Also displayed on these figures are the periods of open-mouth status for the three discharge scenarios. The predicted SCRE water surface elevations exhibit marked differences between the three discharge scenarios and water-year types, which are primarily driven by differences in all the inflows to the SCRE. The maximum water surface elevation as the SCRE approaches a quasi-equilibrium state during filling is higher with less VWRF discharge reduction (see also Figure 5-1). The magnitude of the initial increase in water surface elevation after a mouth closure is primarily due to the wave overwash component of the unmeasured flow (see Section 4.1.9). As previously discussed, the magnitude of the wave overwash is a source of uncertainty and may vary resulting in an observed initial increase in the water surface elevation when the berm breach closes that is different than the increase predicted by the model. However, the results presented are still anticipated to capture the overall SCRE trends in breaching frequency and related changes in habitat conditions associated with the various VWRF discharge scenarios because the influence of the wave overwash component diminishes with time from the berm breach closure, as the SCRE fills and approaches an equilibrium water surface elevation. While the rates of filling and the equilibrium water surface elevation are variable between the modeled VWRF discharge scenarios, the specific timing of the initial berm breach under higher discharge levels is primarily determined in the model by the timing of storms that cause the estuary to fill. The model predicted open-mouth duration generally varies only slightly (e.g., <1 day) across VWRF discharge scenarios for normal and dry years, but during wet year conditions, higher total outflow from the SCRE and variations in the timing of initial breaching can cause the predicted duration of open-mouth conditions to vary between discharge scenarios.

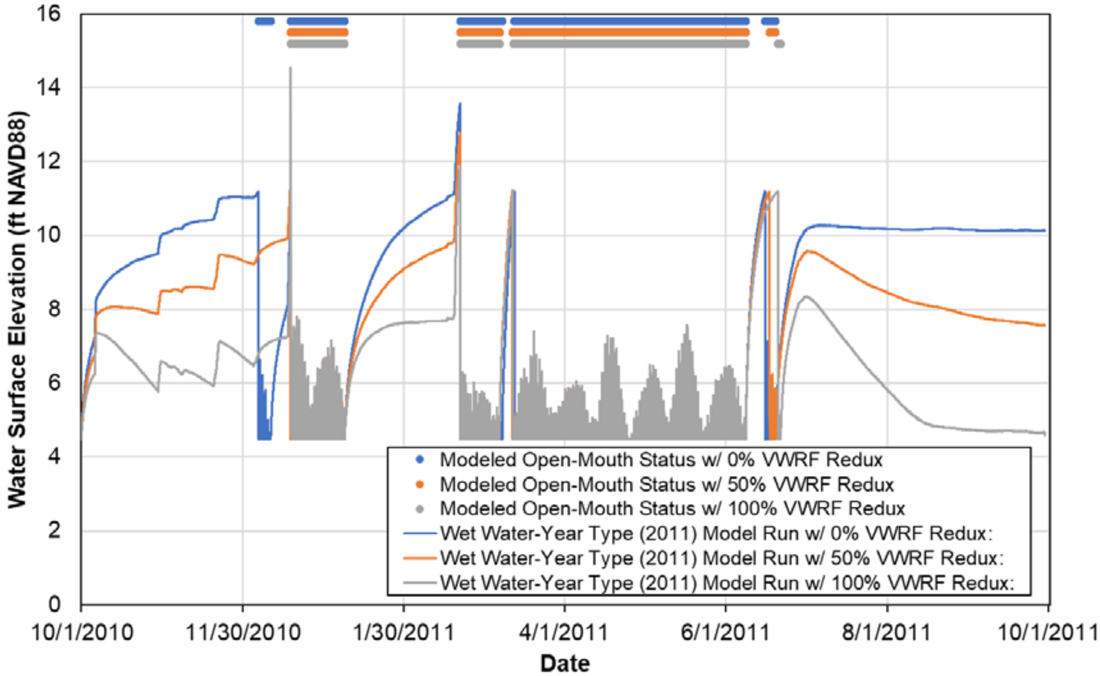


Figure 5-3. Modeled SCRE stage for three VWRF discharge scenarios during the representative wet water-year type (WY 2011).

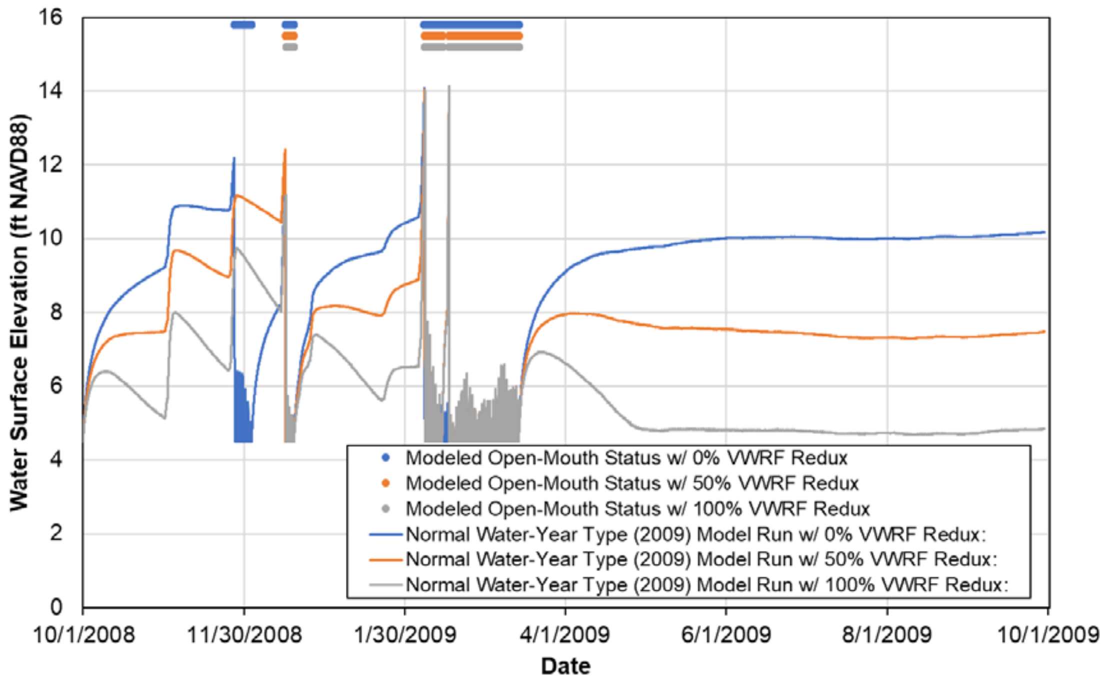


Figure 5-4. Modeled SCRE stage for three VWRF discharge scenarios during the representative normal water-year type (WY 2009).

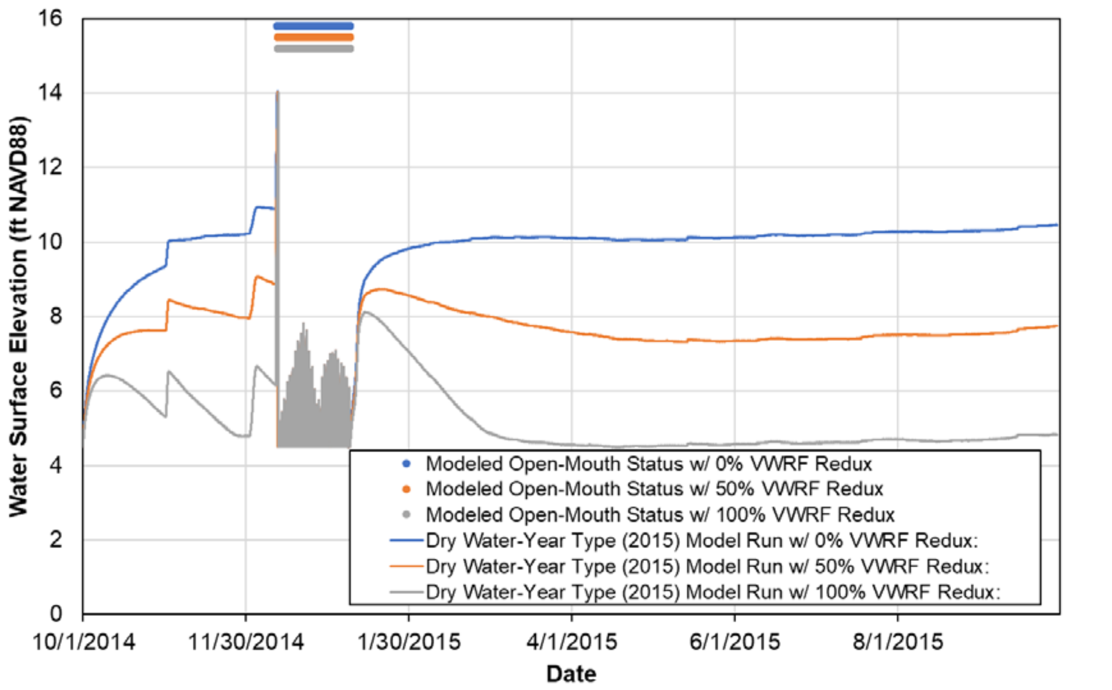


Figure 5-5. Modeled SCRE stage for three VWRF discharge scenarios during the representative dry water-year type (WY 2015).

SCRE beach berm-breaching and the duration of open-mouth conditions are primarily a function of SCRE water surface elevation (volume), storm influence, and tidal conditions, but VWRF discharge levels do influence the timing of berm breaching and the resulting duration of open-mouth conditions, primarily for wet water year conditions. The timing of the initial berm breach is primarily controlled by the timing of storms that cause the SCRE to fill to the level that triggers a breach. Higher VWRF discharge levels are generally predicted to result in slightly earlier berm breaching during wet water years because the additional water volume from VWRF discharge results in the SCRE water surface elevation being at a higher level when a storm event causes rapid filling and triggers breaching. Variations in the timing of berm breaching under different VWRF discharge scenarios for a water year type are usually less than a day for dry and normal water years. The timing of berm breaching under different VWRF discharge scenarios is more variable under wet water year conditions with breaches typically occurring within less than one day for different VWRF discharge scenarios, but modeled results show a December berm breach occurring 8 days later (0% to 10% VWRF reduction) and a June berm breach occurring 5 days later (0% to 100% VWRF reduction). The model only shows the December berm breach occurring during 0% and 10% VWRF reductions with open-mouth conditions lasting approximately 3 days (0% VWRF reduction) or 1 day (10% reduction). The closure of the beach berm following a breaching event usually occurs at the same time for all discharge scenarios during a given water-year type since the timing of berm closure is primarily driven by tidal conditions and SCRE outflows through the breach. Variations in the VWRF discharge do not generally influence the timing of berm closure since VWRF discharges do not alter tidal influences that encourage berm closure and the SCRE outflow that maintains open-mouth conditions during a breach is only negligibly increased by VWRF discharges under most conditions. However, the model did suggest that under wet water year type conditions the small increase in SCRE outflow from VWRF discharge could sometimes contribute to maintaining open-mouth conditions longer.

The magnitude of differences in open-mouth conditions resulting from changes in VWRF discharge scenarios varies with water year type, with wetter water year types having more breaching events and longer open-mouth periods under higher VWRF discharge levels (Figure 5-3 through Figure 5-5 and Table 5-9). In the dry water year type, there is only one breaching event for all discharge scenarios 1-11, and the duration of open-mouth conditions varies by only 9 hours between 0% diversion and 100% diversion. In normal year types, there are between three and four breaching events with the duration of open-mouth conditions ranging from 45 days (0% VWRF diversion) to 36 days (100% VWRF diversion). An initial berm breach under the normal water year type in late November under the higher VWRF discharge Scenarios 1-4 (0% to 40% VWRF diversion) does not occur under lower discharge levels. The duration of open-mouth conditions for normal water year types varies by approximately 3 days between Scenarios 1-4 (i.e., 0% to 40% VWRF diversion), but then the absence of the late-November berm breach when the VWRF discharge decreases further (i.e., > 40% VWRF diversion) results in a decrease in open-mouth conditions by 6 days between Scenarios 4 and 5. There is no difference in the number of breaches for VWRF discharge scenarios 5-11 for a normal water year type and the duration of open-mouth conditions varies by less than 1 day under these scenarios. In the wet water year type, there is an initial berm breach in early December under higher VWRF discharge levels (i.e., 0% and 10% diversion) that does not occur under lower VWRF discharge levels. The duration of open-mouth conditions in wet water years decreases by 4 days between Scenario 1 and 2 (i.e., 0% and 10% VWRF diversion), but then progressively decreases by a day or less with decreases in the VWRF discharge.

Table 5-9. Observed and modeled No. of days SCRE mouth berm open for three VWRF discharge scenarios (MGD) during biologically relevant periods of the three representative water-year types.

Scenario		Season or Time Period: Entire water year (Oct 1–Sept 30)	Steelhead-migration window (Nov 1–May 31)	Summer months (Jun 1–Sept 30)
Wet Weater Year (2011)	Observed	194	132	66
	1	132	120	12
	2	128	117	12
	3	127	116	11
	4	127	116	11
	5	126	115	11
	6	126	115	10
	7	125	115	10
	8	125	115	10
	9	124	115	9
	10	124	115	9
	11	124	115	9
Normal Water Year (2009)	Observed	80	63	17
	1	45	45	0
	2	45	45	0
	3	43	43	0
	4	43	43	0
	5	42	42	0
	6	36	36	0
	7	36	36	0
	8	36	36	0
	9	36	36	0
	10	36	36	0
	11	36	36	0
Dry Water Year (2015)	Observed	3	3	0
	1	27	27	0
	2	27	27	0
	3	27	27	0
	4	27	27	0
	5	27	27	0
	6	27	27	0
	7	27	27	0
	8	27	27	0
	9	27	27	0
	10	27	27	0
	11	27	27	0

Table 5-9 presents the total proportion of time (days) during a given water year when the SCRE mouth is predicted to be open under the different VWRF scenarios. Overall, the open-mouth duration is predicted to be greater during wetter water-year types and higher VWRF discharge scenarios (i.e., lower VWRF diversion). The total number open-mouth condition days during both wet and normal water-year types are predicted to decrease for the lower VWRF discharge scenarios, but open-mouth duration during dry water year types would remain the same regardless of the VWRF discharge scenario. During wet years, the percentage of total time open-mouth conditions occur decreases by 1-2% (6-8 fewer days) with decreasing VWRF discharge across Scenarios 1 through 11. In normal years the percentage of total time open-mouth conditions occur decreases 2% (9 fewer days) with decreasomg VWRF discharge across Scenarios 1-5 (i.e., 0% to 50% VWRF discharge diversion), but there is no additional decrease in the percentage of time open-mouth conditions occur with additional VWRF discharge decreases associated with Scenarios 7-11 (i.e., 60% to 100% VWRF discharge diversion).

Table 5-9 also presents the total duration the mouth would be open during the steelhead migration window (November 1–May 31), representing ocean-lagoon passage opportunities for upstream migration and emigration for each year type without consideration of whether the flows upstream of the SCRE in the Santa Clara River are sufficient to allow passage or support migration of steelhead. In wet years, the duration of open-mouth conditions during the steelhead migration season is predicted to gradually decrease by 5 days across Scenarios 1-5 (i.e., 0% to 40% VWRF discharge diversion), then the duration of open-mouth conditions remains the same with changes in VWRF discharge from Scenario 5-11 (i.e., 40% to 100% VWRF discharge diversion). In normal years, the duration of open-mouth conditions during the steelhead migration window decreases 9 days as VWRF discharge decreases in Scenarios 1-11, but the changes in the open-mouth conditions are not evenly distributed across scenarios. The duration of open-mouth conditions during the steelhead migration window in a normal water year decreases by 3 days across Scenarios 1-5 (i.e., 0% and 40% VWRF discharge diversion), decreases another 6 days between Scenarios 5 and 6 (i.e., 40% to 50% VWRF diversion), and then remains the same with changes in VWRF discharge from Scenario 6-11 (i.e., 50% to 100% VWRF diversion). These reductions in the duration of open-mouth conditions during wet and normal water years are primarily due to a breach during late November/early December occurring during higher VWRF discharge scenarios, but not occurring under lower discharge scenarios. In dry years, the duration of open-mouth conditions during the steelhead migration season does not change with reductions in VWRF discharge.

While the current modeling effort focused specifically on the SCRE ocean-lagoon passage opportunities for steelhead, the United Water Conservation District Multiple Species Habitat Conservation Plan (HCP) Study (Stillwater Sciences 2016) modeled both the steelhead migration opportunities through the SCRE and upstream of the estuary. Both the influence of variations in VWRF discharge and potential variations in upstream diversions were evaluated on the SCRE mouth conditions and steelhead migration opportunities upstream of the lagoon. In that study, the available habitat with passable (> 1 foot) depths during the steelhead migration window did not noticeably change (+/- 3 acres) with reduced VWRF discharge. Similar to the current modeling results, the time the mouth would be open or closed during the steelhead migration would decrease less than 5% under reduced VWRF discharge scenarios. However, the proportion of time the mouth would be open or closed when flows at the "critical riffle[23]" exceed 80 cfs during the steelhead migration window would be unchanged under reduced VWRF discharge scenarios since the SCRE water surface elevation and mouth status during that time period is primarily influenced by river flows rather than VWRF discharges. Variations in VWRF discharge in the SCRE would not alter the annual hydrograph or interannual variability of the hydrograph

[23] Critical riffles are defined by CDFG (2013) as shallow riffles in a river channel where diminished water depth may limit the hydrologic connectivity of river habitats and impede critical life history tactics of salmonids. . . . The current location of the critical riffle on the mainstem Santa Clara River based on recent surveys is in the Oxnard Forebay reach between the Freeman Diversion and the SCRE (Stillwater Sciences 2016).

components upstream of the estuary so the previous modeling results would be expected to still be applicable upstream of the SCRE.

The percent-time the mouth would be open during the typically drier months of a given year (June 1–September 30) is predicted to lack a correlation between water-year type and is only weakly related to reductions in VWRF discharge (Table 5-9). The open-mouth duration in this period shows a gradual 3-day variation associated with changes in the discharge reduction scenario during a wet water-year type, but there is no variation in the open-mouth duration with discharge reduction in other water year types.

Modeled results underpredict summer breach frequencies in all water year types except dry years, when modeled and observed breaches were zero (Table 5-9). Unseasonal breaches are frequently caused by unauthorized manual trenching of the beach berm, as storm-driven breaching is highly uncommon in June–October (Section 3.3.5; Table 3-4). Because such mechanisms are not built into the water balance model, the model does not capture unseasonal breaching (Table 5-9), and it is not a suitable tool for evaluating the effects of VWRF discharge scenarios on unseasonal breaching. Instead, the effects of discharge scenarios on summer breach dynamics were assessed using the difference in elevation between the dry season berm height and equilibrium water surface elevation as a proxy for the likelihood of an unseasonal breach. This metric assumes that likelihood of manual trenching (or any other non-flow related breach triggers) increases as the water surface elevation approaches the berm crest elevation. Water surface elevation time series and recent topographic surveys indicate that wet season (November–May) berm height is typically ~14–17 ft (Section 3.3.5), while breaches during the dry season have occurred at considerably lower water surface elevations, reflecting lower berm heights from reduced summer wave energy, as well as the effects of manual berm trenching (e.g., Table 3-4). Model calibration for 2015 found a low-flow water surface elevation of 11.2 ft as the breach trigger (see Section 4.2.1). Though this value represents the water surface elevation, rather than berm elevation, this value was used to represent a conservative estimate of berm height against which to compare the WSE for estimating the likelihood of unseasonal breaching. Note that actual berm height is highly variable and is thus a source of potential uncertainty in our comparisons of discharge scenarios. However, estimates of the relative risk of unseasonal breaching should be robust despite this potential uncertainty in berm height.

5.4.2 Salinity variations following SCRE breaches

Considering both open-mouth and closed-mouth conditions encountered during grab sampling events between 2012–2016, salinity levels in the SCRE are most notably increased by tidal exchange during open-mouth periods as well as by wave overwash events during refilling of the SCRE. For example, specific conductivity measurements varied from 1,939–50,106 uS/cm during open-mouth conditions and 989–44,188 uS/cm during closed-mouth conditions within the SCRE (Table 3-5 through Table 3-8), or approximately 0.6–33 ppt and 0.6–27 ppt during open- and closed-mouth conditions, respectively. The highest specific conductivity measured in the SCRE occurred either immediately following a mouth closure or from suspected wave overwash events during refilling of the SCRE.

SCRE salinity at the closure of the mouth berm may be higher or lower depending on a number of factors such as the duration of antecedent open mouth conditions, amount of river discharge, as well as whether closure occurs during flood or ebb tides. Modeling of specific conductivity over time indicates reductions in VWRF discharge result in higher SCRE salinity following mouth closure since the wave overwash makes up a higher percentage of the incoming flow. Even assuming initial SCRE salinities are similar to those under equilibrium water surface elevation conditions in the lagoon (Table 5-7), as we have done here, the initial salinity increase from wave overwash under lower VWRF discharge scenarios also results in a longer time period until the estuary approaches the equilibrium water surface elevation conditions. Model calibration of the wave overwash flow component during the water balance suggests the magnitude of the salinity increase from wave overwash may vary seasonally, but the general pattern of increase followed by exponential decline would be consistent over the entire year and is accurately represented by the model (Figure 5-6). In order to examine the effect of VWRF discharge levels upon salinity conditions relevant to individual beneficial uses of the SCRE, the modeling results shown in (Figure 5-6) were replotted as an event duration (days) to show the number of days above salinity thresholds benefiting native estuarine fish (Section 5.5.2.1) and selecting against tidewater goby predators and competitors (Section 5.5.7.1) (Figure 5-7). Additional details about the salinity thresholds considered in Figure 5-7 are found under the relevant beneficial uses in Section 5.5 below.

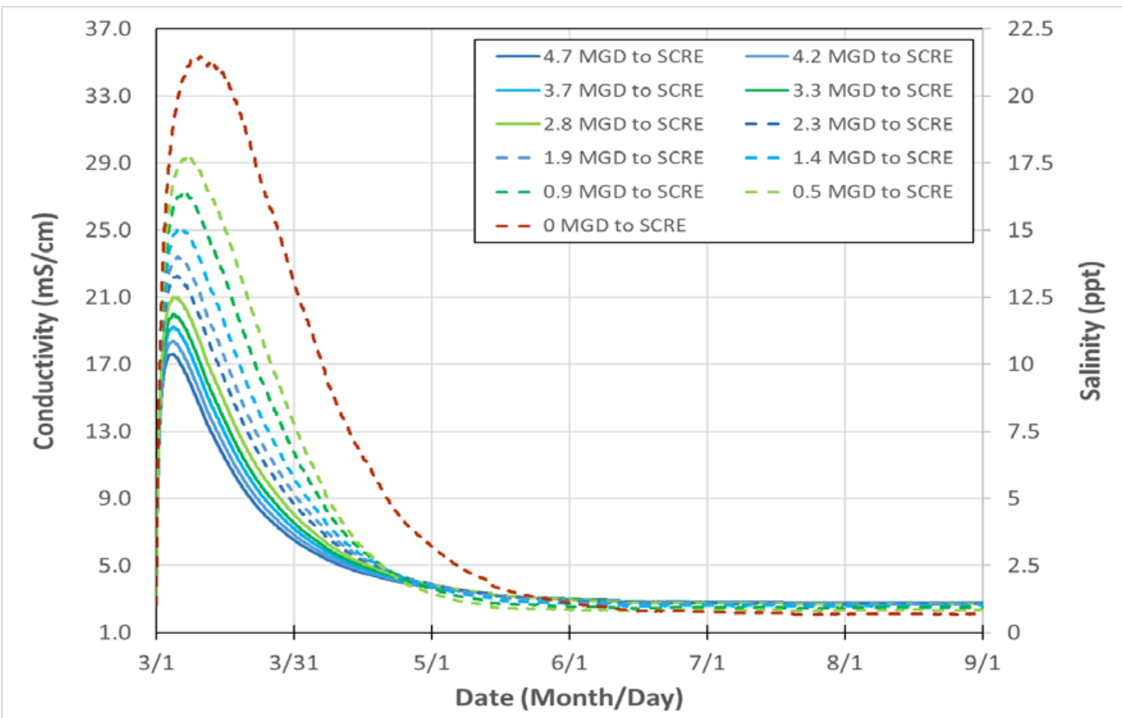


Figure 5-6. Modeled SCRE conductivity under alternative discharge scenario.

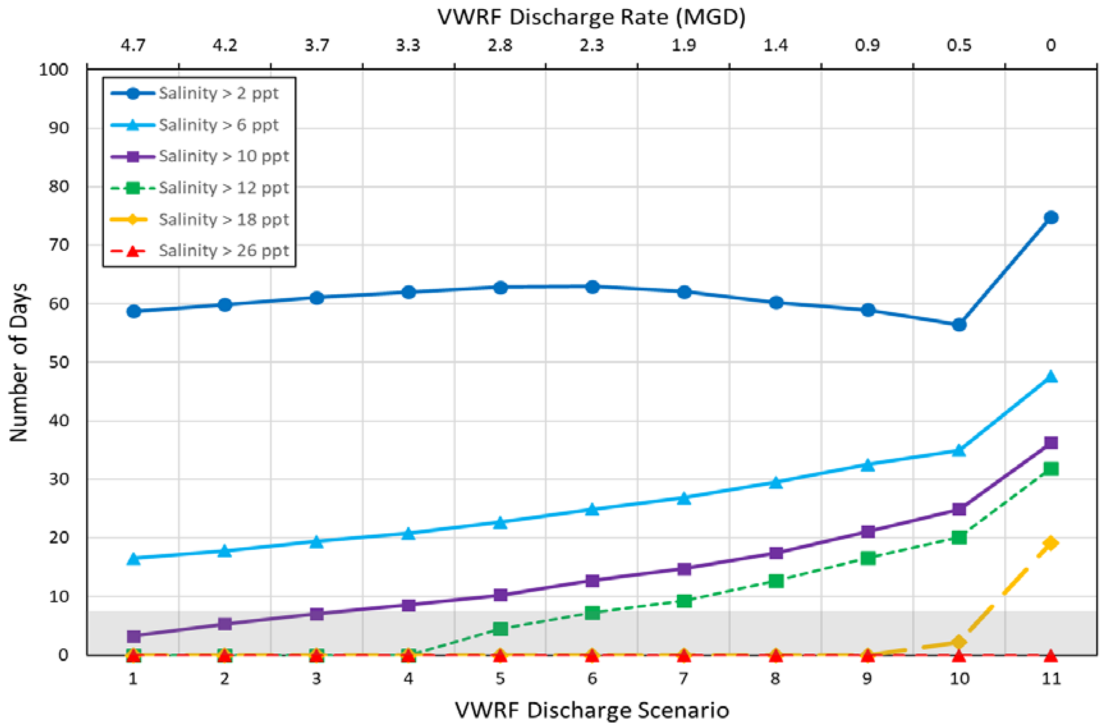


Figure 5-7. Number of modeled days exceeding salinity thresholds following a simulated mouth closure and subsequent estuary filling as a function of discharge scenario. Shaded area shows the first 7 days after mouth closure.

Overall, the model simulations indicate that peak salinity in the SCRE as well as the duration of elevated salinity conditions will both increase under reduced VWRF discharge scenarios since salinity introduced by tidal exchange and overwash will be diluted less and more slowly relative to current conditions by VWRF discharge. These changes will result in the SCRE having longer periods of higher salinity after a mouth breach closes with reduced VWRF discharges. However, the mouth breach frequency and duration of open-mouth conditions are also likely to decrease under reduced VWRF discharge scenarios, at least in normal and wet years, potentially resulting in reduced ocean inputs to the SCRE and potentially moderating salinity increases associated with reduced VWRF discharge (Figure 5-3 through Figure 5-5).

5.5 Scoring of Discharge Scenarios by Beneficial Use

In the following sections, each of 11 SCRE designated beneficial uses, as well as MUN uses in the SCRE watershed are considered in isolation, and the degree to which each VWRF discharge scenario supports the beneficial use is determined by the selected factors or criteria contributing to the beneficial use, and thresholds for those factors used to distinguish whether each discharge scenario would more fully realize (Score of 2), may or may not more fully realize (Score of 1), or would not support (Score of 0) the factor for a given beneficial use. For each beneficial use, the final scores that a VWRF discharge scenario receives are based on the scores it receives for factor considered for the beneficial use, the weight that each factor receives in assessing each beneficial use, and the weighting that each beneficial use receives relative to other beneficial uses. Following analysis and scoring, Section 5.6 presents a synthesis with the final score for each

VWRF discharge alternative representing a weighted summation of these scores across all assessment factors and beneficial uses represented. Beneficial uses are ordered for discussion in this section in alphabetical order consistent with their ordering in the Basin Plan. The weighting of beneficial uses and factors contributing to each beneficial use discussed in the subsections below was determined through the AHP framework with TRT experts, the Resources Agencies, and other stakeholders using their best professional judgment. See Appendix H for a more detailed discussion.

5.5.1 Commercial and Sport Fishing (COMM)

Commercial and Sport Fishing as a beneficial use is defined by the Basin Plan as "[u]ses of water for commercial or recreational collection of fish, shellfish, or other organisms including, but not limited to, uses involving organisms intended for human consumption or bait purposes." COMM was assigned a relative weighting of 1% in the AHP framework (see Section 5.6 for more details).

5.5.1.1 COMM Factors and Associated Metrics for AHP Analysis

Although the SCRE is not extensively used for commercial or sport fishing and COMM is not related to the primary focus of the Phase 3 Studies to protect sensitive native species and habitats, particularly those protected by state and federal law, the following factors were used to assess the relative effects of VWRF discharge on the COMM beneficial use:

1. *Physical habitat amount (i.e., open water area) for sport and bait fish species*. The amount of habitat for sport and bait fish species is quantified by the percent of the maximum open water habitat simulated for each VWRF discharge scenario by the water balance under equilibrium water surface elevation closed-mouth, dry-weather conditions. Although large amounts of open water habitat will generally support greater fish biomass, COMM uses are expected to be supported across a wide range of open water habitat areas, given the broad representation of these species in monitoring data across a range of conditions (Section 3.6). The following thresholds were adopted for comparing the relative extent to which discharge alternatives support habitat area for COMM species:
   - Score of 2: 100% of the maximum modeled open water habitat area
   - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum modeled open water habitat area
   - Score of 0: < 20% of maximum modeled open water habitat area
2. *DO conditions for sport and bait fish species*. While some COMM species are relatively tolerant of low DO conditions (Table 3-32), extended periods of DO < 5 mg/L is expected to be detrimental to the species supporting the COMM beneficial use. Available data do not provide sufficient information for accurate predictive modeling of DO under alternative VWRF discharge scenarios. Because hypoxia events are driven by algal blooms, which in turn depend on elevated nutrient concentrations, the relative potential for low DO events was quantified by comparing the percent reductions in existing nutrient (N, P) loads associated with each VWRF discharge scenario. Given that background nutrient levels in water sources flowing into the SCRE other than VWRF discharge, potentially including groundwater inflows from the surrounding floodplains, are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios, even Scenario 11 (i.e., 100% reduction in VWRF discharge or 0 MGD VWRF discharge). However, nutrient load reductions due to reduced VWRF

discharges in Scenarios 1–11 (i.e., 0% diversion to 100% diversion) might reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in nutrient loading.

- Score of 2: 100% reductions in VWRF N and P loads
- Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
- Score of 1: 0% reductions in VWRF N and P loads

5.5.1.2 Relative Weights of COMM Factors

Physical habitat area was given 83% of COMM weighting and DO conditions were given 17%. Fishing opportunities in the SCRE depend on the abundance of target species. Physical habitat area was weighted more heavily than DO conditions on the basis that most COMM species (Table 3-30) are tolerant of low DO conditions (Table 3-32), so COMM species abundance is not strongly controlled by DO conditions. Rather, abundance is thought to be controlled predominantly by habitat availability for COMM species.

5.5.1.3 Scoring VWRF Discharge Scenarios for AHP Analysis of COMM

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each discharge scenario are summarized in Table 5-10. Open water habitat area is maximized under current VWRF discharge (108 acres) (Scenario 1) and declines nearly linearly with decreasing VWRF discharge (Figure 5-8; Table 5-5). Scores ranged from 2.0 (Scenario 1) to 0.4 (Scenario 11) (Table 5-10). Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing VWRF discharge, with each 10% reduction in VWRF discharge resulting in a 10% decrease from current nutrient loads. Scenario scores for DO conditions (using nutrient loads as a proxy) ranged from 1.0 for Scenario 1 (i.e., 0% diversion/existing VWRF discharge) to 2.0 for Scenario 11 (i.e., 100% diversion/0 MGD VWRF discharge) (Table 5-10). Overall, VWRF discharge scenarios 1 and 2 score the best for supporting COMM at 1.8 with the scores decreasing gradually for Scenarios 3-8 from 1.7 to 1.4 before decreasing more rapidly for Scenarios 9-11 from 1.0 to 0.7.

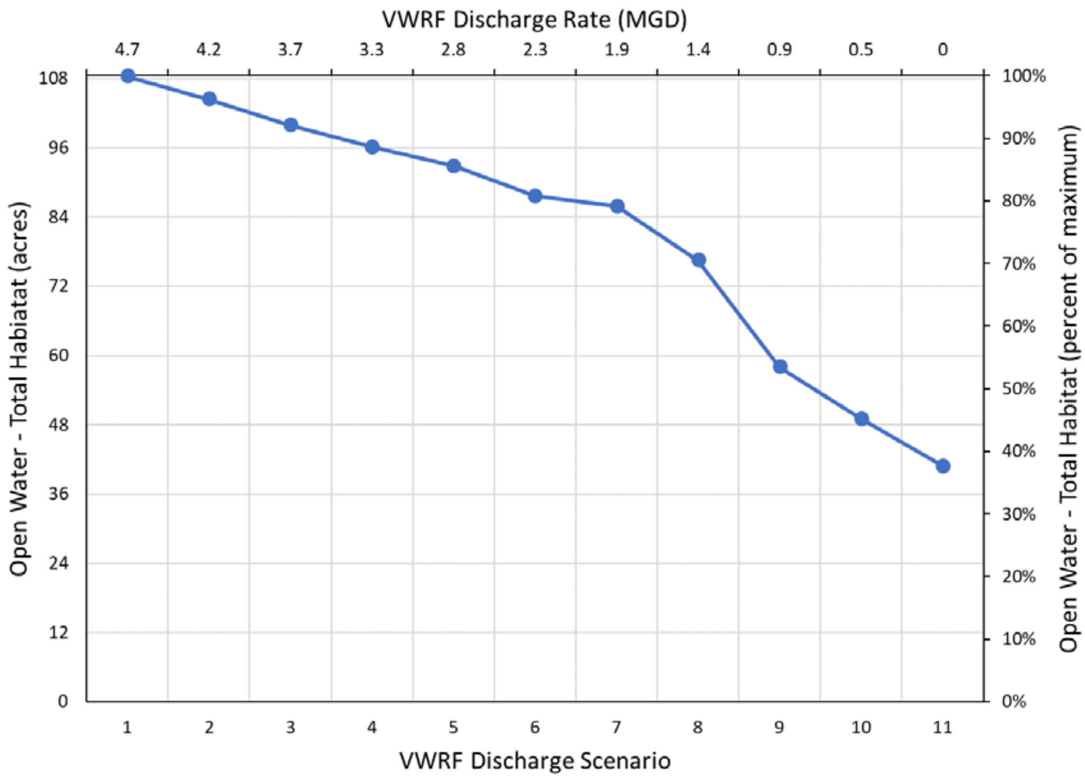


Figure 5-8. SCRE open water habitat area as acres (primary vertical axis) and percent of maximum modeled (secondary vertical axis) by VWRF discharge scenario.

Table 5-10. Scores received by each VWRF discharge scenario for each of the factors affecting the COMM beneficial use.

Factor	Tier 2 Weight	Scores by VWRF Discharge Scenario										
		1	2	3	4	5	6	7	8	9	10	11
Habitat area	83%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
DO conditions	17%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Tier 2 weighted sum	100%	1.8	1.8	1.7	1.6	1.6	1.5	1.5	1.4	1.0	0.8	0.7

5.5.1.4 COMM Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for the COMM beneficial use:

- *Benthic habitat for harvestable shellfish species*: While opportunities for commercial or recreational harvest of shellfish are included in the definition of the COMM beneficial use, the SCRE is not listed for the SHELL beneficial use, which includes harvest of shellfish for human consumption or sport purposes.
- *Salinity conditions suitable for freshwater sport fish following breach events*: While salinity tolerance for sport and bait fish targets varies, freshwater salinities (< 2 ppt) are

suitable for all of the target bait and sport fishes (Table 3-30 and Table 3-32). Modeled duration of salinity conditions exceeding 2 ppt (i.e., unsuitable for COMM species) following a simulated breach event was high (> 55 days) and relatively constant across all VWRF discharge scenarios (Figure 5-7).

- *Long-term equilibrium salinity*: The long-term, dry-weather equilibrium water surface elevation salinity of the SCRE is expected to be freshwater under all discharge scenarios based on the predominance of freshwater inputs other than VRWF discharge during these conditions.
- *Food resources*: Food resources as represented by BMI and prey fish abundance and species composition are considered to be largely controlled by WQ factors (e.g., DO, salinity) already assessed for this beneficial use. However, available data are insufficient for quantitatively predicting food resource abundance and composition under alternative VWRF discharge scenarios. See Section 3.6 for discussion of BMI abundance and composition, diet and feeding habits of COMM and other fish species, and competition for food resources.
- *Water temperature*: COMM species in the SCRE have generally high temperature tolerance. Temperature modeling indicated that the impact of VWRF discharge upon SCRE temperature is likely not biologically relevant (+/- 0.5ºC variability across all scenarios; Section 5.3.2.4). Temperature variability is dominated by solar insolation and radiation.

5.5.2 Estuarine Habitat (EST)

Estuarine Habitat as a beneficial use is defined by the Basin Plan as "[u]ses of water that support estuarine ecosystems including, but not limited to, preservation or enhancement of estuarine habitats, vegetation, fish, shellfish, or wildlife (e.g., estuarine mammals, waterfowl, shorebirds)." EST was assigned a relative weighting of 15% in the AHP framework (see Section 5.6 for more details).

5.5.2.1 EST Factors and Associated Metrics for AHP Analysis

The following factors were used to assess the relative effects of VWRF discharge on the EST beneficial use:

1. *Open water habitat amount for native estuarine fish species*. Open water habitat area was used as a generalized representation of important habitat supporting estuarine fish species since estuarine fish species in the SCRE have wide ranging life histories and habitat requirements (Table 3-32). Open water habitat area was estimated under closed-mouth, dry-weather, equilibrium water surface elevation conditions for each of the alternative VWRF discharge scenarios (Table 5-5). Percent of maximum modeled open water habitat area was used to represent relative availability of open water habitat for native estuarine fish species. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support habitat area for EST species:
    - Score of 2: 100% of the maximum modeled open water habitat area
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum modeled open water habitat area
    - Score of 0: < 20% of maximum modeled open water habitat area

2. *Wetland habitat amount (i.e., freshwater wetland + salt marsh wetland area) for estuarine species*. Wetland habitat area was estimated under closed-mouth, dry-weather, equilibrium water surface elevation conditions for each of the alternative VWRF discharge scenarios (Table 5-5). Wetland habitats provide cover for juvenile fishes, support invertebrate production, and provide trophic subsidies to open water habitats. The sum of freshwater wetland and salt marsh habitat area was used as a generalized representation of wetland habitat for SCRE estuarine species. Percent of maximum modeled wetland habitat area was used for comparing relative wetland habitat availability for SCRE estuarine species. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support freshwater wetland and salt marsh wetland habitat area for EST species:
    - Score of 2: 100% of the maximum modeled sum of the freshwater wetland and salt marsh habitat area
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum modeled sum of the freshwater wetland and salt marsh habitat area
    - Score of 0: < 20% of maximum modeled sum of the freshwater wetland and salt marsh habitat area
3. *DO conditions suitable for native estuarine fish species*. Low DO conditions are considered detrimental to estuarine fishes in the SCRE. While, a number of native estuarine fishes in the SCRE are tolerant of periodic low DO, some (e.g., steelhead) are less tolerant (Table 3-32). Many of the non-native fishes in the SCRE are known to thrive in eutrophic environments with chronically low DO, indicating that low DO conditions may be more detrimental to native species than non-native species (Section 3.6.4.1). Available data do not provide sufficient information for accurate predictive quantitative modeling of DO under alternative VWRF discharge scenarios. Because hypoxia events are driven by algal blooms, which in turn depend on elevated nutrient concentrations, the relative potential for low DO events was quantified by comparing the percent reductions in existing nutrient (N, P) loads associated with each VWRF discharge scenario. Given that background nutrient levels in water sources to the SCRE other than VWRF discharge, potentially including groundwater inflows from the surrounding floodplains, are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios, even Scenario 11 (i.e., 100% reduction in VWRF discharge). However, nutrient load reductions due to reduced VWRF discharges in Scenarios 1-11 (i.e., 0% diversion to 100% diversion) may reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing VWRF discharge. Accordingly, VWRF discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in flow volume, and therefore nutrient loading.
    - Score of 2: 100% reductions in VWRF N and P loads
    - Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
    - Score of 1: 0% reductions in VWRF N and P loads
4. *Salinity conditions benefiting native estuarine fishes.* While some non-native fishes are tolerant of wide ranges of salinity, on average, native estuarine fishes have higher preferred

salinities and greater salinity tolerances ranges than non-native fish (Section 3.6.4.1; see also Table 3-32). >18 ppt was identified as the range of salinities benefiting native estuarine fishes, as that threshold exceeds the tolerance range of several detrimental, introduced, non-native species including common carp, green sunfish, and goldfish; it also exceeds the preferred salinity ranges of Mississippi silverside and shimofuri goby. Based on the dominance of freshwater inputs from sources other than VWRF discharge, the SCRE is expected to remain a predominantly freshwater system under all VWRF discharge scenarios. Within the freshwater regime, deviations from freshwater conditions are driven by the introduction of seawater during open-mouth periods and wave over-wash events. The duration of elevated salinity conditions (>18 ppt) following mouth closure is controlled by the rates of freshwater inputs to the SCRE, and therefore vary with VWRF discharge scenarios (Figure 5-6 and Figure 5-7). 18 ppt was selected as the threshold salinity for EST because it exceeds the tolerance of freshwater non-native species in the SCRE (e.g., common carp, green sunfish, goldfish) and exceeds the preferred salinity of shimofuri goby. The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support suitable salinity conditions favoring EST species:

- Score of 2: > 7 days of salinity > 18 ppt following mouth closure
- Score of 1: 0–7 days of salinity > 18 ppt following mouth closure

5.5.2.2 Relative Weighting of EST Factors

EST factors were weighted in the following order of priority: open water habitat area (~55%), DO conditions (~28%), wetland (freshwater wetland + salt marsh) habitat area (~12%), and salinity (~4%). Open water habitat was deemed important for native fish species on the basis that increasing habitat area increases the total abundance of fish that can be supported by the SCRE (i.e. carrying capacity). This assumes that physical habitat and correlated parameters (e.g. food availability, cover) limit production. DO conditions were the second priority factor. While many of the native estuarine fish (Table 3-30) are tolerant of low DO conditions (Table 3-32), prolonged periods of depressed DO can result in mortality. Furthermore, a number of non-native fishes present in the SCRE have superior tolerance to hypoxia, so low DO conditions may favor non-native fishes. While wetland habitats can be important for providing food, cover, and other benefits to estuarine species, its weighting was reduced here because it is also considered separately in the WET beneficial use. Finally, salinity was considered a low priority factor for the EST beneficial use. While salinity is a critical factor driving habitat suitability, and can be a primary driver of community composition in lagoon and estuarine systems, there is not enough variability in salinity conditions across VWRF discharge scenarios to significantly depress freshwater non-native fishes. That is, the SCRE is a predominantly freshwater system during closed-mouth, dry-weather conditions under all VWRF discharge scenarios. A number of potentially detrimental non-native fishes also have high salinity tolerance (e.g., Mississippi silverside, western mosquitofish; Table 3-32), such that even prolonged periods of elevated salinity are unlikely to significantly benefit native fishes.

5.5.2.3 Scoring VWRF Discharge Scenarios for AHP Analysis of EST

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each discharge scenario are summarized in Table 5-11. The open water habitat area for native estuarine fish species is maximized at 108 acres under VWRF discharge Scenario 1 (i.e., 0% diversion; current discharge) and declines relatively constantly with decreasing VWRF discharge (i.e., Scenarios 2 through 11) (Figure 5-8). The wetland habitat area for native estuarine fish species is maximized at 47 acres under VWRF discharge Scenario 1 (i.e., 0% diversion; current discharge) and declines with decreasing VWRF discharge (i.e., Scenarios 2 through 11) (Figure 5-9). The amount of wetland habitat declines rapidly between Scenario 1 and 3, remains fairly constant in that it fluctuates ± 3 acres between Scenarios 3 through 8, and then declines at a more gradual rate from Scenarios 8 through 11. As with COMM, DO conditions were assessed using VWRF nutrient loads as a proxy for the likelihood of algal blooms. Discharge scenarios were given continuous scores based on the changes in nutrient loading associated with each VWRF discharge scenario (Table 5-12). Assuming constant nutrient concentrations in VWRF discharge across scenarios, the nutrient loading decreases linearly with increasing decreasing VWRF discharge. Only Scenario 11 supports salinity conditions > 18 ppt for more than seven days following mouth closure (score of 2). The remaining scenarios (Scenarios 1–10) receive a score of 1 with respect to salinity. Although only limited variability was observed between scenarios with regards to the selected salinity threshold for EST species, it should be noted that reductions in VWRF discharge decrease the rate at which salinity conditions return to freshwater following saline inputs. Overall, VWRF discharge scenario 1 scores the best for supporting EST at 1.7, with the scores gradually decreasing from Scenarios 2-11 from 1.5 to 0.9.

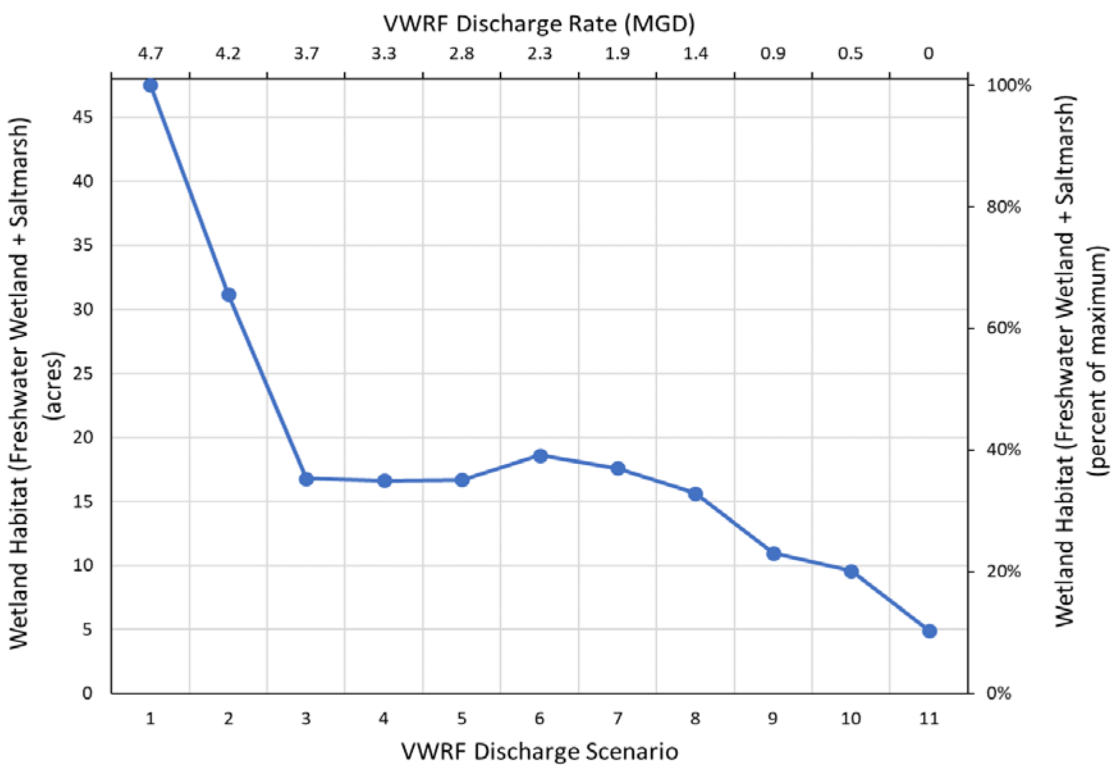


Figure 5-9. SCRE wetland habitat area as acres (primary vertical axis) and percent of maximum modeled (secondary vertical axis) by VWRF discharge scenario.

Table 5-11. Scores received by each VWRF discharge scenario for each of the factors affecting the EST beneficial use.

Factor		Scores by VWRF Discharge Scenario

	Tier 2 Weight	1	2	3	4	5	6	7	8	9	10	11
Open Water Habitat Area	55%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
Wetland Habitat Area	28%	2.0	1.1	0.4	0.4	0.4	0.5	0.4	0.3	0.1	0.0	0.0
DO conditions	12%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Salinity	4%	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	2.0
Tier 2 weighted sum	100%	1.7	1.5	1.4	1.4	1.4	1.4	1.4	1.3	1.0	0.9	0.9

5.5.2.4 EST Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing discharge scenarios for the EST beneficial use:

- *Long-term equilibrium salinity*: Based on the dominance of freshwater inputs to the SCRE, the long-term equilibrium water surface elevation salinity is expected to be freshwater during closed-berm, dry-weather conditions under all discharge scenarios.
- *Food resources*: Food resources as represented by BMI and prey fish abundance and species composition are thought to be largely controlled by WQ factors (e.g., DO, salinity), which, as already assessed for this beneficial use, generally improve with reduced discharges. However, available data are insufficient for quantitatively predicting food resource abundance and composition under alternative VWRF discharge scenarios. See Section 3.6 for discussion of food resources, fish assemblage composition and dynamics, and inter-species interactions.
- *Water temperature*: Temperature modeling indicated that the impact of VWRF discharge upon SCRE temperature is not biologically relevant (+/- 0.5ºC variability across all scenarios; Section 5.3.2.4), as temperature variability is dominated by solar insolation and radiation.

5.5.3 Marine Habitat (MAR)

Marine Habitat as a beneficial use is defined by the Basin Plan as “[u]ses of water that support marine ecosystems including, but not limited to, preservation or enhancement of marine habitats, vegetation such as kelp, fish, shellfish, or wildlife (e.g., marine mammals, shorebirds).” Factors affecting marine species that might use the SCRE are assessed under other beneficial uses, including EST, WILD, and RARE and are not assessed separately. Therefore, Marine Habitat was not included separately in the AHP evaluation.

5.5.4 Migration of Aquatic Organisms (MIGR)

Migration of Aquatic Organisms as a beneficial use is defined by the Basin Plan as “[us]es of water that support habitats necessary for migration, acclimatization between fresh and salt water, or other temporary activities by aquatic organisms, such as anadromous fish.” Migration is assigned a relative weighting of 16% under the AHP framework (see Section 5.6 for more details).

5.5.4.1 MIGR Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting the MIGR beneficial use:

5. *Migration opportunities (i.e., open-mouth conditions) for MIGR species during their migration window.* The migration window (i.e., the time of year during which migration occurs) for steelhead and Pacific lamprey was identified as November 1–May 31 (Section 3.6.3.1). Low occurrence of open-mouth conditions during this period can limit access to upstream spawning locations. The percent of the maximum number of days that the SCRE mouth is open during the steelhead and lamprey migration window for each VWRF discharge scenario, summed across water year types (e.g., dry, normal, or wet) was used to represent migration opportunities into the SCRE for anadromous fish regardless of the availability of Santa Clara River flows enabling migration upstream of the SCRE. Model runs were conducted for each Scenario 1 through 11 (i.e., 0% to 100% reduction in VWRF discharge). The following thresholds were adopted for comparing the extent to which discharge alternatives support the MIGR beneficial use:
    - Score of 2: 100% maximum modeled number of open mouth days
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in open-mouth days relative to the maximum number of days under any modeled scenario.
    - Score of 0: < 50% maximum modeled number of open-mouth days
6. *Suitable DO conditions for MIGR species.* Dissolved oxygen concentrations < 5 mg/L can serve as a barrier to migration for steelhead and Pacific lamprey (Section 3.6.3.1). Available data do not provide sufficient information for accurate predictive modeling of DO under alternative VWRF discharge scenarios. Because hypoxia events are driven by algal blooms, which in turn depend on elevated nutrient concentrations, the relative potential for low DO events was quantified by comparing the percent reductions in existing nutrient (N, P) loads associated with each VWRF discharge scenario. Given that background nutrient levels in water sources to the SCRE other than VWRF discharge are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios, even Scenario 11 (i.e., 100% reduction in VWRF discharge). However, nutrient load reductions due to reduced VWRF discharges in Scenarios 1-11 (i.e., 0% diversion to 100% diversion) may reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing VWRF discharge. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in flow volume, and therefore nutrient loading.
    - Score of 2: 100% reductions in VWRF N and P loads
    - Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
    - Score of 1: 0% reductions in VWRF N and P loads

5.5.4.2 Relative Weights of MIGR Factors

Migration opportunities (i.e. access to the SCRE) during the steelhead and lamprey migration window were given 89% of the weighting for the MIGR beneficial use, with DO conditions given 11%. Given that DO crashes are relatively rare in the SCRE, DO conditions are not expected to limit upstream passage for migratory fish, though reducing the likelihood of DO crashes may benefit migratory fish. Closed mouth conditions prohibiting access to the SCRE may limit opportunities for migration.

5.5.4.3 Scoring VWRF Discharge Scenarios for AHP Analysis of MIGR

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each discharge scenario are summarized in Table 5-12. Model runs were conducted to predict the duration of open-mouth conditions steelhead and Pacific lamprey migration window (Nov 1–May 31) for representative water years for each of three water year types (dry, normal, wet) under Scenarios 1-11 (i.e., 0% to 100% reduced VWRF discharge) producing eleven model runs per water year type and a total of thirty-three model runs. Model estimated average number of days with open-mouth conditions across all water year types during the migration window show decreases in VWRF discharge result in a nearly linear decrease in the water year average number of open-mouth days across Scenarios 1-8 (Figure 5-10). Changes in the average number of days with open-mouth conditions across all water year types during the migration window associated with reductions in VWRF discharge are not evenly distributed, with some reductions in VWRF discharge causing a one-day decrease in open-mouth conditions (i.e., Scenario 1 to 2), while other reductions in VWRF discharge cause no change in the open-mouth conditions (i.e., Scenario 3 to 4). The number of open-mouth days decreases from Scenario 1-8, but then there is no change in the number of open-mouth days with further decreases in VWRF discharge from Scenario 8-11 (Figure 5-10). As with prior beneficial uses, VWRF nutrient loading was used as a proxy for DO conditions in the SCRE and the discharge scenarios were given continuous scores based on the changes in nutrient loading associated with each scenario (Table 5-12). Overall, VWRF discharge scenario 1 scored slightly better at 1.9 than other scenarios for supporting MIGR, with a gradual decrease in the score from 1.8 to 1.7 for Scenarios 2-11.

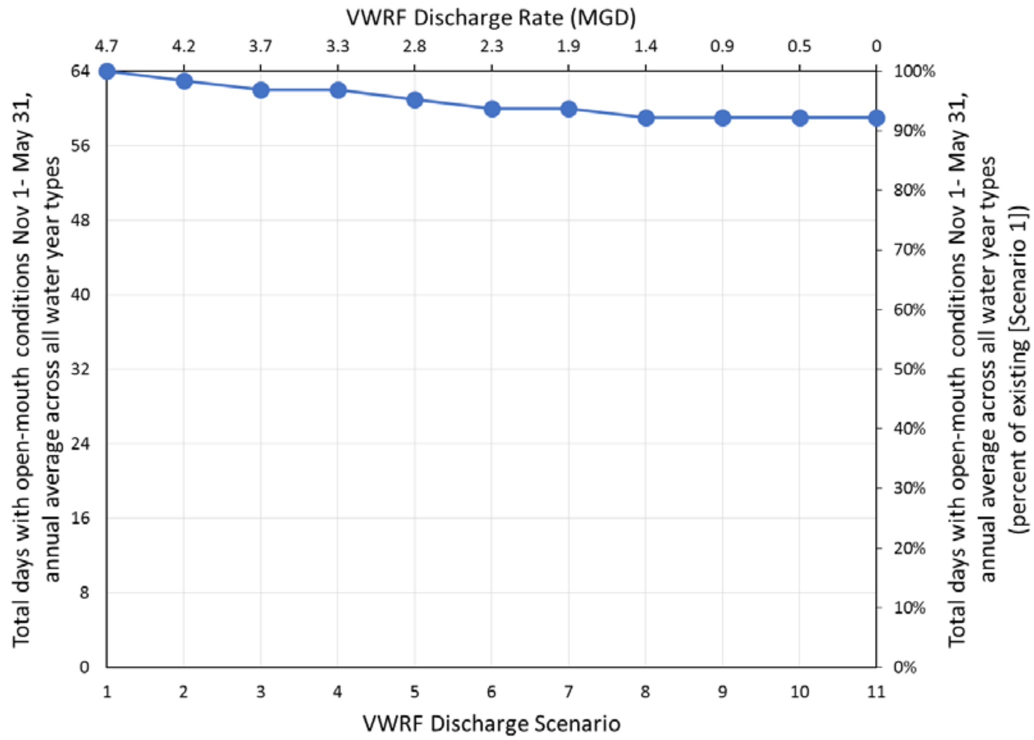


Figure 5-10. Number of days with open-mouth conditions (primary vertical axis) and percent of maximum modeled open-mouth days (secondary vertical axis) during the steelhead and Pacific lamprey migration window (Nov 1–May 31), averaged across all modeled water year types (dry, wet, normal) as a function of VWRF discharge scenario.

Table 5-12. Scores received by each VWRF discharge scenario for each of the factors affecting the MIGR beneficial use.

Factor	Tier 2 Weight	Scores by VWRF Discharge Scenario										
		1	2	3	4	5	6	7	8	9	10	11
Migration opportunities	89%	2.0	1.9	1.9	1.9	1.8	1.8	1.8	1.7	1.7	1.7	1.7
DO conditions	11%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Tier 2 weighted sum	100%	1.9	1.8	1.8	1.8	1.8	1.8	1.8	1.7	1.7	1.7	1.7

5.5.4.4 MIGR Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing discharge scenarios for the MIGR beneficial use:

- *Amount of open water habitat for acclimatization*: Habitat area does not directly relate to opportunity for migration, and habitat area for steelhead smolt acclimatization is evaluated under the EST and RARE beneficial uses.
- *Food resources*: Migratory fish use the SCRE as a migratory corridor with only transient residence. Prior to outmigration, some juveniles or smolt-sized individuals might rear for short periods in the SCRE. Conditions affecting steelhead smolt rearing are evaluated under the EST and RARE beneficial uses, and only factors affecting migration *per se* are evaluated for the MIGR beneficial use.
- *Water temperature*: Temperature modeling indicated that the impact of VWRF discharge upon SCRE temperature is not biologically relevant (+/- 0.5ºC variability across all scenarios; Section 5.3.2.4), as temperature variability is dominated by solar insolation and radiation.
- *Upstream passage flows*: While instream flows for upstream passage are an important factor regulating upmigration timing and success, they are not affected by VWRF discharge.
- *Effects of dissolved copper on steelhead homing*: Information reviewed in Section 3.6.3.1 indicates that dissolved copper concentrations > 5 ug/L can impact the homing abilities of migratory steelhead. However, analysis of trace metals data in the SCRE shows that dissolved copper concentrations in VWRF discharge are comparable or lower than other inflow sources (Table 3-15 through Table 3-18), indicating that concentrations will not vary as a result of variations in VWRF discharge.

5.5.5 Municipal and Domestic Supply (MUN)

Municipal and Domestic Supply is defined by the Basin Plan as "[u]ses of water for community, military, or individual water supply systems including, but not limited to, drinking water supply." MUN is not currently a designated beneficial use of surface waters within the SCRE in the Basin Plan. In recent evaluations of surface water connectivity to the shallow semi-perched aquifer underlying the SCRE, Hopkins (2018) concluded that salinity levels in the shallow groundwaters influenced by the SCRE are too high to directly support MUN uses. The planned diversion of VWRF discharge from the SCRE and the potable reuse of treated effluent, including groundwater injection into deeper aquifers used to augment quantity, quality, and reliability of local water supply, would support and more fully realize MUN uses within the SCRE watershed. Therefore, the impact of the VWRF discharge scenarios on the realization of MUN is also considered in this analysis. MUN is assigned a relative weighting of 5% in the AHP framework (see Section 5.6 for more details).

5.5.5.1 MUN Factors and Associated Metrics for AHP Analysis

The following metric was used to assess the relative effects of VWRF discharge on the MUN beneficial use:

1. *VWRF discharge diverted to municipal reuse*: The City plans to direct reductions in the VWRF discharge to the SCRE associated with Scenarios 2–11 to diversions for municipal reuse as part of the proposed VenturaWaterPure Project. The following thresholds were adopted for comparing the extent to which the VWRF discharge scenarios support the MUN beneficial use:
    - Score of 2: 100% of VWRF discharge diverted to municipal reuse

- Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios that support between the minimum feasible design flow required for construction of an advanced water treatment facility (2.6 MGD) and the maximum diversion volume
- Score of 1: all discharge scenarios that support some water reuse potential (>0% diversion), but fall below the minimum feasible design flow required for construction of an advanced water treatment facility (2.6 MGD)
- Score of 0: 0% of VWRF discharge diverted to municipal reuse

5.5.5.2 Scoring VWRF Discharge Scenarios for AHP Analysis of MUN

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each discharge scenario are summarized in Table 5-13. Scenario 1 does not provide any water for municipal reuse (score of 0). Scenarios 2–6 provide some potential for municipal reuse, but do not meet the minimum target for the WaterPure Project (score of 1). Scenarios 6-11 have linearly increasing scores from 1.0 to 2.0 with Scenario 11 providing the maximum realization of the MUN beneficial use (score of 2) at 100% of current VWRF discharge diverted to municipal reuse.

Table 5-13. Scores received by each VWRF discharge scenario the MUN beneficial use.

Factor	Tier 2 Weighting	VWRF Discharge Scenario										
		1	2	3	4	5	6	7	8	9	10	11
Volume Diverted to MUN Uses (MGD)	100%	0.0	0.5	1.0	1.4	1.9	2.4	2.8	3.3	3.8	4.2	4.7
MUN Score	100%	0.0	1.0	1.0	1.0	1.0	1.0	1.2	1.4	1.6	1.8	2.0

5.5.6 Navigation (NAV)

Navigation as a beneficial use is defined by the Basin Plan as "[u]ses of water for shipping, travel, or other transportation by private, military, or commercial vessels." Because there is no evidence that the SCRE is used extensively for navigational purposes, the NAV beneficial use was not assessed. Further, this beneficial use was not identified as a priority by the Resources Agencies, the TRT, Wishtoyo, HTB, or other stakeholders, particularly considering the focus on protection of listed species occupying or using the SCRE and nearby habitats. Therefore, navigational access and use were not considered further in this assessment.

5.5.7 Rare, Threatened, or Endangered Species (RARE)

Rare, Threatened, or Endangered Species as a beneficial use is defined by the Basin Plan as "[u]ses of water that support habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened, or endangered." Assessment of this beneficial use was considered a priority for the Phase 3 Study, given the listed species occupying and using the SCRE, designated critical habitats within the SCRE and its vicinity, the prohibitions against take of such species imposed by the federal and state Endangered Species Acts, and the importance of these sensitive aquatic and wildlife resources to any determination of an ecologically protective maximum diversion volume or flow (i.e., the MEPDV). Consideration was given to rare aquatic, avian, and plant species in assessment of the RARE beneficial use.

The RARE beneficial use was assigned a relative weight of 35% in the AHP framework (see Section 5.6 for more details). AHP pairwise comparisons resulted in the following weights:

- RARE Aquatic Species: ~75%
- RARE Avian Species: ~19%
- RARE Plant Species: ~6%

Aquatic species were given highest priority consideration for several reasons: aquatic species habitat is strongly affected by VWRF discharge; alternate habitat (i.e., outside the SCRE) is limited for RARE aquatic species; Stakeholder input during Phases 1 and 2 of the Study emphasized aquatic species and related habitat protection. For both of the RARE avian species, significant alternate foraging habitat is available in the vicinity of the SCRE, and they are more mobile than fish species. The avian species were assigned a higher priority than plants based on limited occurrence of rare plants within the SCRE and its vicinity, as well as stakeholder input and regulatory considerations. The sections below detail the assessment factors considered in the AHP analysis for supporting the RARE beneficial use under alternative VWRF discharge scenarios.

5.5.7.1 RARE Aquatic Species

RARE Aquatic Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting RARE aquatic species:

1. *Physical habitat area for steelhead rearing.* Information reviewed in Section 3.6.3.1 identified open water habitat >1.6 ft. in depth as suitable for juvenile steelhead rearing. The following thresholds were adopted for comparing the extent to which discharge alternatives support steelhead rearing habitat:
   - Score of 2: 100% maximum open water area with >1.6 ft. depth
   - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum open water area with >1.6 ft. depth
   - Score of 0: < 20% maximum open water area with >1.6 ft. depth
2. *Physical habitat area for tidewater goby rearing.* Information reviewed in Section 3.6.3.1 identified open water habitat 0.3–6.5 ft in depth as suitable tidewater goby rearing. Because stable tidewater goby populations have been observed in estuaries five acres or greater in area (USFWS 2005), five acres was used to represent the threshold below which physical habitat area was not supported. This was converted to percent of maximum modeled tidewater goby habitat to maintain consistency with other habitat-based metrics. The following thresholds were adopted for comparing the extent to which discharge alternatives support tidewater goby rearing habitat:
   - Score of 2: 100% maximum open water area with 0.3–6.5 ft depth
   - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 5% and 100% of the maximum open water area with 0.3–6.5 ft depth
   - Score of 0: < 5% maximum open water area with 0.3–6.5 ft depth

3. *Physical habitat area for tidewater goby spawning.* Information reviewed in Section 3.6.3.1 identified open water habitat 0.3–6.5 ft in depth with sand substrate as suitable tidewater goby rearing. The following thresholds were adopted for comparing the extent to which discharge alternatives support tidewater goby spawning habitat:
    - Score of 2: 100% maximum open water area with 0.3–6.5 ft depth and sand substrate
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 5% and 100% of the maximum open water area with 0.3–6.5 ft depth and sand substrate
    - Score of 0: < 5% maximum open water area with 0.3–6.5 ft depth and sand substrate
4. *Suitable dissolved oxygen concentrations for steelhead and tidewater goby*: Information reviewed in Section 3.6.3 indicated detrimental effects at DO concentrations < 6 mg/L for steelhead and < 4 mg/L for tidewater goby. Available data do not provide sufficient information for accurate predictive modeling of DO under alternative VWRF discharge scenarios. Because hypoxia events are driven by algal blooms, which in turn depend on elevated nutrient concentrations, the relative potential for low DO events was quantified by comparing the percent reductions in existing nutrient (N, P) loads associated with each VWRF discharge scenario. Given that background nutrient levels in water sources to the SCRE other than VWRF discharge are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios, even Scenario 11 (i.e., 100% reduction in VWRF discharge). However, nutrient load reductions due to reduced VWRF discharges in Scenarios 1-11 (i.e., 0% diversion to 100% diversion) may reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing VWRF discharge. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in flow volume, and therefore nutrient loading.
    - Score of 2: 100% reductions in VWRF N and P loads
    - Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
    - Score of 1: 0% reductions in VWRF N and P loads
5. *Salinity conditions selecting against tidewater goby predators and competitors*: Tidewater goby can tolerate salinities up to, and potentially exceeding 41 ppt, though they are thought to prefer salinities < 12 ppt (see Section 3.6.3.2). The abundance of non-native predators and potential competitors in the SCRE poses a threat to tidewater goby success (see Sections 3.6.3.2 and 3.6.4). Many of these antagonistic species have a considerably lower salinity tolerance than tidewater goby (Table 3-32). Despite the reported preference for salinities <12 ppt, extended periods of elevated salinities (i.e., > 18 ppt) are thought to benefit tidewater goby by reducing competition and predation risks. Based on the dominance of freshwater inputs, the SCRE is expected to remain a predominantly freshwater system under closed-mouth, dry-weather conditions for all VWRF discharge scenarios. Within the freshwater regime, deviations from freshwater conditions are driven by the introduction of seawater during open-mouth periods and wave over-wash events. The duration of elevated salinity conditions (> 18 ppt) following mouth closure is controlled by the rates of freshwater inputs to the SCRE, and therefore vary with changes in VWRF discharge (Figure 5-7). The following thresholds were adopted for comparing

the extent to which alternative VWRF discharge scenarios support suitable salinity conditions favoring tidewater goby and other SCRE species:

- Score of 2: >7 days of salinity > 18 ppt following mouth closure
- Score of 1: 0–7 days of salinity > 18 ppt following mouth closure

6. *Unseasonal breach effects on tidewater goby spawning and juvenile steelhead rearing*: Unseasonal breaching (i.e., June through October) can dewater tidewater goby spawning burrows and displace pelagic tidewater goby larvae (see Section 3.6.3.2). Furthermore, unseasonal breaching may elevate salinity beyond rearing steelhead preference range (10 ppt), potentially reducing growth and increasing the risk of mortality (see Section 3.6.3.1 and Section 3.6.4). Because unseasonal breaching is driven by mechanisms not captured by the water balance model (e.g., manual trenching), the difference in elevation between the dry season berm height and equilibrium water surface elevation (WSE) was used as a proxy for the likelihood of an unseasonal breach (Section 5.4.1). Scoring thresholds for this factor are based on the assumptions that (a) the likelihood of breaching (manually or otherwise) increases as the equilibrium WSE approaches the height of the berm, and (b) when the difference in elevation between berm height and equilibrium WSE is greater than 2 ft, the likelihood of unseasonal breaching decreases due to the difficulty of manually breaching the berm, but when the difference in elevation between berm height and equilibrium WSE is less than 2 ft., manual breaching is easier to accomplish. The following thresholds were adopted for comparing the likelihood of unseasonal breaching across VWRF discharge scenarios:
   - Score of 2: > 3 ft difference between summer berm height and equilibrium WSE
   - Continuous score between 0 and 2: < 3 ft difference between summer berm height and equilibrium WSE
   - Score of 0: Minimum difference between summer berm height and equilibrium WSE

Relative Weights of RARE Aquatic Factors

Of the factors supporting RARE aquatic species, tidewater goby spawning and rearing habitat were given the greatest weighting (~33% and ~22%, respectively). Tidewater goby spend their entire life histories in the SCRE, and have more specific physical habitat requirements relative to steelhead. Unseasonal breach effects and DO conditions were both assigned ~17% weighting. Though tidewater goby are somewhat tolerant of short term depressions in DO concentrations, prolonged periods of hypoxia can be lethal. Furthermore, many of the non-native competitors/predators of tidewater goby were adapted to eutrophic waters with frequent DO crashes, so low DO may be detrimental to tidewater goby's ability to compete or avoid predation. Unseasonal breach events were assigned ~17% weighting because such events can impact the availability of steelheard rearing habitat and dewater tidewater goby spawning burrows and displace pelagic tidewater goby larvae. Because tidewater goby populations are small and have poor dispersal capabilities when displaced, single breach events can have a large impact on tidewater goby populations.

While steelhead are not obligated to rear in the lagoon, this life history pathway can be beneficial. Unlike tidewater goby, steelhead are not obligate lagoon-rearing fish (though lagoons can provide rearing habitat that is beneficial to their survival; Section 3.6.3.1). Accordingly, physical habitat area for steelhead was given some weight, but a lower weigh (~8%) that other factors related to the tidewater goby. Salinity conditions were given a low priority weighting (3%) for several reasons: (a) salinity conditions are largely suitable for both tidewater goby and steelhead under all VWRF discharge scenarios; (b) while some tidewater goby predators/competitors might be suppressed by elevated salinities, several have high salinity tolerance, so benefits to tidewater goby would be expected to be marginal; (c) salinity conditions required to depress freshwater non-native species are also out of the optimum range for steelhead.

Scoring VWRF Discharge Scenarios for AHP Analysis of RARE Aquatic Species

Juvenile steelhead rearing habitat is maximized under Scenario 1 (89 acres) and decreases with decreasing VWRF discharge (Figure 5-11). More than 73% of maximum habitat area is maintained under Scenarios 1–5 (0% to 40% diversion), after which point open water habitat loss accelerates with decreasing VWRF discharge across Scenarios 6-11 (50% to 100% diversion). Scenario scores for steelhead open water rearing habitat area ranged from 2.0 (Scenario 1) to 0.0 (Scenarios 9–11) (Table 5-13). Rearing habitat for tidewater goby is maximized under Scenario 1 (108 acres) and decreases with decreasing VWRF discharge (Figure 5-12). Greater than 70% of maximum tidewater goby rearing habitat is maintained in Scenarios 1–5. Scores for tidewater goby open water rearing habitat area ranged from 2.0 (Scenario 1) to 0.3 (Scenario 11). Spawning habitat for tidewater goby is maximized under Scenario 5 (75 acres) (Figure 5-13) because the area with depth suitable for tidewater goby spawning and a sand substrate is maximized. Greater than 70% of maximum tidewater goby spawning habitat is supported by Scenarios 1–7. Reductions in VWRF discharge associated with Scenarios 8–11 result in accelerated declines in tidewater goby spawning habitat (Figure 5-13). Scores for tidewater goby spawning habitat ranged from 2.0 (Scenarios 3 to 5) to 0.4 (Scenario 11).

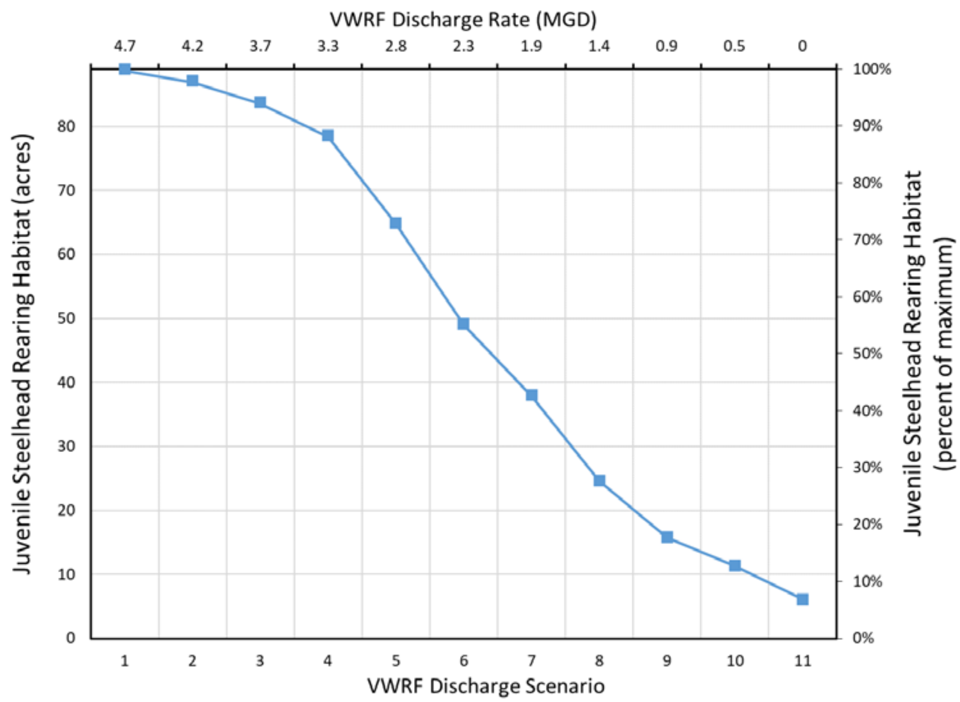


Figure 5-11. Juvenile steelhead rearing habitat area in the SCRE as total acres (primary vertical axis) and percent of maximum modeled habitat area (secondary vertical axis) as a function of VWRF discharge scenario.

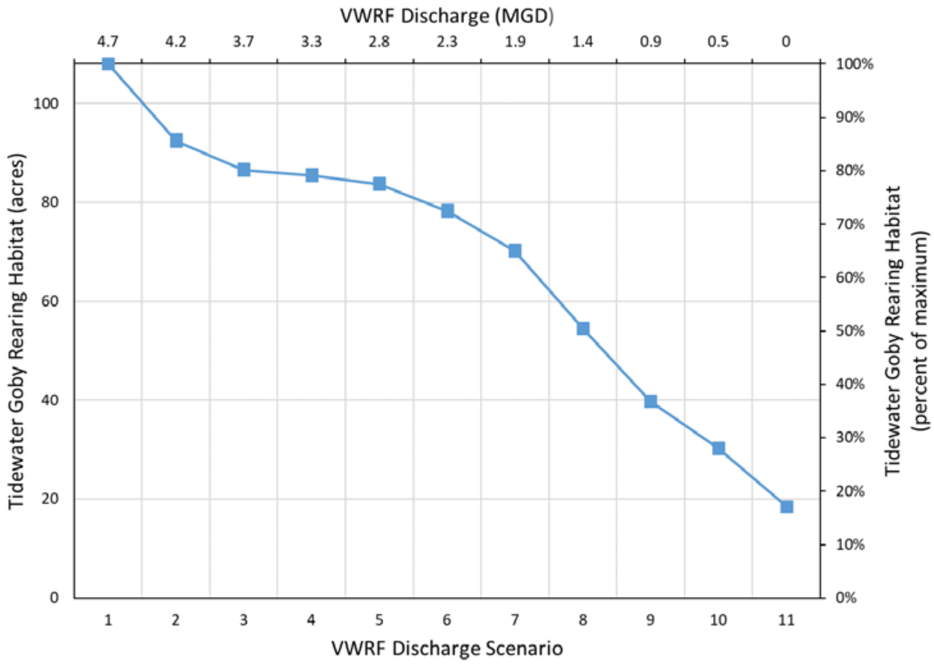


Figure 5-12. Tidewater goby rearing habitat area in the SCRE as total acres (primary vertical axis) and percent of maximum modeled habitat area (secondary vertical axis) as a function of VWRF discharge scenario.

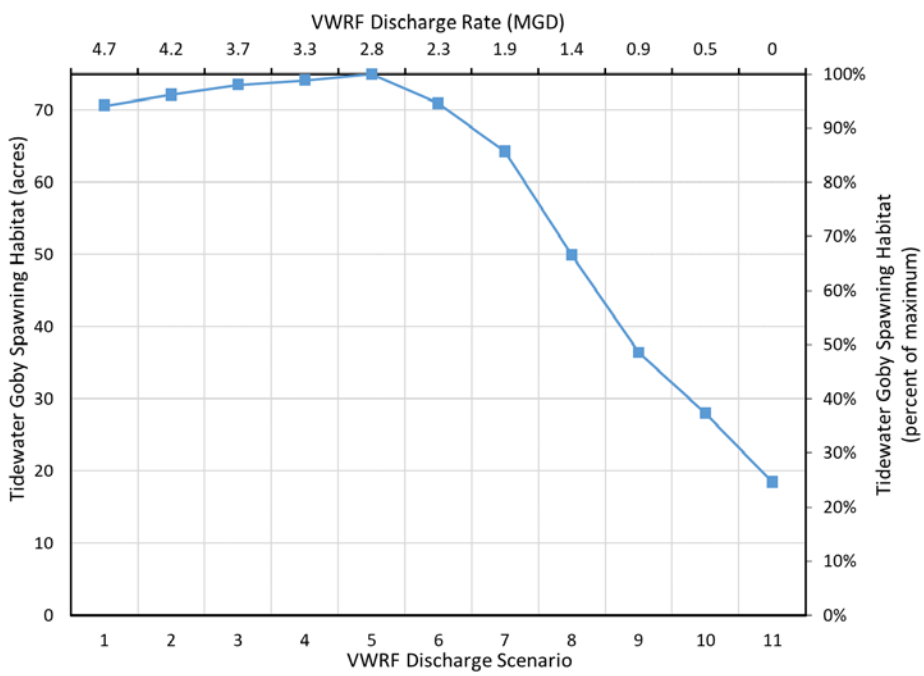


Figure 5-13. Tidewater goby spawning habitat area in the SCRE as total acres (primary vertical axis) and percent of maximum modeled habitat area (secondary vertical axis) as a function of VWRF discharge scenario.

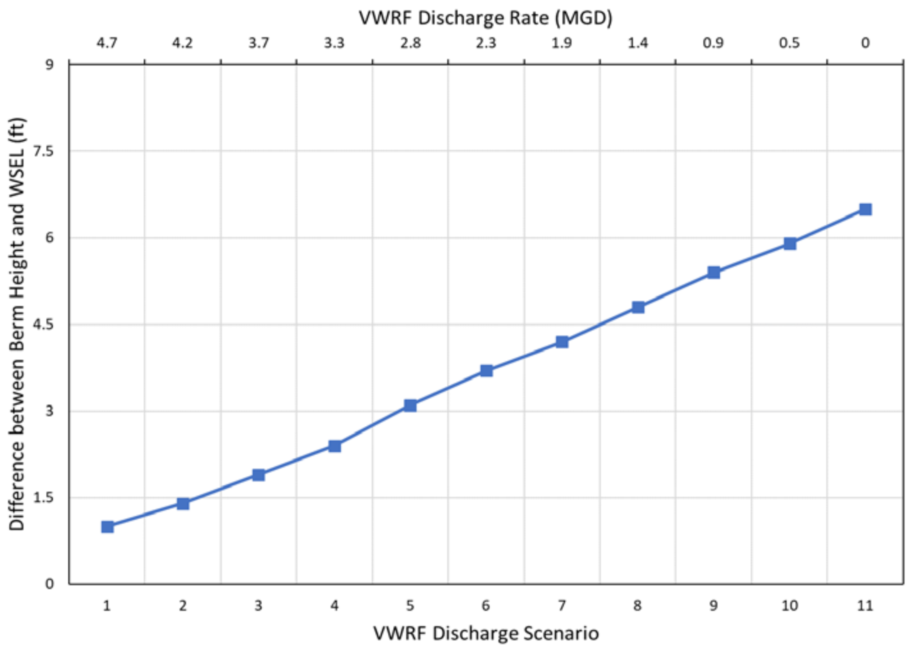


Figure 5-14. Likelihood of unseasonal breaching, represented by the difference between summer berm height and modeled equilibrium water surface elevation (where greater difference corresponds to reduced likelihood of breaching) as a function of VWRF discharge scenario.

As with prior beneficial uses, DO conditions under alternative discharge scenarios were assessed based on nutrient (N, P) loads conveyed to the SCRE. Nutrient loads associates with VWRF discharge scenarios decreased with decreases in the VWRF discharges (Table 5-14). Comparison of discharge scenarios in regards to salinity criteria reflect the competing demands of rearing tidewater goby and steelhead. Salinity conditions following mouth closure exceed the 10 ppt threshold for steelhead for more than 7 days in Scenarios 4–11 (score of 1), but for 7 or fewer days in Scenarios 1–3 (score of 2) (Figure 5-6). For tidewater goby, only Scenario 11 causes salinity to exceed 18 ppt for more than 7 days (score of 2) (Figure 5-7), so Scenarios 1–10 receive a score of 1 (Table 5-14). The difference between summer berm height and equilibrium water surface elevation was used as a proxy for the likelihood of unseasonal breaching which can negatively affect steelhead rearing along with tidewater goby spawning and early development through dewatering of incubating eggs or larval displacement (where the difference between berm height and WSE shares an inverse relationship with the likelihood of breaching). As equilibrium WSE decreases with decreasing VWRF discharge, the difference between berm height and WSE increases linearly (Figure 5-14) so reductions in VWRF discharge decrease the likelihood of unseasonal breaching. Scores for this metric range from 2.0 (Scenario 11) to 0.5 (Scenario 1) (Table 5-14).

Table 5-14. Scores received by each VWRF discharge scenario for each of the factors affecting RARE Aquatic Species.

Factor	Tier 3 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
TWG spawning habitat area	33%	1.9	1.9	2.0	2.0	2.0	1.9	1.7	1.3	0.9	0.7	0.4
TWG rearing habitat area	22%	2.0	1.7	1.6	1.6	1.5	1.4	1.3	1.0	0.7	0.5	0.3
DO conditions	17%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Unseasonal breach effects on STL and TWG	17%	0.0	0.4	0.9	1.4	2.0	2.0	2.0	2.0	2.0	2.0	2.0
STL rearing habitat area	8%	2.0	1.9	1.9	1.7	1.3	0.9	0.6	0.2	0.0	0.0	0.0
Salinity for TWG	3%	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	2.0
Tier 3 weighted sum	100%	1.4	1.4	1.6	1.6	1.7	1.6	1.5	1.3	1.1	1.0	0.9

RARE Aquatic Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing the extent to which discharge scenarios support RARE aquatic species:

- *Other rare fish species (Santa Ana sucker, unarmored threespine stickleback, arroyo chub):* southern California steelhead and tidewater goby were used as focal species, and intended to represent the range of RARE fish species found in the SCRE (see Section 1.3.1) because they have been observed within the SCRE. There are no records of occurrence for Santa Ana sucker, unarmored threespine stickleback (which is not listed for protection) or Arroyo Chub within the SCRE or its vicinity. Tidewater goby and steelhead have narrower aquatic habitat requirements than the other RARE fish species in the SCRE and the habitat requirements of these other sensitive aquatic species that might occupy the SCRE overlap with those of either steelhead or tidewater goby (Table 3-32), making the selected focal aquatic species conservative representatives of all RARE fish species in the SCRE.
- *Food resources for RARE fish species*: Food resource availability for RARE fish species, as represented by BMI abundance and composition, is thought to be largely controlled by WQ factors (e.g., DO, salinity) already assessed for this beneficial use (see Sections 3.6.4.2 and 3.6.3). Available data are insufficient to quantitatively predict food resource abundance and composition under alternative VWRF discharge scenarios.
- *Impacts of ammonia on RARE fish species*: Potential exceedances of Basin Plan criterion continuous concentration (CCC) of ammonia criteria due to pH variations are represented by other WQ considerations (DO and N, P loads) for this beneficial use.
- *Effects of dissolved copper on RARE aquatic species*: Analysis of trace metals data in the SCRE shows that dissolved copper concentrations in VWRF discharge are comparable or lower than other sources (Table 3-15 through Table 3-18), indicating that concentrations will not vary by VWRF discharge.

5.5.7.2 RARE Avian Species

RARE Avian Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting RARE avian species:

1. *Physical habitat area for California least tern foraging.* California least tern use the open water habitat of the SCRE for foraging (3.7.2.2). The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support foraging habitat for California least tern:
    - Score of 2: 100% maximum open water area
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum open water area
    - Score of 0: < 20% maximum open water area
2. *Habitat area for western snowy plover foraging.* Western snowy plover glean invertebrates along the water line in beach habitats, as well as in exposed mudflat habitats. During closed-mouth conditions, western snowy plover are expected to forage in the open beach habitat of the beach berm along the water line in the SCRE, as well as along the water line on the ocean side of the beach berm (Section 3.7.2.1). The length of beach berm that interfaces with the lagoon depends on the equilibrium WSE in the SCRE during closed-mouth, dry-weather conditions, and is used to represent foraging habitat area for western snowy plover during equilibrium closed mouth conditions. The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support foraging habitat for western snowy plover:
    - Score of 2: 100% maximum berm length.
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum berm length.
    - Score of 0: <20% maximum berm length.
3. *Limiting risk of nest flooding for western snowy plover and California least tern nests*: Sandy areas within the SCRE that are exposed (i.e., not wetted) at high tide during open-mouth conditions are likely to invite nesting of California least tern and western snowy plover. If these areas lie below the equilibrium WSE, however, the nests can flood upon mouth closure and subsequent lagoon refilling. The area of potential nesting habitat within the SCRE area that is between the WSE of mean higher high tide and closed-mouth, equilibrium WSE was used as a measure of the risk of nest flooding. The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios limit the risk of nest flooding for western snowy plover and California least tern:
4. Score of 2: minimum modeled nesting habitat area at risk of flooding
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between the minimum and maximum modeled nesting habitat area at risk of flooding
    - Score of 0: maximum modeled nesting habitat area at risk of flooding
5. *Habitat area for riparian-associated special status bird species*: Some special status bird species[24] that have potential to occur in the SCRE, including yellow warbler (*Setophaga*

[24] Note that the SCRE is designated as critical habitat for the southwestern willow flycatcher (*Empidonax traillii extimus*) (federally listed as Endangered [ESA]), although surveys have identified no suitable habitat

petechial)[25] and yellow breasted chat (*Icteria virens*)[16] are associated with riparian habitat, and thus not well-represented by the focal RARE bird species. Riparian habitat area was used to represent habitat for these species. The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support habitat for riparian-associated special status bird species:

- Score of 2: Maximum modeled riparian habitat area
- Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between the maximum modeled riparian habitat area and 0 acres of riparian habitat
- Score of 0: 0 acres of riparian habitat

Relative Weights of RARE Avian Factors

Habitat for California least tern was given ~62% weighting, while western snowy plover foraging habitat was given ~21% weighting. California least tern are federally listed as endangered, while western snowy plover are listed as threatened. Limiting flooding risk for California least tern and western snowy plover nests was given ~9% weighting. Habitat for riparian-associated special status bird species, which are not federally listed, was given ~8% weighting. Because western snowy plover predominantly forage at the water-beach interface, extensive alternate foraging habitat of approximately equal quality to that in the SCRE is available in the nearby area. California least tern forage by sight for fish. Similar to western snowy plover, foraging habitat is abundant near the SCRE in the open waters of the Pacific Ocean. However, a closed lagoon system concentrates prey, making the open waters of the SCRE high quality foraging habitat for California least tern.

Scoring VWRF Discharge Scenarios for AHP Analysis of RARE Avian Species

Open water area for California least tern foraging habitat is maximized in Scenario 1 (i.e., 0% reduction in VWRF discharge/existing discharge) and decreases with decreasing VWRF discharge (Figure 5-8). Scenario scores for least tern foraging range from 2.0 (Scenario 1) to 0.2 (Scenario 11) (Table 5-15). Foraging habitat for western snowy plover is represented by the wetted length of the beach berm, since western snowy plover predominantly forage along water's edge. The wetted length of the berm is maximized under Scenario 1 (4,055 ft), but remains largely constant in Scenarios 1–10 (> 90% maximum modeled) (Figure 5-15). Foraging habitat for western snowy plover drops precipitously under Scenario 11 (50% of maximum modeled) (Figure 5-15). Scenario scores for western snowy plover range from 2.0 (Scenarios 1 to 8) to 0.7 (Scenario 11) (Table 5-15).

---

or southwestern willow flycatchers using or occupying the SCRE. Suitable habitat and listed birds have been identified well upstream of the SCRE. This habitat factor represents the degree to which habitat conditions may develop over time that could potentially be used by the southwestern willow flycatcher as habitat, though their occurance is unlikely in the SCRE.

[25] California Species of Special Concern

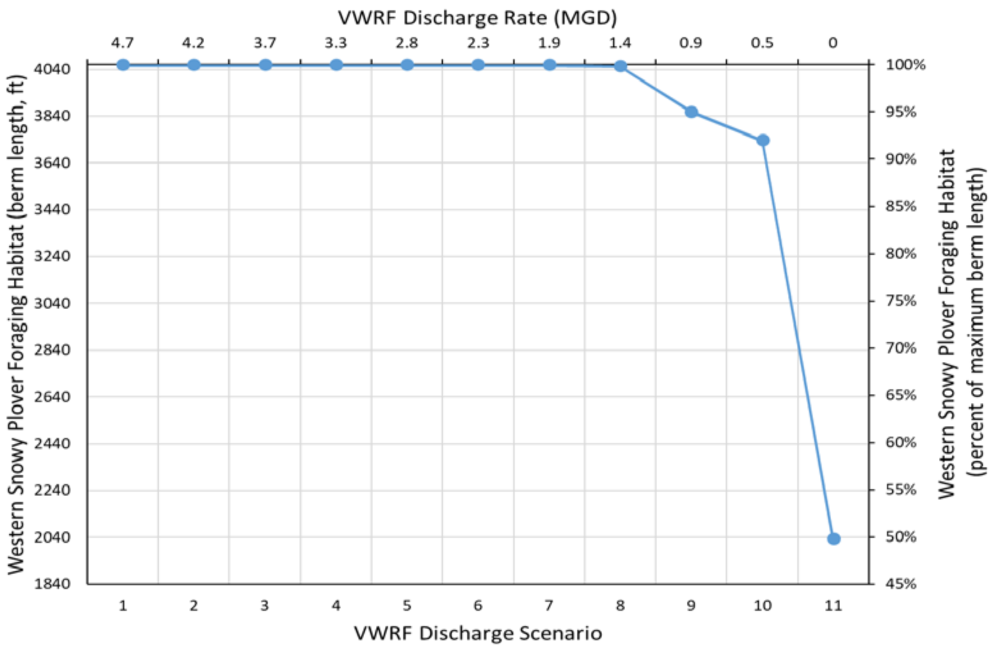


Figure 5-15. Modeled western snowy plover foraging habitat in the SCRE, represented by the length of the beach berm (ft) (primary vertical axis), and percent of maximum modeled berm length (secondary vertical axis) as a function of VWRF discharge scenario.

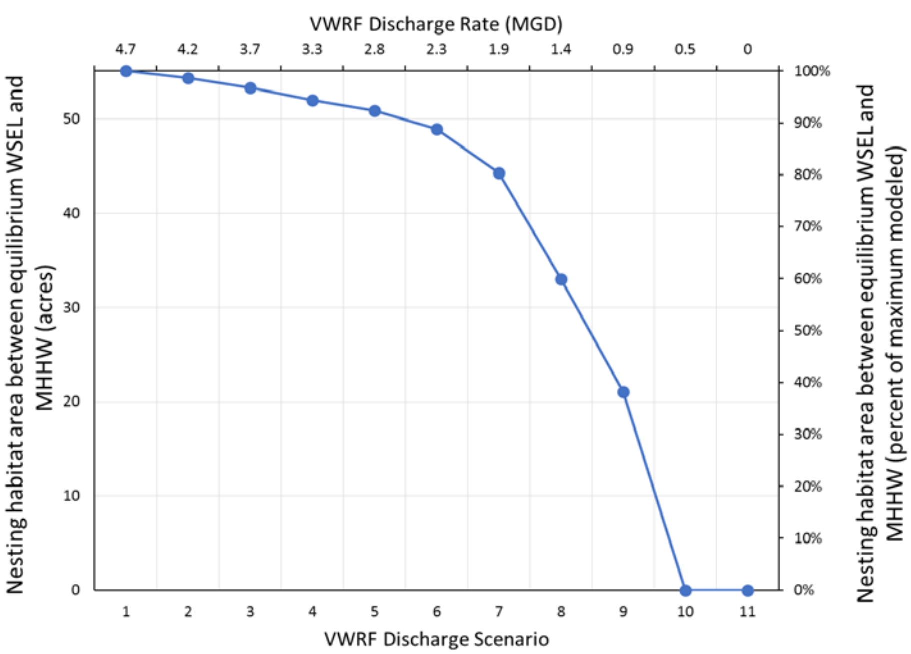


Figure 5-16. Modeled California least tern and western snowy plover nesting habitat area at elevations between mean higher high water (MHHW) and equilibrium water surface elevation (WSEL) in the SCRE, as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis). Risk of flooding is assumed to increase with increasing area.

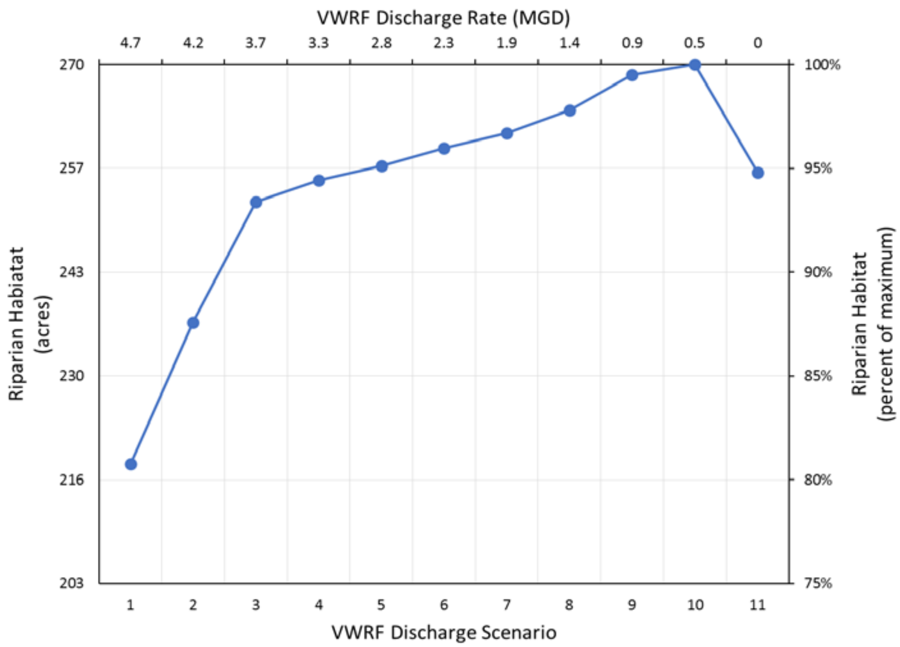


Figure 5-17. Modeled riparian habitat area in the SCRE (an estimate of riparian-associated special status bird habitat) as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) as a function of VWRF discharge scenario.

Table 5-15. Scores received by each VWRF discharge scenario for each of the factors affecting RARE Avian Species.

Factor	Tier 3 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
CLT foraging habitat area	62%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
WSP foraging habitat	21%	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	1.9	1.8	0.7
Limiting risk of WSP and CLT nest flooding	9%	0.0	0.0	0.1	0.1	0.2	0.2	0.4	0.8	1.2	2.0	2.0
Habitat for riparian-associated special status birds	8%	1.6	1.8	1.9	1.9	1.9	1.9	1.9	2.0	2.0	2.0	1.9
Tier 3 weighted sum	100%	1.8	1.7	1.7	1.6	1.6	1.5	1.5	1.5	1.2	1.1	0.7

RARE Avian Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing the extent to which discharge scenarios support RARE avian species:

- *Amount of open beach and foredune habitat for western snowy plover and California least tern nesting*. VWRF discharge has very limited impact on availability of foredune and open beach nesting habitat for listed bird species (Figure 3-53 and Figure 3-55), and extensive alternate habitat is available in nearby areas.

- *Food resources for RARE avian species*. For California least tern, pelagic prey fish abundance is represented by open water habitat area. For least tern, invertebrate abundance is represented by foraging habitat area. Available data are not sufficient to quantitatively predict abundance or composition of prey species under alternative VWRF discharge scenarios.
- *Water temperature*. Temperature modeling indicated that the impact of VWRF discharge upon SCRE temperature is not biologically relevant (+/- 0.5ºC variability across all scenarios; Section 5.3.2.4), as temperature variability is dominated by solar insolation and longwave radiation.
- *Other RARE wildlife species*. Other federally or state listed species that have the potential to occur in the SCRE and surrounding areas have been described in Section 3.7.1. In general, listed species with very low potential to occur in the SCRE, species that utilize habitats that are unaffected by VWRF discharges, or unlisted species that have access to nearby habitats were not included in the analysis. For example, although the Santa Clara River and SCRE has been designated as critical habitat for southwestern willow fly catcher (*Empidonax traillii extimus*) (U.S. Federal Register 2013), the few individuals that have been identified in the limited surveys to date occur well upstream of the SCRE (Labinger et al. 2011). Although riparian scrub habitats in the SCRE may potentially support flycatcher, because of their low potential for occurrence and the broad availability of these habitats along the SCR corridor, use of riparian habitat by these species was considered separately as wildlife habitat (Section 5.5.11).

5.5.7.3 RARE Plant Species

RARE Plant Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting RARE plant species:

1. *Wetland habitat area*: Available data regarding the presence of rare plants within the SCRE and habitat characteristics in the SCRE most conducive to their growth and survival are not sufficient to quantitatively predict changes in abundance of individual RARE plant species. For the purposes of comparing discharge scenarios, abundance of RARE plant species is assumed to be proportional to the habitat type with which species are associated. Eighteen RARE plant species are associated with freshwater and salt marsh wetland habitats in the SCRE (Table 3-28; Section 3.5.3.4). The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support wetland-associated RARE plant species in the SCRE:
    - Score of 2: 100% of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland)
    - Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland)
    - Score of 0: < 20% of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland)
2. *Riparian habitat area:* Available data regarding presence of rare plants within the SCRE and habitat characteristics in the SCRE most conducive to their growth and survival are not sufficient to quantitatively predict changes in abundance of individual RARE plant species. For the purposes of comparing discharge scenarios, abundance of RARE plant species is assumed to be proportional to the habitat type with which species are associated. Seven

RARE plant species are associated with riparian (riparian and riparian riverine) habitats in the SCRE (Table 3-27; Section 3.5.3.3). The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support wetland-associated RARE plant species in the SCRE:

- Score of 2: 100% of the maximum modeled riparian habitat area (riparian + riparian riverwash)
- Continuous score between 0 and 2: a continuous scale was used for scoring discharge scenarios that support between 20% and 100% of the maximum modeled riparian habitat area (riparian + riparian riverwash)
- Score of 0: < 20% of the maximum modeled riparian habitat area (riparian + riparian riverwash)

Relative Weights of RARE Plant Factors

Wetland habitat area was given 67% of the RARE plant weighting, while riparian habitat area was given 33% weighting. Wetland habitats have potential to support a greater number of RARE plant species than riparian habitats in the SCRE (Table 3-28). Furthermore, wetland habitat is only supported in the SCRE, while riparian habitat exists throughout the SCR riparian corridor.

Scoring VWRF Discharge Scenarios for AHP Analysis of RARE Plants

Scenario scores for RARE plant-related factors reflect tradeoffs between wetland and riparian habitat types. Based on the GIS habitat succession rules established for estimating long-term changes in habitat types under alternative VWRF discharge scenarios, dropping equilibrium WSE associated with reduced VWRF discharge scenarios converts wetland habitat to riparian habitat if there is sufficient difference between the WSE and the elevation of the habitat in question (Table 5-4). Wetland habitat is maximized under Scenario 1 (47 acres) and tends to decrease with decreasing discharge (Figure 5-18). More than 60% of the maximum modeled wetland habitat area is maintained in Scenarios 1 and 2, but wetland habitat declines rapidly with discharge reductions. Variations in wetland habitat are less under discharge reductions of between 20% and 70% (Scenarios 3–7), but then wetland habitat declines more rapidly under higher discharge reductions (Scenarios 8–11) (Figure 5-18). Scenario scores for wetland habitat area range from 2.0 (Scenario 1) to 0.0 (Scenarios 10 and 11) (Table 5-16). On the other hand, the total riparian habitat (i.e., riparian and riparian riverwash) tends to increase with VWRF discharge reductions and it is maximized under Scenario 10 (324 acres) (Figure 5-19). However, as large portions of riparian habitat in the SCRE are unaffected by VWRF discharge, greater than 70% of maximum modeled riparian habitat (> 227 acres) is maintained under all scenarios. Scenario scores for support of riparian habitat range from 1.3 (Scenario 1) to 2.0 (Scenarios 10 and 11) (Table 5-16).

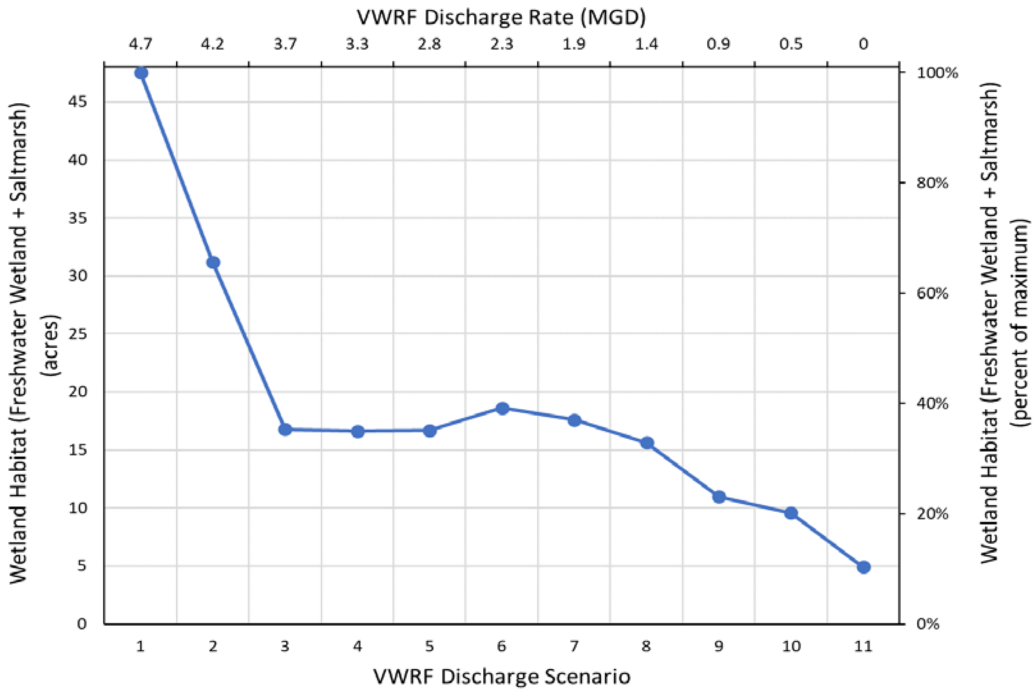


Figure 5-18. Modeled wetland habitat area in the SCRE, a proxy for wetland-associated rare plant abundance, as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) as a function of VWRF discharge scenario.

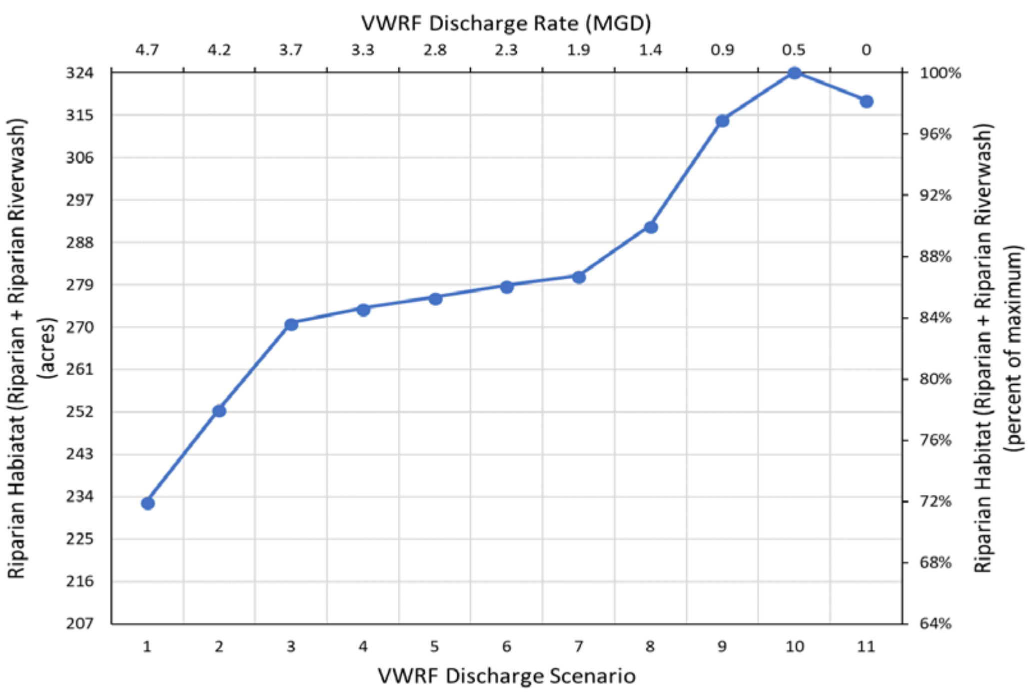


Figure 5-19. Modeled riparian habitat area (including riparian riverwash) in the SCRE, a proxy for the abundance of rare plants associated with riparian habitats, as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) as a function of VWRF discharge scenario.

Table 5-16. Scores received by each VWRF discharge scenario for each of the factors affecting RARE Aquatic Species.

Factor	Tier 3 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
Wetland habitat area	67%	2.0	1.1	0.4	0.4	0.4	0.5	0.4	0.3	0.1	0.0	0.0
Riparian habitat area	33%	1.3	1.4	1.6	1.6	1.6	1.7	1.7	1.7	1.9	2.0	2.0
Tier 3 weighted sum	100%	1.8	1.2	0.8	0.8	0.8	0.9	0.8	0.8	0.7	0.7	0.7

RARE Plant Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing the extent to which discharge scenarios support RARE plant species:

- *Species composition within a habitat type*: The RARE plant assessment was based on the areas of habitat types associated with RARE plant species. We have assumed that RARE plant abundance will be proportional to the habitat types with which they are associated. However, species composition within these habitat types is likely to be heterogeneous, and there are not sufficient data available to predict plant community dynamics within a habitat type.

5.5.7.4 Synthesis of RARE Tier 2 Factors

Overall, scenarios 4–6 (30–50% reductions in VWRF discharge) result in the fullest realization of the RARE beneficial use (Table 5-17). Scenarios 1–7 show only marginal differences in score for the RARE beneficial use (Table 5-17). This optimum at intermediate discharges is largely reflective of maximizing conditions for the benefit of listed aquatic species, which comprise 75% of the RARE beneficial use weighting, while avoiding significant adverse changes to avian habitat conditions for the California least tern and western snowy plover within the SCRE. Estimates of open water aquatic habitat area tend to decrease with reductions in VWRF discharge, while nutrients, water quality (DO), and salinity conditions improve and the risk of unseasonal breaching decreases with increasing diversions/reductions in VWRF discharge (Figure 5-20).

Table 5-17. Scores received by each VWRF discharge scenario for each of the factors affecting the RARE beneficial use.

Factor	Tier 2 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
RARE Aquatic Species	75%	1.4	1.4	1.6	1.6	1.7	1.6	1.5	1.3	1.1	1.0	0.9
RARE Avian Species	19%	1.8	1.7	1.7	1.6	1.6	1.5	1.5	1.5	1.2	1.1	0.7
RARE Plant Species	6%	1.8	1.2	0.8	0.8	0.8	0.9	0.8	0.8	0.7	0.7	0.7
Tier 2 weighted sum	100%	1.5	1.5	1.5	1.6	1.6	1.6	1.5	1.3	1.1	1.0	0.9

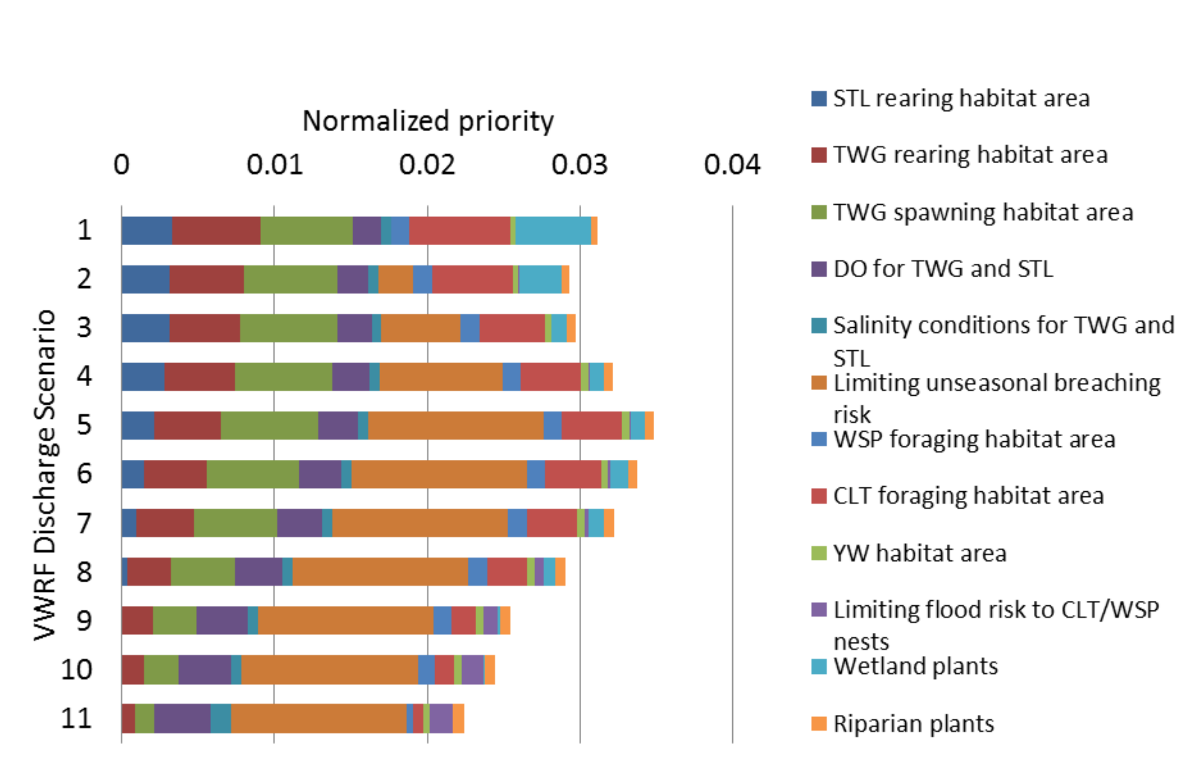


Figure 5-20. Weighted scores (normalized such that the sum of scores across discharge scenarios equals 1) for each discharge scenario with colored bars representing the weighted scores for individual Tier 3 RARE factors.

5.5.8 Water Contact Recreation (REC-1)

Water Contact Recreation as a beneficial use is defined by the Basin Plan as "[u]ses of water for recreational activities involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water-skiing, skin and scuba diving, surfing, white water activities, fishing, or use of natural hot springs." REC-1 is assigned a relative weighting of 1% in the AHP Framework (see Section 5.6 for more details).

5.5.8.1 REC-1 Factors and Metrics for AHP Analysis

While the activities described above are not thought to occur frequently in the SCRE under current conditions, the REC-1 beneficial use was evaluated based on the potential for these activities to occur in the SCRE. The following metric was used to assess the relative effects of VWRF discharge on factors affecting the REC-1 beneficial use:

1. *Open water area*. Open water area was used to represent opportunities for water contact recreation. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support REC-1 uses:
    - Score of 2: 100% of maximum open water area
    - Continuous score from 0-2 based on incremental increases in percent of maximum modeled open water area between 20% and 100%.
    - Score of 0: < 20% of maximum open water area

Because REC-1 has only one factor, open water area is giving full weighting (100%) for the REC-1 beneficial use.

5.5.8.2 Scoring VWRF Discharge Scenarios for AHP Analysis of REC-1

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each scenario are summarized in Table 5-18. Open water area is maximized under Scenario 1 and decreases nearly linearly with decreasing VWRF discharge scenarios (Figure 5-8). Scores for REC-1 ranged from 2.0 (Scenario 1) to 0.24 (Scenario 11). Overall, Scenarios 1 and 2 score the best for supporting REC-1 at 2.0 and 1.9, respectively, due to the reliance on the open water area for the assessment. Scenarios 3 to 7 show only a gradual decrease in scores from 1.8 to 1.5 with decreases in VWRF discharge, then the score decreases rapidly from 1.3 to 0.4 with decreases in VWRF discharge between Scenario 8 and 11.

Table 5-18. Scores received by each VWRF discharge scenario for each of the factors affecting the REC-1 beneficial use.

Factor	Tier 2 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
Open water area	100%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
Tier 2 weighted sum	100%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4

5.5.8.3 REC-1 Factors not Included in AHP Analysis

The following factor was considered, but not included as assessment metrics for comparing discharge scenarios for the REC-1 beneficial use:

- *Bacterial water quality.* Ongoing compliance with Basin Plan bacterial standards is not expected to vary by VWRF discharge scenario.
- *Water contact aesthetics.* Algae blooms can decrease water contact enjoyment of the SCRE by producing unpleasant odors and dissuading contact with the water. The water contact aesthetics overlap significantly with the visual and olfactory aesthetics assessed for REC-2, so they were not assessed separately for REC-1.

5.5.9 Non-contact Water Recreation (REC-2)

Non-contact Water Recreation as a beneficial use is defined by the Basin Plan as "[u]ses of water for recreational activities involving proximity to water, but not normally involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tide-pool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities." REC-2 is assigned a relative weighting of 2% in the AHP framework (see Section 5.6 for more details).

5.5.9.1 REC-2 Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting the REC-2 beneficial use:

1. *Amount of boatable water.* Although recreational boating is not common in the SCRE, opportunities for rowing or inner tube uses exist. Boatable depths for rowing and inner tubes were defined as >1 ft (USFWS 1978). Open water area with depths >1 ft was used to

represent boating opportunities in the SCRE. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support boating opportunities:

- Score of 2: 100% of maximum modeled boatable water area
- Continuous score from 0-2 based on incremental increases in percent of maximum modeled boatable water area between 20% and 100%.
- Score of 0: < 20% of maximum modeled boatable area

2. *Opportunities for camping.* The McGrath State Beach campground provides recreational camping opportunities for visitors to the SCRE. Due to destruction of the campground's protective berm during the 2005 storm events, the SCRE begins to inundate the campground at water surface elevations exceeding 9.5 ft NAVD88 and limits camping opportunities. The percent of campground inundation at the close-mouth, dry-weather equilibrium WSE was estimated by comparing equilibrium stage WSE to the campground elevation, and is used to represent camping opportunities. Equilibrium WSEs below 9.0 ft NAVD88 are assumed to have negligible campground inundation impacts, resulting in a negligible impact on camping activities. Equilibrium WSEs from 9.0 ft NAVD88 to 9.5 ft NAVD88 are assumed to begin to flood the areas surrounding campsites, potentially interfering with camping activities. Equilibrium WSEs greater than 9.5 ft NAVD88 are assumed by the model to inundate campsites as well as the day use parking area, resulting in campground closure so they are assumed to significantly inhibit REC-2 related camping opportunities. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support camping opportunities in the SCRE:
   - Score of 2: Equilibrium stage < 9.0 ft
   - Score of 1: Equilibrium stage 9.0 ft to 9.5 ft
   - Score of 0: Equilibrium stage > 9.5 ft
3. *Physical habitat area for waterfowl and wading birds supporting birding.* Birding is a common recreational use of the SCRE, and is supported by open water habitat for waterfowl, wading birds, and other avian species that use inundated habitat in the SCRE. Open water area is used to represent birding opportunities. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support open water habitat for waterfowl, wading birds, and other avian species:
   - Score of 2: 100% of maximum modeled open water habitat area
   - Continuous score from 0-2 based on incremental increases in percent of maximum modeled open water habitat area between 20% and 100%.
   - Score of 0: < 20% of maximum modeled open water habitat area
4. *Visual and olfactory aesthetics.* Algae blooms can decrease enjoyment of the SCRE by reducing visual aesthetic quality and producing unpleasant odors. Algae blooms are driven by excess nutrient concentrations in the SCRE. Given that background nutrient levels in water sources to the SCRE other than VWRF discharge are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios. However, nutrient (N, P) load reductions may reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing discharge. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in flow volume, and therefore nutrient loading.

- Score of 2: 100% reductions in VWRF N and P loads
- Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
- Score of 1: 0% reductions in VWRF N and P loads

5.5.9.2 Relative Weights of REC-2 Factors

Camping opportunities received ~52% of the weighting for REC-2. By reducing equilibrium WSEs, which in turn is assumed to deter campground inundation even in the absence of the campground's protective berm, reductions in VWRF discharge is predicted to increase REC-2 camping opportunities at the adjacent McGrath State Beach campground. Bird watching is also a popular recreational activity in and around the SCRE and was given ~26% of REC-2 weighting. Birding opportunities are modeled as being affected by open water habitat availability for wading birds and waterfowl, which is predicted to decline with reductions in VWRF discharge. Aesthetic quality was given ~18% of the REC-2 weighting, as most recreational experiences in the SCRE are affected to some degree by the appearance of the SCRE. Algae blooms are unsightly and can produce unpleasant odors; the blooms might decrease somewhat in frequency and/or duration with decreasing nutrient loads, which is partially controlled by VWRF discharge. While boating is a feasible recreational activity in the SCRE, it is not thought to be a common activity, and was thus given a low weighting (~4%).

5.5.9.3 Scoring VWRF Discharge Scenarios for AHP Analysis of REC-2

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each scenario are summarized in Table 5-19. Boatable water area is maximized under Scenario 1 (i.e., 0% reduction in VWRF discharge; existing discharge) due to its reliance on maximization of open water. Boatable water area is largely maintained at > 90% of the maximum through Scenario 4 after which it begins to rapidly decrease with decreasing VWRF discharge (Figure 5-21). Scenario scores for support of boatable water range from 2.0 (Scenario 1) to 0.0 (Scenarios 10 and 11). Inundation of the McGrath State Beach campground is predicted to be minimal under Scenarios 4–11 (i.e., 30% to 100% reduction in VWRF discharge) (score of 2) since the equilibrium WSE remains less than 9.0 ft NAVD88 (Table 5-19) for these scenarios. Inundation of the areas surrounding the campsites is predicted to potentially occur under Scenario 3 since the equilibrium stage is 9.3 ft NAVD88 (score of 1). Scenarios 1 and 2 are predicted by the model to result in significant flooding of the campground since the equilibrium stage is greater than the campground inundation stage that results in campground closures (score of 0) (Figure 5-22). Open water habitat for waterfowl is predicted to decreases nearly linearly with decreasing VWRF discharge (Figure 5-8). Scores for support of open water habitat for waterfowl viewing are maximized at 2.0 for Scenario 1 (i.e., 0% reduction in VWRF discharge; existing VWRF discharge) to 0.4 for Scenario 11 (i.e., 100% reduction in VWRF discharge; 0 MGD VWRF discharge). To account for effects of nutrient loading (and related algal blooms) on estuary aesthetics, VWRF discharge scenarios were given continuous scores based on the nutrient loading ranging from 2.0 for Scenario 11 to 1.0 for Scenario 1 (Table 5-15). Overall, VWRF discharge scenarios 4-7 score the best for supporting REC-2 uses at a score of 1.8. Scenarios 8-11 shows limited variation with scores decreasing from 1.7 to 1.5 with decreases in VWRF discharge, but the scores decrease more rapidly with increases in VWRF discharge for Scenarios 1-3 with the scores ranging from 1.3 in Scenario 3 and 0.8 in Scenario 1 and 2.

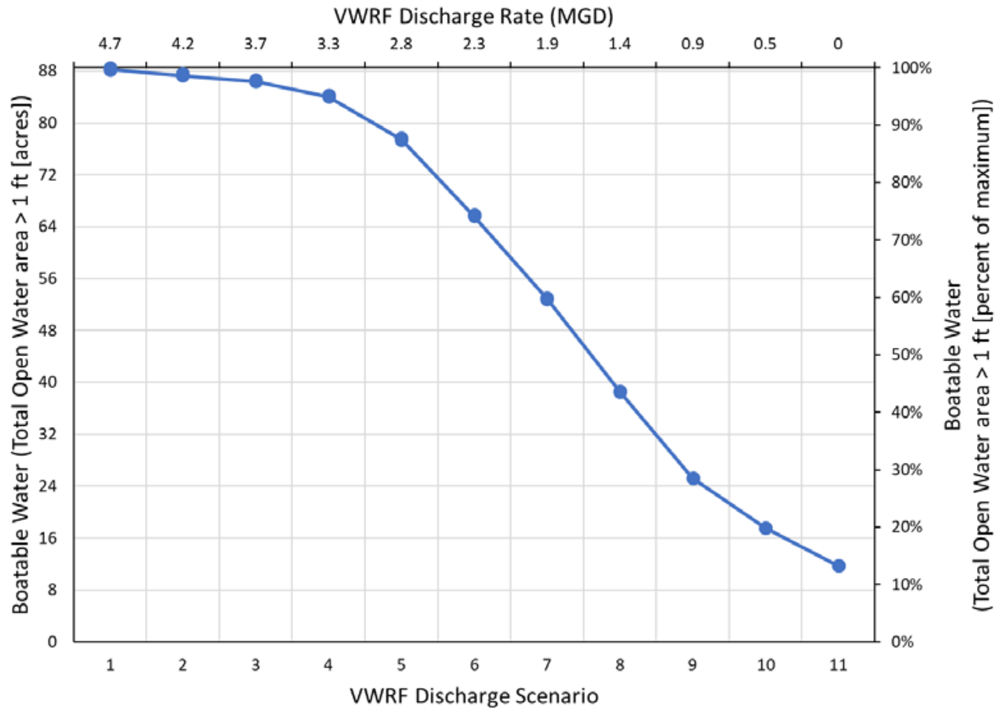


Figure 5-21. SCRE boatable water area as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) by VWRF discharge scenario.

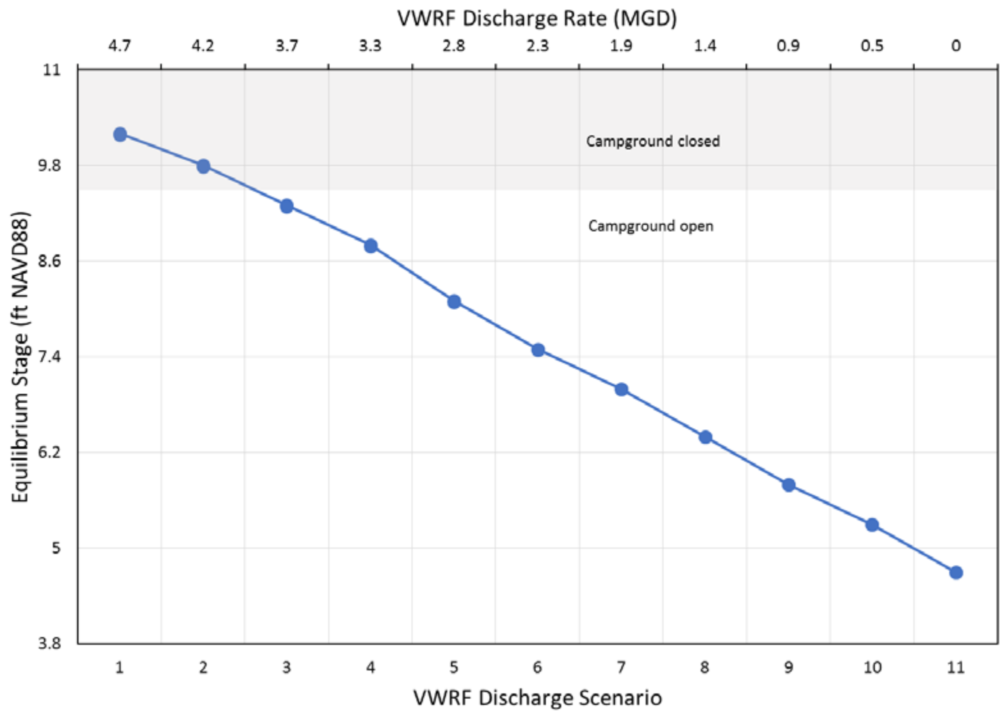


Figure 5-22. Equilibrium WSEL in the SCRE under each VWRF discharge scenario. Gray box shows water surface elevation at which significant flooding causes closure of the McGrath State Beach Campground.

Table 5-19. Scores received by each VWRF discharge scenario for each of the factors affecting the REC-2 beneficial use.

Factor	Tier 2 Weight	Scores by VWRF Discharge Scenario										
		1	2	3	4	5	6	7	8	9	10	11
Camping opportunities	52%	0.0	0.0	1.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Waterfowl habitat	26%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
Aesthetic quality	18%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Boatable water area	4%	2.0	2.0	1.9	1.9	1.7	1.4	1.0	0.6	0.2	0.0	0.0
Tier 2 weighted sum	100%	0.8	0.8	1.3	1.8	1.8	1.8	1.8	1.7	1.6	1.5	1.5

5.5.9.4 REC-2 Factors not Included in AHP Analysis

The following factor was considered, but not included as an assessment metric for the REC-2 beneficial use:

- *Viewing of wildlife species associated with riparian and wetland habitats.* Riparian and wetland habitat is assessed directly as a part of EST and WET beneficial uses (see Section 5.5.11).

5.5.10 Spawning, Reproduction, and/or Early Development (SPWN)

Spawning, Reproduction, and/or Early Development as a beneficial use is defined by the Basin Plan as "[u]ses of water that support high quality aquatic habitats suitable for reproduction and early development of fish." SPWN was assigned a relative weighting of 10% in the AHP framework (see Section 5.6 for more details).

5.5.10.1 SPWN Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting the SPWN beneficial use:

1. *Physical habitat area for spawning and early development of native fish.* A number of native fishes have been documented spawning or have the potential to spawn in the SCRE (Table 3-30), including tidewater goby, as detailed in the RARE beneficial use. Because the species spawning in the SCRE have varied physical spawning habitat preferences, open water habitat area is used to generally represent spawning habitat for SCRE fishes. Tidewater goby spawning habitat needs were more specifically evaluated in RARE. The following thresholds were adopted for comparing the relative extent to which discharge alternatives support habitat area for fish spawning in the SCRE:
   - Score of 2: 100% maximum open water area
   - Continuous score from 0-2 based on incremental increases in percent of maximum modeled open water area between 20% and 100%.
   - Score of 0: < 20% maximum open water area
2. *Suitable DO conditions for spawning fish.* While DO requirements vary by species, low DO conditions are generally unsuitable for incubating eggs and larval fish in the SCRE. DO conditions < 5 mg/L can reduce spawning success and larval fish survival. Available data do not provide sufficient information for accurate predictive modeling of DO under alternative VWRF discharge scenarios. Because hypoxia events are driven by algal blooms, which in turn depend on elevated nutrient concentrations, the relative potential for low DO events was quantified by comparing the percent reductions in existing nutrient (N, P) loads associated with each VWRF discharge scenario. Given that background nutrient levels in water sources to the SCRE other than VWRF discharge are generally well above saturation levels for algal growth (Section 5.3.2.3), it is likely that algal blooms will continue to occur under all discharge scenarios, even Scenario 11 (i.e., 100% reduction in VWRF discharge). However, nutrient load reductions due to reduced VWRF discharges in Scenarios 1-11 (i.e., 0% diversion to 100% diversion) may reduce the frequency and duration of algal blooms and associated depressed DO in the SCRE. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge decreases linearly as a function of decreasing VWRF discharge. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 based on incremental reductions in flow volume, and therefore nutrient loading.
   - Score of 2: 100% reductions in VWRF N and P loads

- Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
- Score of 1: 0% reductions in VWRF N and P loads

3. *Unseasonal breach effects on egg incubation and early larval development*: Unseasonal (i.e., June through October) mouth breaching and estuary draining can dewater incubating eggs and can displace pelagic larval fish, significantly reducing larval survival and subsequent recruitment. Because unseasonal breaching is driven by mechanisms not captured by the water balance model (e.g., manual trenching), the difference in elevation between the dry season berm height and equilibrium water surface elevation (WSE) was used as a proxy for the likelihood of an unseasonal breach (Section 5.4.1). Scoring thresholds for this metric are based on the assumptions that (a) the likelihood of breaching (manually or otherwise) increases as equilibrium WSE approaches the height of the berm, and (b) when the difference in elevation between berm height and equilibrium WSE is greater than 3 ft, the likelihood of unseasonal breaching decreases more than when the difference in elevation between berm height and equilibrium WSE is less than 3 ft. due to difficulty in manually implementing a breach. The following thresholds were adopted for comparing the likelihood of unseasonal breaching across VWRF discharge scenarios:
    - Score of 2: > 3 ft difference between summer berm height and equilibrium WSEL.
    - Continuous score between 0 and 2: < 3 ft difference between summer berm height and equilibrium WSEL.
    - Score of 0: Negligible difference between summer berm height and equilibrium stage WSE

5.5.10.2 Relative Weights of VWRF Discharge Scenarios for AHP Analysis of SPWN

Unseasonal breaching impacts were given the majority of weighting for the SPWN beneficial use (~70%) because unseasonal breaches may dewater eggs and displace pelagic larval fish. DO conditions were given ~21% weighting, as eggs and larval fish can be sensitive to DO conditions. Open water habitat area for spawning was given low weighting (~9%) since our generalized representation of spawning habitat does not vary from estuarine rearing habitat, which is accounted for under the EST beneficial use.

5.5.10.3 Scoring VWRF Discharge Scenarios for AHP Analysis of SPWN

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each scenario are summarized in Table 5-20. Open water habitat (a generalized proxy for SCRE spawning habitat) is maximized under Scenario 1 (108 acres) (i.e., 0% reduction in VWRF discharge; existing discharge) and decreases with decreasing VWRF discharge across Scenarios 2-11 (Figure 5-8). As with prior beneficial uses, for assessing DO conditions, discharge scenario were given continuous scores based on the percent decrease in nutrient loading (Table 5-20). The difference between summer berm height and equilibrium water surface elevation was used as a proxy for the likelihood of unseasonal breaching, which can disrupt spawning and early development through dewatering of incubating eggs or larval displacement (where the difference between berm height and WSE shares an inverse relationship with the likelihood of breaching—the bigger the difference in height and WSE, the smaller the likelihood of unseasonal breach). As equilibrium WSE decreases with decreasing VWRF discharge, the difference between berm height and WSE increases (Figure 5-14). Accordingly, reductions in VWRF discharge decrease the likelihood of unseasonal breaching. Scores for this metric range from a maximum of 2.0 for the least likelihood of unseasonal breach associated with Scenario 11 (i.e., 100% reduction in VWRF discharge; 0 MGD VWRF discharge) to 0.0 for the greatest likelihood of unseasonal breach associated with Scenario 1 (i.e., 0% reduction in VWRF discharge; existing VWRF discharge) (Table 5-20). Overall, VWRF discharge scenarios 7, 8, and 11 scored the best for supporting SPWN at 1.9, Scenarios 5, 6, 9, and 10 scored only slightly below at 1.8, and Scenarios 1-4 scored the lowest for support of SPWN, with scores ranging from 0.4 to 1.4.

Table 5-20. Scores received by each VWRF discharge scenario for each of the factors affecting the SPWN beneficial use.

Factor	Tier 2 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
Unseasonal breach effects on spawning and early development	70%	0.0	0.4	0.9	1.4	2.0	2.0	2.0	2.0	2.0	2.0	2.0
DO conditions	21%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Spawning habitat area	8%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
Tier 2 weighted sum	100%	0.4	0.7	1.0	1.4	1.8	1.8	1.9	1.9	1.8	1.8	1.9

5.5.10.4 SPWN Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing discharge scenarios for the SPWN beneficial use:

- *Water temperature:* Temperature modeling indicated that the impact of VWRF discharge upon SCRE temperature is not biologically relevant (+/- 0.5ºC variability across all VWRF discharge scenarios; Section 5.3.2.4), as temperature variability is predominantly controlled by solar insolation and radiation.
- *Salinity conditions initiating spawning behavior*: Some fish species spawning in the SCRE have been reported to have narrower salinity preferences for spawning than for rearing. These ranges suggest salinity might trigger spawning events for some fishes. However, a causal link between salinity and spawning behavior for these fishes has not been identified in literature and the observed narrower salinity range for spawning may be an epiphenomenon of other drivers of spawning behavior (e.g., phenology). Furthermore, reported preferred salinity ranges vary across the species likely to occupy the SCRE, making it difficult to identify a normative metric defining “good” salinity conditions for the range of native species potentially spawning in the SCRE (see Table 3-30).

5.5.11 Wetland Habitat (WET)

Wetland Habitat as a beneficial use is defined by the Basin Plan as “[u]ses of water that support wetland ecosystems, including, but not limited to, preservation or enhancement of wetland habitats, vegetation, fish, shellfish, or wildlife, and other unique wetland functions which enhance water quality, such as providing flood and erosion control, stream bank stabilization, and filtration and purification of naturally occurring contaminants." Riparian areas serve many of the functions that define the WET beneficial use, including bank stabilization, erosion control, and other water quality functions typically attributed to wetlands. The SWRCB has recognized multiple habitat types contributing to wetland functions, including freshwater, estuarine, and saltwater marshes, swamps, mudflats, as well as riparian areas (SWRCB Resolution Number 2008-0026). Additionally, because riparian areas within the SCRE were mapped as wetland habitats in the most recent wetland delineation of the SCRE (WRA 2014a), the evaluation of the WET beneficial use below considers both marsh habitats (freshwater and salt marsh) as well as riparian areas. The WET beneficial use was assigned a relative weighting of 7% (see Section 5.6 for more details).

5.5.11.1 WET Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting the WET beneficial use:

1. *Wetland habitat area*. Freshwater and salt marsh wetland habitat area was estimated under equilibrium closed-mouth, dry-weather conditions for each of the alternative VWRF discharge scenarios (Table 5-5). Freshwater wetland and salt marsh are represented by a single metric because modeled predictions of salt marsh habitat area do not vary across VWRF discharge scenarios. Adding salt marsh habitat to estimates of freshwater wetland habitat area was justified because both habitat types serve similar wetland related ecological functions. The following thresholds were adopted for comparing the extent to which discharge alternatives support freshwater and salt marsh wetland habitat in the SCRE:
    - Score of 2: 100% of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland).
    - Continuous score from 0-2 based on incremental increases in percent of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland) between 20% and 100%.
    - Score of 0: < 20% of the maximum modeled total wetland habitat area (freshwater wetland + salt marsh wetland).
2. *Riparian habitat area*. Riparian habitat area (riparian + riparian riverwash) was estimated under equilibrium closed-mouth, dry-weather conditions for each of the alternative VWRF discharge scenarios (Table 5-5). The consideration of riparian habitat area estimates as part of WET was justified because riparian areas generally and within the SCRE serve many of the same ecological functions as those identified as part of the WET beneficial use and riparian areas provide important habitat for native sensitive species. The following thresholds were adopted for comparing the extent to which discharge alternatives support riparian habitat in the SCRE:
    - Score of 2: 100% of the maximum modeled riparian habitat area (riparian + riparian riverwash).
    - Continuous score from 0 and 2: based on incremental increases in percent of maximum modeled riparian habitat area (riparian + riparian riverwash) between 20% and 100%.
    - Score of 0: < 20% of the maximum modeled riparian habitat area (riparian + riparian riverwash).

3. *Nutrient conditions supporting native wetland species*. Previous studies in the Santa Clara River watershed and elsewhere have shown that elevated nutrient conditions can degrade wetland and riparian habitats by fostering invasion by *Arundo* (Section 3.5.2.3). Given that background nutrient levels in water sources to the SCRE other than VWRF discharge are generally elevated well above levels supporting *Arundo* invasion (Section 5.3.2.2 and 3.5.2.3), large reductions in nutrient loading from VWRF sources would be required to establish nutrient-limiting conditions sufficient to potentially reduce the success of *Arundo* invasion in the SCRE. However, even incremental reductions in nutrient loading may confer some benefit to native species. Assuming constant VWRF nutrient concentrations across discharge scenarios, nutrient loading from VWRF discharge is assumed to decrease linearly with decreases in VWRF discharge. Accordingly, discharge scenarios are assigned a score based on a continuous scale from 1–2 with the best score for nutrient loads being assigned to the lowest VWRF discharge and the worst score for nutrient loads being assigned to the highest VWRF discharge. The following thresholds were adopted for comparing the extent to which discharge alternatives support nutrient conditions favorable to native species:
    - Score of 2: 100% reductions in VWRF N and P loads
    - Continuous score between 1 and 2: a continuous scale was used for scoring discharge scenarios based on their incremental reductions in VWRF N and P loads relative to current conditions
    - Score of 1: 0% reductions in VWRF N and P loads
4. *Flood control capacity*: Localized flood control capacity is a critical function of wetland systems. The area of wetland habitat between closed-mouth equilibrium WSE and the elevation of the McGrath State Beach Campground (9.5 ft NAVD88) was used to represent localized flood control capacity of wetlands. The following thresholds were adopted for comparing the extent to which alternative VWRF discharge scenarios support localized flood control capacity for the WET beneficial use:
    - Score of 2: Maximum modeled wetland habitat area between equilibrium WSE and elevation of McGrath State Beach Campground
    - Continuous score between 0 and 2: A continuous scale was used for discharge scenarios supporting between 0 acres and the maximum modeled wetland habitat area between equilibrium WSE and elevation of McGrath State Beach Campground
    - Score of 0: 0 acres of wetland habitat area between equilibrium WSE and elevation of McGrath State Beach Campground

5.5.11.2 Relative Weights of WET Factors

Freshwater and salt marsh habitat were given ~51% of the weighting for the WET beneficial use. These habitat types are locally more rare than riparian habitat within the SCRE and the Santa Clara River riparian corridor. Wildlife support functions of riparian habitat were separately assessed under the WILD beneficial use. Riparian habitat was given ~17% of the weighting for WET. Although it is a locally more abundant habitat type than wetlands, riparian habitat performs important functions related to WET, including bank stabilization, erosion control, and other water quality functions typically attributed to wetlands. Localized flood control capacity, measured as the amount of wetland habitat available to buffer surrounding areas during storm events, was given ~28% weighting. Nutrient conditions supporting native wetland species was given a low weighting (~4%) because Arundo is likely to remain a formidable competitor even under reduced nutrient load conditions due to the many inflow sources other than VWRF discharge contributing nutrients to the SCRE, and direct assessment of VWRF discharge nutrient loads on species using WET habitats is performed for other beneficial uses, including MIGR, SPWN, RARE and WILD.

5.5.11.3 Scoring VWRF Discharge Scenarios for AHP Analysis of WET

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each Scenario are summarized in Table 5-21. Total wetland habitat area (freshwater wetland + salt marsh wetland) is maximized under Scenario 1 at 47 acres (i.e., 0% reduction in VWRF discharge; existing VWRF discharge), and initially declines precipitously for VWRF discharge reductions of 0% to 20% (Scenarios 1–3) (Figure 5-23) due largely to the conversion of wetland habitat to riparian habitat. At VWRF discharge reductions of between 20% and 60% (Scenarios 3–7), there is less than 5% variation in the wetland habitat area conversion, but then wetland habitat begins to decline more steeply again under higher VWRF discharge reductions (Scenarios 8–11). Scenario scores for total wetland habitat range from 2.0 (Scenario 1) to 0.0 (Scenarios 10 and 11). Flood control capacity tends to increase with decreases in VWRF discharge with it being maximized under Scenario 11 (Figure 5-24). Scores for flood control capacity range from 0.0 (Scenarios 1–2) to 2.0 (Scenario 11) (Table 5-21). The total riparian habitat (riparian + riparian riverwash) tends to increase with decreasing VWRF discharge with it being maximized under Scenario 10 (324 acres) (Figure 5-25). Large portions of riparian habitat in the SCRE are unaffected by changes in VWRF discharge, so greater than 70% of maximum modeled riparian habitat (>227 acres) is maintained under all VWRF discharge scenarios. Scenario scores for support of riparian habitat range from 1.3 for Scenario 1 (i.e., 0% reduction in VWRF discharge; existing discharge) to 2.0 for Scenarios 10 and 11 (i.e., 90% to 100% reduction in VWRF discharge) (Table 5-21). Low nutrient loading is achieved by reducing flow, so discharge scenarios associated with increasing VWRF discharge reductions scored best for low nutrient loading (Table 5-21). Overall, Scenario 1 results in the greatest realization of the WET beneficial use with a score of 1.3, Scenarios 3 and 4 result in the lowest scores of 0.7, and scores for the remaining scenarios ranging between 0.8 and 1.0 (Table 5-21).

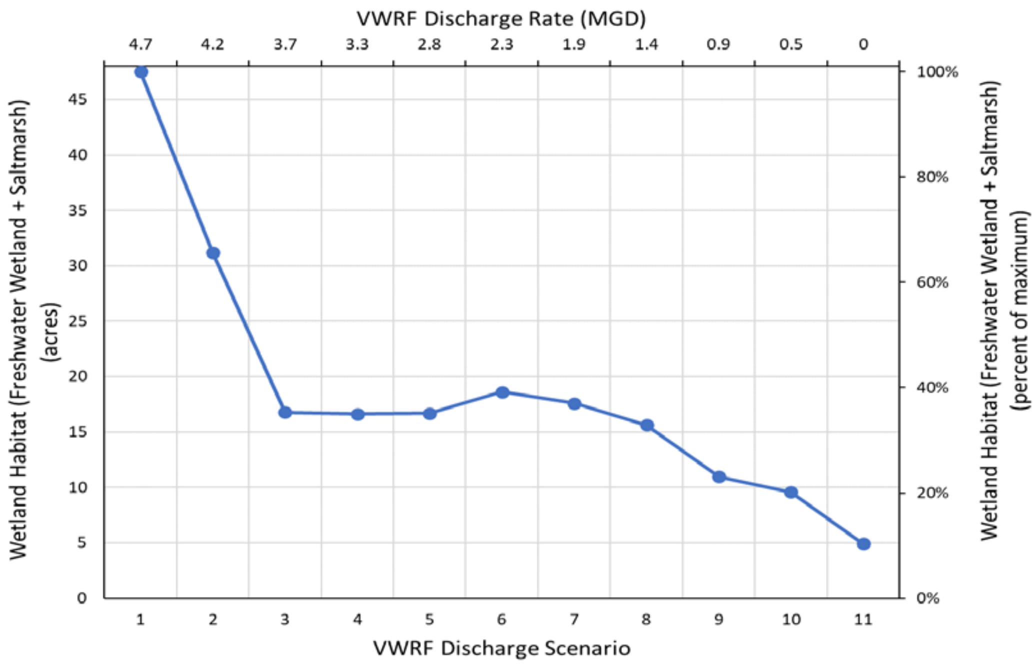


Figure 5-23. Wetland habitat area in the SCRE as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) as a function of VWRF discharge scenario.

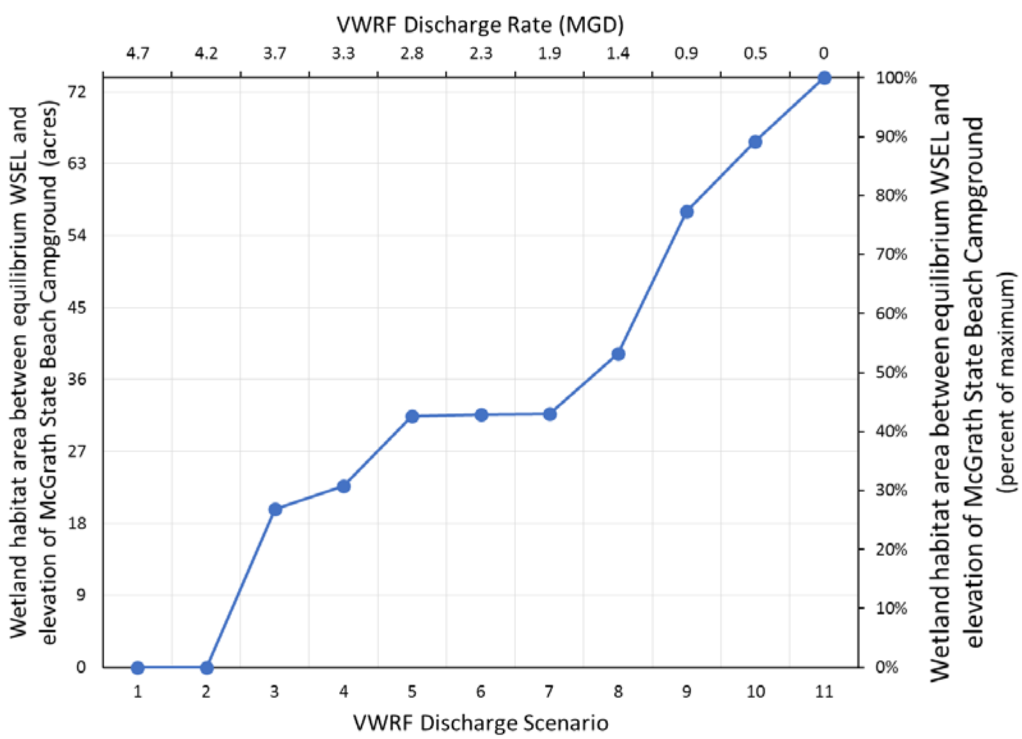


Figure 5-24. Flood control capacity in the SCRE as acres of wetland habitat for flood control (primary vertical axis) and percent of maximum modeled flood control habitat area (secondary vertical axis) as a function of VWRF discharge scenario.

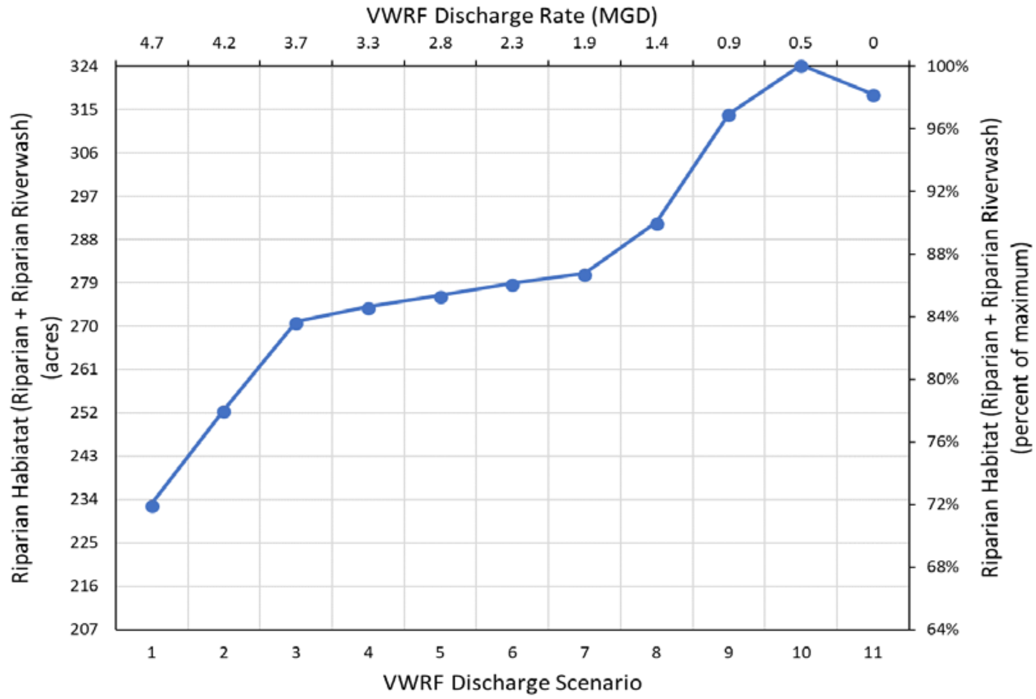


Figure 5-25. Riparian habitat area in the SCRE as acres (primary vertical axis) and percent of maximum modeled area (secondary vertical axis) as a function of VWRF discharge scenario.

Table 5-21. Scores received by each VWRF discharge scenario for each of the factors affecting the WET beneficial use.

Factor	Tier 2 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
Total wetland habitat area	51%	2.0	1.1	0.4	0.4	0.4	0.5	0.4	0.3	0.1	0.0	0.0
Flood control capacity	28%	0.0	0.0	0.5	0.6	0.9	0.9	0.9	1.1	1.5	1.8	2.0
Riparian habitat area	17%	1.3	1.4	1.6	1.6	1.6	1.7	1.7	1.7	1.9	2.0	2.0
Nutrient conditions affecting *Arundo* success	5%	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
Tier 2 weighted sum	100%	1.3	0.9	0.7	0.7	0.8	0.9	0.8	0.8	0.9	0.9	1.0

5.5.11.4 WET Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing discharge scenarios for the WET beneficial use:

- *Water surface elevation impacts on Arundo invasion*: *Arundo* abundance as a function of equilibrium WSE is expected to follow patterns of modeled freshwater wetland habitat.

Species composition within the freshwater wetland habitat may be impacted by nutrient and salinity conditions, as described above for this beneficial use.

- *Salinity conditions selecting against Arundo: Arundo* has a high salinity tolerance, such that only salinities exceeding 26 ppt for extended periods would be expected to shift competitive dynamics in favor of native wetland species. The SCRE is expected to remain a predominantly freshwater system under all closed-mouth, dry-weather VWRF discharge scenarios. Further, salinity modeling suggests that salinity in the SCRE will not exceed 26 ppt for extended periods of time under any of the VWRF discharge scenarios (Figure 5-6).

5.5.12 Wildlife Habitat (WILD)

Wildlife Habitat as a beneficial use is defined by the Basin Plan as "[u]ses of water that support terrestrial ecosystems including, but not limited to, preservation and enhancement of terrestrial habitats, vegetation, wildlife (e.g., mammals, birds, reptiles, amphibians, invertebrates), or wildlife water and food sources." The WILD beneficial use was assigned a relative weighting of 7% (see Section 5.6 for more details).

5.5.12.1 WILD Factors and Metrics for AHP Analysis

The following metrics were used to assess the relative effects of VWRF discharge on factors affecting the WILD beneficial use:

1. *Riparian habitat area*. Riparian areas provide habitat for a variety of wildlife that occupy and use the SCRE, contributing to its ecological value, including birds, mammals, and herpetofauna (Section 3.7.1). Riparian habitat area (riparian + riparian riverwash) was estimated under equilibrium closed-mouth, dry-weather conditions for each of the VWRF discharge scenarios (Table 5-5). Percent of maximum modeled riparian habitat area was used to represent relative availability of habitat for riparian species. The following thresholds were adopted for comparing the extent to which discharge alternatives support riparian habitat in the SCRE:
    - Score of 2: 100% of the maximum modeled total riparian habitat area (riparian + riparian riverwash).
    - Continuous score from 0-2: based on incremental increases in percent of the maximum modeled total wetland habitat area (riparian + riparian riverwash) between 20% and 100%.
    - Score of 0: < 20% of the maximum modeled total riparian habitat area (riparian + riparian riverwash).
2. *Mudflat habitat area.* Mudflat habitat provides foraging opportunities for a variety of birds and other wildlife (Section 3.7.1), including the western snowy plover that is more specifically assessed under RARE. Tidally exposed mudflat habitat was estimated under open-mouth conditions for each of the alternative VWRF discharge scenarios (Table 5-5). Percent of maximum modeled mudflat habitat area was used to represent relative availability of habitat for species foraging in mudflats. The following thresholds were adopted for comparing the extent to which discharge alternatives support riparian habitat in the SCRE:

- Score of 2: 100% of the maximum modeled mudflat habitat area
- Continuous score from 0 and 2 based on incremental increases in percent of maximum modeled mudflat habitat area between 20% and 100%.
- Score of 0: < 20% of the maximum modeled mudflat habitat area

3. *Open water habitat area.* Open water provides foraging opportunities for a variety of bird and other wildlife species (Section 3.7.1), including the California least tern that is more specifically assessed under RARE. Open water habitat area was estimated under equilibrium closed-mouth, dry-weather conditions for each of the alternative VWRF discharge scenarios (Table 5-5). Percent of maximum modeled open water habitat area was used to represent relative availability of open water foraging habitat for wildlife species. The following thresholds were adopted for comparing the extent to which discharge alternatives support open water habitat in the SCRE:
    - Score of 2: 100% of the maximum modeled open water habitat area
    - Continuous score from 0-2 based on incremental increases in percent of maximum modeled open water habitat area between 20% and 100%.
    - Score of 0: < 20% of the maximum modeled open water habitat area

5.5.12.2 Relative Weights of WILD Factors

Riparian habitat is given the majority of weighting (~69%) for the WILD beneficial use. Riparian habitats comprise large portions of the SCRE and support multiple life history stages of a variety of native and sensitive wildlife species. Riparian habitats in the SCRE also serve as a proxy for potential southwestern willow flycatcher habitat that, per the USFWS critical habitat designation, may over time develop the physical or biological features necessary to support the listed bird. Mudflat habitat is given ~22% weighting, while open water habitat is given ~9%. Mudflat habitats can provide important foraging habitat for a variety of wildlife species, including the western snowy plover and other shorebird wildlife species that are not explicitly included in evaluations of other beneficial uses. Open water habitat supports foraging for the California least tern, as well as a variety of other birds, herpetofauna, and wildlife species not explicitly included in evaluations of other beneficial uses. However, open water weighting is somewhat reduced in this assessment based on the overlapping function of open water habitats in the EST and RARE beneficial use assessments.

5.5.12.3 Scoring of VWRF Discharge Scenarios for AHP Analysis of WILD

VWRF discharge scenarios were evaluated and scored based on the criteria described above. Scores for each Scenario are summarized in Table 5-22. Riparian habitat area is maximized under Scenario 10 (324 acres) and generally decreases with increasing VWRF discharge (Figure 5-25). Scenarios 3–11 (i.e., 20% to 100% reduction in VWRF discharge) support over 80% of the maximum modeled riparian habitat area. VWRF discharge has limited impact on the area of riparian habitats relative to other habitat types, as greater than 70% of the maximum modeled riparian habitat area is supported under all VWRF discharge scenarios (Figure 5-25). Scenario scores for support of riparian habitat range from 2.0 (Scenario 10 and 11) to 1.3 (Scenario 1). Tidally exposed mudflat habitat area is maximized under Scenario 1 (i.e., 0% reduction in VWRF discharge; existing VWRF discharge) at ~66 acres at low tide and decreases with reductions in VWRF discharge (Figure 5-26). Scenarios 10 and 11 support only 17% and 4% of maximum modeled mudflat habitat area, respectively. Scenario scores for support of mudflat habitat range from 2.0 (Scenarios 1 and 2) to 0.0 (Scenarios 10 and 11). Open water habitat area for wildlife species is maximized under Scenario 1 (108 acres) and decreases nearly linearly with decreasing VWRF discharge (Figure 5-8). Scenario scores for support of open water habitats for wildlife species range from 2.0 for Scenario 1 to 0.4 for Scenario 11. Overall, VWRF discharge scenarios 1, 3, 4, 6, and 7 score the highest for support of WILD at 1.5, while all the remaining scenarios score only slightly less at 1.4.

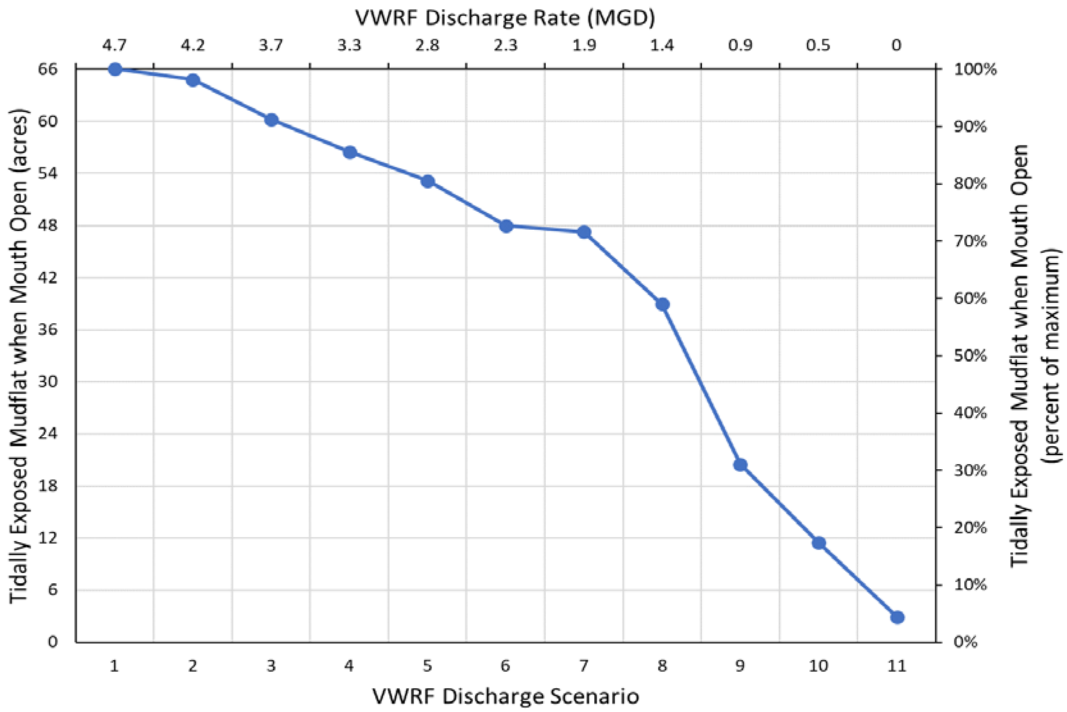


Figure 5-26. Tidally exposed mudflat habitat area in the SCRE as total acres (primary vertical axis) and percent of maximum habitat area (secondary vertical axis) as a function of VWRF discharge scenario.

Table 5-22. Scores received by each VWRF discharge scenario for each of the factors affecting the WILD beneficial use.

Factor	Tier 2 weight	Scores by VWRF discharge scenario										
		1	2	3	4	5	6	7	8	9	10	11
Open water habitat area	69%	2.0	1.9	1.8	1.7	1.6	1.5	1.5	1.3	0.8	0.6	0.4
Mudflat habitat area	22%	2.0	2.0	1.8	1.6	1.5	1.3	1.3	1.0	0.3	0.0	0.0
Riparian habitat area	9%	1.3	1.4	1.6	1.6	1.6	1.7	1.7	1.7	1.9	2.0	2.0
Tier 2 weighted sum	100%	1.5	1.4	1.5	1.5	1.4	1.5	1.5	1.4	1.4	1.4	1.4

5.5.12.4 WILD Factors not Included in AHP Analysis

The following factors were considered, but not included as assessment metrics for comparing discharge scenarios for the WILD beneficial use:

- *Open beach and foredune habitat area*. Though wildlife species, including birds, mammals, and herpetofauna use open beach and foredune habitat for foraging and nesting, extensive habitat is available in nearby areas and VWRF discharge has only minimal impact on available habitat areas.
- *Food resources*: Availability and composition of food resources in foraging habitat areas is assumed to be proportional to habitat area and otherwise not significantly impacted by VWRF discharge. Available data are not sufficient for direct, quantitative predictions of the changes in food resources for the wide variety of wildlife species under alternative VWRF discharge scenarios.
- *Freshwater wetland habitat*: Freshwater wetland habitat for wildlife species is assessed directly as the WET beneficial use, so the assessment was not duplicated for WILD.

5.6 Synthesis of Beneficial Use Analysis

In Sections 5.5.1 through 5.5.12, VWRF discharge scenarios were scored for each beneficial use in isolation by considering each factor on the terminal tier of the Analytic Hierarchy, with relative weights applied at tiers 2 and 3 (i.e., for factors contributing to each individual beneficial use). In this Section, we apply the relative weights of each beneficial use (i.e. tier 1 of the analytic hierarchy) to evaluate the extent to which each VWRF discharge scenario supports the weighted balance of all beneficial uses. Weights for beneficial uses were developed in the exercise of best professional judgment, with the input of TRT experts, Resources Agencies, and stakeholders, as determined through the Analytic Hierarchy Process (AHP) described in Section 5.1 and Appendix H.

The beneficial use weighting prioritizes the protection of native listed species and designated critical habitats, as well as other ecological functions of the SCRE (Table 5-23), consistent with stakeholder feedback during prior study phases and regulatory requirements (including prohibitions against "take" of state and federally listed endangered and threatened species and adverse modification of designated critical habitat).

Table 5-23. Prioritized ranking and relative weighting of beneficial uses as determined through the Analytic Hierarchy Process.

Rank	Beneficial use	Relative Weighting
1	Rare, Threatened, or Endangered Species (RARE)	35%
2	Migration of Aquatic Organisms (MIGR)	16%
3	Estuarine Habitat (EST)	15%
4	Spawning, Reproduction, and/or Early Development (SPWN)	10%
5	Wildlife Habitat (WILD)	7%
6	Wetland Habitat (WET)	7%
7	Municipal and Domestic Use (MUN)	5%
8	Non-contact Water Recreation (REC-2)	2%
9	Commercial and Sport Fishing (COMM)	1%
10	Water Contact Recreation (REC-1)	1%
N/A	Navigation (NAV)	0%
N/A	Marine Habitat (MAR)	0%

The weighting shown in Table 5-23 was applied to the scores received by each discharge scenario for each beneficial use. These scores were then normalized and summed across beneficial uses (Table 5-24). This normalized priority score is a generalized metric of the extent to which each discharge scenario supports the balance of beneficial uses in the SCRE. Because beneficial uses related to protection of the SCRE's native listed species and related critical habitats were weighted most heavily in the analysis, these priority scores generally represent the extent to which those functions are supported as well. Figure 5-27 shows the normalized priority score color-coded by beneficial use. Figure 5-28 shows the normalized priority score as a percent of maximum to simplify comparison of VWRF discharge scenarios.

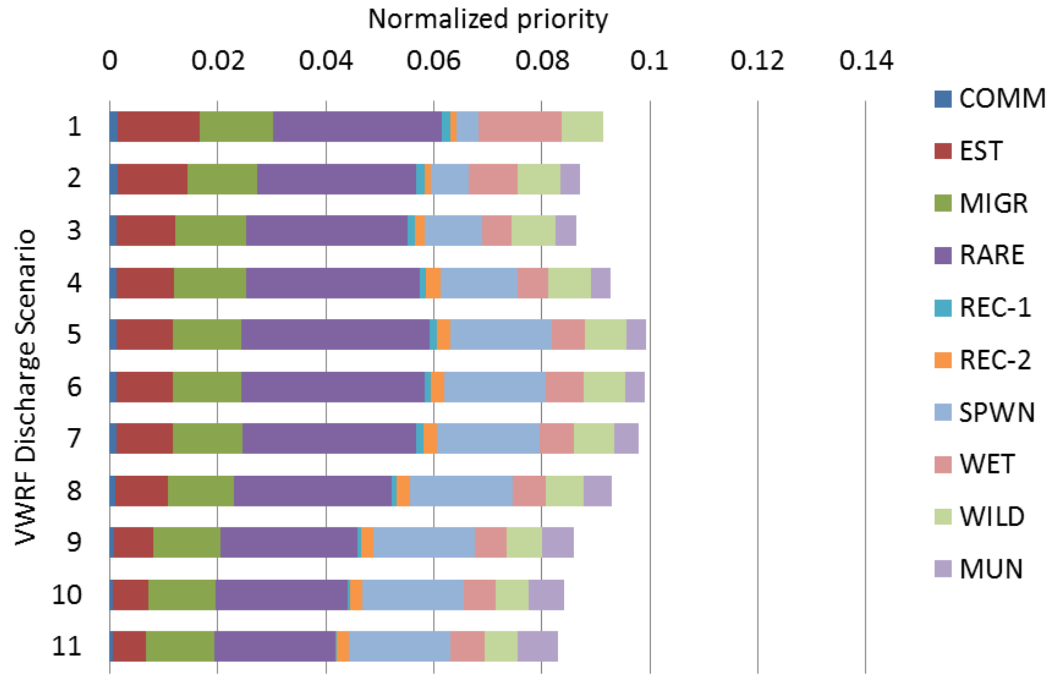


Figure 5-27. Weighted scores (normalized such that the sum of scores across discharge scenarios equals 1) for each discharge scenario with colored bars representing the weighted scores for individual beneficial uses, as affected by VWRF discharge.

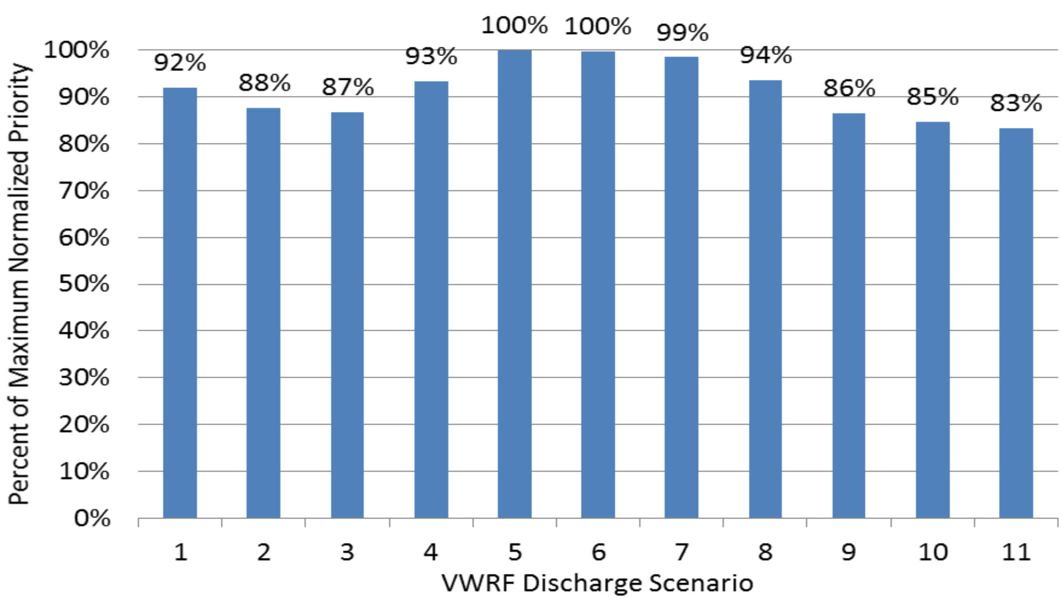


Figure 5-28. Percent of maximum normalized priority for each VWRF discharge scenario, where 100% represents the maximum realization of the balance of beneficial uses in the SCRE, as affected by VWRF discharge.

Table 5-24. Normalized weighted scores by beneficial use, normalized priority (i.e., normalized cumulative weighted score), and priority (% of maximum normalized priority) for each VWRF discharge scenario.

	VWRF Discharge Scenario										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
Discharge to SCRE (MGD)	4.7	4.2	3.7	3.3	2.8	2.3	1.9	1.4	0.9	0.5	0
Percent reduction from current VWRF discharge	0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Modeled Equilibrium SCRE Stage (ft, NAVD88)	10.2	9.8	9.3	8.8	8.1	7.5	7	6.4	5.8	5.3	4.7
Normalized Weighted Scores for Beneficial Uses											
COMM	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
EST	0.015	0.013	0.011	0.011	0.010	0.011	0.011	0.010	0.007	0.006	0.006
MIGR	0.014	0.013	0.013	0.013	0.013	0.013	0.013	0.012	0.012	0.013	0.013
MUN	0.000	0.004	0.004	0.004	0.004	0.004	0.004	0.005	0.006	0.007	0.007
RARE	0.031	0.029	0.030	0.032	0.035	0.034	0.032	0.029	0.025	0.024	0.022
REC-1	0.002	0.002	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.000	0.000
REC-2	0.001	0.001	0.002	0.003	0.003	0.003	0.003	0.002	0.002	0.002	0.002
SPWN	0.004	0.007	0.011	0.014	0.019	0.019	0.019	0.019	0.019	0.019	0.019
WET	0.016	0.009	0.005	0.006	0.006	0.007	0.006	0.006	0.006	0.006	0.006
WILD	0.008	0.008	0.008	0.008	0.008	0.008	0.008	0.007	0.006	0.006	0.006
Normalized Score and Priority (% of maximum score) Across All Beneficial Uses											
Cumulative Score	0.091	0.087	0.086	0.093	0.099	0.099	0.098	0.093	0.086	0.084	0.083
Priority (% of maximum)	92%	88%	87%	93%	**100%**	**100%**	**99%**	94%	86%	85%	83%
Normalized Score and Priority (% of maximum score) Across Ecological Beneficial Uses (COMM, EST, MIGR. RARE, WET, WILD)											
Cumulative Score	0.089	0.08	0.079	0.085	0.092	0.093	0.09	0.084	0.076	0.075	0.073
Priority (% of maximum)	96%	86%	85%	91%	**99%**	**100%**	**97%**	90%	82%	81%	78%

5.6.1 Recommendations regarding beneficial use enhancement under the enclosed bays and estuaries policy

As stated in Section 1.2, a primary goal of this Report is to recommend the Enhancement Discharge Levels, i.e., the levels of VWRF discharge, if any, that provide fuller realization of beneficial uses of the SCRE and surrounding subwatershed relative to the absence of all discharge. This recommendation is necessary to comply with requirements of the NPDES Permit, and to facilitate the RWQCB's determination of "enhancement," as defined by EBE Policy, associated with different levels of discharge. As required by the EBE Policy, in evaluating the potential effects of these 11 VWRF discharge scenarios on beneficial uses, the potential effects of each of the Scenarios 1–11 were not only compared against one another, but each of Scenarios 1-10 were also compared to Scenario 11 (i.e., the absence of continued discharge). Accordingly, the recommendation below regarding Enhancement Discharge Levels constitutes the conclusions of our assessment of the degree to which the various VWRF discharge scenarios either result in a "new beneficial use," or "fuller realization of an existing designated beneficial use" that would occur in the absence of all point source discharges. (SWRCB, WQ 79-20, p. 10 [May 17, 1979]).

The results of the beneficial use analyses above indicate that in comparison to the absence of VWRF discharge to the SCRE (Scenario 11), continued tertiary treated effluent discharges for Scenarios 5 through 7 in the amounts ranging from 1.9–2.8 MGD (i.e., 40–60% of current levels) constitute Enhancement Discharge Levels, providing fuller realization of the balance of beneficial uses important to protection of the SCRE relative to the absence of all VWRF discharge (Figure 5-27). This also holds true when considering only those beneficial uses with ecological benefit (Table 5-24). Although beyond the scope of EBE Policy requirements, the results of the analysis support the conclusion that continued discharge of tertiary treated effluent in amounts ranging from 1.9 to 2.8 MGD associated with Scenarios 5 through 7 optimize realization of beneficial uses as compared to all other VWRF discharge scenarios assessed (Figure 5-28).

To provide additional context and transparency regarding the recommended Enhancement Discharge Levels, information from the preceding sections show that VWRF discharges in the amounts of 1.9 to 2.8 MGD contribute to fuller realization of existing designated beneficial uses because, relative to the absence of discharge, this discharge flow range is associated with the following attributes:

- Increased physical aquatic habitat areaassociated with rearing, foraging, and spawning for native fish, including the listed tidewater goby and steelhead, and native wildlife, including the listed snowy plover, currently using and occupying the SCRE (COMM, EST, RARE, SPWN);
- Increased freshwater wetland providing important functions including habitat for wildlife species, bank stabilization, erosion control, localized flood control capacity, and associated water quality functions (WET, WILD);
- Increased opportunities for water contact and non-contact recreation (REC-1, REC-2).

Although the recommended Enhancement Discharge Levels support realization of each of the beneficial uses on balance, it should be noted that not all of the factors and attributes contributing to each beneficial use were improved by the Enhancement Discharge Levels of 1.9–2.8 MGD. For example, it should be noted that the City proposes to divert to planned potable reuse all of the tertiary treated VWRF effluent not discharged to the SCRE to improve quantity, quality, and reliability of local water supplies, resulting in realization of the municipal water supply (MUN) uses within the SCRE sub-watershed. Because this Report recommends Enhancement Discharge Levels of 1.9 to 2.8 MGD, those flows may not be available for diversion (depending on determinations made by the RWQCB and other Resources Agencies) to water reclamation uses that would otherwise benefit municipal water supply (MUN) uses. Further, reducing VWRF discharges more than permitted by maintenance of the Enhancement Discharge Levels, as proposed in Scenarios 8-11, would increase potable reuse and further enhance MUN uses as shown by the assessment of Scenario 11 (100% diversion, 0 MGD discharge) effects on MUN (Section 5.5.5). Additionally, nutrient loading was directly considered as one of the assessment factors for each of the beneficial uses related to habitat for aquatic species (COMM, EST, MIGR, RARE, SPWN) and VWRF discharges contribute significantly to nutrient loads in the SCRE during closed-mouth, dry-weather conditions. Although background nutrient levels within the Santa Clara River and in shallow groundwater within portions of the SCRE sub-watershed are above saturating conditions for algal uptake and growth, additional nutrients associated with VWRF discharges may exacerbate the frequency and intensity of algal blooms and associated periods of unsuitable DO conditions for aquatic species. Lastly, the amounts of riparian habitats affecting several beneficial uses (RARE, WET, WILD), as well as the duration of elevated salinity condtions variations following breaching events (EST, RARE) would be greater in the absence of VWRF discharge. Based upon the weighting and scoring developed with input from the TRT and Resources Agencies (see Sections 5.1, 5.5, and Appendix H), however, the benefits associated with the recommended Enhancement Discharge Levels of 1.9–2.8 MGD outweigh the potential adverse effects (Table 5-24 and Figure 5-28).

The potential for adverse impacts to factors related to individual beneficial uses discussed above, as well as uncertainties in the assessments summarized in Section 5.6.3 below, indicate the need for adaptive management measures (Section 5.6.5) to ensure that listed species are not adversely impacted and that an appropriate balance of beneficial uses is attained.

5.6.2 MEPDV recommendations

As stated in Section 1.2, the second and third goals of this report are to recommend the maximum ecologically protective diversion volume (MEPDV) that may be diverted from the VWRF discharge, as well as the Continued Discharge Levels, if any, needed to protect the SCRE's existing ecological resources, particularly native species and habitats with a focus on listed species and related critical habitats protected under the state and federal Endangered Species Acts. For the purposes of identifying the recommended MEPDV, the beneficial use assessment conducted for this study indicates that the balance of all ecological beneficial uses (as determined by weighted AHP scores for COMM, EST, MIGR, RARE, SPWN, WET, and WILD) is maximized by Scenarios 5 through 7, with Scenarios 1, 4 and 8 exhibiting intermediate scores, and Scenarios 2, 3, 9, 10 and 11 exhibiting the lowest scores (Table 5-24). When compared with reduced VWRF discharge scenarios considered in this report, current VWRF discharge (Scenario 1) and discharge reductions of 10–30% (Scenarios 2-4) are associated with increased risks of unseasonal breaching (Figure 5 14), increased potential for nest flooding (Figure 5-16), increased nutrient loads and related algal bloom effects such as DO and pH variability (e.g., Table 5 10), reduced riparian habitat availability (Figure 5 17), and reduced duration of elevated salinity conditions following breach events (Figure 5 7), among other effects. Based upon review of these results and the underlying ecological considerations reviewed in this report, we consider diversions of up to 60% from current discharge to water reclamation uses (Scenario 7) to be ecologically protective (Table 5-24; Figures 5-27; Figure 5-28) and recommend:

- An MEPDV of 60% of VWRF discharges, corresponding to a diversion of 2.8 MGD under current conditions; and
- Correspondingly, a minimum Continued Discharge Level of 1.9 MGD to be maintained to the SCRE during closed-mouth, dry-weather conditions.

It should be noted that if additional tertiary-treated effluent becomes available in the future, the diversion flow volume would be expected to increase above the 2.8 MGD MEPDV recommendatin in this report, as long as the Continued Discharge Level of 1.9 MGD to the SCRE is maintained during closed-mouth, dry-weather conditions.

Although we recommend an MEPDV of 60% and corresponding Continued Discharge Level of 1.9 MGD, it should be noted that weighted AHP scores indicate increased support for some beneficial uses under VWRF discharge Scenario 1 (4.7 MGD) representing current conditions (COMM, WET), whereas other (e.g., MIGR) are largely unaffected relative to reduced discharges. For example, freshwater marsh decreases rapidly with modeled 10–20% reductions in VWRF discharge (Scenarios 2 and 3); although the estimated marsh habitat area remains largely unchanged between 20–70% flow reductions. Further, while the total area of habitat types supporting WET beneficial uses (i.e., the summed area of freshwater marsh, saltmarsh, and riparian habitat) generally increases with increasing flow reductions relative to current conditions, the increase is largely due to increases in riparian habitat, while freshwater marsh declines and saltmarsh remains constant. Although riparian habitats support wetland functions and values (See Section 5.5.11), these areas may be of lower habitat value than freshwater marsh areas for particular species, and may not provide the same level of erosion control, bank stabilization, and, particularly water quality treatment value provided by freshwater marsh or wetland habitat types. Nevertheless, based on the Phase 3 Studies assessments, the risks related to maintenance of current VWRF discharge levels outweigh the benefits to some uses (Table 5-24).

In examining the potential for reductions in current VWRF discharges to result in adverse modification of designated critical habitat and incidental "take" of listed species evaluated in this study, we reviewed the description of primary constituent elements (PCEs) included in the individual critical habitat designations along with available basin-specific and regional information. These information sources were considered in selection of criteria used to assess potential effects of VWRF discharge reductions upon listed species. PCEs and the more recently used term "physical or biological features" (PBFs) established in updated Critical Habitat regulations (81 FR 7414) are often generalized into categories such as: (1) Space for individual and population growth and for normal behavior; (2) Food, water, air, light, minerals, or other nutritional or physiological requirements; (3) Cover or shelter; (4) Sites for breeding, reproduction, or rearing (or development) of offspring; and (5) Habitats that are protected from disturbance or are representative of the historical, geographical, and ecological distributions of a species. Information regarding PCEs in the critical habitat designations was then used to develop more specific assessment factors and metrics for determining effects of the different VWRF discharge scenarios for steelhead (Section 3.6.3.1), tidewater goby (Section 3.6.3.2), western snowy plover (Section 3.7.2.1), and California least tern (Section 3.7.2.2). Based on review of the PCEs/PBFs for the listed species in the SCRE in the sections below, and with consideration that discharge reductions may adversely affect some PCEs/PBFs, but benefit others, we recommend that an MEPDV of 60% reduction in existing VWRF discharge, with a Continued Discharge Level of 1.9 MGD during closed-mouth, dry-weather conditions will not result in take of listed species.

5.6.2.1 Southern California steelhead

After review of general life history requirements for southern California steelhead (Section 3.6.3.1), the extent to which listed PCEs (Table 3-32) are realized in the existing condition within the SCRE, their relationship with VWRF discharge, and the feasibility of realizing certain PCEs based on the changes from historical habitat conditions in the Santa Clara River and SCRE were all considered in evaluating the potential for take under reduced discharge scenarios. PCEs related to spawning and early rearing (PCE 1; Table 3-32) are unaffected by VWRF discharge since this occurs well upstream of the SCRE. Similarly, PCE 2(i)i is not assessed here since floodplain habitat is largely absent in the SCRE, and water quality conditions supporting juvenile growth are assessed under PCE 2(ii). As described in Section 3.6.3.1, depth provides the predominant source of cover (PCE 2iii) for rearing steelhead, since the SCRE lacks large woody debris and other structural components that comprise natural cover in some other systems. Analysis of the intersection of the modeled equilibrium WSEs for the alternative VWRF discharge scenarios (Table 5-5) with the underlying bathymetric surface (Figure 3-2) shows the recommended 60% MEPDV represents a total of 38 acres that correspond to water depths equal to or exceeding 1.5 ft (0.5 m), and approximately 21 acres would correspond to water depths equal to or exceeding 2.5 ft (0.75 m)(PCE 2). Recognizing that a range of factors affect the potential for steelhead rearing in the SCRE (Section 3.6.3.1), the areas estimated above indicate that there is sufficient physical habitat to support a steelhead rearing population far larger than has been recently observed in the SCRE.

Water quality conditions for steelhead rearing and migration through the SCRE are expected to improve under reduced discharge scenarios, including the recommended MEPDV. Although lagoon water temperatures are marginally suitable for rearing (Section 3.6.3.1), they are largely unaffected by VWRF discharges (Section 5.3.2.4). Reductions in physical habitat area for steelhead associated with reduced VWRF discharge scenarios would be accompanied by reductions in nutrient loading from the VWRF, decreasing the likelihood of adverse dissolved oxygen conditions. Forage (PCEs 2i, 2ii, and 4iii; Table 3-32) is not expected to be adversely affected by the recommended MEPDV. Food conditions are difficult to predict given available data, but improved water quality conditions under the recommended MEPDV may improve food abundance and/or quality. Additionally, the likelihood of unseasonal breaching is expected to decrease under the proposed MEPDV relative to current conditions, potentially reducing juvenile mortality risk, and improving conditions for the physiological salinity transitions (PCE 4i; Table 3-32) of any steelhead using the SCRE.

Modeling of breaching dynamics conducted as part of this study indicated that reductions in VWRF current discharges by the 60% results in minor reductions migratory access by steelhead as represented by the modeled duration of open mouth conditions following breaching (Figure 5-10). Given that connectivity with upstream habitats is largely dependent on the presence of Santa Clara River flows capable of breaching the SCRE mouth berm, the small reductions in modeled open mouth duration are not considered to represent an obstruction to juvenile or adult freshwater migration (PCE 3; Table 3-32). Although VWRF discharge reductions are not expected to reduce the presence of predatory fish species, aquatic predators capable of preying on smolt-sized steelhead are not abundant in the SCRE, and areas with sufficient depth cover from predation by wading birds are predicted to be maintained under 60% reductions in VWRF current discharge (PCE 3 and 4; Table 3-32). Lastly, although human-related disturbance is not specifically listed as a PCE, the likelihood of unseasonal breaching is predicted to decrease under the proposed 60% MEPDV relative to current conditions.

Based on these assessments, we recommend that the MEPDV of 60% reduction in current VWRF discharges, with a Continued Discharge Level of 1.9 MGD in critical, dry weather, closed mouth conditions is not likely to adversely affect steelhead currently using and occupying the SCRE.

5.6.2.2 Tidewater goby

The extent to which listed PCEs for tidewater goby (Table 3-33) are realized in the existing condition within the SCRE, their relationship with VWRF discharge, and the feasibility of realizing certain PCEs based on the changes from historical habitat conditions in the Santa Clara River and SCRE were all considered in evaluating the potential for take under reduced discharge scenarios. Persistent, shallow, still-to-slow-moving, aquatic habitat with salinity below 12 ppt (PCE 1) is expected to be abundant under 60% reductions in existing VWRF discharge. Modeling of the Continued Discharge Level of 1.9 MGD discharge associated with the recommended 60% MEPDV represents a total of 70 acres of physical habitat with suitable depths for tidewater goby rearing (Figure 5-12), well within the range considered as supporting stable populations by USFWS (2005). In regard to habitat quality, salinity conditions are unlikely to change, but could slightly improve with 60% reductions in VWRF existing discharge such that no impact or a slight benefit to tidewater goby (see Figure 5-7) may be predicted to result from implementation of the MEPDV. DO conditions are also likely to improve as a result of reduced nutrient loading associated with implementation of the MEPDV. Burrowing substrate availability (PCE 2) within the preferred tidewater goby depth range is expected to increase with implementation of the MEPDV 60% reductions in VWRF existing discharge relative to current conditions.

Given the small size of the tidewater goby population currently supported in the SCRE despite the abundant habitat availability, it is likely that predation and competition, not habitat area, are controlling tidewater goby populations in the SCRE. Although non-native predators/competitors are expected to be abundant under all VWRF discharge scenarios, submerged aquatic vegetation that may be used as cover (PCE 3) is expected to be available under 60% reductions in VWRF existing discharge, though margin areas associated with freshwater marsh habitat may be slightly reduced relative to current conditions. Competition and predation by non-native species is likely to still occur under reduced VWRF discharge scenarios (Section 5.5.7.1). Lastly, the likelihood of unseasonal breaching is predicted to decrease under the proposed 60% MEPDV relative to current conditions (PCE 4).

Based on these assessments, we recommend that the MEPDV of 60% reduction in current VWRF discharges, with a Continued Discharge Level of 1.9 MGD in critical, dry weather, closed mouth conditions is not likely to adversely affect tidewater goby currently using and occupying the SCRE.

5.6.2.3 Western snowy plover

The extent to which listed PCEs for western snowy plover (Table 3-35) are realized in the existing condition within the SCRE, their relationship with VWRF discharge, and the feasibility of realizing certain PCEs based on the changes from historical habitat conditions in the Santa Clara River and SCRE were all considered in evaluating the potential for take under reduced discharge scenarios. Acordingly, VWRF flow reductions of 60% associated with implementation of the MEPDV are not expected to affect western snowy plover breeding or foraging habitat within the SCRE. As modeled in this Study, a 60% reduction in VWRF existing discharge results in almost no reduction in foraging habitat for western snowy plover Figure 5-15) or nesting habitat (Figure 5-16). For this reason, PCEs listed in Table 3-35 related to nesting of plover such as areas between heavily vegetated or developed areas and the daily high tide lines (PCE 1), wind-driven wrack such as seaweed and driftwood along the surf-line (PCE 3), as well as potential forage habitat along the lagoon margin of the SCRE mouth berm (PCE 2) are largely unaffected by VWRF existing discharge reductions on the order of 60%. Lastly, while some reduction in the potential for nest flooding will be accompanied by the recommended MEPDV (PCE 4), other sources of disturbance from the presence of humans, pets, vehicles, or human-attracted predators are otherwise not expected to vary with VWRF discharge reductions.

Based on these assessments, we recommend that the MEPDV of 60% reduction in current VWRF discharges, with a Continued Discharge Level of 1.9 MGD in critical, dry weather, closed mouth conditions is not likely to adversely affect snowy plover currently using and occupying the SCRE.

5.6.2.4 California least tern

Although no specific PCEs have been developed for California least tern, based upon the generalized PCEs in Table 3-37 neither the potential changes in foraging habitat (PCEs 1 and 2) within the habitat analysis boundary of the SCRE nor the 40% reduction in open water rearing habitat (Figure 5-8) accompanying a MEPDV 60% VWRF flow reduction are expected to adversely impact California least tern. These reductions in potential foraging habitat area are not expected to restrict access to food resources because of the large areas of nearby foraging habitat in the Pacific Ocean, McGrath Lake, and Ventura Harbor. While some reduction in the potential for nest flooding will be accompanied by the recommended MEPDV (PCE 5), available cover and shelter (PCE 3), access to available nesting sites shown in Figure 5-16 (PCE 4), and other sources of disturbance (PCE 5) is largely unaffected by variations in VWRF discharge.

Based on these assessments, we recommend that the MEPDV of 60% reduction in current VWRF discharges, with a Continued Discharge Level of 1.9 MGD in critical, dry weather, closed mouth conditions is not likely to affect snowy plover currently using and occupying the SCRE.

5.6.3 Caveats and uncertainty

The analysis presented in this report relies upon data and literature review to develop conceptual models and modeling tools for the assessment of existing as well as potential future conditions under alternative VWRF discharge levels. Uncertainties discussed in this report include limitations in modeling tools and factors affecting suitability, quantitative and qualitative assessments of beneficial uses using the Analytic Hierarchy Process, as well as the assessment of future conditions. While we have made every effort to incorporate the most important factors into the quantitative assessments in Section 5.3 through 5.5, the conceptual models and modeling tools developed in this report each have limitations and associated uncertainty. The models were parameterized based on site-specific observations, extensive data collection, and literature review. However, as with any model, there is some uncertainty resulting from predictions of the future based on past observations; this is particularly true in a geomorphically and hydrologically dynamic system like the SCRE. These challenges are magnified by the ambitious range of conditions evaluated by this study (i.e., habitat conditions for multiple species, conditions for a variety of human uses, etc.). The following sections identify areas of uncertainty in the analysis and provide guidance on interpretation of the results in light of these limitations. These uncertainties, combined with the potential for each discharge scenario to adversely affect certain beneficial uses or their contributing factors while benefitting others, indicate the need for adaptive management measures to accompany implementation of diversions consistent with the MEPDV.

5.6.3.1 Limitations of modeling tools

Because of complex interactions of abiotic, biotic, and anthropogenic processes affecting the beneficial uses of the SCRE, the ability to accurately and precisely predict future conditions under hypothetical future VWRF discharge scenarios is limited by a number of factors, including modeling constraints and data limitations. While we have made every to incorporate the most important factors into the quantitative assessments, conceptual models and modeling tools developed in this report, each have limitations and associated uncertainty. The water balance model, modeling of habitat evolution, and predictions of water quality conditions considered the influence of current and alternative VWRF discharge scenarios that may be adopted as part of ongoing NPDES permit evaluations as well as those that may be diverted (i.e., the MEPDV and 100% diversion/0 discharge scenarios) to water reclamation uses as part of the VenturaWaterPure Project. In addition to the uncertainties presented in Section 4.2.2, there are several uncertainties in the presentation of future habitat conditions in the SCRE based upon necessary model assumptions or limitations in available data. First among these is related to uncertainties in the water balance (Section 4.2.2). Sensitivity testing showed that the calibrated water balance is sensitive to assumptions in berm position and geometry which affect subsurface flows. Though the SCRE is morphologically dynamic system, water balance modeling assumed static lagoon and berm morphology for the purposes of comparing VWRF discharge scenarios. For example, short-term changes in berm seepage due to dredge deposits as well as longer-term changes in berm length and height are not readily represented, which may cause disparities between future observations and model predictions. Additionally, the water balance model assumes a constant flow path length through the berm for all scenarios, so SCRE seepage losses through the berm may be overestimated at lower VWRF discharge scenarios when the SCRE water level decreases potentially resulting in an increase in the flow path length through the berm. The dynamic nature of the SCRE morphology with flood events also has strong influences upon SCRE mouth position and berm height (Section 3.2.1), which can lead to large uncertainties in future estimates of berm outflow as well as equilibrium water levels in the SCRE (Section 4.2.2). The effect of sustained flow reduction on factors such as shallow groundwater levels and shifts in the vegetation of the SCRE may also affect changes in the SCRE morphology in response to future flood events, altering the modeled stage-habitat relationships used for this study.

Furthermore, changes in the morphology may have significant effects on habitat characteristics in the SCRE. Because water quality was found to meet Basin Plan objectives at sites within open water portions of the SCRE (Section 3.4.1), predictions of physical habitat areas for focal fish species, for example, were based upon current lagoon morphology and filtered based on depth and sediment suitability criteria. Morphological changes resulting from sediment deposition, scour, or landward migration of the beach berm are likely to alter stage-depth relationships, and therefore suitable habitat area for focal fish species.

The seasonally dynamic hydrology of the SCRE may impact the short-term distribution of relative habitat types (e.g. wetland, riparian, open water) through flood events and scour. The simplified habitat succession rules used to model future habitat distribution assumes a fixed morphology with qualitative changes in riparian vegetation at distances from the areas near the channel thalweg (Section 5.3.1). However, large flood events can alter the morphology including the thalweg location, and have been shown to scour areas within the entire riparian and lagoon corridor (Appendix C). In addition to changes in berm and lagoon morphology, potential distribution of vegetation and habitat types to these large events cannot be accurately predicted by the vegetation succession model employed for this study. Accordingly, the results of this model should be used to evaluate general trends in vegetation changes with changing water levels averages over the long term solely for purposes of comparing VWRF discharge scenarios, rather than as absolute estimates of habitat areas at a particular point in time under each scenario.

5.6.3.2 Inability to model some factors affecting habitat suitability

Habitat assessment under alternative VWRF discharge scenarios considered factors that (a) were expected to vary by VWRF discharge and (b) could be readily assessed or modeled based on the available data. While every effort was made during the study planning phases to collect the most relevant data for modeling habitat suitability, it is clear that some factors that may be important determinants of habitat suitability are not directly represented in the habitat assessment. For example, the spatial and temporal resolution of temperature data limited the scope of predictive temperature modeling in this study. Relative changes in average water temperatures under alternative VWRF discharge scenarios were modeled using a whole-lagoon heat balance model. Results of this modeling indicated only minor variations in equilibrium temperature (±0.5°C) with changing VWRF discharge, as evaporation and insolation are the primary drivers of lagoon water temperature. While the lagoon is generally well-mixed and apparent differences in surface and bottom Sondes at the south side of the SCRE were not corroborated over multiple water column profile measurements (see Section 2.2.3), there is some potential for the development of local stratification that may afford thermal refugia for fish species. Conceptually, changes in mean depth associated with alternative VWRF discharge scenarios might be expected to result in some changes not captured by the heat balance model, including potential changes in the heat content of the water column affecting the rate of response of lagoon water temperature due to changing air temperature.

In addition to simplified water temperature modeling approaches, predictions of water quality parameters for habitat suitability under current and alternative VWRF discharge scenarios were limited in their scope. For example, the factors contributing to "DO crises" are complex, relating to seasonal and breach related algal bloom dynamics that greatly complicate predictive modeling of actual DO concentrations under alternative VWRF discharge. Physical and chemical water quality in the SCRE has been shown to be highly variable (Section 3.4) and the nutrient balance model is only suitable for estimation of idealized nutrient concentration based upon loadings and average uptake or decomposition (e.g., denitrification) rates (Section 5.3.2.2). DO levels as well as ammonia are affected by not only the VWRF, but also by indirect nutrient pathways resulting from groundwater inputs as well as many point and non-point discharge sources to the Santa Clara River. For example, future ammonia concentrations in the SCRE depend on the complex interactions of many variables other than the low ammonia levels measured in the VWRF Pond discharge (Section 3.4.1.5). Because ammonia may be formed by decomposition of plant proteins, multiple nitrogen sources to the SCRE may potentially contribute to dissolved ammonia levels in the future, including water column algae, benthic algae, floating, submersed and emergent plants, as well as sediment associated plant litter from prior years. In the water column, conversion of ammonium to free ammonia levels related to toxicity is controlled by pH and water temperature, which cannot be feasibly modeled to represent conditions under future VWRF discharge scenarios. Based upon the discussion above, an assumption was adopted that reductions in VWRF discharges will decrease nutrient loading to the SCRE and thereby reduce the frequency and severity of algal blooms. Scoring of scenarios regarding DO as well as ammonia levels relies on a linear inverse relationship between nutrient loads and score. This is admittedly crude, as the relationship between nutrient loading and these metrics is likely to be non-linear and it is also likely that adverse water quality conditions may periodically occur in the future even in the complete absence of VWRF discharges.

Another important factor not directly assessed in the comparison of discharge scenarios is the short-term impact of changing VWRF discharge on fish community interactions. Most of the selected metrics for evaluating habitat conditions relied on estimates of equilibrium conditions under a discharge scenario. However, the effects of changing from current discharge to a different hypothetical discharge is a perturbation that is likely to trigger a community response not captured by the static comparison of hypothetical conditions. For example, reductions in habitat area from current conditions are likely to result in a short-term increase in the density of aquatic species, magnifying density-dependent effects on inter- and intraspecies interactions. Nevertheless, the lack of predictive modeling of the changes in community interactions should be taken into account when interpreting study results and conclusions. Qualitative analysis of interspecies interactions is provided in Section 3.6.4.

Lastly, available data did not allow for predictive modeling of food availability for aquatic species under alternative VWRF discharge scenarios. Instead, food conditions are assumed to correlate with habitat area and suitable water quality conditions for native fish species (note also the limits of predictions of water quality, described below). Though existing BMI data provided some insight into the composition of benthic food resources in the SCRE, the factors controlling seasonal, annual, and spatial variability of BMI resources are not readily decipherable from available data (Section 3.6.1). Additionally, data collection primarily focused on benthic infauna, while epibenthic and pelagic resources are arguably more important for focal fish species in the SCRE. The ability to predict food resource availability is further hampered by the lack of site-specific diet or resource use data. Characterization of native fish species' diets (Table 3-32) were based on observations of species in other systems. However, systems with different community compositions have qualitatively different food web structures; variation in a fish species diet across systems might be expected as a result of varying pressures from competition or predation (i.e. resource partitioning). The unique fish assemblage in the SCRE likely results in partitioning of food resources that is difficult to predict based on observations from other systems. Given the challenges of characterizing even current food resource conditions for the variety of target fish species in the SCRE, it is infeasible to directly assess food resources under hypothetical future conditions. Qualitative discussion of food resources is provided in Section 3.6.4.2.

5.6.3.3 Interpretation of results from the Analytic Hierarchy Process

The AHP implemented for the Phase 3 study is one example of a multi-criteria decision-making tool for ranking and choosing amongst a range of alternatives. In this application, the AHP is used as a means of weighing the potentially competing demands of multiple beneficial uses in order to make discharge recommendations for the VWRF. It is also used as a means of incorporating input from the Resources Agencies, TRT experts and other stakeholders into the process of identifying, prioritizing and determining the relative weight of various factors, attributes and beneficial uses potentially affected by differences in discharge scenarios, as well as the metrics used to assess those effects.

While there is uncertainty associated with multi-criteria decision-making tools like the AHP and their outcomes, the use of the AHP to solve complex problems that are marked by conflicting interests is still valid (Mardani et al. 2015). Despite some criticisms of AHP and other such methods regarding alternative rank reversals when analyses are repeated with the substitution of new alternatives (Dyer 1990, Weber 1997), the method is mathematically as well as conceptually simple and reduces cognitive errors by decomposing a decision problem into its constituent parts and end users generally understand the pairwise comparisons used to derive the factor weighting. Numerous modifications to the AHP have been studied to assist in evaluating the uncertainty associated with it (Reynolds 2001, Mendoza and Prabhu 2001, Eskandari and Rabelo 2007, Sadiq and Tesfamariam 2009, Karimi et al. 2011), but uncertainty associated with the outcome cannot be eliminated. Sensitivity testing of the pairwise comparisons provides one method to transparently evaluate these inherent uncertainties of the AHP and the resulting priority scores and to provide insight into the robustness of priority scores.

Principal among assumptions of the AHP is that the influences of selected factors are additive and for this reason comparison of negatively and positively scoring factors have the potential to cancel one another out. Factors included in the beneficial use assessment (Section 5.5, Appendix H) were reduced and simplified to those that can be reliably modeled or qualitatively related to VWRF discharges. For example, a highly weighted factor supporting the estuarine beneficial use (EST) is physical habitat area for estuarine fish. The metric used to represent physical habitat area was based on open water habitat area in the SCRE. While open water habitat is certainly the foremost component, it is not the only factor affecting physical habitat for estuarine fish. Similarly, as discussed above, assessment of dissolved oxygen conditions was conducted using simplified metrics based on VWRF nutrient loading because actual dissolved oxygen conditions could not be readily modeled.

While every effort was made to consider all TRT and Resources Agency input provided and to assign representative and reasonable scoring thresholds to the metrics used to assess VWRF discharge scenarios in Section 5.5.1 through 5.5.11, the selected thresholds are, in some cases, somewhat arbitrary. To reduce the sensitivity of scoring outcomes to uncertain threshold criteria that could not be well assigned from data or literature sources, scoring was assigned on a continuous basis rather than discrete assignments related to exceedance of a particular threshold criterion. For example, continuous scores were assigned based on incremental reductions in VWRF nutrient loading as a representation of DO conditions because we have only a qualitative understanding of how DO conditions will change with reduced nutrient loads. Similarly, it is not clear from available data how reductions in habitat area will affect overall habitat conditions for the species considered, as the data are insufficient for identifying factors limiting individual life stage progression or overall population size supportable in the estuary. As such, VWRF discharge scenarios were scored continuously based on the incremental reductions in habitat area they supported relative to the maximum modeled amount.

While the AHP was used to provide a transparent and repeatable framework for comparing and discharge alternative upon a wide range of factors considered in this study, recognizing the potential that discharge recommendations may be more subjective, a sensitivity evaluation was conducted to compare individual weighting and scenario rankings. The results of the AHP analysis and the ranking of alternative scenarios are dependent on the pairwise judgments. Therefore, it might be expected that the outcome of these judgments and subsequent analysis might vary depending on who made the comparisons. In order to evaluate the variability in priority scores and ranking of VWRF discharge scenarios associated with alternative pairwise judgements within the AHP framework, we conducted a sensitivity test in which Resource Agency personnel and TRT experts with knowledge of this study and the SCRE were asked to perform AHP pairwise judgements (see Appendix H for more detail). We received four completed worksheets with pairwise judgements, which were used to calculate weights, and ultimately priority scores for each of the VWRF discharge scenarios. Results of this analysis show high variability in priority scores for the lowest and highest discharge reduction scenarios

(Scenarios 1–2 and 9–11, respectively), but low variability for intermediate discharge scenarios (Scenarios 4–8) (Figure 5-29). Additionally, the results for the composite of the five weightings closely matches the Stillwater results for both priority scores (Figure 5-29) and the ranking of discharge scenarios (Table 5-25). This indicates that our recommendation of 40–60% reductions in VWRF existing discharge is robust when considering alternative perspectives in weighting.

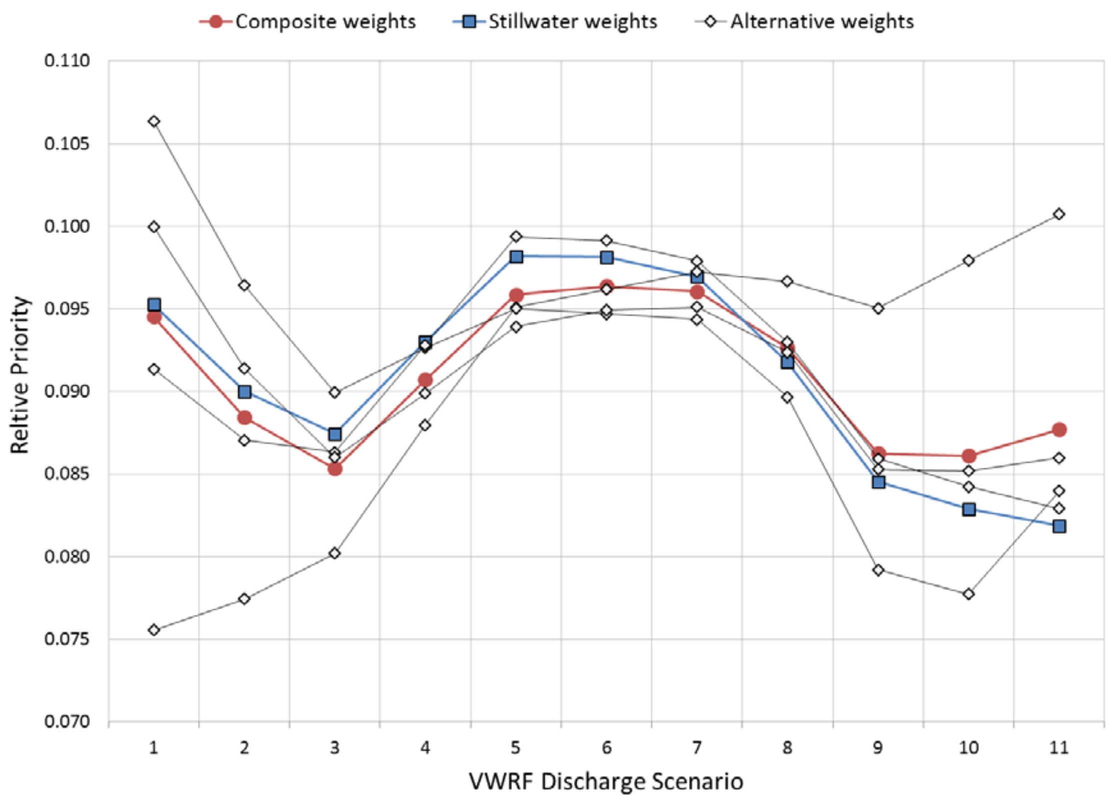


Figure 5-29. Sensitivity of relative priority results to alternative weighting, with "Stillwater weights" representing the weights for results presented above and "Composite weights" representing the average of the four alternatives and the Stillwater weights.

Table 5-25. Sensitivity of the ranking of VWRF discharge scenarios to alternative weightings.

Weighting	Rank of VWRF Discharge Scenario										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
Stillwater	4	7	8	5	1	2	3	6	9	10	11
ALT-1	11	10	9	8	6	5	3	4	7	2	1
ALT-2	1	2	7	6	3	4	5	8	10	11	9
ALT-3	6	7	8	5	1	2	3	4	9	10	11
ALT-4	1	6	8	7	4	3	2	5	10	11	9
Composite	4	7	11	6	3	1	2	5	9	10	8

While the AHP was used to give rigor to the decision making process, the caveats and uncertainties discussed above dictate caution when interpreting final weighted scores for each

VWRF discharge scenario (Figure 5-23 and Figure 5-24). These values, which essentially represent the extent to which each discharge scenario supports the balance of beneficial uses in the SCRE relatable to VWRF discharge, should not be used as an absolute prediction of future conditions in the SCRE, but rather as a ranking of discharge scenarios relative to one another. As such, management decisions should not be based on small differences in scoring values across scenarios, but rather on general trends and large differences across groups of scenarios. The sensitivity testing described above showed that the flow recommendation from the AHP for intermediate discharge scenarios is robust even with alternative weightings. Nonetheless, the assumptions and limitations of the quantitative analysis presented in the previous sections should be taken into account when interpreting the results. Quantitative results should be considered in the context of the qualitative analysis presented in Section 3. The AHP is intended as a tool to aid in decision making, rather than as the final decision-maker itself. The recommendations of this study should be viewed holistically, and best professional judgement should ultimately remain central to decision-making process. Lastly, it should be noted that the AHP is intended as a tool to aid in decision making, rather than as the final decision-maker itself. The recommendations of this study should be viewed holistically, and best professional judgement should ultimately remain central to decision-making process.

5.6.4 Assessment of future conditions due to climate change

In making future VWRF discharge recommendations in the sections above, it is important to note that the VWRF discharges as well as other future flow contributions to the SCRE will vary at seasonal and interannual time scales based on future variations in hydrologic conditions, geomorphic change, as well as human population changes. Other than potential changes in VWRF discharge as well as simplified assumptions regarding evolution of vegetation types in the SCRE in the future, no other future changes in lagoon morphology or habitat conditions were considered as part of this study. Although historical variability in SCRE morphology and vegetation was considered for this study, several future changes in the SCRE habitat conditions due to global climate change effects are discussed below.

Based on climate change assessments for the SCRE vicinity (Carollo Engineers 2011) as well as more recent assessments presented in cbec (2015), sea levels for the California coast are projected to rise by 1.5 to 5.5 ft by year 2100 (relative to year 2000) and cause changes to the coastline, including vertical and landward movement of the shoreline (NRC 2012). SCRE berm height is expected to rise at a similar rate to mean sea level with ongoing climate change (Hanslow 2000) which will result in increased SCRE water surface levels during closed-mouth conditions (cbec 2015).

An analysis of historical precipitation patterns documents the average annual precipitation amount at the SCRE was approximately 15 in (as recorded at Ventura's National Weather Station No. 049285) between 1900 and 2010 and upstream of the SCRE (as recorded at the Santa Paula-UWCD Station No. 049245a), the average annual precipitation is approximately 17 inches between 1820 and 2010. Projected total average annual precipitation changes have varied widely across models and emissions scenarios (Kiparsky and Gleick 2003, Madsen and Figdor 2007) with model accuracy decreasing for finer geographic resolutions (e.g., regional or metropolitan level). 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). 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. By the year 2100, northern California is projected to experience an increase in both low and high intensity events (Dettinger 2005). It is projected that extreme precipitation pattern events for the SCRE are likely to increase by a factor of 3 by year 2100 (Carollo Engineers 2011). The consequences of these predictions may include increased frequency of open-mouth conditions due to berm breaching caused by extreme precipitation events. In addition, because of channel confinement by levees along the lower SCR (Stillwater Sciences 2007b), particularly upstream of Harbor Blvd., the estuary will be both deeper and smaller than under current conditions. Depending upon the pace of these changes, VWRF flow modifications may be considered in the future to maintain protection of the ecological functions of the SCRE, particularly those that support of native species.

5.6.5 Recommendations for adaptive management

Considering the potential for discharge scenarios to both benefit and adversely affect competing beneficial uses, and the factors and attributes that contribute to them, the uncertainties associated with model predictions and AHP analysis described in Section 5.6.3, as well as potential future changes in the physical and hydrologic conditions in the SCRE due to climate change or anthropogenic alteration (Section 5.6.4), it is recommended that an adaptive management plan be implemented concurrent with changes to VWRF discharge to the SCRE based on the MEPDV and Continued Discharge Level is recommended.

An adaptive management plan would be intended to ensure that anticipated benefits of the recommended diversions/continued discharge for native species are realized, and any unanticipated detrimental effects be mitigated. As examples of potentially adverse outcomes, adaptive management could be used in the future to address higher than expected water levels at the equilibrium “full” stage in the SCRE that could contribute to flooding of the campground in the absence of its protective berm, as well as occurrences of man-made unseasonal breaches. Similarly, effects of decreased aquatic habitat amounts associated with the recommended MEPDV may result in or magnify density dependent effects on intra- and interspecies interactions (Section 5.6.3.3), which cannot be readily predicted from available data. While some improvements in water quality conditions related to lower variability in ambient DO conditions (Section 5.3.2.3) as well as greater variability in ambient salinity conditions (Section 5.4.2) are expected, the effects of these changes on species outcomes are less certain, given the wide-ranging water quality tolerances for both native and non-native species in the SCRE. Accordingly, it is recommended that adaptive management actions be based on data and analysis, and that any adaptive management plan consider system constraints.

It is recommended that the following guidelines, considerations and constraints be taken into account in the development and implementation of an adaptive management program:

- **Future Monitoring** – In addition to routine NPDES monitoring conducted by the City, it is expected that future adaptive management be informed by monitoring of water quality conditions as well as periodic assessment of conditions affecting aquatic and avian species with metrics and monitoring frequencies determined during permitting. While there is some value in implementation monitoring to determine whether expected conditions are achieved, some caution should be exercised in applying model results as performance standards, given the dynamic nature of the SCRE.

- **Response Actions** – Adaptive management actions should rely on a science based process, including testable hypotheses regarding the timeline and detectability of future system responses as well as seperability of contributing factors. As detailed in this Report, the SCRE is naturally a highly variable system on seasonal and annual timescales and because of this variability, changes in environmental and ecological conditions concurrent with changes in VWRF discharges may not necessarily be related. In addition, adaptive management actions that do not involve changes in VWRF discharge should be considered. For example, many management responses related to invasive plant and invasive fish species removal may be undertaken without adjustments in discharge. Lastly, the potential impact/benefit and the uncertainty of achieving the desired system response, should be carefully considered before making adaptive management decisions involving VWRF discharge flow. Any significant adaptive management actions should be based on evaluation from qualified individuals and should involve approval from regulatory and resources agencies.
- **Consideration of Regional Benefits and Construction and Operational Constraints -** Because adoption of the VWRF flow requirements from the NPDES permitting process will be accompanied by large infrastructure investments to provide for diversion of the MEPDV to water reclamation uses, including potable reuse, it is anticipated that future flow recommendations consider existing and proposed water supply commitments before making changes in the VWRF flow or water quality discharge requirements or flow- or water quality-related adaptive management measures. It is recommended that discharge and adaptive management requirements consider reasonable expectations of operability of the VWRF and VenturaWaterPure municipal reuse project, and afford flexibility and sufficient time schedules for funding, planning, design, environmental review, permitting etc. as applicable to both flow-related responses and non-flow management measures that may be considered in the future.

6 CONCLUSIONS

Pursuant to the December 30, 2016 update of the VWRF Combined Workplan for the Phase 3 Estuary, Nutrient, Toxicity and Groundwater Special Studies (Workplan), this Phase 3 Estuary Study incorporates quantitative information from historical and present day monitoring of discharge and estuary stage, groundwater monitoring data, mapping of habitat types, aquatic and terrestrial species monitoring data, as well as historical and present day compilations of water quality data. As required in the December 12, 2014 letter from the Regional Board, the report summarizes the results of Phase 1 and 2 studies and reviews the data gaps analysis used to define the scope of the Phase 3 studies. The requirements of Order R4-2013-0174, § VI.c.2.b.i are specifically addressed in Sections 3 through 5 through the development of a water balance as well as evaluation of water quality, and habitat suitability in the SCRE.

Our approach covered the following key areas to compare current and alternative discharge scenarios to qualitatively and quantitatively assess the enhancement of designated beneficial uses associated with VWRF discharges to the SCRE:

- Directly assessed individual beneficial uses.
- Utilized existing information sources and model predictions to compare current and alternative discharge scenarios.

- Focused primarily on equilibrium dry-weather, closed-mouth conditions when VWRF discharges exert a greater influence on habitat conditions within the SCRE.
- Incorporated assessment of potential changes in mouth breaching dynamics (during both dry and wet periods) as well as duration of higher salinity conditions following mouth berm closure.
- Used a collaborative and iterative Analytic Hierarchy Process (AHP) to assign weights to the various factors under consideration.

As required by the EBE Policy, the NPDES Permit and the Consent Decree, determining whether alternative VWRF discharge scenarios enhance beneficial uses of the SCRE and recommending Enhancement Discharge Levels, an MEPDV, and a Continued Discharge Level were accomplished using a combination of water balance modeling, nutrient and heat balance models, habitat succession model and through a collaborative Analytic Hierarchy Process (AHP) adapted to the Phase 3 Estuary Study that was used to assign weights to the various beneficial uses, and their factors and attributes under consideration. Based upon these assessments we conclude that:

- As required by the NPDES permit and EBE Policy, we recommend continued VWRF discharges in accordance with Scenarios 5–7 in the amounts ranging from 40–60% of VWRF discharge (1.9 to 2.8 MGD) during closed-mouth, dry-weather conditions provide a fuller realization of the balance of beneficial uses of the SCRE and MUN uses within the SCRE watershed relative to the absence of all VWRF discharge, and therefore provide “enhancement.”
- As required by the Consent Decree and permitted by the NPDES Permit, we recommend that an MEPDV of 60%, corresponding to a diversion of 2.8 MGD under current conditions, will be protective of the ecological functions of the SCRE while providing for improvement of local water supply by diversion of VWRF effluent to water reclamation uses. This recommended diversion of 2.8 MGD is expected to increase if additional wastewater flow becomes available, provided that the Continued Discharge Level of 1.9 MGD (see below) is maintained during closed-mouth, dry-weather conditions.
- As required by the NPDES Permit, we recommend that a Continued Discharge Level 1.9 MGD during closed-mouth, dry-weather conditions would be protective of estuary native species and habitats, particularly those that are listed for protection under the state and federal Endangered Species Acts.

In addressing questions detailed in Order R4-2013-0174, § VI.c.2.b.ii and § VI.c.2.b.iii regarding potential causes of nutrient, dissolved oxygen and toxicity impairments in the SCRE to be addressed as part of the *Nutrient, Dissolved Oxygen and Toxicity Special Study*, it should be noted that although the Phase 3 Study results do not indicate any toxicity impairments, the VWRF discharge has been shown to contribute to elevated nutrient levels in the SCRE. Although nutrient levels of flow sources to the SCRE other than the VWRF are well above saturation thresholds for algal uptake, the VWRF discharge has been shown to contribute to elevated nutrient levels in the SCRE. Because the Consent Decree between Wishtoyo, HTB and the City included provisions for treatment wetlands to reduce nutrient loading to the SCRE for any VWRF discharge that is not able to be diverted for recycled water purposes, it should be expected that the recommended 60% MEPDV and discharge recommendation of 1.9 MGD in conjunction with treatment wetlands or other measures adopted by the City for any continued discharges to the SCRE will result in additional reductions in nutrient loading to the SCRE, thus improving water quality above that associated with VWRF flow reductions alone.

The recommendations in this Report regarding the Enhancement Discharge Levels, the MEPDV, and the Continued Discharge Level will be further considered and peer reviewed by three scientific experts appointed to the "Scientific Review Panel" (SRP). The SRP will prepare a report of their findings regarding these levels, and will recommend an alternative MEPDV if they disagree with the recommendation set forth in this Report. The RWQCB, NMFS, USFWS, and CDFW, as Resources Agencies with jurisdiction to environmentally review, consult with respect to, certify, approve, condition, or otherwise permit continued VWRF discharges to the SCRE and/or modifications to existing VWRF discharges, may then use this Report and the SRP Report to inform their regulatory determinations.

7 REFERENCES

ABC (Aquatic Bioassay and Consulting) Laboratories. 2004. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2003.

ABC Laboratories. 2005. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2004.

ABC Laboratories. 2006. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2005.

ABC Laboratories. 2007. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2006.

ABC Laboratories. 2008. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2007.

ABC Laboratories. 2009. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2008.

ABC Laboratories. 2010. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2009.

ABC Laboratories. 2011. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2010.

ABC Laboratories. 2012. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2011.

ABC Laboratories. 2013. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2012.

ABC Laboratories. 2014. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2013.

ABC Laboratories. 2015. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2014.

ABC Laboratories. 2016. Santa Clara River Estuary, Macroinvertebrate Bioassessment Survey, Annual Report 2015.

Airphoto USA. 2002. 2002 Digital Ortho Mosaic.

Allen, J.R.M. and R.J. Wootton. 1982. The effect of ration and temperature on the growth of the three-spined stickleback, *Gasterosteus aculeatus* L. Journal of Fish Biology, 20(4), pp.409-422.

Allen, L. G., M. Yoklavich, G. M. Calliet, and E. S. Horn. 2006. Bays and estuaries. Pages 119–148 *in* L. G. Allen, D. J. Pondella, and M. H. Horn, editors. The ecology of marine fishes: California and adjacent Waters. University of California Press, Berkeley.

Anderson, T., C. Clark, Z. Croyle, J. Maas-Baldwin, K. Urquhart, and F. Watson. 2007. Carmel Lagoon Water Quality and Steelhead Soundings: Fall 2007.

Ascough, N. J., J. Fawell, G. Franklin and W. Young. 2000. Review of human pharmaceuticals in the environment. R&D Technical Report P390. Environmental Agency.

Baldwin, D. 2015. Effect of salinity on the olfactory toxicity of dissolved copper in juvenile salmon. Contribution #754. Prepared by National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center, Seattle, Washington for the San Francisco Estuary Institute, Regional Monitoring Program. Richmond, California.

Baldwin, B. G., D. H. Goldman, D. J. Keil, R. Patterson, T. J. Rosatti, and D. H. Wilken, editors. 2012. The Jepson manual: vascular plants of California, second edition. University of California Press, Berkeley.

Baldwin, D., J. Sandahl, J. Labenia, and N. Scholz. 2003. Sublethal effects of copper on coho salmon: Impacts on nonoverlapping receptor pathways in the peripheral olfactory nervous system. Environmental Toxicology and Chemistry 22: 2,266–2,274.

Barbour, M. G., T. Keeler-Wolf, and A. A. Schoenherr, editors. 2007. Terrestrial vegetation of California. Third edition. University of California Press, Berkeley.

Barletta, M., A. Barletta-Bergan, U.S.G.H. Saint-Paul, and G. Hubold. 2005. The role of salinity in structuring the fish assemblages in a tropical estuary. Journal of fish biology, 66(1), pp.45-72.

Barnard, P. L., Revell, D. L., Hoover, D., Warrick, J., Brocatus, J., Draut, A.E., Dartnell, P., Elias, E., Mustain, N., Hart, P. E., and Ryan, H. F. 2009. Coastal processes study of Santa Barbara and Ventura counties, California. U.S. Geological Survey Open-File Report 2009-1029. http://pubs.usgs.gov/of/2009/1029/.

Barron, J.J., C. Ashton, and L. Geary. 2006. The effects of temperature on pH measurement. 57th Annual Meeting of the International Society of Electrochemistry, Edinburgh, Scotland.

Baxter, R., K. Hieb., S. DeLeon, K. Fleming, and J. Orsi. 1999. Report on the 1980-1995 fish, shrimp, and crab sampling in the San Francisco Estuary, California.

Begnudelli, L. and Sanders, B. F. 2007. Simulation of the St. Francis Dam-Break Flood. ASCE Journal of Engineering Mechanics 133: 1,200–1,212.

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society, Bethesda, Maryland.

Behrens, D.K., F.A. Bombardelli, J.L. Largier and E. Twohy. 2013. Episodic closure of the tidal inlet at the mouth of the Russian River – a small bar-built estuary in California. Geomorphology, http://dx.doi.org/10.1016/j.geomorph.2013.01.017

Bell, M. C., editor. 1991. Fisheries handbook of engineering requirements and biological criteria. Fish Passage Development and Evaluation Program, U. S. Army Corps of Engineers, North Pacific Division, Portland, Oregon.

Beller, E.E., R.M. Grossinger, M.N. Salomon, S.J. Dark, E.D. Stein, B.K. Orr, P.W. Downs, T.R. Longcore, G.C. Coffman, A.A. Whipple, R.A. Askevold, B Stanford, and J.R. Beagle. 2011. Historical ecology of the lower Santa Clara River, Ventura River, and Oxnard Plain: an analysis of terrestrial, riverine, and coastal habitats. Prepared for the State Coastal Conservancy. A report of SFEI's Historical Ecology Program, SFEI Publication #641, San Francisco Estuary Institute, Oakland, CA.

Bjornn, J. R., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. In W. R. Meehan (ed.), Influence of forest and rangeland management on salmonid fishes and habitats, p. 83–138. Special Publ. 19. American Fisheries Society, Bethesda, MD.

Boelee, N.C., H. Temmink, M. Janssen, C. Buisman, and R. Wijffels. 2011. Nitrogen and phosphorus removal from municipal wastewater effluent using microalgal biofilms. Water Res. 45, 5923-5933

Bond, M. H. 2006. Importance of estuarine rearing to central California steelhead (*Oncorhynchus mykiss*) growth and marine survival. Master of Arts thesis, University of California Santa Cruz.

Boughton, D.A., L.R. Harrison, A.S. Pike, J.L. Arriaza, and M. Mangel. 2015. Thermal Potential for Steelhead Life History Expression in a Southern California Alluvial River. Transactions of the American Fisheries Society, 144:2, 258-273.

Boynton, W.R., and W. M. Kemp. 2008. Nitrogen in estuaries. pp. 809-866, *in:* D.G. Capone, D.A.Bronk, M.R. Mulholland, E.J. Carpenter (eds.) Nitrogen in the marine environment. Academic Press, New York

Boynton, W. R., W. M. Kemp, and C. W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. Pages 69–90 *in* V. S. Kennedy, editor. Estuarine comparisons. Academic Press, New York.

Bricker, S. B., C. G. Clement, D. E. Pirhalla, S. P. Orlando, D. R. G. Farrow. 1999. National estuarine eutrophication assessment. Effects of nutrient enrichment in the nation's estuaries. NOAA—NOS Special Projects Office.

Bruner de Miranda, L., F. P. Andutta, B. Kjerfve, Castro Filho, B. M. de. 2017. Fundamentals of Estuarine Physical Oceanography. Springer, Singapore.

Brungs, W.A.. 1971. Chronic effects of low dissolved oxygen concentrations on the fathead minnow (*Pimephales promelas*). Journal of the Fisheries Board of Canada, 28(8), pp.1119-1123.

Burgner, R. L., J. T. Light, L. Margolis, T. Okazaki, A. Tautz, and S. Ito. 1992. Distribution and origins of steelhead trout (*Oncorhynchus mykiss*) in offshore waters of the North Pacific Ocean. International North Pacific Fisheries Commission Bulletin 51: 92 p.

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, Oregon, and California. National Oceanographic and Atmospheric Administration Technical Memorandum NMFS-NWFSC-27. National Marine Fisheries Service, Seattle, Washington.

Byers, H. R., H. Moses, and P. J. Harney. 1949. Measurement of rain temperature. Journal of Meteorology 6, 51-55.

California State Parks. 2013. Western Snowy Plover Annual Report 2013. Channel Coast District Western Snowy Plover Program.

California State Parks 2014. Western Snowy Plover Annual Report 2014. Channel Coast District Western Snowy Plover Program.

California State Parks 2015. Western Snowy Plover Annual Report 2015. Channel Coast District Western Snowy Plover Program.

California State Parks. 2017. Telephone conversation between Alexis Frangis, Environmental Scientist, California State Parks, Channel Coast District and Noah Hume, Stillwater Sciences.

Capelli, M. H. 1997. Tidewater goby (*Eucyclogobius newberryi*) management in California estuaries. Pages 1–18 *in* Proceedings of the California and world ocean conference.

Cardno/Entrix. 2010. Survey for tidewater goby, *Eucyclogobius newberryi*, in Santa Clara River Estuary. Prepared by Cardno/Entrix, Santa Barbara, California for City of Ventura, California.

Caro, T.M. and G. O'Doherty. 1999. On the use of surrogate species in conservation biology. Conservation Biology 13(4): 805-814.

Carollo Engineers 2010. City of Ventura treatment wetlands feasibility report. Final. Prepared by Carollo Engineers, Walnut Creek, California with assistance from Stillwater Sciences, Berkeley, California.

Carollo Engineers. 2011. Estimate of future conditions due to climate change. Final Report. Prepared by Carollo Engineers, Walnut Creek, California for City of Ventura, California.

Carollo Engineers. 2014. Amended Estuary Special Studies Phase 2: Facilities Planning Study for Expanding Recycled Water Delivery. Prepared by Carollo Engineers, Walnut Creek, California for City of Ventura, California. September.

Carollo Engineers. 2017. City of Ventura Diversion Infrastructure Projects Study. Final Draft. Prepared by Carollo Engineers, Walnut Creek, California for City of Ventura, California.

Castleberry, D.T. and J.J. Cech. 1986. Physiological responses of a native and an introduced desert fish to environmental stressors. Ecology, 67(4), pp.912-918.

Cavallo, B., R. Kurth, J. Kindopp, A. Seesholtz and M. Perrone. 2003. Distribution and habitat use of steelhead and other fishes in the lower Feather River, 1999–2001. Interim Report SP-F10, Task 3a. Prepared for California Department of Water Resources, Division of Environmental Services, Sacramento, California.

Cayan, D. R., P. D. Bromirski, K. Hayhoe, M. Tyree, M. D. Dettinger, and R. E. Flick. 2008. Climate change projections of sea level extremes along the California coast. Climatic Change 87: S57–S73.

cbec, Inc., WRA, Inc., and M. Podlech. 2015. Santa Clara River Estuary Habitat Restoration and Enhancement Feasibility Study. Existing Conditions Technical Report. Prepared for Wishtoyo Foundation and Wishtoyo Foundation's Ventura Coastkeeper Program.

CDFG (California Department of Fish and Game). 1988. A Guide to Wildlife Habitats of California. Edited by Mayer, K.E. and W.F. Laudenslayer, Jr. State of California, Resources Agency, Department of Fish and Game. Sacramento, CA. 166 pp.

CDFG. 2010. List of vegetation alliances and associations. Vegetation Classification and Mapping Program, California Department of Fish and Game, Sacramento, California. September 2010.

CDFG. 2011. California least tern breeding survey, 2010 season. California Department of Fish and Game, Nongame Wildlife Program, 2011-06. Sacramento, California.

CDFG. 2012. California least tern breeding survey, 2011 season. California Department of Fish and Game, Nongame Wildlife Program, 2012-1. Sacramento, California.

CDFW (California Department of Fish and Wildlife). 2013. California least tern breeding survey, 2012 season. California Department of Fish and Game, Nongame Wildlife Program, 2013-01. Sacramento, California.

CDFW. 2014. California least tern breeding survey, 2013 season. California Department of Fish and Game, Nongame Wildlife Program, 2014-01. Sacramento, California.

CDFW. 2015. California least tern breeding survey, 2014 season. California Department of Fish and Game, Nongame Wildlife Program, 2015-01. Sacramento, California.

CDFW. 2016. Special vascular plants, bryophytes, and lichens list. Quarterly publication. Prepared by CDFW, Sacramento, California.

California Department of Parks and Recreation. 1979. Santa Barbara/Ventura Coastal State Park System General Plan, McGrath.

CDWR (California Department of Water Resources). 2006a. Santa Clara River Valley Basin, Mound Subbasin. California's Groundwater Bulletin 118. California Department of Water

Resources, Sacramento, California. Available online at: http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/4-4.03.pdf.

CDWR. 2006b. California's groundwater: Bulletin 118: Santa Clara River Valley Basin, Oxnard Valley Subbasin. California Department of Water Resources, Sacramento, California. Available at: http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/4-4.02.pdf

CDWR and USBR (U.S. Bureau of Reclamation). 2000. Effects of the Central Valley Project and State Water Project on Steelhead and Spring-Run Chinook Salmon. California Department of Water Resources; U.S. Bureau of Reclamation.

CNPS (California Native Plant Society), Rare Plant Program. 2017. Inventory of Rare and Endangered Plants (online edition, version 8-02). California Native Plant Society, Sacramento, California. [Accessed March 2017].

Collins, M.R. 1985. Species Profiles. Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (South Florida). Striped Mullet. Florida Cooperative Fish and Wildlife Research Unit Gainesville.

Chamberlain, C. D. 2006. Environmental variables of northern California lagoons and estuaries and the distribution of tidewater goby (*Eucyclogobius newberryi*). Arcata Fisheries Technical Report TR 2006-04. U.S. Fish and Wildlife Service, Arcata, California.

Chapman, G.A.; and D. Stevens. 1978: Acutely lethal levels of cadmium copper and zinc to adult male coho salmon and steelhead. Transactions of the American Fisheries Society 107(6): 837-840

Chapman, P. M. and F. Wang. 2001. Assessing sediment contamination in estuaries. Environmental Toxicology and Chemistry. 20: 3–22.

Chase D. A., E. Flynn, and A. Todgham. 2016. Survival, growth and stress response of juvenile tidewater goby, *Eucyclogobius newberryi*, to interspecific competition for food. Conservation Physiology 4(1).

City of Oxnard. 1982. Oxnard Coastal Plan. Planning and Environmental Services. February.

Clemento A., Anderson E.C., Boughton D., Girman D., Garza J.C. 2009. Population genetic structure and ancestry of *Oncorhynchus mykiss* populations above and below dams in south-central California. Conserv Genet 10: 1,321–1,336.

Coffman, G. C., R. F. Ambrose, and P. W. Rundel. 2010. Wildfire promotes dominance of invasive giant reed (*Arundo donax*) in riparian ecosystems. Biological Invasions 12: 2,723–2,734.

Cohen, S.E. and S.M. Bollens. 2008. Diet and growth of non-native Mississippi silversides and yellowfin gobies in restored and natural wetlands in the San Francisco Estuary. Marine Ecology Progress Series, 368, pp.241-254.

Cordone, A. J., and D. W. Kelley. 1961. The influences of inorganic sediment on the aquatic life of streams. Calif. Fish and Game 47: 189–228.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. Biological Services Program Report FWS/OBS-79/31. Prepared for U. S. Fish and Wildlife Service, Office of Biological Services, Washington, D. C. http://www.fws.gov/nwi/Pubs_Reports/Class_Manual/class_titlepg.htm.

Cox, R.A., F. Culkin, and J. Riley. 1967. The electrical conductivity/salinity/temperature relationship in natural water. Deep-Sea Res. 14: 203-220.

Daniels, M., D. Frank, R. Holloway, B. Kowalski, P. Krone-Davis, S. Quan, E. Stanfield, A. Young, A., and F. Watson. 2010. Evaluating good water quality habitat for steelhead in Carmel Lagoon: fall 2009. Publication No. WI-2010-03. The Watershed Institute, California State Monterey Bay.

D. Detmer. 2017. E-mail transmittal from Dan Detmer, United Water Conservation District to Noah Hume, Stillwater Sciences. December 4, 2017

Dettinger, M. D. 2005. From climate-change spaghetti to climate-change distributions for 21st century California. San Francisco Estuary and Watershed Science 3: Article 4.

DiTomass, J.M. and E.A. Healy. 2003. Aquatic and Riparian Weeds of the West. University of California Agriculture and Natural Resources Publication 3421: Oakland, CA. 442 p.

DiTomoso, J. M. and E. Healy. 2007. Weed of California and other western states. University of California Press, Berkeley, California.

Donohoe, C. J., P. B. Adams, and C. F. Royer. 2008. Influence of water chemistry and migratory distance on ability to distinguish progeny of sympatric resident and anadromous rainbow trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences 65: 1,060–1,075.

Downs, P. W., S. R. Dusterhoff, and W. A. Sears. 2013. Reach-scale channel sensitivity to multiple human activities and natural events: lower Santa Clara River, California, USA. Geomorphology 189: 121–134.

Dunne, T., and L. B. Leopold. 1978. Water in environmental planning. W. H. Freeman and Company, San Francisco.

Dyer, J. S. 1990. Remarks on the Analytic Hierarchy Process, Management Science, 36, 3, pp. 249-258.

Emmett, R. L. 1997. Estuarine survival of salmonids: the importance of interspecific and intraspecific predation and competition. Pages 147–158 in R. L. Emmett and M. H. Schiewe, eds. Estuarine and ocean survival of northeastern Pacific salmon. NOAA Tech. Memo. NMFS-NWFSC-29, Seattle, WA.

Entrix, Inc. 2002a. Metals translator study, Santa Clara River Estuary Ventura Water Reclamation Facility. NPDES Permit No. CA0053651, CI-1822. Prepared by Entrix, Inc. Ventura, California for City of San Buenaventura, Ventura, California.

Entrix, Inc. 2002b. Resident species study. Santa Clara River Estuary. Ventura Water Reclamation Facility NPDES permit number CA 0053561, CI-1822. Prepared by Entrix, Inc. Ventura, California for the City of San Buenaventura, California.

Entrix, Inc. 2004. VWRF Discharge Beneficial Uses on the Distribution and Utilization of SCRE Tidewater Goby. September

Entrix Inc. 2007a. Resident Species Study Santa Clara River Estuary Ventura Reclamation Facility. September.

Entrix Inc. 2007b Metals Translator Study Santa Clara River Estuary Ventura Reclamation Facility.

Entrix, Inc. 2008. Results of Survey for Tidewater Gobies, *Eucyclogobius newberryi*, in the Santa Clara River Lagoon with Aquatic Bioassay, Inc. on October 14, 2008 for City of Ventura NPDES Assessment. November 21.

Entrix, Inc. 2010a. April bird survey and Santa Clara River Estuary and wildlife treatment ponds. Technical Memorandum from Carly Spahr to Dan Pfeifer, May 7, 2010.

Entrix, Inc. 2010b. June bird survey and Santa Clara River Estuary and wildlife treatment ponds. Technical Memorandum from Carly Spahr to Dan Pfeifer, June 27, 2010.

ESA (Environmental Science Associates). 2003. McGrath State Beach natural resources management plan. Final. Prepared by ESA, San Francisco, California for California Department of Parks and Recreation, Channel Coast District.

Eskandari, H. and L. Rabelo. 2007. Handling uncertainty in the Analytic Hierarchy Process: a stochastic approach. Int. J. Info. Tech. Dec. Mak. 6(1), pp.177.

EURA (European Union Risk Assessment). 2009. European Union Risk Assessment Report, TRIS (2-Chloroethyl) Phosphate, TCEP, European Union Risk Assessment Report. EUR 2371592 EN.

European Commission[26]. 2003. On the prohibition of organotin compounds on ships. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=%20OJ:L:2003:115:0001:0011:SV:PDF%202008-06-15.

Faust, M. A. and D. L. Correll. 1976. Comparison of bacterial and algal utilization of orthophosphate in an estuarine environment. Marine Biology, 34, pp. 151–162.

Fetter, C. W. 2001. Applied hydrogeology. Prentice-Hall, Inc., New Jersey.

Fetscher, A.E., M. Sutula, A. Sengupta, and N. Detenbeck. 2014. Linking nutrients to alterations in aquatic life in California wadeable streams. EPA/600/R-14/043. U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, Rhode Island.

[26] Citation translate from Swedish and may contain errors. Take as cited in Karlsson 2007

Feldmeth, C.R. and J.N. Baskin. 1976. Thermal and respiratory studies with reference to temperature and oxygen tolerance for the unarmored stickleback *Gasterosteus aculeatus williamsoni* Hubbs. Bulletin of the Southern California Academy of Sciences, 75(2), pp.127-131.

Feldmeth, C.R. and J.P. Waggoner. 1972. Field measurements of tolerance to extreme hypersalinity in the California killifish, *Fundulus parvipinnis*. Copeia, 1972(3), pp.592-594.

Froese, R. and D. Pauly. Editors. 2017. FishBase. World Wide Web electronic publication. www.fishbase.org, (02/2017)

Goldstein, R. M. 1978. Quantitative comparison of seining and underwater observation for stream fishery surveys. The progressive fish-culturist, 40: 108–111.

Graf, W. L. 1988. Fluvial processes in dryland rivers. Springer-Verlag, New York.

Gawlik, D. E. 2002. The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72: 329-346.

Greenwald, G. M., C. L. Snell, G. S. Sanders, and S. D. Pratt. 1999. Santa Clara River Estuary ecological monitoring program, 1997–1999. Prepared by the U. S. Fish and Wildlife Service, Ventura Fish and Wildlife Office for the California Department of Parks and Recreation, Channel Coast District, Santa Barbara, California.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummings. 1991. An ecosystem perspective of riparian zones. Bioscience 41: 540–551.

Grinnell, J. 1917. Field tests of theories concerning distributional control. The American Naturalist 51: 115–128.

Groffman, P. M., D. J. Bain, L. E. Band, K. T. Belt, G. S. Brush, J. M. Grove, R. V. Pouyat, I. C. Yesilonis, and W. C. Zipperer. 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment 1: 315–321.

Grossinger, R. M., E. D. Stein, K. N. Cayce, R. A. Askevold, S. Dark, and A. A. Whipple. 2011. Historical wetlands of the southern California coast: an atlas of U.S. coast survey T-sheets, 1851–1889. San Francisco Estuary Institute Contribution #586 and Southern California Coastal Water Research Project Technical Report #589.

Haglund, T.R., J.N. Baskin, and C.C. Swift. 2003. Results of the year 3 implementation of the Santa Ana sucker conservation program for the Santa Ana River. Unpublished report for Santa Ana Watershed Project Authority.

Hallock, R. J. 1989. Upper Sacramento River steelhead (*Oncorhynchus mykiss*), 1952–1988. Prepared for U.S. Fish and Wildlife Service, Sacramento, California.

Hanslow, D.J., Davis, G.A., You, B.Z, Zastawny, J. 2000. Berm Height at Coastal Lagoon Entrances in SW. 10th NSW Coastal Conference.

Hanson, R. T., P. Martin, and K. M. Koczot. 2003. Simulation of ground-water/surface-water flow in the Santa Clara–Calleguas Ground-water Basin, Ventura County, California. U.S. Geological Survey, Water-Resources Investigations Report 02-4136.

Harrison, R. L., E. A. Keller, E. Kelley, and L. A. K. Mertes. 2006. Minimum flow requirements for southern steelhead passage on the lower Santa Clara River, CA. Prepared for the Nature Conservancy, Arlington, Virginia.

Hart, J.L.. 1973. Pacific fishes of Canada. Bull. Fish. Res. Board Can. 180:740

Hayes, S. A., M. H. Bond, C. V. Hanson, E. V. Freund, J. J. Smith, E. C. Anderson, A. J. Ammann, and R. B. MacFarlane. 2008. Steelhead growth in a small Central California watershed: upstream estuarine rearing patterns. Transactions of the American Fisheries Society 137: 114–128.

Hayes, S. A., M. H. Bond, C. V. Hanson, A.W. Jones, A.J. Ammann, J.A. Harding, A.L. Collins, J. Perez, and R.B. MacFarlane. 2011. Down, up, down and "smolting" twice? Seasonal movement patterns by juvenile steelhead (*Oncorhynchus mykiss*) in a coastal watershed with a bar closing estuary. Canadian Journal of Fisheries and Aquatic Sciences, 68(8):1341-1350.

Healey, M. C. 1982. Juvenile Pacific salmon in estuaries: the life support system. Pages 343–364 *in* V. S. Kennedy, editor. Estuarine Comparisons. Academic Press.

Hecht, S., D. Baldwin, C. Mebane, T. Hawkes, S. Gross, and N. Scholz. 2007. An overview of sensory effects on juvenile salmonids exposed to dissolved copper: Applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-83, 39 p.

Hieb, K.A. 2000. Arrow goby. Goals Project. Baylands ecosystem species and community profiles: life histories and environmental requirements of key plants, fish and wildlife. Oakland, California: San Francisco Bay Area wetlands ecosystem goals project. San Francisco Bay Regional Water Quality Control Board. pp.136-138.

Hoffman, K. and Kunze, R. 1971. Linear Algebra, 2nd ed. Prentice-Hall, Englewood Cliffs, NJ:

Hopkins (Hopkins Groundwater Consultants, Inc.) 2018. Phase 3 Santa Clara River Estuary groundwater special study, Ventura, California. Preliminary hydrogeological study. Prepared for City of San Buenaventura. Ventura, California. February

Hornberger, G. M., J. P. Raffensperger, P. L. Wiberg, and K. N. Eshleman. 1998. Elements of physical hydrology. The Johns Hopkins University Press, Baltimore, Maryland.

Horne, A. J. and C. R. Goldman. 1994. Limnology. McGraw-Hill, New York.

Hubbs, C., H.B. Sharp, and J.F. Schneider. 1971. Developmental rates of Menidia audens with notes on salt tolerance. Transactions of the American Fisheries Society, 100(4): 603–610.

Inman, D. L., and S. A. Jenkins. 1999. Climate change and the episodicity of sediment flux of small California rivers. Journal of Geology 107:251–270.

IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

Irwin, J. F., and D. L. Soltz. 1984. The natural history of the tidewater goby, *Eucyclogobious newberryi*, in the San Antonio and Shuman Creek systems, Santa Barabara county, California. Contract Order No. 11310-0215-2 US Fish and Wildlife Service, Sacramento Endangered Species Office, Sacramento, California.

ISU (Iowa State University). 2008. Managing Iowa fisheries water quality. In cooperation with the U.S. Department of Agriculture. Originally published by J. Morris, Iowa State University Extension aquaculture specialist and updated by R. Clayton.

Japan MOE (Ministry of Environment). 2016. Profiles of the Initial Environmental Risk Assessment of Chemicals. Ministry of the Environment Government of Japan. Downloaded May 1, 2017 at:
http://www.env.go.jp/en/chemi/chemicals/profile_erac/profile9/pf1-04.pdf;
http://www.env.go.jp/en/chemi/chemicals/profile_erac/profile9/pf1-12.pdf;
https://www.env.go.jp/en/chemi/chemicals/profile_erac/profile4/pf1-10.pdf.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report. Prepared by California Academy of Sciences, Department of Herpetology, San Francisco and Portland State University, Department of Biology, Portland, Oregon for California Department of Fish and Game, Inland Fisheries Division, Rancho Cordova.

Jowett, I. G. 1997: Instream flow methods: a comparison of approaches. Regulated rivers 13: 115–127.

Kamman Hydrology. 2007. Santa Clara River Estuary Water Budget and Salinity Assessment. Technical Memorandum prepared by Kamman Hydrology & Engineering, Inc. San Rafael, California, for City of San Buenaventura, California.

Karimi, A. R., N. Mehrdadi, S. J. Hashemian, G. R. Nabi Bidhendi, R. Tavakkoli Moghaddam. 2011. Selection of wastewater treatment process based on the analytical hierarchy process and fuzzy analytical hierarchy process methods. Int. J. Environmental Science and Technology. 8:267-280.

Karlsson, C., 2007. Risk assessment of compounds that could impair the aquatic environment. Support for the Swedish Environmental Protection Agency in their work to propose new priority substances within the water Framework Directive, p.55.

Kelley, E. 2004. Information synthesis and priorities regarding steelhead trout (*Oncorhynchus mykiss*) on the Santa Clara River. The Nature Conservancy, Santa Barbara, California.

Kelley, E. 2008. Steelhead smolt survival in the Santa Clara and Santa Ynez River estuaries. Prepared by University of California, Santa Barbara for California Department of Fish and Game.

Kennedy/Jenks Consultants. 2002. McGrath Lake Preliminary Watershed Study: Final Report. Prepared for the McGrath State Beach Area Trustee Council.

Kennedy/Jenks Consultants. 2007. City of San Buenaventura Recycled Water Market Assessment. Technical Memorandum prepared by Kennedy/Jenks Consultants.

Kennish, M. J. 1990. Ecology of estuaries, Volume II biological aspects. CRC Press, Boca Raton, Florida.

Kharin, V. V., and F. W. Zwiers. 2005. Estimating extremes in transient climate change simulations. Journal of Climate 18: 1,156–1,173.

Kharin, V. V., F. W. Zwiers, X. Zhang, and G. C. Hegerl. 2007. Changes in temperature and precipitation extremes in the IPCC ensemble of global coupled model simulations. Journal of Climate 20: 1,419–1,444.

Kiparsky, M. and P. Gleick. 2003. Climate change and California water resources: a survey and summary of the literature. Pacific Institute for Studies in Development, Environment, and Security, Oakland, California.

Kondolf, G. M., and W. V. G. Matthews. 1991. Unmeasured residuals in sediment budgets: A cautionary note. Water Resources Research 27: 2,483–2,486

Küçük, S. 2013. The effects of salinity on growth of goldfish, *Carassius auratus* and crucian carp, *Carassius carassius*. African Journal of Biotechnology, 12(16).

Küpper, C., J. Augustin,A, Kosztolányi,T. Burke,J Flguerola, and T Székely. 2009. Kentish versus snowy plover: phenotypic and genetic analyses of *Charadrius alexandrinus* reveal divergence of Eurasian and American subspecies. The Auk 126: 839–852

LaBaugh, J. W. 1985. Uncertainty in phosphorous retention. Water Resources Reseach 21: 1,684–1,692

Labinger, Z., J. Greaves, and E. Gevirtz. 2011. Avian populations of the Santa Clara River in 2005 and 2006. An evaluation and monitoring tool for habitat restoration. Prepared for the Santa Clara River Trustee Council, U.S. Fish and Wildlife Service. Ventura, California.

Lafferty, K. D., C. C. Swift, and R. F. Ambrose. 1999. Postflood persistence and recolonization of endangered tidewater goby populations. North American Journal of Fisheries Management 19: 618–622.

Lambeck, R.J. 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology 11: 849-856.

RWQCB (Los Angeles Regional Water Quality Control Board). 2014. The Water Quality Control Plan: Los Angeles Region Basin Plan for the coastal watersheds of Los Angeles and Ventura counties. California Regional Water Quality Control Board, Los Angeles Region (4).

RWQCB (Los Angeles Regional Water Quality Control Board). 2017. 2016 Clean Water Act Sections 305(b) and 303(d) Integrated Report for the Los Angeles Region Staff Report. California Regional Water Quality Control Board, Los Angeles Region (4).

Laudier, N.A., Thornton, E.B., MacMahan, J. 2011. Measured and modeled wave overwash on a natural beach. Coastal Engineering, 58 (9), 815–825.

Lindenmayer, D.B. and J. Fischer. 2003. Sound science or social hook—a response to Brooker's application of the focal species approach. Landscape and urban planning, 62(3), pp.149-158.

Luo, Y., W. Guo, H. Ngo, L. Nghiem, F. Hai. 2014. A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. Science of the Total Environment, 473-474 (March), 619-641.

Madsen, T., and E. Figdor. 2007. When it rains, it pours global warming and the rising frequency of extreme precipitation in the United States. Prepared by Environment California Research & Policy Center, Los Angeles, California.

Magney, D. L. 2017. Checklist of Ventura County Rare Plants. 4 January 2017, 22nd Edition. California Native Plant Society, Channel Islands Chapter, Ojai, California.

Mardani, A., A. Jusoh, K.M.D. Nor, Z. Khalifah, N. Zakwan, and A. Valipour. 2015. Multiple criteria decision-making techniques and their applications - A review of the literature from 2000 to 2014. Ekonomska Istrazivanja, 28(1), 516-571.

Martino, E.J. and K.W. Able. 2003. Fish assemblages across the marine to low salinity transition zone of a temperate estuary. Estuarine, Coastal and Shelf Science, 56(5), pp.969-987.

McCabe, G. T., Jr., W. D. Muir, R. L. Emmett, and J. T. Durkin. 1983. Interrelationships between juvenile salmonids and non-salmonid fish in the Columbia River estuary. California Department of Fish and Game Fish Bulletin 81: 815–826.

McIntyre, J., D. Baldwin, J. Meador, and N. Scholz. 2008. Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions. Environmental Science & Technology 42: 1,352–1,358.

Meehl, G. A., J. M. Arblaster, and C. Tebaldi. 2005. Understanding future patterns of increased precipitation intensity in climate model simulations. Geophysical Research Letter 32: L18719.

Mitsch, W.J. and J.G. Gosselink. 2000. Wetlands, third edition. John Wiley & Sons, New York, NY.

Mendoza, G. and R. Prabhu. 2001. A fuzzy analytic hierarchy process for assess biodiversity conservation. In. Schmoldt, D. L., J. Kangas, G. A. Mendoza, and M. Pesonen (eds). The Analytic Hierarchy Process in Natural Resource and Environmental Decision Making. Springer Science.

Monsen N.E., J.E. Cloern, and LV. Lucas. 2002. A comment on the use of flushing time, residence time, and age as transport time scales. Limnology and Oceanography, 47(5) 1545-1553.

Morgan J.D. and G.K. Iwama. Effects of Salinity on Growth, Metabolism, and Ion Regulation in Juvenile Rainbow and Steelhead Trout (*Oncorhynchus mykiss*) and Fall Chinook Salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences, 1991, 48(11): 2083-2094.

Moriasi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Bingner, R.D. Harmel, and T.L. Veith. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed similations. Transactions of the ASABE, 50(3): 885-900.

More, J. 2008. California State Parks, Channel Coast District Western Snowy Plover Program Annual Report: September 2007–September 2008.

Moyle, P. B. 2002. Inland fishes of California. Revised edition. University of California Press, Berkeley.

Moyle, P. B., and J. E. Williams. 1990. Biodiversity loss in the temperate zone: decline of the native fish fauna of California. Conservation Biology 4: 275–284.

Myrick, C. A., and J. J. Cech. 2000. Swimming performances of four California stream fishes: temperature effects. Environmental Biology of Fishes 58: 289–295.
Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28: 621–658.

NAIP (National Agriculture Imagery Program). 2009. 2009 Raster Digital Data. ortho_1-2_1n_s_ca111_2009_1.sid.

NAIP. 2016. USDA-FSA-APFO Digital Ortho Mosaic. Salt Lake City, Utah.

Nautilus Environmental. 2005. Comprehensive analysis of enhancements and impacts related with discharge of treated effluent from the Ventura Water Reclamation Facility to the Santa Clara River Estuary: Toxicology, Ecology, and Hydrology. Prepared by Nautilus Environmental, Gardena, California, with assistance from Kamman Hydrology & Engineering, Inc., San Rafael, California, for City of San Buenaventura Ventura Water Reclamation Facility, California.

Nautilus Environmental. 2009. Summary report 2008 tidewater goby surveys, Santa Clara River Lagoon, CA. Prepared by Nautilus Environmental, Gardena, California for Ventura Water Reclamation Facility, Ventura, California.

Needham, P. R. 1939. Quantitative and qualitative observations on fish foods in Waddell Creek Lagoon. Transactions of the American Fisheries Society 69: 178–186.

Nelson, J., E. Baglivio, and T. Kahles. 2009. Santa Rosa Creek steelhead habitat and population survey, 2005. Prepared by California Department of Fish and Game and California Conservation Corps.

NMFS (National Marine Fisheries Service). 1996. Endangered and threatened species; proposed endangered status for five ESUs of steelhead and proposed threatened status for five ESUs of steelhead in Washington, Oregon, Idaho, and California. Federal Register 61: 4,1541–4,1561.

NMFS. 1997. Status review of West Coast steelhead from Washington, Oregon and California. NOAA Technical Memorandum NMFS-NWFSC-27. National Marine Fisheries Service. Seattle Washington.

NMFS. 2006. Endangered and threatened species: final listing determinations for 10 Distinct Population Segments of west coast steelhead. Federal Register 71: 834–862.

NMFS. 2009. Southern California Steelhead Recovery Plan. Public Review Draft version July 2009. Southwest Regional Office, Long Beach. California.

NOAA (National Oceanic and Atmospheric Administration). 2005. Endangered and threatened species; designation of critical habitat for seven evolutionarily significant units of Pacific salmon and steelhead in California, final rule. Federal Register (70): 52487-52627.

NOAA. 2017. El Niño/Southern Oscillation (Enso) Diagnostic Discussion. Climate Prediction Center/NCEP/Nws and the International Research Institute for Climate and Society. May, 2017.

Noble Consultants. 1989. Coastal Sand Management Plan, Santa Barbara, Ventura County Coastline. Irvine, California, Prepared for BEACON (Beach Erosion Authority for Control Operations and Nourishment).

O'Hirok, L. S. 1985. Barrier beach formation and breaching, Santa Clara River mouth, California. Master’s thesis. University of California, Los Angeles.

Orme, A.R. 1982. Temporal variability of a summer shorezone. In Thorne, C.E. ed., Space and time in geomorphology, Allen and Unwin, London, 285- 313.

Orth, D.J. 1987. Ecological considerations in the development and application of instream flow habitat models. Regul. Rivers: Res. Mgmt. 1: 171–181.

Page, G. W., L. E. Stenzel, J. S. Warriner, J. C. Warriner, and P. W. Paton. 2009. Snowy plover (*Charadrius alexandrinus*). *In* A. Poole, editor. The birds of North America online. Cornell Lab of Ornithology, Ithaca, New York.

Park, K., D.W. Hood, and H.T. Odum. 1958. Diurnal pH variation in Texas bays, and its application to primary production estimation. Publication of the Institute for Marine Sciences, Texas 5: 47-64.

Patten, D. T. 1998. Riparian ecosystems of semi-arid North America: Diversity and human impacts. Wetlands 18: 498–512.

Pearse, D. E., S. A. Hayes, M. H. Bond, C. V. Hanson, E. C. Anderson, R. B. MacFarlane, and J. C. Garza. 2009. Over the falls? Rapid evolution of ecotypic differentiation in steelhead/rainbow trout (*Oncorhynchus mykiss*). Journal of Heredity 100: 515–525.

Peterson, C. H. 1979. Predation, competitive exclusion, and diversity in the soft-sediment benthic communities of estuaries and lagoons. Pages 233–264 *in* Ecological processes in coastal and marine systems. Springer, U.S.

Pörtner, H.O., P.M. Schulte, C.M. Wood, and F. Schiemer. 2010. Niche dimensions in fishes: an integrative view. Physiological and Biochemical Zoology, 83(5), pp.808-826.

Powell, A. N. 2001. Habitat characteristics and nest success of snowy plovers associated with California least tern colonies. The Condor 103: 785–792.

PRISM. 2017. PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu

Rauch-Williams, T., E. Dickenson, J. Drewes, and S. Snyder. 2016. Framework for Assessing the Financial, Environmental, and Social Costs and Benefits for Managing Contaminants of Emerging Concern (CEC) in Surface Water. Water Research Foundation Report.

Reid, S.B. 2015. Assessment of occupancy and potential habitat for Pacific Lamprey in the Santa Clara River drainage. Report to United Water Conservation District, Santa Paula CA.

Reynolds, K.M. 2001. Prioritizing salmon habitat restoration with the AHP, SMART, and uncertain data. In. Schmoldt, D. L., J. Kangas, G. A. Mendoza, and M. Pesonen (eds). The Analytic Hierarchy Process in Natural Resource and Environmental Decision Making. Springer Science.

Richards, J.E. and F.W.H. Beamish. 1981. Initiation of feeding and salinity tolerance in the Pacific lamprey *Lampetra tridentata*. Marine Biology, 63(1), pp.73-77.

Richards, A. B., and D. C. Rogers. 2011. List of freshwater macroinvertebrate taxa from California and adjacent states including standard taxonomic effort levels. Southwest Association of Freshwater Invertebrate Taxonomists (SAFIT).

Ripley, B. D. 2001. Spatial statistics. R. R News 1: 14–15.

Rushton, S. P., S. J. Ormerod, and G. Kerby. 2004. New paradigms for modelling species distributions? Journal of Applied Ecology 41: 193–200.

Saaty, T. L. 1980. The analytic hierarchy process. New York: McGraw-Hill.

Saaty, T.L., 2008. Decision Making with the Analytic Hierarchy Process. Int. J. Services Sciences, 1(1), p.83-98.

Sadiq, R. and S. Tesfamariam. 2009. Environmental decision-making under uncertainty using intuitionistic fuzzy analytic hierarchy process (IF-AHP). Stochastic Environmental Research and Risk Assessment. 23(1): 75-91.

Saiki, M.K. 2000. Water quality and other environmental variables associated with variations in population densities of the Santa Ana sucker. Unpubl report to SAWPA from USGS/BRD, Western Fisheries Research Center. 117p.

Sandahl, J., D. Baldwin, J. Jenkins, and N. Scholz. 2007. A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science & Technology 41: 2,998–3,004.

Sawyer, J.O., T. Keeler-Wolf, and J.M. Evens. 2009. A manual of California vegetation. Second edition. California Native Plant Society Press, Sacramento, California.

Schemel, L. 2001. Simplified conversions between specific conductance and salinity units for use with data from monitoring stations. IEP Newsletter 14: 16–18.

Schwartzberg, B. and P. Moore. 1995. A history of the Santa Clara River, Santa Clara River enhancement and management plan.

SCR (Santa Clara River) Project Steering Committee. 1996. Santa Clara River enhancement and management plan study. Biological Resources, Volume 1.

Sully, M.E. 2010. Wind modulation of dissolved oxygen in Chesapeake Bay. Estuaries and Coasts 33: 1164-1175.

Seitzinger, S. P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnology & Oceanography 33: 702–724.

Shapovalov, L., and A. C. Taft. 1954. The life histories of the steelhead rainbow trout (*Salmo gairdneri gairdneri*) and silver salmon (*Oncorhynchus kisutch*) with special reference to Waddell Creek, California, and recommendations regarding their management. California Department of Fish and Game Fish Bulletin 98.

Simenstad, C. A. and J. R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecological Engineering 15: 283–302.

Simons, Li & Associates. 1983. Hydraulic, erosion and sedimentation study of the Santa Clara River Ventura County, California. Prepared for Ventura County Flood Control District, Ventura, California.

Smith, J. J. 1990. The effects of sandbar formation and inflows on aquatic habitat and fish utilization in Pescadero, San Gregorio, Waddell, and Pomponio Creek estuary/lagoon systems, 1985–1989. Prepared by San Jose State University, Department of Biological Sciences, San Jose, California for California Department of Parks and Recreation.

Smith, R. 2008. California least tern breeding survey, 2008. McGrath State Beach, Ventura County, Season Report.

Smith, R. 2009. McGrath and Mandalay State Beaches: California and western snowy plover nesting 2009. Report for the California State Parks from Ventura Audubon Society.

Smith, R., L. O'Neill, S. Tucker, D. Pereksta. 2007. Birds of Ventura County: May, 2007. http://www.venturaaudubon.org/BirdList.htm

Speers-Roesch, B., M. Mandic, D.J. Groom, and J.G. Richards. 2013. Critical oxygen tensions as predictors of hypoxia tolerance and tissue metabolic responses during hypoxia exposure in fishes. Journal of experimental marine biology and ecology, 449, pp.239-249.

Spies, B.T.; B.C. Tarango, and M.A. Steele. 2014. Larval duration, settlement, and larval growth rates of the endangered tidewater goby (*Eucyclogobius newberryi*) and the arrow goby (*Clevelandia ios*) (Pisces, Teleostei). Bulletin of the Southern California Academy of Sciences 113(3): 165-175.

Spina, A. P. 2007. Thermal ecology of juvenile steelhead in a warm-water environment. Environmental Biology of Fishes 80: 23-34.

Steele, M. A., S. C. Schroeter, and H. M. Pace, 2006. Experimental evaluation of biases associated with sampling estuarine fishes with seines. Estuaries and Coasts 29: 1,172–1,184.

Stillwater Sciences. 2007a. Focal species analysis and habitat characterization for the lower Santa Clara River and major tributaries, Ventura County, California. Santa Clara River Parkway Floodplain Restoration Feasibility Study. Prepared by Stillwater Sciences for the California State Coastal Conservancy and the Santa Clara River Trustee Council.

Stillwater Sciences. 2007b. Assessment of geomorphic processes for the Santa Clara River watershed, Ventura and Los Angeles Counties, California. Santa Clara River Parkway Floodplain Restoration Feasibility Study. Prepared by Stillwater Sciences, Berkeley, California for the California State Coastal Conservancy, Oakland, California.

Stillwater Sciences. 2009. City of Ventura special studies—estuary subwatershed study. Final 2009–2010 Monitoring Plan. Prepared by Stillwater Sciences, Berkeley, California, with assistance from Carollo Engineers, Walnut Creek, California, for the City of Ventura, California.

Stillwater Sciences 2010. City of Ventura special studies: estuary subwatershed study. Final Year One Data Summary and Assessment. Prepared by Stillwater Sciences, Berkeley, California for City of Ventura, California.

Stillwater Sciences. 2011a. City of Ventura special studies: Estuary Subwatershed Study assessment of the physical and biological condition of the Santa Clara River Estuary, Ventura County, California. Amended Final Synthesis Report. Prepared by Stillwater Sciences, Berkeley, California for City of Ventura, California. September.

Stillwater Sciences. 2011b. Geomorphic assessment of the Santa Clara River watershed: synthesis of the lower and upper watershed studies, Ventura and Los Angeles counties, California. Prepared by Stillwater Sciences, Berkeley, California for Ventura County Watershed Protection District, Los Angeles County Department of Public Works, and the U.S. Army Corps of Engineers–L.A. District.

Stillwater Sciences. 2015. City of Ventura special studies – Phase 3 Draft Monitoring Plan. Prepared by Stillwater Sciences, Berkeley, California for City of Ventura, California. January.

Stillwater Sciences. 2016. United Water Conservation District Multiple Species Habitat Conservation Plan Study: Effects of Freeman Diversion on Habitat Conditions in the Santa Clara River Estuary. Prepared by Stillwater Sciences, Berkeley, California for United Water Conservation District, Santa Paula, California.

Stillwater Sciences and URS Corporation. 2007. Analysis of riparian vegetation dynamics for the lower Santa Clara River and major tributaries, Ventura County, California. Prepared for the California State Coastal Conservancy, Oakland, California.

Stoecker, M. W. 2002. Steelhead assessment and recovery opportunities in southern Santa Barbara County, California. Conception Coast Project. Santa Barbara, California.

Straub, J. 2013. An environmental risk assessment for human-use trimethoprim in European surface waters. Antibiotics 2:115–162. doi:10.3390/antibiotics2010115

Straub, J. 2016. Aquatic environment risk assessment for human use of the old antibiotic sulfamethoxazole in Europe. Environmental Toxicology and Chemistry 35: 767–779.

Strongin, K., C.M. Taylor, M.E. Roberts, W.H. Neill, and F. Gelwick. 2011. Food habits and dietary overap of two silversides in the Tennessee-Tombigbee waterway: The invasive *Menidia audens* versus the native *Labidesthes sicculus*. The American Midland Naturalist, 166(1), pp.224-233.

Sutula, M., K. Kamer, J. Cable, H. Collis, W. Berelson, J. and Mendez. 2006. Sediments as an internal source of nutrients to Upper Newport Bay, California. Southern California Coastal Water Research Project. Westminster, CA. Technical Report 482.

Sutula, M., H. Bailey, and S. Poucher. 2012. Science supporting dissolved oxygen objectives in California estuaries. Southern California Coastal Water Research Project. Westminster, CA. Technical Report 684.

Swanson, M. L., M. Josselyn, and J. McIver. 1990. McGrath State Beach Santa Clara River Estuary Natural Preserve: restoration and management plan. Prepared for California Department of Parks and Recreation.

Swenson, R. O. 1995. The reproductive behavior and ecology of the tidewater goby *Eucyclogobius newberryi* (Pisces: Gobiidae). Doctoral dissertation. University of California, Berkeley.

Swenson, R.O. 1997. Sex-role reversal in the tidewater goby, *Eucyclogobious newberryi*. Environmental Biology of Fishes 50: 27–40.

Swenson, R. O. 1999. The ecology, behavior, and conservation of the tidewater goby, *Eucyclogobius newberryi*. Environmental Biology of Fishes 55: 99–114.

Swenson, R. O., and A. T. McCray. 1996. Feeding ecology of the tidewater goby. Transactions of the American Fisheries Society 125: 956–970.

Swift, C.C. 2001. The Santa Ana sucker in the Santa Ana River: distribution, relative abundance, spawning areas, and impact of exotic predators. Final report submitted by Larry Munsey International to the Santa Ana Water Project Authority, Riverside, California, USA.

Swift, C. C., P. Duangsitti, C. Clemente, K. Hasserd, and L. Valle. 1997. Biology and distribution of the tidewater goby (*Eucyclogobius newberryi*) on Vandenberg Air Force Base, Santa Barbara County, California. Final report, USNBS Cooperative Agreement 1445-007-94-8129. U.S. National Biological Service.

Swift, C. C., T. R. Haglund, M. Ruiz, and R. N. Fisher. 1993. The status and distribution of the freshwater fishes of southern California. Bulletin of the Southern California Academy of Sciences 92: 101–167.

Swift, C. C., J. L. Nelson, C. Maslow, and T. Stein. 1989. Biology and distribution of the tidewater goby, *Eucyclobius newberryi* (Pisces: Gobiidae) of California. Los Angeles County Museum of Natural History Contributions in Science 404: 1–19.

Swift, C.C., S. Howard, J. Mulder, D. Pondella, and T.P. Keegan. 2014. Expansion of the non-native Mississippi Silverside, Menidia audens (*Pisces, Atherinopsidae*), into fresh and marine waters of coastal southern California. Bulletin of the Southern California Academy of Sciences: 113(3).

Sylvester, J.R., C.E. Nash, and C.R. Emberson. 1975. Salinity and oxygen tolerances of eggs and larvae of Hawaiian striped mullet, *Mugil cephalus* L. Journal of Fish Biology, 7(5), pp.621-629.

Taylor, J. R. 1982. An Introduction to Error Analysis: The Study of Uncertainties in Physical Measurements. University Science Books.

Teledyne Geotronics. 1977. Ortho imagery Flight TG-7700.

Tetra Tech, Inc. 2000. Final report intensive habitat survey for Lake Earl and Lake Talawa, Del Norte County, California. Prepared by Tetra Tech, Inc., San Francisco, California for US Army Corps of Engineers, San Francisco District, San Francisco, California.

Thelander, C., editor. 1994. Life on the edge: a guide to California's endangered natural resources. BioSystem Books, Santa Cruz, California.

Theurer, F.D., K.A. Voos, and W.J. Miller. 1984. Instream water temperature model. Instream Flow Information Paper 16. U.S. Fish and Wildlife Service, Fort Collins, Colorado. FWS/OBS-85/15.

Thomson, J.M. 1963. Synopsis of biological data on the grey mullet, Mugil cephalus Linneaus, 1758. CSIRO Fish. Synopsis 1. 65 pp.

Thompson, K. G., E. P. Bergersen, and R. B. Nehring. 1997. Injuries to brown trout and rainbow trout induced by capture with pulsed direct current. North American Journal of Fisheries Management 17: 141–153.

Tilley, D. 2012. Plant guide for hardstem bullrush (*Schoenoplectus acutus*). USDA-Natural Resources Conservation Service, Idaho Plant Materials Center. Aberdeen, ID.

Titus, R. G., D. C. Erman, and W. M. Snider. In prep. History and status of steelhead in California coastal drainages south of San Francisco Bay.

TNC (The Nature Conservancy). 2013. Coastal resilience Ventura, technical report for coastal hazards mapping. Prepared for TNC by ESA PWA.

Trussell, R., H. A. Anderson, E. G Archuleta. J. Crook, J. Drewes, D. Fort, C. Haas, B. Haddad, D. Huggett, S. Jiang, D. Sedlak, S. Snyder, M. Whittaker, D. Washington. 2012. Water reuse: potential for expanding the nation's water supply through reuse of municipal wastewater.

Turner, J. M. 1975. Ventura County water resources management study—aquifer delineation in the Oxnard-Calleguas area, Ventura County: technical information record, January 1975, Ventura County Department of Public Works Flood Control District.

TVA (Tennessee Valley Authority). 1972. Heat and mass transfer between a water surface and the atmosphere. Water Resources Research, Laboratory Report No. 14. Tennessee Valley Authority, Norris, Tennessee.

URS (URS Corporation). 2005. McGrath Lake Watershed Management Study. Prepared for the U.S. Army Corp of Engineers, 66 pp. plus Appendices.

USFWS (U.S. Fish and Wildlife Service). 1978. Methods of Assessing Instream Flows for Recreation, Instream Flow Information Paper No. 6, FWS/OBS-78/34, U.S. Fish and Wildlife Service, Fort Collins, Colorado.

USFWS. 1999. Santa Clara River Estuary, Ecological Monitoring Program. *In* G. M. Greenwald, L. S. Snell, G. S. Sanders, and S. D. Pratt, editors. USFWS, Ventura, California.

USFWS. 2005. Recovery plan for the tidewater goby (*Eucyclogobius newberryi*). USFWS, Portland, Oregon.

USFWS. 2006. California least tern (*Sternula antillarum browni*) Five-year review: summary and evaluation.

USFWS. 2007. Recovery plan for the Pacific Coast population of the western snowy plover (*Charadrius alexandrinus nivosus*). USFWS, Sacramento, California.

USFWS. 2013. Endangered and threatened wildlife and plants; designation of critical habitat for tidewater goby; final rule. Federal Register (78):8764- 8819.

USGCRP (U.S. Global Change Research Program). 2009. Global climate change impacts in the United States. *In* T. R. Karl, J. M. Melillo, and T. C. Peterson, editors. United States Global Change Research Program. Cambridge University Press, New York..

Ventura County Watershed Protection District (VCWPD). 2016. Streamflow Data for Victoria Ave. Gage #723 http://www.vcwatershed.net/fws/VCAHPS/php/ahps.php?gage=723

VCWPD 2017. Rainfall data for downtown Ventura Courthouse. Gage no 66E http://www.vcwatershed.net/hydrodata/php/getstation.php?siteid=066E#top

Ventura (City of San Buenaventura). 2012. Annual Report of Analysis 2011, Ventura Water Reclamation Facility, City of San Buenaventura, Public Works Department, Ventura, California.

Ventura. 2013. Annual Report of Analysis 2012, Ventura Water Reclamation Facility, City of San Buenaventura, Public Works Department, Ventura, California.

Ventura. 2014. Annual Report of Analysis 2013, Ventura Water Reclamation Facility, City of San Buenaventura, Public Works Department, Ventura, California.

Ventura. 2015. Annual Report of Analysis 2014, Ventura Water Reclamation Facility, City of San Buenaventura, Public Works Department, Ventura, California.

Ventura. 2016. Annual Report of Analysis 2015, Ventura Water Reclamation Facility, City of San Buenaventura, Public Works Department, Ventura, California.

Ventura Water. 2015. 2015 Urban Water Management Plan for City of Ventura. Final Report. Prepared for City of Ventura by Kennedy Jenks Consultants, Oxnard California. June.

Ventura Water. 2016. Drinking Water Consumer Confidence Report. City of Ventura. Available online at: http://www.cityofventura.net/files/file/WaterQualityRptCCR2016Final,Final(1).pdf

Ventura Water. 2017. 2017 Comprehensive Water Resources Report. Final Report. Prepared for City of Ventura by RBF Consulting, Camarillo, California. May.

Verhille C.E., K.K. English, D.E. Cocherell, A.P. Farrell, N.A. Fangue. 2016. High thermal tolerance of a rainbow trout population near its southern range limit suggests local thermal adjustment. Conserv Physiol 4(1): cow057; doi:10.1093/conphys/cow057.

Waite, I. R., and R. A. Barnhart. 1992. Habitat criteria for rearing steelhead: a comparison of site-specific and standard curves for use in the Instream Flow Incremental Methodology. North American Journal of Fisheries Management 12: 40–46.

Warrick, J.A. and L.A.K. Mertes. 2009. Sediment yield from the tectonically active semi-arid Western Transverse Ranges of California Geol. Soc. Am. Bull., 121 (7/8) (2009), pp. 1050–1070.

Weber, M. 1997. Remarks on the paper "On the Measurement of Preferences in the Analytic Hierarchy Process", Journal of Multicriteria Decision Analysis, 6, pp. 320-321.

WRA, Inc. 2014a. Wetland delineation technical report. Santa Clara River Estuary Restoration Project, Oxnard, Ventura County, California. Prepared for Wishtoyo Foundation. November.

WRA, Inc. 2014b. Biological resources technical report. Santa Clara River Estuary Restoration Project, Oxnard, Ventura County, California. Prepared for Wishtoyo Foundation. December.

YCE. 1998. Emergency Lake Level Lowering Permit. YCE Civil Engineering, Land Surveying, and Planning. 23 June 1998.

Zhou, H, T. Ying, X. Wang and J. Liu. 2016. Occurrence and preliminarily environmental risk assessment of selected pharmaceuticals in the urban rivers, China. Scientific Reports 6: 34,928.

Zimmerman, C. E, G. W. Edwards, and K. Perry. 2009. Maternal origin and migratory history of steelhead and rainbow trout captured in rivers of the Central Valley, California. Transactions of the American Fisheries Society 138: 280–291.

Appendices