State of San Francisco Bay 2011
Technical Appendices

Introduction
This document comprises the technical appendices to *The State of San Francisco Bay 2011* (SFEP 2011), a science-based assessment of the health of the San Francisco Bay. The following appendices provide detailed descriptions of the background and rationale, data sources, and methods of calculation for the indicators used to evaluate the health of the Bay. In addition, an indicator that could be utilized in future evaluations of the Bay by SFEP is described.

Indicator Screening Process
The ecological indicators described in *The State of San Francisco Bay 2011* were selected from a group of indicators identified by previous authors (Gunther and Jacobson, 2002; Thompson and Gunther, 2004; The Bay Institute 2003; 2005), and additional potential indicators identified by the authors of *The State of San Francisco Bay 2011*. These candidate indicators were screened using a modified version of the Watershed Assessment Framework developed by the US Environmental Protection Agency. The framework was developed to evaluate the suitability of potential indicators for assessing watershed conditions and trends and to relate management program goals and objectives to ecological conditions. The framework identifies six key attributes that describe the features of an ecological system:

- Landscape condition
- Biotic condition
- Chemical/physical characteristics
- Hydrology/geomorphology
- Ecological processes
- Natural disturbance

The first level of indicator selection criteria evaluated the conceptual relevance of the proposed indicator to the above attributes and to the goals and objectives of the Comprehensive Conservation and Management Plan (SFEP 1993). The results of the initial screening process are presented in *Assessment Framework as a Tool for Integrating and Communicating Watershed Health Indicators for the San Francisco Estuary* (SFEIT 2011), available via the SFEI website (http://www.sfei.org/documents/assessment-framework-tool-integrating-and-communicating-watershed-health-indicators-san-fr).

The selection criteria for indicators are shown in Table 1. It is important to note that the "Transferability" category employed in the initial evaluation of indicators to be selected for calculation in SFEIT (2011) was not relevant to the final evaluation of indicators to be included in *The State of San Francisco Bay 2011*. Therefore, indicators that were excluded for calculation based solely on non-transferability were not excluded from *The State of San Francisco Bay 2011*. For example, the Estuarine Open Water Habitat indicator was not calculated in SFEIT (2011) because it could not be transferred to other watersheds, but was included in *The State of San Francisco Bay 2011* as an important regional indicator.

Peer Review
The data, methods, and analysis used to compile *The State of San Francisco Bay 2011* were drawn from many different sources. These include papers in the peer-reviewed literature, ongoing monitoring programs, or publications from respected organizations in the region. These various sources utilize peer review to ensure the credibility and authority of their products, and

so it was not deemed necessary (nor economically feasible) for *The State of San Francisco Bay 2011* to be developed using a completely independent peer review process. However, SFEP sought peer review of all the methods and analysis to provide a document that was authoritative and useful.

Consequently, the peer review of the material presented in the following technical reports took many forms. For some of the indicators described, published methods and analysis in peer-reviewed literature were used in the evaluation. Other indicators were evaluated using methods that had been developed with input from scientific advisory panels for previous assessments of the Bay, such as the *Ecological Scorecard* (The Bay Institute 2003; 2005). Some ongoing monitoring programs, such as the Regional Monitoring Program, have technical review committees and periodically empanel independent reviewers to assure their methods and analysis are credible (Bernstein and O'Connor, 1997; Berger *et al*. 2004). In addition, the authors of the individual technical reports that follow sought review by knowledgeable colleagues from the Bay Area scientific community.

SFEP invites any readers of these Technical Appendices who have specific comments to forward these in writing to Judy Kelly, Executive Director of SFEP. It is fully expected that this report will generate a wide array of comments, and SFEP expects to integrate these comments into preparations for a future State of the Bay report in an effort to continue to refine indicator selection and analysis.

Table 1: Selection criteria for watershed assessment indicators for the San Francisco Estuary.

(Indicator Name)				
	Result (yes or no)	**WAF category**	**CCMP Goal**	**Comments**
Conceptual Relevance				
Fits with WAF category (ecological function)				
Fits with CCMP (management objectives)				
Data Availability and Adequacy				
Data available				
Data suitable quality				
Responsiveness				
Driver-outcome linkage				
Sensitivity				
Response time frame				
Spatial sampling frame				
Interpretation				
Goals, thresholds or reference conditions defined				
Meaningful to public				
Transferability				
Scalable				
Transferable to other watershed				

References

Berger, R., J. Conomos, P. Herrgesell, A. Mearns, J.R. Schubel, and S. Weisberg. 2004. Report of the 2003 Program Review, Regional Monitoring Program for Trace Substances in the San Francisco Estuary. SFEI Contribution 303. San Francisco Estuary Institute, Oakland, CA.

Bernstein, B. and J. O'Connor. 1997. Five-year program review: Regional Monitoring Program for Trace Substances in the San Francisco Estuary. Prepared for the San Francisco Estuary Institute, Richmond,CA.

Gunther, A. and L. Jacobson. 2002. Evaluating the Ecological Condition of the South Bay: A Potential Assessment Approach. Prepared for the City of San Jose Department of Environmental Services.

[SFEIT] San Francisco Estuary Indicators Team. 2011. Assessment Framework as a Tool for Integrating and Communicating Watershed Health Indicators for the San Francisco Estuary. Report by J.N. Collins, J.A. Davis, R. Hoenicke, T. Jabusch, C. Swanson, A. Gunther, N. Nur, and P. Trigueros. SFEI Contribution 634. San Francisco Estuary Institute, Oakland, CA.

[SFEP] San Francisco Estuary Project.1993. Comprehensive Conservation and Management Plan.

[SFEP] San Francisco Estuary Project. 2011. The State of San Francisco Bay.

The Bay Institute. 2003. Ecological Scorecard: San Francisco Bay Index 2003. Available at: http://www.bay.org/publications/-ecological-scorecards

The Bay Institute. 2005. Ecological Scorecard: San Francisco Bay Index 2005. Available at: http://www.bay.org/publications/-ecological-scorecards

Thompson, B., and A. Gunther. 2004. Development of Environmental Indicators of the Condition of San Francisco Estuary: A Report to the San Francisco Estuary Project. SFEI Contribution 113. San Francisco Estuary Institute, San Francisco, CA.

State of San Francisco Bay 2011
Appendix A

WATER - Water Quality
Technical Appendix

Jay Davis and John Ross, San Francisco Estuary Institute
Michael Kellogg, City and County of San Francisco
Andrew Cohen, Center for Research on Aquatic Bioinvasions

GENERAL CONSIDERATIONS

Clean water is essential to the health of the San Francisco Bay ecosystem and to many of the beneficial uses of the Bay that Bay Area residents enjoy and depend on. Billions of dollars have been invested in management of the wastewater and other pollutant sources that impact Bay water quality, and as a result the Bay is in much better condition than it was in the 1970s. However, thousands of chemicals are carried into the Bay by society's waste streams, and significant and challenging water quality problems still remain.

The Bay Area is fortunate to have one of the best water quality monitoring programs in the world (the Regional Monitoring Program for Water Quality in the San Francisco Estuary) in place to track conditions in the Bay and to provide the information that water quality managers need to address the remaining problems. This report card on Bay water quality is based largely on information generated by the Regional Monitoring Program. Other valuable sources of information are also available and were also considered.

The availability of appropriate assessment thresholds (i.e., water quality objectives or fish tissue contamination guidelines) is fundamentally important to evaluating the condition of the Bay. For many pollutants such guidelines are not available. Pollutants can be placed into three categories with regard to the availability of assessment thresholds.

The first group includes pollutants that historically have posed the greatest threats to water quality and that have been the subject of intense scrutiny by managers. Guidelines have been established for these pollutants that are generally based on extensive information on their effects on target organisms and that are accepted by regulators and scientists. This report card pays greater attention to these pollutants as they are a primary focus of water quality regulators and scientists. Mercury and PCBs, for example, are two of the greatest concerns in the Bay, and highly scrutinized cleanup plans (TMDLs) have been incorporated into the Basin Plan for the Bay (http://www.swrcb.ca.gov/rwqcb2/basin_planning.shtml) in an effort to reduce their impacts on Bay water quality.

A second group consists of pollutants where guidelines exist but the degree of concern is low. Many pollutants with established assessment thresholds are present at concentrations that are far below the threshold and do not threaten to approach those thresholds in the foreseeable future. Some of these pollutants used to be problems in the past, but now do not pose a threat because of effective management. While it is important to recognize this category of pollutants and to continue monitoring them to make sure they stay below thresholds, this report card focuses on the pollutants that are the current focus of managers and where progress is most needed.

A third, and very large, group consists of pollutants where assessment thresholds are not available. Some of these pollutants are suspected to potentially be causing impairment in the Bay, but regulators have not yet established thresholds either due to a lack of scientific information or resources to address the long list of pollutants of potential concern. While quantitative assessment of these pollutants is not possible, they are still addressed in a qualitative manner.

EVALUATION SCHEME

The water quality indicators presented in this report card were evaluated using a scheme that takes into account both 1) the distance of the data distribution relative from the relevant guideline in terms of the estimated length of time expected for the indicators to reach the desired condition and 2) the severity of the impairment of water quality.

This water quality element of the Bay report card addresses the three main beneficial uses of the Bay that are affected by water pollution and protected by the Clean Water Act, addressing three key questions that are posed in a manner intended to be easily understood by the public:

1. Is the Bay safe for aquatic life?
2. Are fish from the Bay safe to eat?
3. Is the Bay safe for swimming?

Suites of indicators were identified to answer each of these questions. The basic approach to answering each of these questions is described below.

QUESTION 1: IS THE BAY SAFE FOR AQUATIC LIFE?

A varied group of indicators is most appropriate for addressing question 1. This group includes a target from the Mercury TMDL for methylmercury concentrations in small fish, a qualitative narrative objective that applies to the occurrence of toxicity in Bay sediments, and numeric water quality objectives that are based on measurement of concentrations in water.

For each parameter, the distribution of the data for each sampling year is compared to the target. The degree of risk for pollutants in this category are based on assessments in published studies and other considerations discussed below for each pollutant. A second measure for pollutants that do not meet the goal is the estimated recovery time. A quantitative recovery time estimate is available for methylmercury. For others, the estimates are based on conceptual considerations.

QUESTION 2: ARE FISH FROM THE BAY SAFE TO EAT?

For question 2, the appropriate indicators are concentrations of pollutants of concern in the tissue of fish species that are popular for consumption by Bay anglers. The Regional Monitoring Program has conducted systematic monitoring of Bay sport fish on a triennial basis since 1994, providing a solid foundation for assessing this question.

Thresholds for evaluating fish tissue concentrations have been developed by the California Office of Environmental Health Hazard Assessment (OEHHA) (Klasing and Brodberg 2008). OEHHA is the agency responsible for establishing safe eating guidelines for wild fish caught from California water bodies, including San Francisco Bay. OEHHA issued consumption guidelines for the Bay in response to the first sport fish survey in 1994 (OEHHA 1994). OEHHA completed an update of these guidelines in 2011 (Gassel et al. 2011). OEHHA has developed thresholds called advisory tissue levels (ATLs) that are a component of their complex process of data evaluation and interpretation in the development of safe eating guidelines. Other factors are also considered in this process, such as omega-3 fatty acid concentrations in a given species in a water body, and risk communication needs. OEHHA uses ATLs as a framework, along with best professional judgment, to provide fish consumption guidance on an ad hoc basis that best combines the needs for health protection and ease of communication for each site. Given their role in development of safe eating guidelines, ATLs are used in this report for assessing fish tissue data with respect to question 2. Consistent with the description of ATLs above, however, it is important to note that the comparisons to ATLs presented in this report are general indications of potential levels of risk, and are not intended to represent consumption advice. The updated safe eating guidelines for the Bay represent the definitive statement for the public on the safety of consuming Bay fish. The intent of using ATLs in the State of the Bay Report is to convey a message to the public that is consistent with and supports the safe eating guidelines.

OEHHA has not developed thresholds for interpreting dioxin concentrations. In the absence of OEHHA thresholds, a screening value developed by the San Francisco Bay Regional Water Quality Control Board as part of the PCB TMDL (SFBRWQCB 2008) was used.

For evaluating question 2, time series plots are presented that show the average concentration for selected indicator species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay. ATLs are used as a frame of reference to indicate the general degree of risk posed by each pollutant. OEHHA has established ATLs for different levels of consumption. The ATLs used include the concentrations above which no consumption may be indicated ("no consumption ATLs") and concentrations below which consumption of up to three eight ounce (prior to cooking) servings per week may be indicated. Estimated recovery times for methylmercury and PCBs are based on analyses presented in the TMDLs.

QUESTION 3: IS THE BAY SAFE FOR SWIMMING?

For question 3, the best available indicator is concentrations of bacteria in water near popular bathing beaches.

To protect beach users from exposure to fecal contamination California has adopted standards developed for high use beaches and applies them during the prime beach season from April through October at beaches with more than 50,000 annual visitors that are adjacent to a storm drain that flows in the summer; these requirements are only mandatory in years that the legislature has appropriated monies sufficient to fund the monitoring. County Public Health and other agencies routinely monitor fecal indicator bacteria (FIB) concentrations at Bay beaches where water contact recreation is common and provide warnings to the public when concentrations exceed the standards (Table 1). FIB are enteric bacteria common to the digestive systems of mammals and birds and are indicators of fecal contamination. While not generally pathogenic themselves, FIB are used because they correlate well with the incidence of human illness in epidemiology studies at recreational beaches and can be enumerated more quickly and cost effectively than can pathogens directly.

Heal the Bay, a Santa Monica-based non-profit, provides comprehensive evaluations of over 400 California bathing beaches in both Annual and Summer Beach Report Cards as a guide to aid beach users' decisions concerning water contact recreation. Higher grades are considered to represent less health risk to swimmers than are lower grades. The Heal the Bay grades for Bay beaches were used as the primary indicator of whether the Bay is safe for swimming.

IS THE BAY SAFE FOR AQUATIC LIFE?

POLLUTANTS WITH APPROPRIATE THRESHOLDS

1. Methylmercury in Prey Fish

In addition to posing risks to humans who eat Bay fish, methylmercury poses significant risks to Bay wildlife. Extensive studies in Forster's Terns have concluded that 48% of birds in the breeding season in this species were at high risk of reproductive impairment due to methylmercury exposure (Eagles-Smith et al. 2009). They also estimated substantial, but lower risk, to Caspian Terns, Black-necked Stilts, and American Avocets. Methylmercury is also considered to pose significant risks to two endangered bird species in the Bay. The federally endangered California Clapper Rail has poor reproductive success that may be related to methylmercury. An estimated 15–30% of the observed reduction below normal hatchability in this subspecies has been attributed to contaminants, with methylmercury principal among them (Schwarzbach et al. 2006). In the evaluation of risks to wildlife for the Mercury TMDL, the greatest concern was for the federally endangered California Least Tern, based on an assessment by the U.S. Fish and Wildlife Service, and a prey fish tissue target to protect aquatic life was developed based on protection of this species (SFBRWQCB 2006). Other species where possible effects have been less thoroughly examined but the degree of exposure suggests potential risks to reproduction include the Black Rail and tidal marsh Song Sparrow (Grenier and Davis 2010).

Gathering information on where and when methylmercury enters the food web has been a top priority in the RMP over the past several years. In addition to their value as an indicator of wildlife exposure, small fish have been sampled extensively because they are a valuable indicator for obtaining this information. The young age and restricted ranges of small fish allow the timing and location of their mercury exposure to be pinpointed with a relatively high degree of precision.

Based on the Mercury TMDL, methylmercury in prey fish tissue is the key regulatory target for protection of aquatic life. The primary fish species upon which the opportunistic California Least Tern prey are whole fish in the size range of 3-5 cm, so the target is based on this class of fish. The target to protect reproduction in the Least Tern as well as other aquatic life is 0.03 ppm as an average concentration. These parameters were used to define and assess the indicator for methylmercury impact on aquatic life.

Data Source The methylmercury in prey fish indicator was calculated using data from the RMP. A summary report on the extensive prey fish sampling that has been conducted in recent years is in preparation.

The RMP began monitoring methylmercury in prey fish in 2005 as part of a three-year pilot study. This study sampled 10 or fewer sites per year. In 2008, the RMP began more extensive small fish monitoring in a concerted effort to determine patterns in food web uptake. This second three-year effort sampled approximately 50 sites per year. The sampling has focused on two species: Mississippi silverside and topsmelt. Samples have been collected in all of the regional embayments.

Methods and Calculations The aquatic life methylmercury indicator (Figure 1) was calculated using available data from the RMP for Mississippi silverside and topsmelt in the 3-5 cm size range. The time series plot shows the distribution of the data for each year sampled. The distribution is described with percentiles (25^{th}, 50^{th}, and 75^{th}).

Goals, Targets, and Reference Conditions The target established by the TMDL to protect reproduction in the Least Tern as well as other aquatic life is 0.03 ppm as an average concentration in prey fish in the 3-5 cm size range.

Results

In the most recent sampling year, methylmercury concentrations in prey fish exceeded the 0.03 ppm target in approximately 95% of the samples collected. Similar results were obtained in 2008, the other year with a larger sample size. Results from the pilot study in 2005-2007 were lower, but the distributions for those years are based on a very small sample size. The Baywide median concentration in 2009 was 0.051 ppm.

Evaluation of spatial and temporal trends should focus on data from 2008 and 2009, which are based on larger sample sizes. Median concentrations in each region in 2009 ranged from a high of 0.081 in South Bay to a low of 0.035 ppm in Suisun Bay.

As discussed below in the Methylmercury in Sport Fish section, methylmercury concentrations in the Bay food web have not changed perceptibly over the past 40 years, and it is not anticipated that they will decline significantly in the next 30 years. Extensive studies on risks to Bay birds have concluded that substantial portions of some populations are facing very high risk of reproductive impairment. However, the species facing the greatest risks, the Forster's Tern, forages primarily in salt ponds. These relatively highly managed habitats may offer opportunities for intervention in the methylmercury biogeochemical cycle to reduce exposure of wildlife. It is therefore plausible that ways of reducing Forster's Tern exposure and risk may be identified and implemented within the next 30 years. While exposure of wildlife to methylmercury may be a somewhat tractable problem, it will be difficult to reduce exposure in other habitats (open Bay and tidal marsh) in the next 30 years. The summary rating for methylmercury risk to aquatic life is therefore one star (Figure 2).

2. Sediment Toxicity

The frequent occurrence of toxicity in sediment samples from the Bay is a significant concern. In every year since sampling began in 1993, at least 26% of sediment samples have been determined to be toxic to one or more test species. In 2009, 67% of the samples were found to be toxic to at least one of the two test species. No long-term trend is apparent in this time series. These toxicity tests indicate that pollutant concentrations in Bay sediments are high enough to affect the abundance of aquatic invertebrates. The pollutants causing this persistent toxicity have not yet been identified. Until the stressors driving this toxicity are reduced, this problem will persist into the future.

The State Water Board is in the process of developing quantitative sediment quality objectives (SQOs) for protection of aquatic life in enclosed bays and estuaries in California (SFEI 2009). Attainment of these objectives is to be assessed using a combination of data on sediment chemistry, sediment toxicity, and benthic community composition (the sediment quality triad). SQOs have been established for polyhaline (marine) habitats, and are still in development for lower salinity habitats, such as those that are present throughout much of San Francisco Bay. Assessments of triad data from the Bay using the SQO framework have concluded that some degree of impact was considered possible in 96% of the ecosystem (SFEI 2009). Most of the Bay (73%) was classified as "possibly impacted." Sediment toxicity was the primary driver of these assessment results. Until SQOs that cover the entire Bay are established, the incidence of sediment toxicity is an appropriate indicator of Bay sediment quality.

In the meantime, a narrative water quality objective in the Basin Plan applies to sediment toxicity. The objective states: "No toxic or other deleterious substances shall be present in receiving waters in concentrations or quantities which will cause deleterious effects on aquatic biota, wildlife, or waterfowl or which render any of these unfit for human consumption either at levels created in receiving waters or as a result of biological concentration." The implicit quantitative goal associated with this objective is a 0% incidence of toxicity in Bay samples.

Data Source The sediment toxicity indicator is based on data from the RMP, available on the RMP website (www.sfei.org/rmp/data). The RMP measures sediment toxicity annually at 27 stations throughout the Bay. Most of the samples are collected at randomly selected locations, with a few fixed stations included to continue long-term time series. Two types of sediment bioassays are conducted at each station. Homogenized whole sediment is tested for toxicity using the amphipod *Eohaustorius estuarius* in a 10 day amphipod survival test. Sediment-water interface (SWI) cores are tested using the bivalve *Mytilus galloprovincialis* in a 48 hour static embryolarval development toxicity test.

Methods and Calculations The sediment toxicity indicator (Figure 3) is simply the percentage of the samples tested in each year that were determined to be toxic to at least one of the test organisms. Samples are considered to be toxic if they meet two criteria: 1) statistically significant difference from controls, and 2) a difference from controls that is of sufficient magnitude in absolute terms.

Goals, Targets, and Reference Conditions As discussed above, the implicit goal associated with the narrative objective pertaining to sediment toxicity is 0% incidence of toxicity in Bay samples.

Results

On the whole Bay scale, the incidence of sediment toxicity has ranged from a low of 26% of samples in 2004 to a high of 85% in 2007 (Figure 3). In most years the incidence has been higher than 50%. In 2009, 67% of samples were toxic to the test organisms. The incidence of toxicity has shown no indication of a decline.

The incidence of sediment toxicity varies among the embayments. The incidence has been highest in Suisun Bay, where frequently 100% of samples have been toxic. South Bay has had the second highest incidence, with 50% or more samples toxic in all but two years. The incidence of toxicity has been lower in San Pablo Bay and Central Bay, where fewer than 50% of samples have been toxic in most years.

In most of these cases where toxicity has been observed, the degree of toxicity has not been severe, with severe toxicity defined as mortality rates for *Eohaustorius* approaching 100% or rates of abnormal development in *Mytilus* larvae approaching 100%. The observed degree of toxicity is considered to be moderate. In terms of the assessment scheme used in this report, this corresponds to the "moderate concern" category. The incidence of sediment toxicity has shown no sign of declining since 1993, and until the causes of the toxicity are identified it is not possible to say whether the goal of 0% toxicity will be attained in 30 years. These considerations place sediment toxicity in the "rapid progress unlikely" category. The summary rating for sediment toxicity is therefore two stars.

3. Copper in Water

Background and Rationale Copper pollution was a major concern in the Estuary in the 1990s, as concentrations were frequently above the water quality objective. An evaluation of the issue by the Water Board and stakeholders led to new site-specific water quality objectives for copper in the Bay (less stringent but still considered fully protective of the aquatic environment), pollution prevention and monitoring activities, and the removal of copper from the 303(d) List in 2002. Along with the new objectives, a program has been established to guard against future increases in concentrations in the Bay. The program includes actions to control known sources in wastewater, urban runoff, and use of copper in shoreline lagoons and on boats. More aggressive actions to control sources can be triggered by increases in copper or nickel concentrations. A remaining concern regarding possible impacts of copper on olfaction in salmonids is currently being investigated by the National Oceanographic and Atmospheric Administration's Northwest Fisheries Science Center.

Concentrations of copper in water are the key impairment indicator for this pollutant.

Data Source The copper indicator was calculated using data from water sampling conducted by the RMP. The data are available from the RMP website (www.sfei.org/rmp/data).

Methods and Calculations The copper indicator was calculated for each year of RMP monitoring from 1993 to 2009 (Figure 4). The time series plot shows the distribution of the data (dissolved concentrations in water) for each year sampled. The distribution is described with percentiles (5^{th}, 25^{th}, 50^{th}, 75^{th}, and 95^{th}).

Goals, Targets and Reference Conditions Two different site-specific copper objectives have been established for the Bay. For Lower San Francisco Bay south of the line representing the

Hayward Shoals shown and South San Francisco Bay the objective is 6.9 ug/L. For the portion of the delta located in the San Francisco Bay Region, Suisun Bay, Carquinez Strait, San Pablo Bay, Central San Francisco Bay, and the portion of Lower San Francisco Bay north of the line representing the Hayward Shoals the objective is 6.0 ug/L. The objectives are for dissolved concentrations.

Results Copper concentrations in the Bay have been below the site-specific objectives for all samples measured from 1993 to 2009. Due to the remaining uncertainty regarding the possible impact of copper on salmon olfaction, copper was placed in the "low concern/rapid progress likely" category.

4. Dissolved Oxygen in Water

Background and Rationale Enforcement of the Clean Water Act and other environmental laws over the past 39 years has resulted in tremendous improvements in overall Bay water quality, solving serious problems related to organic waste, nutrients, and silver contamination. In the early 1970s the Bay suffered from severely degraded water quality. The discharge of poorly treated wastewater, primarily from publicly-owned treatment works (POTWs) serving the Bay Area's growing population, was the cause of large and frequent fish kills, unsafe levels of bacteria in water and shellfish, and a notoriously foul stench (Krieger et al 2007). The Clean Water Act provided a major impetus toward cleaning up the Bay by setting clear goals and supplying over a billion dollars that supported construction of POTWs. In response, POTWs and industrial wastewater dischargers achieved significant reductions in their emissions of pollutants into the Bay, and the most noticeable problems of the 1970s have been solved. Inputs of organic waste and nutrients have been greatly reduced and no longer cause fish kills or odor problems.

Some concerns remain with regard to dissolved oxygen concentrations in the Bay. Low dissolved oxygen resulting indirectly from the large amount of freshwater input to the Bay in 2006 was considered a possible cause of a fish kill in June of that year. Dissolved oxygen and nutrient concerns still exist for salt ponds, lagoons, and other areas around the edges of the Bay. Recent observations of increasing transparency in the Bay due to declining suspended sediment concentrations (Schoellhamer 2009) and increasing chlorophyll concentrations (SFEI 2009) are raising concerns that dissolved oxygen concentrations could again decline to problematic levels.

Concentrations of dissolved oxygen in water are a key impairment indicator for organic waste and nutrients.

Data Source The dissolved oxygen indicator was calculated using data from water sampling conducted by the RMP. The data are available from the RMP website (www.sfei.org/rmp/data).

Methods and Calculations The dissolved oxygen indicator was calculated for each year of RMP monitoring from 1993 to 2009 (Figure 5). The time series plot shows the distribution of the data (dissolved concentrations in water) for each year sampled. The distribution is described with percentiles (5^{th}, 25^{th}, 50^{th}, 75^{th}, and 95^{th}).

Goals, Targets, and Reference Conditions There are two objectives for dissolved oxygen in the Bay. An objective of 5 mg/L applies to waters downstream of the Carquinez Strait. The objective for Suisun Bay is 7 mg/L.

Results Dissolved oxygen concentrations in the Bay have exceeded the objective for almost all samples measured from 1993 to 2009 (Figure 5). No pattern of declining dissolved oxygen is evident in the time series for each embayment. The overall score for dissolved oxygen is therefore "goal attained" (five stars). It should be noted, however, that increasing phytoplankton abundance in the South Bay has raised concern that concentrations could potentially decline again to problematic levels.

5. Silver in Water

Background and Rationale Enforcement of the Clean Water Act and other environmental laws over the past 35 years has resulted in tremendous improvements in overall Bay water quality, solving serious problems related to organic waste, nutrients, and silver contamination. In the 1970s the Bay had the highest silver concentrations recorded for any estuary in the world, but the closure of a major photo processing plant and improved wastewater treatment led to a reduction in concentrations in South Bay clams from 100 ppm in the late 1970s to 3 ppm in 2003, eliminating adverse impacts on clam reproduction. With the continued vigilance of regulators and treatment plant operators, broad-scale adverse impacts of dissolved oxygen, nutrients, and silver on Bay water quality are not likely.

Concentrations of silver in water are the key impairment indicator for this pollutant.

Data Source The silver indicator was calculated using data from water sampling conducted by the RMP. The data are available from the RMP website (www.sfei.org/rmp/data).

Methods and Calculations The silver indicator was calculated for each year of RMP monitoring from 1993 to 2009 (Figure 6). The time series plot shows the distribution of the data (dissolved concentrations in water) for each year sampled. The distribution is described with percentiles (5^{th}, 25^{th}, 50^{th}, 75^{th}, and 95^{th}).

Goals, Targets, and Reference Conditions The water quality objective for silver in the Bay is 1.9 ug/L (SFBRWQCB 2007). The objective applies to dissolved concentrations.

Results Silver concentrations in the Bay have been far below the objective for all samples measured from 1993 to 2009, and are not expected to increase. The overall score for dissolved oxygen is therefore "goal attained" (five stars).

6. Other Priority Pollutants

In addition to the pollutants mentioned above, the RMP monitors many other pollutants that are present at concentrations below water quality objectives and are considered to pose low risk to Bay aquatic life. In the 1970s, USEPA established a list of 129 pollutants that were identified as priorities for regulation. Objectives and analytical methods for these "priority pollutants" were developed and they became widely monitored. California has its own set of water quality criteria for these pollutants that was promulgated in 2000 under the "California Toxics Rule." These criteria apply to all inland surface waters in California, including the Bay.

The RMP measures many of the priority pollutants, either routinely or through special studies. A large number of these priority pollutants are present in the Bay at concentrations that are well below water quality criteria. These pollutants all fall in the "goals attained" category. Some of these pollutants are listed below by class:

- metals - arsenic, cadmium, cobalt, chromium, iron, manganese, nickel, lead, zinc, alkyltins;
- pesticides - diazinon, chlorpyrifos, dachthal, lindanes, endosulfans, mirex, oxadiazon;
- industrial chemicals - phthalates, hexachlorobenzene;
- nutrients - nitrate, nitrite, phosphate, ammonium;
- others – cyanide.

POLLUTANTS WITHOUT APPROPRIATE THRESHOLDS

1. Exotic Species

Exotic species released from ship ballast water are considered a water pollutant under the Clean Water Act, and they are included on the 303(d) list of impaired water bodies due to their disruption of benthic communities, their disruption of food availability to native species, and their alteration of pollutant availability in the food web. San Francisco Bay is considered one of the most highly invaded estuaries in the world (Cohen and Carlton 1998), and the ecological impacts of exotic species have been immense. Introductions of hundreds of exotic species have irreversibly altered the Bay ecosystem in fundamental ways. Nonnative species introduced to the Bay have reduced or eliminated populations of many native species so that in some regions and habitats virtually 100% of the organisms are introduced. They have also interfered with water withdrawals, boating, fishing (though also providing sport and forage fish), water contact recreation, and probably have eroded marshes in some areas though also accreting marsh elsewhere. Recently adopted state interim ballast discharge regulations to be phased in over 2010-2016, if rigorously implemented and enforced, would essentially resolve one major introduction pathway. Several other pathways - including introductions due to aquaculture activities and importations of live bait, aquarium organisms, ornamental plants, live educational/research organisms and live seafood - could also be better managed by thoughtful regulation, or by a combination of regulations and public education and outreach.

Exotic species introductions do not fit neatly into the assessment framework used for this report card. Successful invasions of nonnative species are essentially irreversible, so to a significant degree goals of restoring native species are not achievable. Attention is best focused on a goal that is achievable in the near term: reducing the rate of introductions. California's new ballast discharge regulations could have a significant impact in this regard, if rigorously enforced; and the USEPA is currently developing a revised Vessel General Permit (to be issued in 2013) that will include limits on organism concentrations in ballast water discharges into US waters; appropriate limits, effectively enforced, would be a tremendous help.

Focusing on the significant achievable goals mentioned above, exotic species fall in the "rapid progress likely" category. With regard to the degree of risk, this is hard to quantify but no pollutants have had a higher degree of impact on the ecology of the Bay than exotic species, and if invasions are allowed to continue additional large impacts are likely. This places exotic species in the "high concern" category. The summary rating for exotic species is therefore two stars.

2. Trash

Trash is a continuing problem in the Bay both as an aesthetic nuisance and as a threat to aquatic life. Data suggest that plastic from trash persists for hundreds of years in the environment and can pose a threat to wildlife through ingestion, entrapment and entanglement, and this plastic can leach potentially harmful chemicals to the aquatic environment. Trash is a concern at both a macro scale, with the aesthetic, ingestion, and entanglement associated with visible trash items. Trash is also a concern at a micro scale, as larger trash items degrade to small fragments that are not visible but may have significant impacts on small aquatic life through ingestion and through exposure of small aquatic life to the chemical constituents that leach from the particles, as well as the organic pollutants from other sources that accumulate on the particles.

In recognition of the risks posed by trash, Central Bay and a portion of South Bay (in addition to many urban creeks) have been recommended for inclusion on the 303(d) List (SFBRWQCB 2009). Beneficial uses adversely impacted by trash are supported by narrative water quality objectives and prohibitions in the Basin Plan regarding solid waste, floating material, and settleable material. An established numerical goal for trash abundance in the Bay does not exist.

Trash has recently been receiving increased attention from Bay Area water quality managers. Extensive requirements relating to trash were included in the municipal regional permit for stormwater (MRP) issued in 2010. The trash reduction requirements in the MRP are multifaceted and focus both on short-term actions to remove trash from known creek and shoreline hot spots and long-term actions to significantly reduce trash discharged from municipal storm drain systems. During this permit term, municipalities are required to develop and implement a Short-Term Trash Load Reduction Plan to attain a 40% reduction of trash loads by 2014. Municipalities are then required to use their short-term experiences and lessons learned to develop and begin implementation of a Long-Term Trash Load Reduction Plan, to attain a 70% reduction in trash loads by 2017 and 100% by 2022. Attaining these goals should greatly reduce the input of trash into Bay waters and hopefully allow the abundance of trash and microplastics to dissipate.

The severity of the trash problem is difficult to quantify and not well-characterized but a plausible argument can be made that trash in the Bay is a moderate concern in regard to impacts on aquatic life. Aggressive requirements in the MRP should significantly reduce inputs in the next 30 years, and hopefully this will rapidly reduce the amount of trash and microplastic particles in the Bay. The summary rating for trash is therefore three stars.

3. Other Suspected Threats

There are several other pollutants that are suspected to possibly pose moderate to high risks to Bay aquatic life, but for which appropriate thresholds have not yet been developed. A few of the most prominent examples are briefly described below.

Selenium

Selenium concentrations found in Bay biota are thought to exceed levels that can cause reproductive impacts in white sturgeon and are often higher than levels considered safe for fish and other wildlife species in the Estuary. Concern for risks to aquatic life is the primary impetus for the North Bay Selenium TMDL that is in development (SFBRWQCB 2011). Thresholds to protect aquatic life are in development that will be more appropriate than existing water quality criteria.

Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are included on the 303(d) List for several Bay locations. There is also concern that PAH concentrations in sediment across much of the Bay exceed thresholds for impacts on early life stages of fish and on benthic invertebrates. PAH concentrations over the past 20 years have held fairly constant. Increasing population and motor vehicle use in the Bay Area are cause for concern that PAH concentrations could increase over the next 20 years. On the other hand, PAH concentrations in Bay Area air have declined over the past ten years, and if PAH inputs to the Bay can be decreased concentrations are expected to drop quickly.

Polybrominated Diphenyl Ethers (PBDEs)

PBDEs are considered a potential risk to Bay wildlife. However, a regulatory goal has not yet been established for PBDEs in aquatic life. The RMP is currently conducting a study to better understand threshold for risks to birds.

Perfluorooctanesulfonate (PFOS)

PFOS is also considered a potential risk to Bay wildlife. A regulatory goal has not yet been established for PFOS in aquatic life. RMP monitoring has found concentrations of PFOS in bird eggs that approach levels associated with adverse impacts seen in studies elsewhere.

4. Contaminants of Emerging Concern

As discussed above relative to risks to human health, in addition to the specific pollutants that pose threats to aquatic life, there are thousands of other chemicals used by society, including pesticides, industrial chemicals, and chemicals in consumer products, and many of these make their way from our homes, businesses, and watersheds into the Bay. Due to inadequate screening of the hazards of these chemicals, some may cause toxicity in Bay biota, either through direct exposure to contaminated water or sediment or through accumulation in the Bay food web and dietary exposure in species at higher trophic positions. As understanding advances, some of these contaminants emerge as posing risks to the health of humans and wildlife.

The RMP actively monitors contaminants of emerging concern that pose the greatest known threats to water quality. However, as mentioned above, these monitoring efforts to protect Bay water quality are severely hampered by the lack of information on the chemicals present in commercial products, their movement in the environment, and their toxicity. Ultimately, the reduction of use of toxic chemicals in products is the ideal way to prevent further additions to the list of legacy contaminants that is passed on to future generations of humans and wildlife that depend upon the Bay.

ARE BAY FISH SAFE TO EAT?

POLLUTANTS WITH APPROPRIATE THRESHOLDS

1. Methylmercury in Sport Fish

Background and Rationale

Methylmercury is one of four pollutants (the others are PCBs, exotic species, and trash) that are classified as having significant impacts on Bay water quality because the entire Bay is considered impaired by these pollutants, and the degree of risk is above established thresholds of concern.

Methylmercury is arguably the Bay's most serious water quality concern. Methylmercury is a primary driver of the fish consumption advisory for the Bay (OEHHA 1994, Hunt et al. 2008), and also is suspected to be adversely affecting wildlife populations, including the endangered California Clapper Rail and California Least Tern, as well as the Forster's Tern (Schwarzbach et al. 2006, Eagles-Smith et al. 2009). Due to these concerns, the first TMDL for the Bay has been developed for mercury (SFBRWQCB 2006).

Methylmercury typically represents only about 1% of total mercury, but is the specific form that accumulates in aquatic life and poses health risks to humans and wildlife. Methylmercury is a neurotoxicant, and is particularly hazardous for fetuses and children and early life-stages of wildlife species as their nervous systems develop. The sources of methylmercury in the Bay, particularly the methylmercury that actually gets taken up into the food web, are not well understood. Methylmercury concentrations in the Estuary (as indicated by accumulation in striped bass) have been relatively constant since the early 1970s (Hunt et al. 2008), but could quite plausibly increase, remain constant, or decrease in the next 30 years. Wetlands are often sites of methylmercury production, and restoration of wetlands in the Bay on a grand scale is now beginning, raising concern that methylmercury concentrations could increase across major portions of the Bay. However, methylmercury cycling is not yet well understood, and recent findings suggest that some wetlands actually trap methylmercury and remove it from circulation.

Concentrations of methylmercury in sport fish tissue represent a key regulatory target for this pollutant. The mercury TMDL for the Bay established a water quality objective for mercury based on concentrations in the five most commonly consumed fish species in the Bay (striped bass, California halibut, jacksmelt, white sturgeon, and white croaker). Concentrations in these five species therefore provide a reasonable basis for a methylmercury indicator for the Bay. The concentrations were compared to OEHHA thresholds, as described previously.

Data Source The methylmercury in sport fish indicator was calculated using data from the Regional Monitoring Program for Water Quality in the San Francisco Estuary (RMP) (www.sfei.org/rmp). The data are available from the RMP website (www.sfei.org/rmp/data). The RMP measures contaminant concentrations in Bay sport fish every three years. Monitoring began with a pilot study in 1994 (Fairey et al. 1997), and has continued to the present (Davis et al. 2002, Greenfield et al. 2005, Davis et al. 2006, Hunt et al. 2008, Davis et al. 2011).

The RMP collects sport fish from five popular fishing locations in the Bay (Figure 7). The monitoring is specifically directed at assessing trends in potential human exposure to contaminants in fish tissue. Sampling in Suisun Bay was attempted in the early years of the program, but was discontinued due to the low catch per unit sampling effort in that region, and the correspondingly low fishing pressure. The species targeted and the pollutant analyte list have varied slightly over the years. The five most commonly consumed species that are designated by the mercury water quality objective for the Bay (striped bass, California halibut, jacksmelt, white sturgeon, and white croaker) have been inconsistently sampled (Figure 2). In the most recent sampling in 2009, methylmercury was analyzed in striped bass, California halibut, and jacksmelt, but not white sturgeon or white croaker.

Methods and Calculations The sport fish methylmercury indicator (Figure 8) was calculated using whatever data for these species that were available for each sampling year. The RMP sampling targets specific size ranges of each species (Hunt et al. 2008) to control for variation of concentrations of methylmercury and other pollutants with fish size. Methylmercury concentrations in striped bass have been analyzed over the years in individual fish, making it possible to normalize the concentrations to fish length. Statistics for striped bass are therefore based on results normalized to a standard size of 60 cm, using methods described in Greenfield et al. (2005). The time series plots show the average concentration for each species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay.

Goals, Targets and Reference Conditions OEHHA has developed separate ATLs for methylmercury that apply to the most sensitive population (women of child-bearing age - 18-45 years - and children aged 1-17 years) and that apply to women over 45 years and men (Klasing and Brodberg 2008). The values for the most sensitive population are used in this report. The no consumption ATL for methylmercury is 0.44 ppm. The level below which OEHHA considers recommending consumption of up to three eight ounce servings per week is 0.07 ppm.

Results

In the most recent sampling year, the three species sampled (striped bass, California halibut, and jacksmelt) all had average concentrations between 0.07 and 0.44 ppm. Concentrations of the five indicator species have fluctuated over the years, but no trend over the 15-year period of record is evident for any species. Spatial and temporal trends within San Pablo Bay, Central Bay, and South Bay have been similar to those observed at the whole Bay scale. Striped bass are a particularly important indicator species for methylmercury because they are the most popular fish species consumed from the Bay and a time series for methylmercury in Bay-Delta striped bass dates back to 1970. Comparisons of recent striped bass data to data from 1970 also indicate no decline (Davis et al. 2011). Preliminary modeling included in the Mercury TMDL suggested that recovery would take more than 100 years. Our current conceptual understanding of methylmercury sources and cycling in the Bay also indicates that reducing concentrations of methylmercury in the Bay food web poses a considerable challenge that is likely to take many decades.

Overall, all of the methylmercury indicator species had average concentrations between the no consumption ATL of 0.44 ppm and the two serving per week ATL of 0.07 ppm; this corresponds to the “moderate concern” category in Table 1. Methylmercury concentrations in the Bay food web have not changed perceptibly over the past 40 years, and it is not anticipated that they will decline significantly in the next 30 years. The summary rating for methylmercury in Bay sport fish is therefore two stars (Figure 9).

2. PCBs in Sport Fish

Background and Rationale

Polychlorinated biphenyls (PCBs) are also in the class of pollutants considered to have the most severe impacts on Bay water quality because the entire Bay is considered impaired, and the degree of risk is above established thresholds of concern.

The term “polychlorinated biphenyl” refers to a group of hundreds of individual chemicals (“congeners”). Due to their resistance to electrical, thermal, and chemical processes, PCBs were used in a wide variety of applications (e.g., in electrical transformers and capacitors, vacuum pumps, hydraulic fluids, lubricants, inks, and as a plasticizer) from the time of their initial commercial production in 1929 (Brinkmann and de Kok, 1980). In the U.S. PCBs were sold as mixtures of congeners known as “Aroclors” with varying degrees of chlorine content. By the 1970s a growing appreciation of the toxicity of PCBs led to restrictions on their production and use. In 1979, a final PCB ban was implemented by USEPA, prohibiting the manufacture, processing, commercial distribution, and use of PCBs except in totally enclosed applications (Rice and O’Keefe, 1995). A significant amount of the world inventory of PCBs is still in place in industrial equipment (Rice and O’Keefe, 1995). Leakage from or improper handling of such equipment has led to PCB contamination of runoff from industrial areas. Other sources of PCBs to the Estuary are atmospheric deposition, effluents, and remobilization from sediment (Davis et al. 2007).

Like methylmercury, PCBs are highly persistent, bound to sediment particles, and widely distributed throughout the Bay and its watershed. PCBs reach high concentrations in humans and wildlife at the top of the food chain where they can cause developmental abnormalities and growth suppression, endocrine disruption, impairment of immune system function, and cancer. PCBs are another significant driver of the fish consumption advisory for the Bay (OEHHA 1994, Hunt et al. 2008). PCB concentrations in sport fish are above thresholds of concern for human health. There is also concern for the effects of PCBs on wildlife, including species like harbor seals (Thompson et al. 2007) and piscivorous birds (Adelsbach and Maurer 2007) at the top of the Bay food web and sensitive organisms such as young fish. General recovery of the Bay from PCB contamination is likely to take many decades because the rate of decline is slow and concentrations are so far above the threshold for concern. Due to concerns about PCB impacts, a PCBs TMDL for the Bay has been developed and incorporated into the Basin Plan (SFBRWQCB 2008a,b).

Concentrations of PCBs in sport fish tissue are the key regulatory target for this pollutant. The PCBs TMDL for the Bay (SFBRWQCB 2008a,b), approved by USEPA in 2010, established a fish tissue target for PCBs in the Bay for protection of both human health (and the fishing beneficial use) and wildlife (the preservation of rare and endangered species, estuarine habitat and wildlife habitat beneficial uses). The target applies to two commonly consumed fish species in the Bay that accumulate relatively high concentrations of PCBs: white croaker and shiner surfperch. Average concentrations for these two species therefore provide a reasonable basis for a PCB indicator for the Bay. Average concentrations were compared to OEHHA thresholds, as described previously.

Data Source The PCBs indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. The two key indicator species for PCBs have been sampled consistently over the years (Figure 10).

Methods and Calculations The sport fish PCBs indicator (Figure 10) is based on whatever data for shiner surfperch and white croaker were available for each sampling year. In the PCBs TMDL, comparison of these two species of fish to thresholds is considered to be protective and provide a margin of safety, because PCBs concentrations in these species are the highest of the fish species measured and sport recreational fishers likely consume a variety of fish species, including those with lower PCBs concentrations. The time series plots show the average concentration for each species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay. PCB concentrations expressed as the sum of all reported congeners were used in the evaluation. Values for congeners reported as below the limit of detection were set to zero.

Goals, Targets and Reference Conditions The no consumption ATL for PCBs is 120 ppb. The level below which OEHHA considers recommending consumption of up to three eight-ounce servings per week is 21 ppb.

Results

In the most recent sampling year, both of the PCB indicator species had average concentrations between 21 ppb and 120 ppb (Figure 10). The Bay-wide average for shiner surfperch in 2009 (118 ppb) was just below the 120 ppb threshold. The average for white croaker (51 ppb) was closer to the two serving ATL of 21 ppb.

No clear pattern of long-term decline in PCB concentrations has been evident in these species. Concentrations in white croaker in 2009 were the lowest observed since monitoring began in 1994. This does not, however, signal a decline in PCB contamination in the Bay. The principal reason for the lower average in 2009 was that the RMP switched from analyzing white croaker fillets with skin to analyzing white croaker fillets without skin. This change was made to achieve consistency with OEHHA advice on fish preparation and with how white croaker are processed in other programs in California, and to reduce variability associated with the difficulty of homogenizing skin. Another reason for the low average concentration in white croaker in 2009 was the unusually low average fat content of the croaker collected in 2009. PCBs and other organic contaminants accumulate in fat, so concentrations rise and fall with changing fat content. Concentrations in shiner surfperch in 2009 were also lower than in most other years, but the time series does not suggest a trend. The time series for shiner surfperch in San Pablo Bay, however, does suggest a decline from an average of 103 ppb in 1994 to 38 ppb in 2009. A regression of these data was significant (R^2=0.84). Continued sampling will help establish whether this represents an actual decline and not simply interannual variation.

Significant regional variation in PCBs in shiner surfperch was observed in 2009, and consistently over the 1994-2009 period. Average concentrations in 2009 in Central Bay (147 ppb) and South Bay (107 ppb) were higher than the average in San Pablo Bay (38 ppb). Similar differences were also observed in earlier rounds of sampling. White croaker did not show variation among regions.

One of the key PCB indicator species, shiner surfperch, had an average concentration in 2009 just below the no consumption ATL. Based on the data for shiner surfperch, the new safe eating guidelines for the Bay recommend no consumption of any surfperch species by anyone eating Bay fish. Given this determination by OEHHA, PCBs were placed in the "high concern" category. The Baywide average PCB concentration in shiner surfperch did not decline over the period 1994-2009. The Baywide average concentration in white croaker was lower in 2009, but this was a function of low lipid and a shift to analyzing samples without skin. The model used in the PCB TMDL to forecast recovery (Davis et al. 2007) indicates that declines sufficient to bring fish concentrations down below 21 ppb are likely to take more than 30 years, placing PCBs in the "rapid progress unlikely" category. The summary rating for PCBs in Bay sport fish is therefore one star.

3. Dioxins in Sport Fish

Background and Rationale

Dioxins (including chlorinated dibenzodioxins and dibenzofurans) are a third member of the class of pollutants considered to have the most severe impacts on Bay water quality because the entire Bay is above thresholds for concern, and the degree of impairment is well above those thresholds (Connor et al. 2004a).

Dioxins have many similarities to PCBs. They are highly persistent, strongly associated with sediment particles, and widely distributed throughout the Bay and its watershed. Dioxins also reach high concentrations in humans and wildlife at the top of the food chain. The human and wildlife health risks of dioxins are similar to those for PCBs. Dioxins have not received as much attention from water quality managers because there are no large individual sources in the Bay Area and concentrations in the Bay are among the lowest measured across the U.S. Nevertheless, concentrations in sport fish are well above the threshold for concern and the entire Bay is included on the 303(d) List. Dioxins are similar to PCBs in their persistence and distribution throughout the Bay and its watershed, and are unlikely to decline significantly in the next 20 years.

Concentrations of dioxins in sport fish tissue are the key regulatory indicator for this pollutant. Connor et al. (2004a) discussed screening values and impairment relative to those values. The San Francisco Bay Regional Water Quality Control Board (Water Board) has not established a target for dioxins. A TMDL for dioxins is currently in the early development stage. In the absence of a Water Board target, a screening value for use in this report was calculated using the same parameters for consumption rate and risk that were employed in the PCBs TMDL. White croaker is the species that has been monitored for dioxins in Bay fish – the dioxins index is therefore based on data for this species.

Data Source The dioxins indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish index. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. White croaker have been sampled consistently over the years (Figure 11). Shiner surfperch have also been sampled intermittently.

Methods and Calculations The dioxins in sport fish index was calculated for each year of RMP monitoring. The time series plot shows the distribution of the data for each year sampled. Consistent with the evaluation scheme described under "Background and Rationale," the distribution is described with percentiles (5^{th}, 25^{th}, 50^{th}, 75^{th}, 95^{th}). Dioxins concentrations expressed as the sum of the dioxin toxic equivalents (TEQs) were calculated for comparison to the screening value, following USEPA guidance (USEPA 2000). TEQs express the potency of a mixture of dioxin-like compounds relative to the potency of 2,3,7,8-TCDD, the most toxic dioxin congener. The sum of TEQs for all of the congeners is the overall measure of the dioxin-like potency of a sample. Values for congeners reported as below the limit of detection were set to zero.

Goals, Targets, and Reference Conditions The calculated screening value to protect human health is a concentration of 0.14 pg/g wet weight in the tissue of white croaker. The same size class specified in the PCBs TMDL for white croaker (20 to 30 cm in length) was used. Comparison of white croaker and shiner surfperch data to the screening value is a conservative approach because these species are likely to have the highest concentrations among the species that are popular for consumption, and anglers likely consume a variety of fish species, including species with lower concentrations.

This screening value represents the maximum level that is considered to be safe for people consuming Bay fish at a rate less than the 95^{th} percentile rate (32 g/day, or 8 ounces per week) for all Bay fish consumers (Connor et al. 2004a).

Results

Nearly all of the white croaker and shiner surfperch samples analyzed since 1994 have been higher than the dioxin TEQ screening value of 0.14 parts per trillion (Figure 11). Median dioxin TEQ concentrations in white croaker have been over ten times higher than the target. Without no consumption ATLs for dioxins from OEHHA, however, there is an insufficient basis for determining that dioxins should be categorized as a high concern. Therefore dioxins were placed in the "moderate concern" category. No pattern of long-term decline has been evident in the dioxin time series, and there is no conceptual reason to expect a rapid decline. The overall assessment for dioxins was therefore two stars.

3. Dieldrin in Sport Fish

Background and Rationale Dieldrin is an organochlorine insecticide that was widely used in the U.S. from 1950 to 1974, primarily on termites and other soil-dwelling insects, as a wood preservative, in moth-proofing clothing and carpets, and on cotton, corn, and citrus crops (USEPA, 1995a). Restrictions on dieldrin use began in 1974. Most uses in the U.S. were banned in 1985. Dieldrin use for underground termite control continued until voluntarily canceled by industry in 1987 (USEPA, 1995a). Dieldrin and two other organochlorine pesticides (DDTs and chlordanes) are often referred to as "legacy pesticides" (Connor et al. 2004b).

Dieldrin and the other legacy pesticides have similar properties, and are also similar in many ways to PCBs and dioxins. They are highly persistent, strongly associated with sediment particles, widely distributed throughout the Bay and its watershed, and reach high concentrations in humans and wildlife at the top of the food chain. The human and wildlife health risks of the legacy pesticides are similar to those for PCBs. However, concentrations of the legacy pesticides in sport fish are not as elevated relative to their thresholds for concern.

Concentrations of dieldrin and the other legacy pesticides in sport fish tissue are the key indicators for these pollutants. The San Francisco Bay Regional Water Quality Control Board (Water Board) has not established targets for the legacy pesticides. A TMDL for legacy pesticides is currently in the early development stage. In the absence of a Water Board target, the same indicator species used for the PCBs TMDL (white croaker and shiner surfperch) were used.

Data Source The dieldrin indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. White croaker and shiner surfperch, the key indicator species for the legacy pesticides, have been sampled consistently over the years (Figure 12).

Methods and Calculations The sport fish dieldrin indicator (Figure 12) is based on available data for shiner surfperch and white croaker each sampling year. As in the PCBs TMDL, comparison of these two species of fish to thresholds is protective and provides a margin of safety, because dieldrin concentrations in these species are the highest of the fish species measured and sport recreational fishers likely consume a variety of fish species, including those with lower dieldrin concentrations. The time series plots show the average concentration for each species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay.

Goals, Targets and Reference Conditions The no consumption ATL for dieldrin is 46 ppb. The level below which OEHHA considers recommending consumption of up to three eight ounce servings per week (the two serving ATL) is 15 ppb.

Results

In the most recent sampling year, both of the dieldrin indicator species had average concentrations well below the two serving ATL of 15 ppb (Figure 12). The Bay-wide averages for shiner surfperch and white croaker in 2009 were 1.1 ppb and 0.5 ppb, respectively.

No clear pattern of long-term decline in dieldrin concentrations has been evident in these species. Concentrations in white croaker in 2009 were the lowest observed since monitoring began in 1994, but this was due to the switch to analyzing white croaker fillets without skin (discussed further in the PCBs section above) and the unusually low average fat content of the croaker collected in 2009. Concentrations in shiner surfperch in 2009 were moderate compared to past years, and the time series does not suggest a trend. Dieldrin concentrations in mussels in the Bay declined sharply in the 1980s (Gunther et al. 1999), but have not declined appreciably in either sport fish or bivalves over the past 20 years (Davis et al. 2007).

No distinct differences among the three regions sampled were evident, although concentrations in the South Bay were more variable. The time series for shiner surfperch in San Pablo Bay suggests a possible downward trend.

The two dieldrin indicator species had Baywide average concentrations well below the two serving per week ATL of 15 ppb, corresponding to the “goal attained” category in Figure 9. Dieldrin concentrations can be expected to continue to gradually decline. The summary rating for dieldrin in Bay sport fish is therefore five stars.

4. DDTs in Sport Fish

Background and Rationale DDT is an organochlorine insecticide that was used very extensively in home and agricultural applications in the U.S. beginning in the late 1940s and continuing in the U.S. until the end of 1972, when all uses, except emergency public health uses, were canceled (USEPA 1995). The primary sources of DDT to the Bay are probably continuing transport of contaminated soils and sediments from urban and agricultural sites of historic use, and remobilization of residues from Bay sediments. The terms DDT or DDTs are often used to refer to a family of isomers (i.e., p,p’-DDT and o,p’-DDT) and their breakdown products (p,p’-DDE, o,p’-DDE, p,p’-DDD, and p,p’-DDD). DDT data are often expressed as the sum of these six components, and this approach is recommended by USEPA (2000). DDT and its metabolites DDE and DDD are neurotoxic and are also classified by USEPA as probable human carcinogens (USEPA 1995).

Concentrations of DDTs in sport fish tissue are the key impairment indicator for this pollutant. Other considerations regarding thresholds were described above in the Dieldrin section.

Data Source The DDTs indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. White croaker and shiner surfperch, the key indicator species for the legacy pesticides, have been sampled consistently over the years (Figure 13).

Methods and Calculations The sport fish DDTs indicator (Figure 13) is based on available data for shiner surfperch and white croaker each sampling year. As in the PCBs TMDL, comparison of these two species of fish to thresholds is protective and provides a margin of safety, because DDT concentrations in these species are the highest of the fish species measured and sport recreational fishers likely consume a variety of fish species, including those with lower DDT concentrations. The time series plots show the average concentration for each species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay.

Goals, Targets and Reference Conditions The no consumption ATL for DDTs is 2100 ppb. The level below which OEHHA considers recommending consumption of up to three eight ounce servings per week (the two serving ATL) is 520 ppb.

Results

In the most recent sampling year, both of the DDT indicator species had average concentrations well below the two serving ATL of 520 ppb (Figure 13). The Bay-wide averages for shiner surfperch and white croaker in 2009 were 22 ppb and 9 ppb, respectively.

No clear pattern of long-term decline in DDT concentrations has been evident in these species. Concentrations in white croaker in 2009 were the lowest observed since monitoring began in 1994, but this was due to the switch to analyzing white croaker fillets without skin (discussed further in the PCBs section above) and the unusually low average fat content of the croaker collected in 2009. Concentrations in shiner surfperch in 2003, 2006, and 2009 were low relative to past years, possibly suggesting a trend. DDT concentrations in the Bay have declined since the ban in 1972 (Davis et al. 2007), and are expected to continue on a downward trajectory.

DDT concentrations and trends were similar in the three regions sampled. The time series for all three regions indicate relatively low concentrations from 2003-2009; continued monitoring will determine whether this is actually represents a decline.

The two DDT indicator species had Baywide average concentrations well below the two serving ATL of 520 ppb, corresponding to the “goal attained” category in Figure 9. DDT concentrations can be expected to continue to gradually decline. The summary rating for DDTs in Bay sport fish is therefore five stars.

5. Chlordanes in Sport Fish

Background and Rationale Chlordane is another organochlorine insecticide that was used extensively in home and agricultural applications (including corn, grapes, and other crops) in the U.S. for the control of termites and many other insects (USEPA 1995). Like PCB, chlordane is a term that represents a group of a large number (140) of individual compounds (Dearth and Hites 1991). Restrictions on chlordane use began in 1978, and domestic sales and production ceased in 1988 (USEPA 1995). As for DDT, the primary sources of chlordane to the Bay are probably continuing transport of soils and sediments from urban and agricultural sites of historic use and remobilization of residues from Bay sediments.

Chlordane data are usually expressed as the sum of several of the five most abundant and persistent components and metabolites of the technical chlordane mixture. Chlordane is neurotoxic and is classified by USEPA as a probable human carcinogen (USEPA 2000). Like PCBs and DDT, chlordane compounds are very persistent in the environment, resistant to metabolism, have a strong affinity for lipid, and biomagnify in aquatic food webs (Suedel et al. 1994).

Concentrations of chlordanes in sport fish tissue are the key impairment indicator for this pollutant. Other considerations regarding thresholds were described above in the Dieldrin section.

Data Source The chlordanes indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. White croaker and shiner surfperch, the key indicator species for the legacy pesticides, have been sampled consistently over the years (Figure 14).

Methods and Calculations The sport fish chlordanes indicator (Figure 14) is based on available data for shiner surfperch and white croaker each sampling year. As in the PCBs TMDL, comparison of these two species of fish to thresholds is protective and provides a margin of safety, because chlordane concentrations in these species are the highest of the fish species measured and sport recreational fishers likely consume a variety of fish species, including those with lower chlordane concentrations. The time series plots show the average concentration for each species for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay.

Goals, Targets and Reference Conditions The no consumption ATL for chlordanes is 560 ppb. The level below which OEHHA considers recommending consumption of up to three eight ounce servings per week (the two serving ATL) is 190 ppb.

Results

In the most recent sampling year, both of the chlordane indicator species had average concentrations well below the two serving ATL of 190 ppb (Figure 14). The Bay-wide averages for shiner surfperch and white croaker in 2009 were 7 ppb and 2 ppb, respectively.

No clear pattern of long-term decline in chlordane concentrations has been evident in these species. Concentrations in white croaker in 2009 were the lowest observed since monitoring began in 1994, but this was due to the switch to analyzing white croaker fillets without skin (discussed further in the PCBs section above) and the unusually low average fat content of the croaker collected in 2009. Chlordane concentrations in shiner surfperch in 2009 were similar to past years. Other Bay species have generally declined since the ban in 1988 (Davis et al. 2007), and chlordanes generally are expected to continue on a gradual downward trajectory.

The two chlordane indicator species had Baywide average concentrations well below the two serving ATL of 190 ppb, corresponding to the “goal attained” category in Figure 9. Chlordane concentrations can be expected to continue to gradually decline. The summary rating for chlordanes in Bay sport fish is therefore five stars.

6. Selenium in Sport Fish

Background and Rationale San Francisco Bay has been on the 303(d) List since 1998 for selenium because bioaccumulation of this element has led to recurring health advisories for local hunters against consumption of diving ducks. Moreover, elevated selenium concentrations found in biota often exceed levels that can cause potential reproductive impacts in white sturgeon and are often higher than levels considered safe for fish and other wildlife species in the Estuary. Sources and pathways leading to the possible impairment in northern and southern segments of the Bay differ significantly and therefore a separate approach to addressing the problem in these segments is being followed. Thus, a TMDL is being developed for the North San Francisco Bay segments only, which include a portion of the Sacramento/San Joaquin Delta, Suisun Bay, Carquinez Strait, San Pablo Bay, and Central Bay. This North Bay Selenium TMDL project was initiated in 2007 to assess the current state of impairment in the North Bay, identify pathways for bioaccumulation, enhance understanding of the relationship between sources of selenium and fish and wildlife exposure, and establish site-specific water quality targets protective of aquatic biota. In developing the TMDL, the Water Board, with support from stakeholders, is conducting a series of analysis to refine understanding of the behavior of selenium in the Estuary that will help formulate a strategy for attaining water quality standards. A Preliminary TMDL Project Report was published in January 2011 (SFBRWQCB 2011). As part of this information gathering effort, the RMP measured selenium concentrations in all eight sport fish species sampled in 2009 (Davis et al. 2011).

Concentrations of selenium in sport fish tissue are an impairment indicator of secondary importance for this pollutant. Risks to aquatic life are greater and are the impetus for the TMDL. The sport fish species of greatest concern is white sturgeon, which accumulates higher selenium concentrations than other sport fish due to its preference for *Corbula*, an abundant clam species that has a strong tendency to accumulate selenium (Stewart et al. 2004).

Data Source The selenium indicator was calculated using data from the same RMP sport fish monitoring program described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data). Additional details on this sampling were provided in the methylmercury section. White sturgeon, the key indicator species for selenium, has been sampled consistently over the years (Figure 15).

Methods and Calculations The sport fish selenium indicator (Figure 15) is based on available data for white sturgeon for each sampling year. Focusing on this species is protective and provides a margin of safety because it has the highest selenium concentrations among the fish species measured and sport recreational fishers likely consume a variety of fish species, including those with lower selenium concentrations. The time series plots show the average concentration for each year sampled. Data are presented for the Bay as a whole and for the three segments of the Bay that have consistently been sampled over the years: San Pablo Bay, Central Bay, and South Bay.

Goals, Targets and Reference Conditions The no consumption ATL for selenium is 15 ppm. The level below which OEHHA considers recommending consumption of up to three eight-ounce servings per week (the two serving ATL) is 2.5 ppm.

Results

In the most recent sampling year, white sturgeon had a Baywide average concentration (1.4 ppm) well below the two serving ATL of 2.5 ppm (Figure 15). Concentrations measured in seven other sport fish species in the Bay in 2009 were much lower than in white sturgeon (Davis et al. 2011). No clear pattern of long-term decline in selenium concentrations has been evident in Bay white sturgeon. Recent results for *Corbula* in the North Bay indicate declines (Stewart, USGS, personal communication). No differences among the three Bay regions sampled were evident.

White sturgeon had a Baywide average concentration well below the two serving per week ATL of 2.5 ppm, corresponding to the "goal attained" category in Figure 15. The summary rating for selenium in Bay sport fish is therefore five stars.

7. PBDEs in Sport Fish

PBDEs, a class of bromine-containing flame retardants that was practically unheard of in the early 1990s, increased rapidly in the Bay food web through the 1990s and are now pollutants of concern. They have not been placed on the 303(d) List, but information on them is lacking and they are being studied through the RMP to better understand their spatial distribution, temporal trends, and the risks they pose to wildlife and humans. The California Legislature has banned the use of two types of PBDE mixtures ("penta" and "octa") in 2006, but one mixture remains in use ("deca").

In May 2011, OEHHA published thresholds for PBDEs (Klasing and Brodberg 2011). PBDE concentrations in all samples were far below the lowest OEHHA threshold (the 100 ppb 2 serving ATL), indicating that PBDE concentrations in Bay sport fish are not a concern with regard to human health. The Baywide average for shiner surfperch, the species with the highest concentrations in 2009, was 8 ppb.

The Baywide average for shiner surfperch in 2009 was lower than the averages observed in 2003 and 2006. A decline might be anticipated in response to the bans on the penta and octa mixes, but how quickly the decline would occur as the overall inventory in the watersheds is reduced is unknown. Given the short time series available and a potential lack of comparability due to the switch to a new method in 2009, it is unclear whether the lower concentrations in 2009 are a sign of a real decline or not. Continued monitoring of sport fish and other matrices in the Bay will be needed to determine whether the bans of the penta and octa mixtures are indeed reducing PBDE concentrations in the Bay food web.

POLLUTANTS WITHOUT APPROPRIATE THRESHOLDS

Contaminants of Emerging Concern

In addition to the pollutants discussed above, there are thousands of other chemicals used by society, including pesticides, industrial chemicals, and chemicals in consumer products, and many of these make their way from our homes, businesses, and watersheds into the Bay. Due to inadequate screening of the hazards of these chemicals, some may accumulate in the Bay food web and cause exposure in people who consume Bay fish. As understanding advances, some of these contaminants emerge as posing risks to the health of humans and wildlife.

The RMP monitors contaminants of emerging concern that pose the greatest known threats to water quality. One important class of emerging contaminants monitored in 2009 was perfluorinated chemicals (PFCs). PFCs have been used extensively over the last 50 years in a variety of products including textiles treated with stain-repellents, fire-fighting foams, refrigerants, and coatings for paper used in contact with food products. As a result of their chemical stability and widespread use, PFCs such as perfluorooctane sulfonate (PFOS) have been detected in the environment. PFOS and related PFCs have been associated with a variety of toxic effects including mortality, carcinogenity, and abnormal development. PFCs have been detected in sport fish fillets in other studies. Sampling has been fairly extensive in Minnesota, where concentrations have been high enough that the state has established thresholds for issuing consumption guidelines (Delinsky et al. 2010). Neither OEHHA nor the Water Board have developed thresholds for evaluating the risks to humans from consumption of contaminated sport fish from San Francisco Bay. In 2009 only four samples had detectable PFOS concentrations. The highest concentration was 18 ppb in a leopard shark composite.

Other chemicals among the thousands in commerce may also be entering the Bay, accumulating in the Bay food web, and leading to human exposure and risk through consumption of Bay sport fish. Past experience has shown that the Bay is a sensitive ecosystem that is very slow to recover from contamination by persistent pollutants. Cleaning up this type of contamination is very challenging and very costly. Given these lessons learned, the RMP has placed a priority on early identification of emerging water quality threats so they can be addressed before they affect sensitive species or are added to the pollutant legacy that we leave for future generations. However, these monitoring efforts to protect Bay water quality are severely hampered by the lack of information on the chemicals present in commercial products, their movement in the environment, and their toxicity. Screening of chemical properties and toxicity is currently required for many chemicals, but this could be improved. Furthermore, much of the information that does exist is not made readily available to the public. Measuring chemicals in environmental samples at the low concentrations that can cause toxicity is challenging and requires customized analytical chemistry methods. When the identities of the potentially problematic chemicals are not known, it is exceptionally challenging. Ultimately, the reduction of use of toxic chemicals in products is the ideal way to prevent environmental contamination.

IS THE BAY SAFE FOR SWIMMING?

Background and Rationale

Recreation, including water sports, provides numerous physical, social, and psychological benefits to participants and spectators. Every year countless Bay Area residents and visitors are drawn to Bay waters to engage in water contact recreation. Swimming, surfing, windsurfing, kite boarding, and stand-up paddling all have their enthusiasts. Water contact sports in the Bay carry numerous inherent dangers including drowning, hypothermia, danger of collision with vessel traffic, exposure to marine life (jellyfish stings, parasites, sea lion bites, etc.), and waterborne diseases or infection from the ingestion of Bay water contaminated with fecal material. With the exception of information on cercarial dermatitis or swimmer's itch caused by parasites (Brant, et al. 2010), morbidity rates associated with water-contact recreation in the Bay are lacking. Exposure to water contaminated by fecal matter can result in numerous diseases and illnesses including gastro-intestinal illnesses, respiratory illness, skin rashes and infections, and infections of the ears, nose, and throat. In order to transmit infectious disease the infectious agent must be present, and in sufficient quantities to produce the infection and/or disease, and the susceptible individual must come into contact with the pathogen (Cooper 1991). Although a wide variety of pathogens have been identified in raw wastewater relatively few types appear to be responsible for the majority of waterborne illnesses caused by pathogens of wastewater origin (Soller, et al. 2010a). Further, and most importantly, reliable and effective wastewater treatment occurs consistent with State and Federal standards throughout the San Francisco Bay Area.

To protect beach users from exposure to fecal contamination California has adopted standards developed for high use beaches and applies them during the prime beach season from April through October at beaches with more than 50,000 annual visitors that are adjacent to a storm drain that flows in the summer; these requirements are only mandatory in years that the legislature has appropriated monies sufficient to fund the monitoring. County Public Health and other agencies routinely monitor fecal indicator bacteria (FIB) concentrations at Bay beaches where water contact recreation is common and provide warnings to the public when concentrations exceed the standards (Table 1). FIB are enteric bacteria common to the digestive systems of mammals and birds and are indicators of fecal contamination. While not generally pathogenic themselves, FIB are used because they correlate well with the incidence of human illness in epidemiology studies at recreational beaches and can be enumerated more quickly and cost effectively than can pathogens directly.

Heal the Bay, a Santa Monica-based non-profit, provides comprehensive evaluations of over 400 California bathing beaches in both Annual and Summer Beach Report Cards as a guide to aid beach users' decisions concerning water contact recreation (Heal the Bay 2011). Higher grades are considered to represent less health risk to swimmers than are lower grades. The Heal the Bay grades for Bay beaches were used as the primary indicator of whether the Bay is safe for swimming.

The frequency at which Bay beaches are posted or closed is another valuable indicator of whether the Bay is safe for swimming. This additional metric was also examined and is discussed below, along with other supplemental information relating to beach water quality.

Data Source Whether the Bay is safe for swimming was assessed using the FIB monitoring data from the counties, described above. Bay county public health and other agencies monitor bacteria at 30 Bay beaches. These agencies collect and analyze samples, then post the necessary health warnings to protect public health. Data from these agencies are used to generate the Heal the Bay report card grades.

Methods and Calculations Heal the Bay (2011) presents the methods used to generate the grades that appear in the statewide annual beach report card. The grading system takes into consideration the magnitude and frequency of an exceedance above indicator thresholds over the course of the specified time period. Those beaches that exceed multiple indicator thresholds (if applicable) in a given time period receive lower grades than those beaches that exceeded just one indicator threshold. Water quality typically drops dramatically during and immediately after a rainstorm but often rebounds to its previous level within a few days. For this reason, year-round wet weather data throughout California are analyzed separately in order to avoid artificially lowering a location's year-round grade and to provide better understanding of statewide beach water quality impacts. Wet weather data are comprised of samples collected during or within three days following the cessation of a rainstorm. Heal the Bay's annual and weekly Beach Report Cards utilize a definition of a 'significant rainstorm' as precipitation greater than or equal to one-tenth of an inch (>0.1").

Goals, Targets and Reference Conditions California standards for fecal indicator bacteria established by the Department of Public Health are shown in Table 2.

Results

Overall, the monitoring data and resulting grades (Table 3) indicate that most Bay beaches are safe for swimming in the summer, but that bacterial contamination is a concern at a few beaches in the summer, and at most beaches in wet weather.

Data for the summer beach season in 2010 are available for 27 of the 30 beaches that have been monitored over the past five years. In 2010, 19 of the 27 monitored beaches received an A or A+ grade, reflecting minimal exceedance of standards. Ten of these beaches received an A+: Coyote Point, Alameda Point South, Bath House, Windsurf Corner, Sunset Road, Shoreline Drive, Hyde Street Pier, Crissy Field East, Crissy Field West, and Schoonmaker Beach. Most Bay beaches, therefore, are quite safe for swimming in the summer.

Seven of the 27 beaches monitored in the summer in 2010 had grades of B or lower, indicating varying degrees of exceedance of bacteria standards. Keller Beach was the one beach receiving an F. Five beaches received a D, including one in Contra Costa County, two in San Mateo County, and two in San Francisco County. These low grades indicate an increased risk of illness or infection.

Overall, the average grade for the 27 beaches monitored from April-October was a B (Table 3).

During wet weather, which mostly occurs from November-March, water contact recreation is less popular but is still enjoyed by a significant number of Bay Area residents. Bacteria concentrations are considerably higher in wet weather making the Bay less safe for swimming. This pattern is evident in Heal the Bay report card grades for wet weather. In wet weather, only five of 22 beaches with data received an A. Six of these 22 beaches, on the other hand, received an F. The average grade for these beaches in wet weather was a C+ (Table 3).

Additional Discussion

Beach Closure Data

The frequency of beach closures is another informative metric for evaluating how safe the Bay is for swimming. Based upon the number of days beaches were closed or posted with advisories warning against water contact recreation, Bay beaches were open 80% to 100% of the time during the prime beach season of April through October from 2006 through 2010 (Figures 16-20). Monitoring data from the City and County of San Francisco, required to monitor and apply the high-use standards year-round by NPDES permit, illustrate a pattern found throughout the Bay: bacteria water quality is generally very good during dry weather, but tends to degrade during wet weather (Figure 21).

For beach users trying to decide whether or not to engage in water contact recreation at a particular beach, the recent monitoring data provided by some Bay counties (Table 1) along with the Heal the Bay on-line grades (Table 1) represent the easiest, most robust, and consistent means of evaluating beach water quality. The Heal the Bay grading system incorporates established water quality thresholds and has been endorsed by regulators and beach managers. Beach users concerned about exposure to elevated FIB should heed beach closures and advisory warnings and avoid water contact recreation during and for up to 72 hours after rainstorms, especially at beaches with flowing creeks, storm drains, or combined sewer discharges.

FIB Sources

The sources of FIB at specific sites are often unknown, but potentially include fecal contamination from humans (leaky sewer systems, sanitary sewer overflows, storm water, combined sewer discharges during wet weather, septic tanks, illegal boat discharges, and babies and other people defecating directly at the beach); fecal contamination from non-humans (dogs and other pets, wildlife such as birds, seals, sea lions, deer, etc., and cattle, horses and other agrarian land uses); and non-enteric environmental FIB.

Environmental FIB are a relatively new discovery in California. Recent studies have shown that marine sands in California can serve as a reservoir of FIB that can contribute to water column concentrations (Ferguson, et al. 2005, Lee, et al. 2006, Yamahara, et al. 2007, 2009, Halliday, et al. 2010). Other sources of environmental FIB include beach wrack, salt marshes, and upland soils (Imamura, et al. in press, Grant, et al. 2001, Whitman, et al. 2006). One study has found an increased risk of enteric illness with sand contact at marine beaches (Heaney, et al. 2009). Several recent studies have used quantitative microbial risk assessments to estimate the health risks to humans from exposure to recreational waters contaminated by non-human fecal sources

(EPA 2010a, Schoen and Ashbolt 2010, Soller, et al. 2010b). They have found that the risk of illness ranged from similar for cattle impacted waters to substantially lower for chicken, pig, and seagull impacted waters than the risk from exposure to human impacted waters based upon current EPA (1986) Recreational Water Quality Criteria. Similar studies are needed for non-human sources likely important in the Bay such as dogs and marine mammals.

Currently used culture-based methods for identifying and enumerating FIB do not allow differentiation of the various possible sources. In addition, culture-based methods require 18 - 24 hours before results are available so warnings to the swimming public can only happen after elevated FIB concentrations occur. Moreover, FIB concentrations in recreational waters are highly variable from year to year as well as spatially, during time of day, and tidal cycle (Boehm and Weisberg 2005, Boehm 2007, Boehm, et al. 2009).

Best Management Practices

Best management practices for beach managers should include sanitary surveys to identify and mitigate contamination sources where possible. Low impact design installations may be possible at some sites to retain and treat storm water before it reaches beaches. Diversion of storm water away from bathing beaches where possible may provide another solution. Repair and replacement of defective and aging sanitary sewer systems will be necessary in many instances before human fecal sources are considered controlled. Sanitary surveys should also inform monitoring plans as should site-specific knowledge of how FIB concentrations vary diurnally, are affected by tidal forcing, and by photoperiod.

Future Directions

Beach monitoring is transitioning from the traditional culture-based methods for determining FIB to more rapid molecular methods that will allow same-day notification to the public. The EPA is currently involving scientists and stakeholders in a process to produce revised or new standards for recreational waters by October 2012. Possible changes could include adding new, human-specific indicators and rapid methods for detecting FIB. The rapid methods use quantitative polymerase chain reaction (qPCR) to amplify and identify genetic material present in the sample and thus also have the additional potential of source identification. The rapid techniques are not without logistical and technical constraints however.

Logistically, the qPCR sample analysis time of 2 to 4 hours can only begin once samples have arrived in the laboratory. Sample preparation, QA/QC, and data analysis are additional steps that add time to delivery of results to the public. Sample collection can take several hours, especially for large counties with monitoring sites on both ocean and bay coasts. Sample collection at first light would help decrease the time to notification, but that raises potential safety concerns for sample collectors and would not account for inactivation of FIB by sunlight (Boehm, et al. 2009) nor FIB contributions from the public directly at the beach. In addition, notification by late morning or mid-day, while an improvement, will not serve many local users who enjoy early morning water recreation. A demonstration project applying qPCR to beach monitoring in southern California successfully achieved notification by mid-day for a period of eight weeks (Griffith and Weisberg 2011). They overcame many of the logistical problems by limiting the number of sites to which the rapid methods were applied, modifying the sample collection routine for those sites, prudent method selection and automation of data analysis, and augmenting normal beach posting procedures with social media, the internet, and remotely operated electronic message boards at the test beaches. Interestingly, they recognized that adoption of rapid methods will create expectations of more frequent sampling.

Research and development will no doubt overcome the technological constraints of qPCR as applied to beach monitoring. One of the most important is the current inability of the method to distinguish between viable and non-viable microbes. This makes the technique problematic at beaches influenced by disinfected wastewater effluent and for FIB inactivated by sunlight. Methods are being developed to overcome this limitation, but they will necessitate longer processing and analysis times. Despite these constraints, qPCR represents a valuable new tool in beach monitoring, particularly because of the reduced analysis time and source identification potential.

Another probable outcome of the EPA revision of water quality standards for recreational waters is the incorporation of modeling into beach advisory decisions, at least at some beaches. One such model is the USEPA's Virtual Beach 2.0 which uses multiple linear regressions (USEPA 2011). A partial least squares regression model has been demonstrated effective at a beach in southern California (Hou et al. 2006).

Genetic microarrays are another tool that are just beginning to be applied to shoreline monitoring. Microarrays use genetic probes to match gene sequences present in the sample. The main advantage of a microarray is the ability to detect thousands of microbial taxa in a single sample. They can be customized to look for specific taxa, including strains of a single taxon (e.g., virulent and non-virulent strains of *E. coli*), and it should be possible to configure hybrid microarrays that detect both bacteria and viruses simultaneously so that pathogens can be detected directly rather than through indicators. Microarrays are not rapid technology (PCR is a preliminary step) and the bioinformatics involved in assessing the results are formidable, but tracking a larger proportion of the microbial community present at a beach may allow beach managers to customize warnings and advisories to the specific risks at each site. For example, the loading and seasonality of non-enteric environmental FIB on beaches can be significantly different from those of fecal pollution events (USEPA 2010b).

Adoption of new methods and new indicators will increase protection of public health by providing same day notification in many cases and identification of FIB sources, as well as targeting human-specific indicators and eventually pathogens directly. Sanitary surveys in conjunction with source identification and determining the timing and seasonality of environmental FIB vs. fecal contamination events, perhaps in combination with robust modeling, will allow beach managers to properly balance public health protection and access to recreational opportunity on a site by site basis.

Acknowledgments

Peer review of the water quality report card was provided by members of the RMP Technical Review Committee, the RMP Steering Committee, Water Board staff, and others on the RMP distribution list. Robert Cooper (BioVir Laboratories), Dean Peterson (San Mateo County Environmental Health Division), Alexandria Boehm (Stanford University), Steve Weisberg (Southern California Coastal Water Research Project), Mark Gold and Amanda Griesbach (Heal the Bay), John S. Roddy (City and County of San Francisco), Tom Hall (EOA, Inc.), and Al Hom and Cynthia Bartus Jepsen (Alameda County Department of Environmental Health), all provided comments and/or references that improved this report. In addition Adam Olivieri and Robert Cooper provided valuable guidance on exposure and applicability of standards. Shoreline monitoring data for the City and County of San Francisco were obtained directly from the San Francisco Public Utilities Commission. At the recommendation of the SWRCB's Northern California Beach Monitoring Workgroup, data for San Mateo, Marin, Alameda, and Contra Costa counties were obtained from the State database and provided by Larry Cooper, Southern California Coastal Water Research Project. Hal MacLean, East Bay Regional Park District, provided posting and additional bacteria data for Alameda and Contra Costa Counties. Rainfall data were obtained from Golden Gate Weather Services through the courtesy of Jan Null and are used here with permission. Richard Looker, San Francisco Bay Regional Water Quality Control Board, generously computed forward looking 30-day geometric means of Enterococci data using a program he created in Perl.

REFERENCES

Adelsbach, T.L. and T. Maurer. 2007. Dioxin Toxic Equivalents, PCBs, and PBDEs in Eggs of Avian Wildlife of San Francisco Bay. U.S. Fish and Wildlife Service, Sacramento, CA.

Boehm, A.B. and S.B. Weisberg 2005. Tidal forcing of enterococci at marine recreational beaches at fortnightly and semidiurnal frequencies. Environ. Sci. Technol. 39(15):5575-5583.

Boehm, A.B. 2007. Enteococci concentrations in diverse coastal environments exhibit extreme variability. Environ. Sci. Technol. 41(24):8227-8232.

Boehm, A.B., K.M. Yamahara, D.C. Love, B.M. Peterson, K. McNeil, and K.L. Nelson 2009. Covariation and photoinactivation of traditional and novel indicator organisms and human viruses at a sewage-impacted marine beach. Environ. Sci. Technol. 43(21):8046-8052.

Brant, S.V., A.N. Cohen, D. James, L. Hui, A. Hom, and E.S. Loker 2010. Cercarial dermatitis transmitted by exotic marine snails. Emerging Infectious Diseases 16(9):1357-1365.

Connor, M., D. Yee, J.A. Davis, and C. Werme. 2004a. Dioxins in San Francisco Bay Conceptual Model/Impairment Assessment. SFEI Contribution 309. Prepared for the Clean Estuary Partnership.

Connor, M., J.A. Davis, J. Leatherbarrow, and C. Werme. 2004b. Legacy Pesticides in San Francisco Bay Conceptual Model/Impairment Assessment. SFEI Contribution 313. Prepared for the Clean Estuary Partnership.

Cooper, R.C. 1991. Public health concerns in wastewater reuse. Water Science & Technology 24(9):55-65.

Davis, J.A., M.D. May, B.K. Greenield, R. Fairey, C. Roberts, G. Ichikawa, M.S. Stoelting, J.S. Becker, and R.S. Tjeerdema. 2002. Contaminant concentrations in sport fish from San Francisco Bay, 1997. Mar. Pollut. Bulletin. 44: 1117-1129.

Davis, J.A., J.A. Hunt, B.K. Greenfield, R. Fairey, M. Sigala, D.B. Crane, K. Regalado, and A. Bonnema. 2006. Contaminant Concentrations in Fish from San Francisco Bay, 2003. SFEI Contribution #432. San Francisco Estuary Institute, Oakland, CA.

Davis, J.A., F. Hetzel, and J.J. Oram. 2007a. Polychlorinated biphenyls (PCBs) in San Francisco Bay. Environmental Research 105: 67-86.

Davis, J.A., J.L. Grenier, A.R. Melwani, S.N. Bezalel, E.M. Letteney, E.J. Zhang, M. Odaya. 2007. Bioaccumulation of Pollutants in California Waters: A Review of Historic Data and Assessment of Impacts on Fishing and Aquatic Life. A Report of the Surface Water Ambient Monitoring Program (SWAMP). California State Water Resources Control Board, Sacramento, CA.

Davis, J.A., K. Schiff, A.R. Melwani, S.N. Bezalel, J.A. Hunt, R.M. Allen, G. Ichikawa, A. Bonnema, W.A. Heim, D. Crane, S. Swenson, C. Lamerdin, and M. Stephenson. 2011. Contaminants in Fish from the California Coast, 2009: Summary Report on Year One of a Two-Year Screening Survey. A Report of the Surface Water Ambient Monitoring Program (SWAMP). California State Water Resources Control Board, Sacramento, CA.

Dearth, M.A. and R.M. Hites. 1991. Complete analysis of technical chlordane using negative ionization mass spectrometry. Environ. Sci. Technol. 25(2): 245–254.

Eagles-Smith, C.A., J.T. Ackerman, S.E.W. de la Cruz, and J.Y. Takekawa. 2009. Mercury bioaccumulation and risk to three waterbird foraging guilds is influenced by foraging ecology and breeding stage. Environmental Pollution 157: 1993–2002.

Fairey, R., K. Taberski, S. Lamerdin, E. Johnson, R. P. Clark, J. W. Downing, J. Newman, and M. Petreas. 1997. Organochlorines and other environmental contaminants in muscle tissues of sportfish collected from San Francisco Bay. Marine Pollution Bulletin 34:1058-1071.

Ferguson, D.M., D.F. Moore, M.A. Getrich, and M.H. Zhowandai 2005. Enumeration and speciation of enterococci found in marine and intertidal sediments and coastal water in southern California. Journal of Applied Microbiology 99:598-608.

Grant, S.B., B.F. Sanders, A.B. Boehm, J.A. Redman, J.H. Kim, R.D. Mrse, A.K. Chu, M. Gouldin, C.D. McGee, N.A. Gardiner, B.H. Jones, J. Svejkovsky, G.V. Leipzig, and A. Brown 2001. Generation of enterococci bacteria in a coastal saltwater marsh and its impact on surf zone water quality. Environ. Sci. Technol. 35(12):2407-2416.

Greenfield, B. K., J. A. Davis, R. Fairey, C. Roberts, D. Crane and G. Ichikawa. 2005. Seasonal, interannual, and long-term variation in sport fish contamination, San Francisco Bay. Science of the Total Environment 336: 25– 43.

Griffith, J.F. and S.B. Weisberg 2011. Challenges in implementing new technology for beach water quality monitoring: lessons from a California demonstration project. Marin Technology Society Journal 45(2):65-73.

Gunther, A.J., J.A. Davis, D. Hardin, J. Gold, D. Bell, J. Crick, G. Scelfo, J. Sericano, M. Stephenson. 1999. Long term bioaccumulation monitoring with transplanted bivalves in San Francisco Bay. Marine Pollution Bulletin 38(3): 170-181.

Halliday, E., J.F. Griffith, and R.J. Gast 2010. Use of an exogenous plasmid standard and quantitative PCR to monitor spatial and temporal distribution of Enterococcus spp. in beach sands. Limnol. and Oceanogr.: Methods 8:146-154.

Heal the Bay 2007. 17th Annual Report 2006 – 2007: Beach Report Card. Heal the Bay, Santa Monica, CA; 81 pp.

Heal the Bay 2008. 18th Annual Report 2007 – 2008: Beach Report Card. Heal the Bay, Santa Monica, CA; 83 pp.

Heal the Bay 2009. 19th Annual Report 2008 – 2009: Beach Report Card. Heal the Bay, Santa Monica, CA; 81 pp.

Heal the Bay 2010. 20th Annual Report 2009 – 2010: Beach Report Card. Heal the Bay, Santa Monica, CA; 77 pp.

Heal the Bay 2011. 21st Annual Beach Report Card 2010 – 2011. Heal the Bay, Santa Monica, CA; 98 pp.

Heaney, C.D., E. Sams, S. Wing, S. Marshall, K. Brenner, A.P. Dufour, and T.J. Wade 2009. Contact with beach sand among beachgoers and risk of illness. Amer. J. Epidemiolgy 170(2):164-172.

Hou, D., S.J.M. Rabinovici, and A.B. Boehm 2006. Enterococci predictions from partial least squares regression models in conjunction with a single-sample standard improve the efficacy of beach management advisories. Environ. Sci. Technol. 40(6):1737-1743.

Hunt, J.A., J.A. Davis, B.K. Greenfield, A. Melwani, R. Fairey, M. Sigala, D.B. Crane, K. Regalado, and A. Bonnema. 2008. Contaminant Concentrations in Fish from San Francisco Bay, 2006. SFEI Contribution #554. San Francisco Estuary Institute, Oakland, CA.

Imamura, G.J., R.S. Thompson, A.B. Boehm In Press. Wrack promotes the persistence of faecal indicator bacteria in marine sands and seawater. FEMS Microbiology Ecology [accepted, but not yet published]

Klasing, S. and R. Brodberg. 2011. Development of Fish Contaminant Goals and Advisory Tissue Levels for Common Contaminants in California Sport Fish: Polybrominated Diphenyl

Ethers (PBDEs). California Office of Environmental Health Hazard Assessment, Sacramento, CA.
Krieger, F., M. Pla, and C. Weir. 2007. The History of Municipal Wastewater Treatment: 35 Years of POTWs Protecting the Bay. In SFEI. 2007. The Pulse of the Estuary: Monitoring and Managing Water Quality in the San Francisco Estuary. SFEI Contribution 532. San Francisco Estuary Institute, Oakland, CA.
Lee, C.M., T.Y. Lin, C.-C. Lin, G.A. Kohbodi, A. Bhatt, R. Lee, and J.A. Jay 2006. Persistence of fecal indicator bacteria in Santa Monica Bay beach sediments. Water Res. 40(14):2593-2602.
OEHHA. 1994. Health advisory on catching and eating fish: Interim sport fish advisory for San Francisco Bay. Office of Environmental Health Hazard Assessment, California Environmental Protection Agency, Sacramento, CA.
Schoellhamer, D. 2007. Suspended Sediment in the Bay: Past a Tipping Point. In SFEI. 2009. The Pulse of the Estuary: Monitoring and Managing Water Quality in the San Francisco Estuary. SFEI Contribution 583. San Francisco Estuary Institute, Oakland, CA.
Schoen, M.E. and N.J. Ashbolt 2010. Assessing pathogen risk to swimmers at non-sewage impacted recreational beaches. Environmental Science and Technology 44(7):2286-2291.
Schwarzbach, S.E., J.D. Albertson, and C.M. Thomas. 2006. Effects of predation, flooding, and contamination on reproductive success of California Clapper Rails (*Rallus longirorostrus obsoletus*) in San Francisco Bay. The Auk 123(1):45–60.
SFBRWQCB (San Francisco Bay Regional Water Quality Control Board). 2006.Mercury in San Francisco Bay Proposed Basin Plan Amendment and Staff Report for
Revised Total Maximum Daily Load (TMDL) and Proposed Mercury Water Quality Objectives. San Francisco Bay Regional Water Quality Control Board, Oakland, CA.
SFBRWQCB (San Francisco Bay Regional Water Quality Control Board). 2007. San Francisco Bay Basin (Region 2) Water Quality Control Plan (Basin Plan). San Francisco Bay Regional Water Quality Control Board, Oakland, CA. http://www.swrcb.ca.gov/sanfranciscobay/basin_planning.shtml
SFBRWQCB (San Francisco Bay Regional Water Quality Control Board). 2008a. Exhibit A: Proposed Basin Plan Amendment. San Francisco Bay Regional Water Quality Control Board, Oakland, CA. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/TMDLs/sfbaypcbstmdl.shtml
SFBRWQCB (San Francisco Bay Regional Water Quality Control Board). 2008b. Total Maximum Daily Load for PCBs in San Francisco Bay: Final Staff Report for Proposed Basin Plan Amendment. San Francisco Bay Regional Water Quality Control Board, Oakland, CA. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/TMDLs/sfbaypcbstmdl.shtml
SFBRWQCB (San Francisco Bay Regional Water Quality Control Board). 2008c. Adopted Basin Plan Amendment. http://www.swrcb.ca.gov/sanfranciscobay/basin_planning.shtml

SFEI. 2000. San Francisco Bay Seafood Consumption Study. San Francisco Estuary Institute, Oakland, CA.
SFEI. 2009. The Pulse of the Estuary: Monitoring and Managing Water Quality in the San Francisco Estuary. SFEI Contribution 583. San Francisco Estuary Institute, Oakland, CA.

Soller, J.A., T. Bartrand, N.J. Ashbolt, J. Ravenscroft, and T.J. Wade 2010a. Estimating the primary etiologic agents in recreational freshwaters impacted by human sources of faecal contamination. Water Research 44:4736-4747.

Soller, J.A., M.E. Schoen, T. Bartrand, J.E. Ravenscroft, N.J. Ashbolt 2010b. Estimated human health risks from exposure to recreational waters impacted by human and non-human sources of faecal contamination. Water Research 44:4674-44691.

Suedel, B.C., J.A. Boraczek, R.K. Peddicord, P.A. Clifford, and T.M. Dillon. 1994. Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. Rev. Environ. Contam. Toxicol. 136: 21–89.

Thompson, B., T. Adelsbach, C. Brown, J. Hunt, J. Kuwabara, J. Neale, H. Ohlendorf, S. Schwarzbach, R. Spies, K. Taberski. 2007. Biological effects of anthropogenic contaminants in the San Francisco Estuary. Environmental Research 105: 156–174.

USEPA 1986. Ambient water quality criteria for bacteria – 1986. U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division, EPA 440/5-84-002; 18 pp.

USEPA. 1995. Guidance for assessing chemical contaminant data for use in fish advisories: Volume 1, Fish sampling and analysis. EPA 823-R-93-002, U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

USEPA. 2000. Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories: Volume 1, Fish Sampling and Analysis, Third Edition. EPA 823-R-93-002B-00-007. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

USEPA 2010a. Quantitative microbial risk assessment to estimate illness in freshwater impacted by agricultural animal sources of fecal contamination. U.S. Environmental Protection Agency, Office of Water, EPA 822-R-10-005; 61 pp. + appendices.

USEPA 2010b. Sampling and consideration of variability (temporal and spatial) for monitoring of recreational waters. U.S. Environmental Protection Agency, Office of Water, EPA 823-R-10-005; 63 pp.

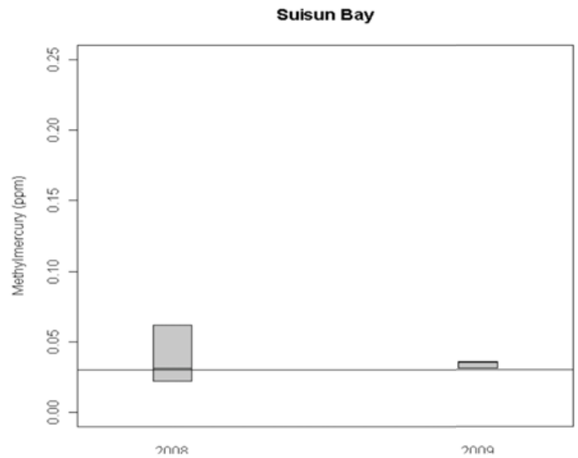
USEPA 2011. Exposure assessment models: virtual beach (VB). U.S. Environmental Protection Agency, Center for Exposure Assessment Modeling, <http://www.epa.gov/ceampubl/swater/vb2/> [accessed: 4/17/2011]

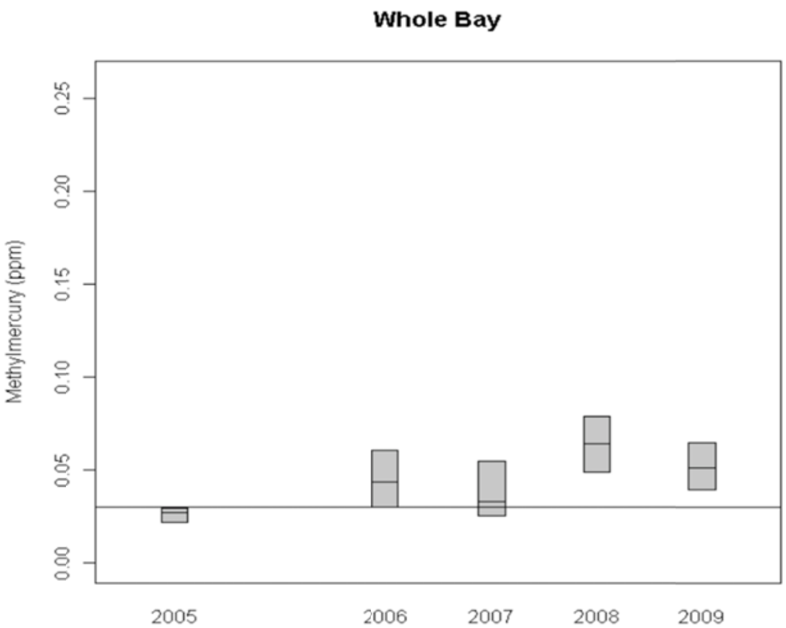
Whitman, R.L., M.B. Nevers, and M.N. Byappanahalli 2006. Examination of the watershed-wide distribution of Escherichia coli along southern Lake Michigan: an integrated approach. Applied Environ. Microbiol. 72(11):7301-7310.

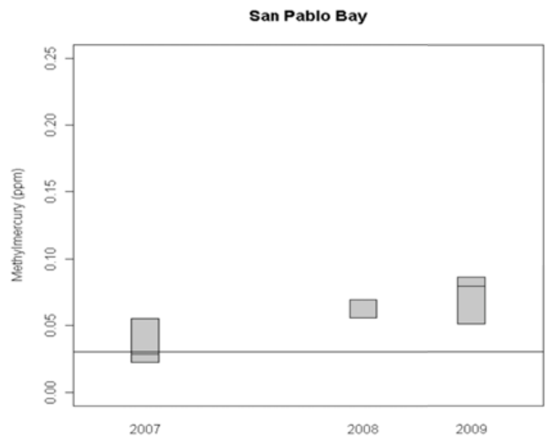
Yamahara, K.M., B.A. Layton, A.E. Santoro, and A.B. Boehm 2007. Beach sands along the California coast are diffuse sources of fecal bacteria to coastal waters. Environ. Sci. Technol. 41(13):4515-4521.

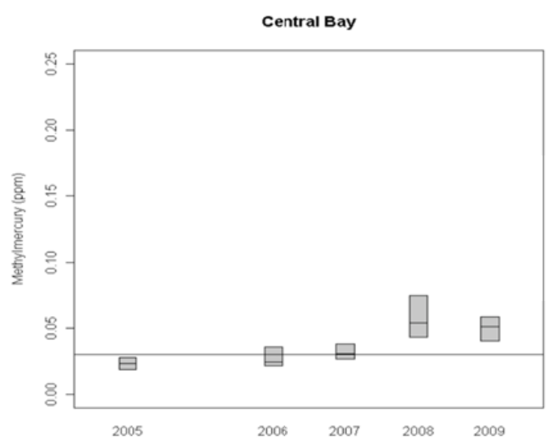
Yamahara, K.M., S.P. Walters, and A.B. Boehm 2009. Growth of enterococci in unaltered, unseeded beach sands subjected to tidal wetting. Appl. Environ. Microbiol. 75:1517-1524.

Figure 1. Methylmercury concentrations in small fish. Plots indicate the 25th, 50th, and 75th. Data for Mississippi silversides and topsmelt in the 3-5 cm size range sampled by the RMP. Reference line is the 0.030 ppm target from the Mercury TMDL.









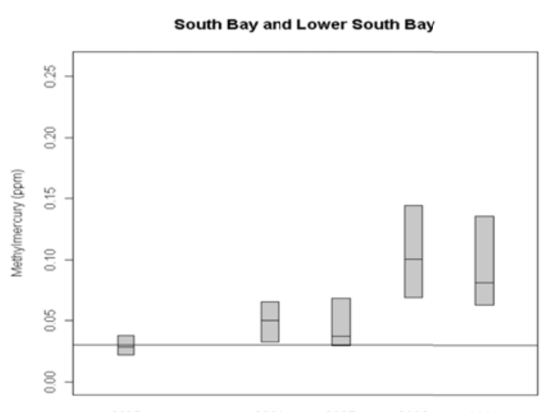


Figure 2. Summary assessment related to the "safe for aquatic life" question. The two key dimensions of water quality problems are their severity (degree of concern) and how quickly the Bay is anticipated to respond to pollution prevention actions (whether rapid progress is likely or not). The overall assessment scores indicated by the stars are based on a combination of these two factors.

	High Concern	*Moderate Concern*	*Low Concern*
Rapid Progress Likely	★★ Exotic Species	★★★ Trash	★★★★ Copper
Rapid Progress Unlikely	★ Methyl-mercury	★★ Sediment Toxicity	

Figure 3. Percent of Bay sediment samples exhibiting toxicity in laboratory assays. Sediment samples are tested in the RMP using amphipods and mussel larvae at xx stations each year.

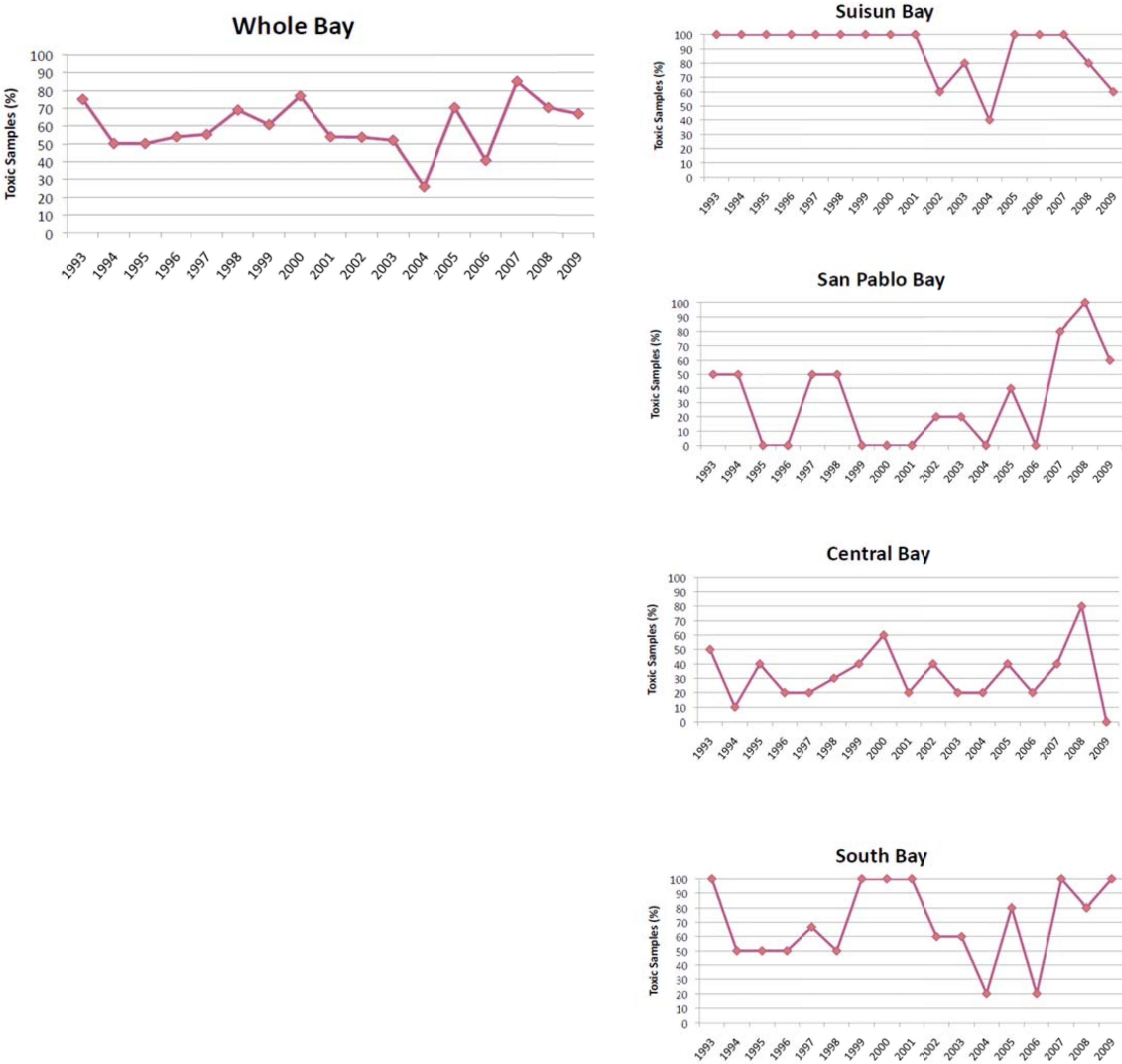


Figure 4. Dissolved copper concentrations in Bay water. Box and whisker plots indicate the 5th, 25th, 50th, 75th, and 95th percentiles. The water quality objective is a maximum of 6.9 ug/L in South Bay, and 6.0 ug/L in the other embayments.

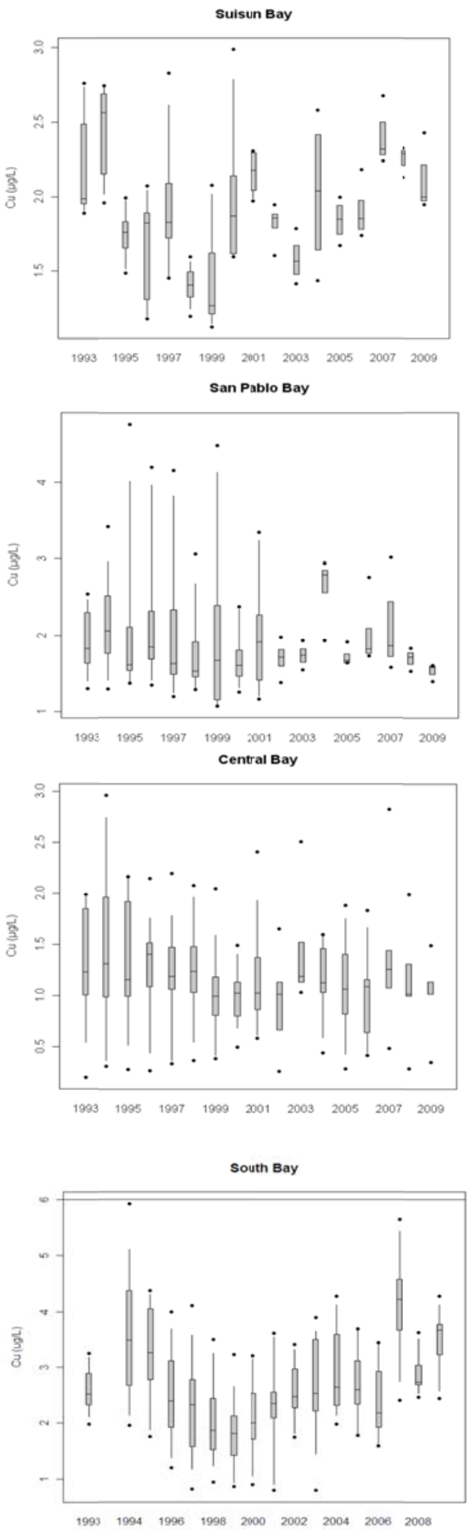


Figure 5. Dissolved oxygen concentrations in Bay water. Box and whisker plots indicate the 5th, 25th, 50th, 75th, and 95th percentiles. Water quality objectives are a minimum of 5 mg/L downstream of Carquinez Strait and a minimum of 7 mg/L upstream of Carquinez Strait.

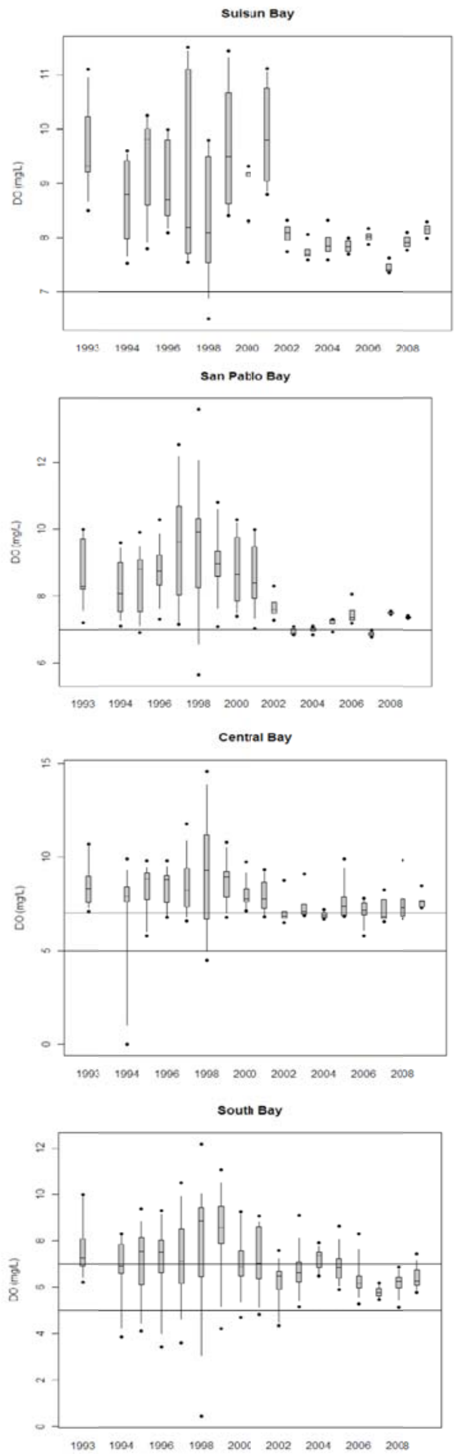


Figure 6. Dissolved silver concentrations in Bay water. Box and whisker plots indicate the 5th, 25th, 50th, 75th, and 95th percentiles. The water quality objective is a maximum of 1.9 ug/L.

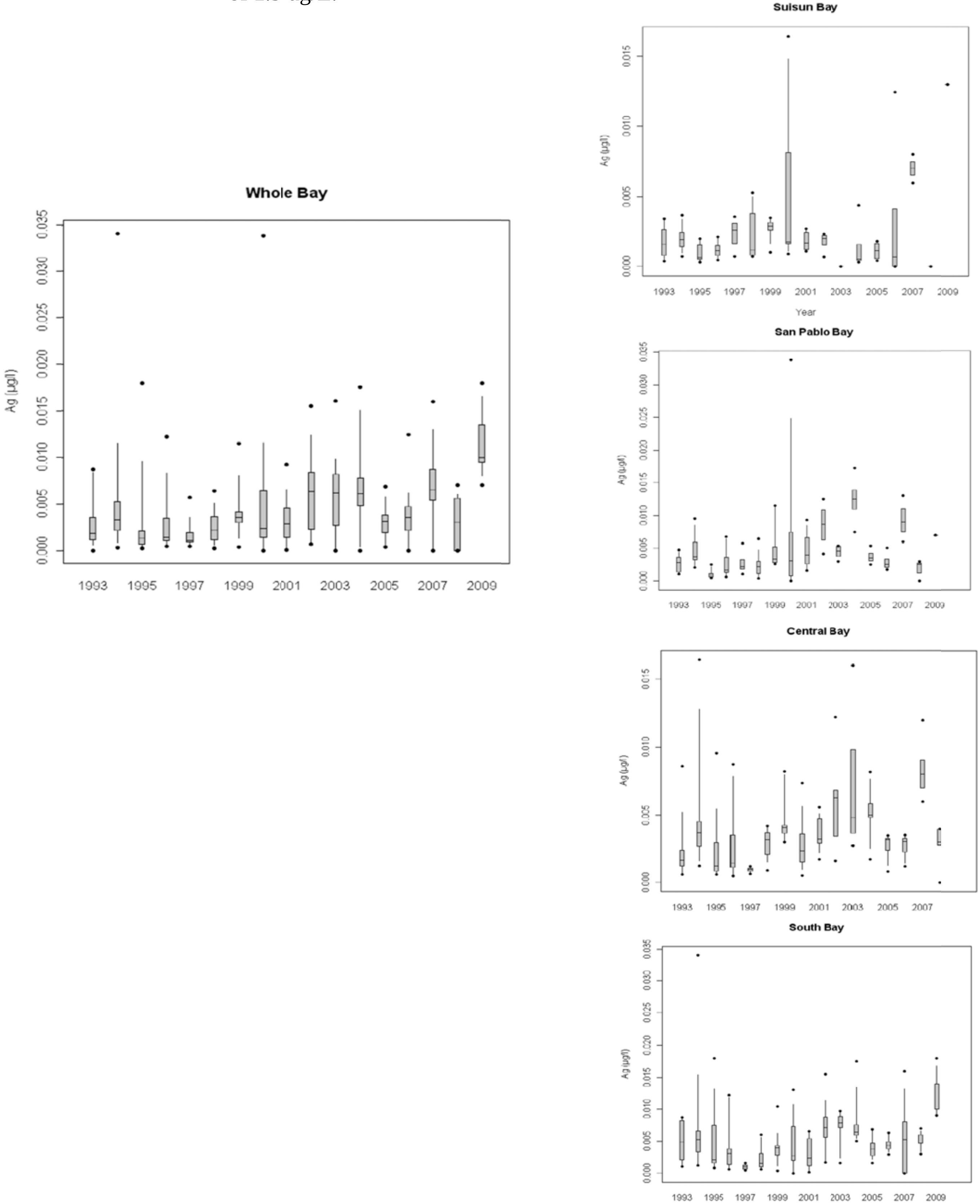


Figure 7. Locations of the five sampling stations in RMP sport fish monitoring.

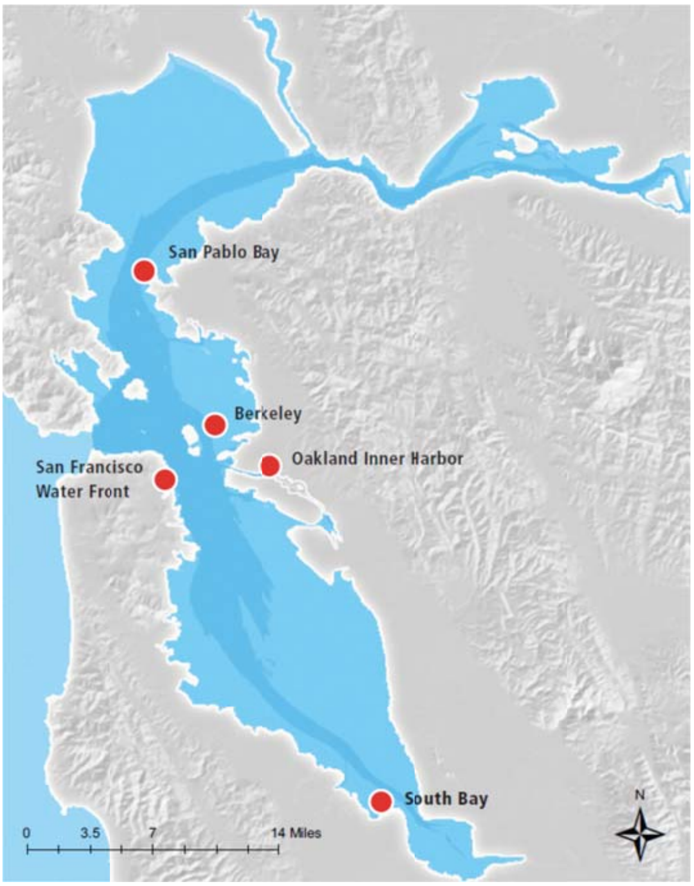


Figure 8. Average methylmercury concentrations in sport fish indicator species. Averages for striped bass based on concentrations for individual fish normalized to 60 cm. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for mercury is 0.44 ppm, and the two serving advisory tissue level is 0.07 ppm. Average concentrations for each species in the most recent sampling were between these two thresholds.

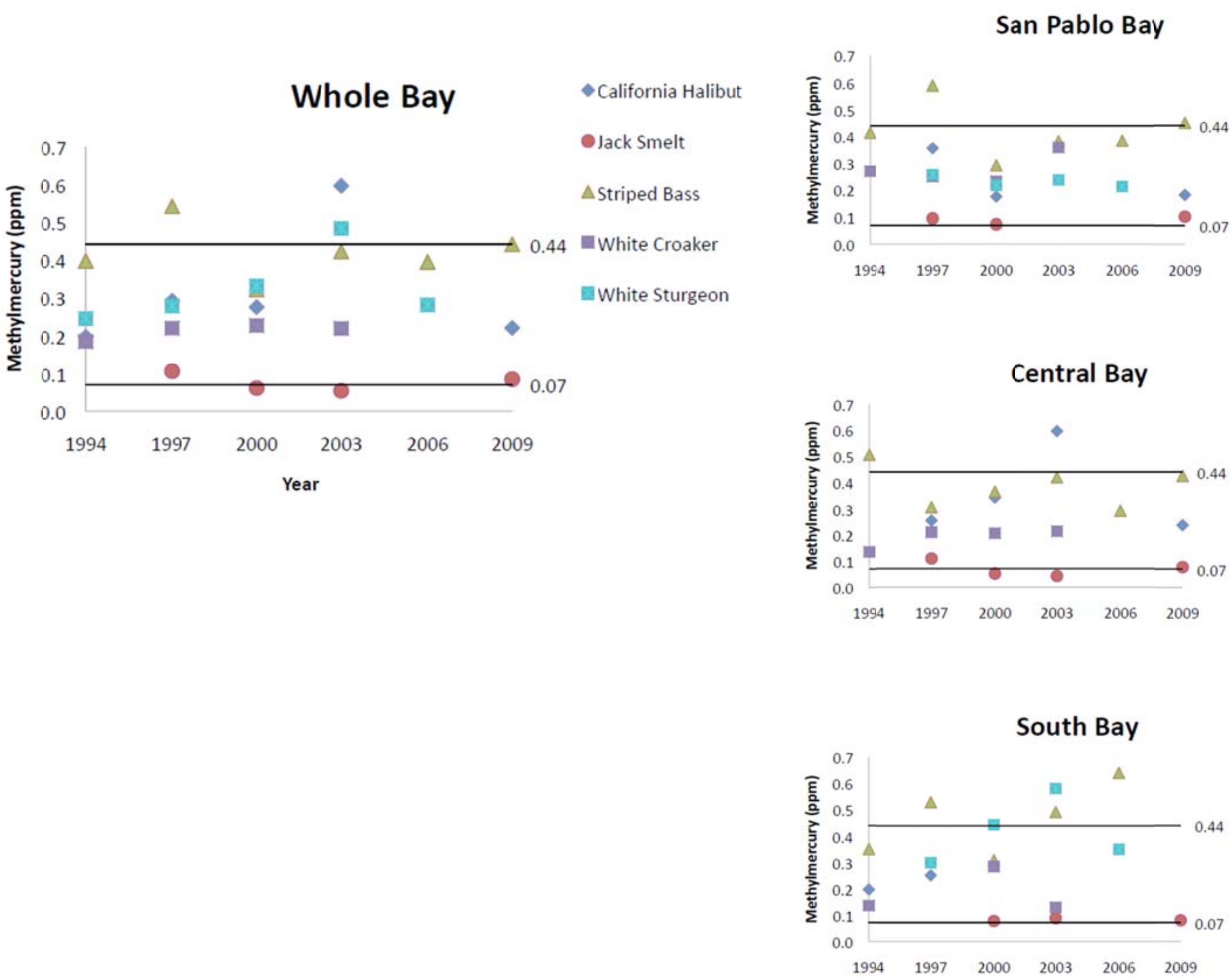


Figure 9. Summary assessment related to the “safe to eat” question. The two key dimensions of water quality problems are their severity (degree of concern) and how quickly the Bay is anticipated to respond to pollution prevention actions (whether rapid progress is likely or not). The overall assessment scores indicated by the stars are based on a combination of these two factors.

	High Concern	*Moderate Concern*	*Low Concern*
Rapid Progress Likely	★★	★★★	★★★★
Rapid Progress Unlikely	★ PCBs	★★ Methyl-mercury Dioxins*	

Footnote: * Dioxins were assessed using a San Francisco Bay Regional Water Quality Control Board target, rather than the Office of Environmental Health Hazard Assessment thresholds used for the other pollutants.

Figure 10. Average PCB concentrations in sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for PCBs is 120 ppb, and the two serving advisory tissue level is 21 ppb. Average concentrations for both species in the most recent sampling were between these two thresholds. Concentrations in shiner surfperch in San Pablo Bay had a declining trend. White croaker were analyzed with skin from 1994-2006, and without skin in 2009.

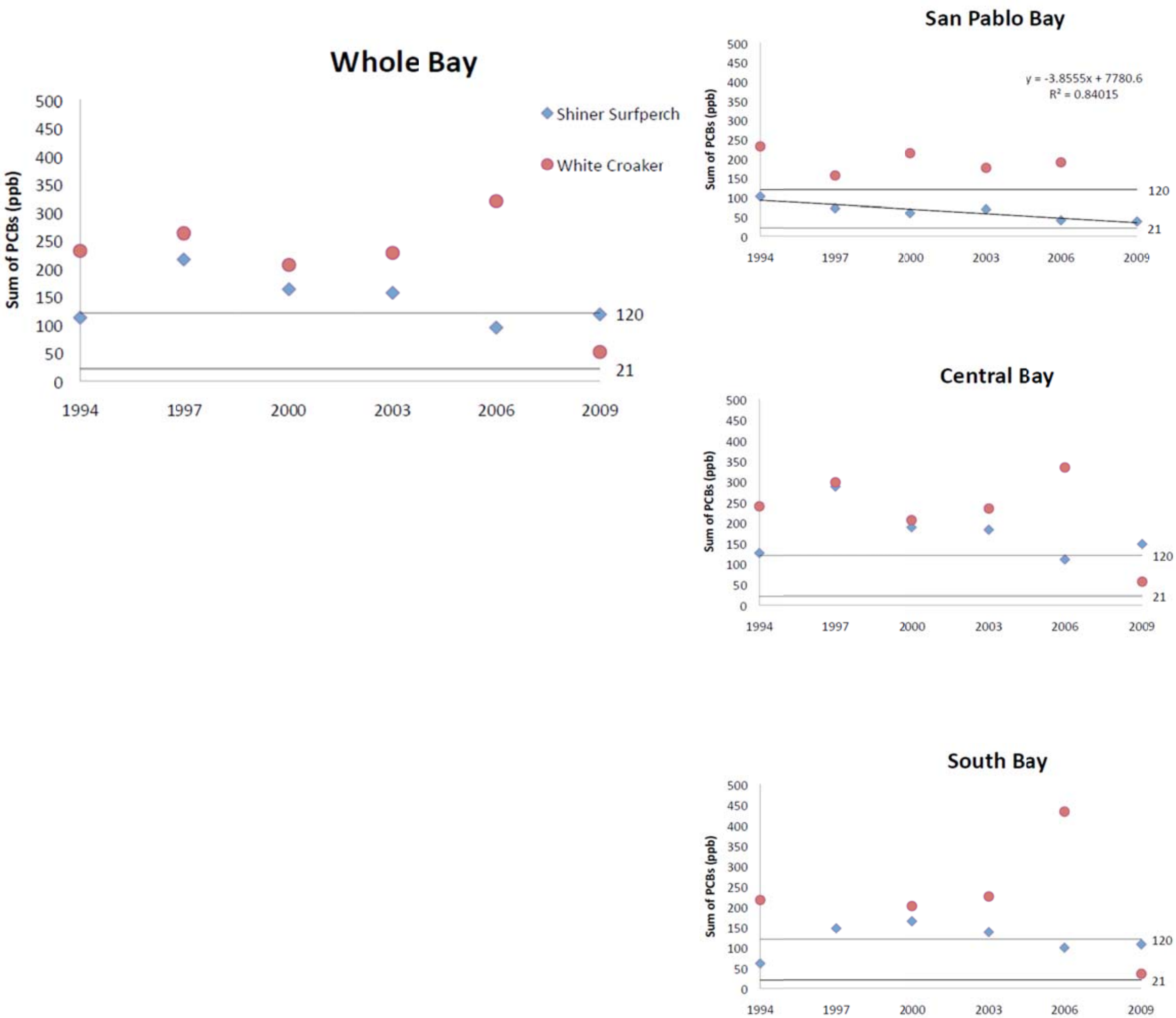


Figure 11. Average dioxin TEQ concentrations in shiner surfperch and white croaker, the key sport fish indicator species for organic pollutants. Sport fish are not routinely sampled in Suisun Bay. OEHHA has not established ATLs for dioxin TEQs. The San Francisco Bay Water Quality Control Board has developed a screeniung value for dioxin TEQs 0.14 parts per trillion (ppt). White croaker were analyzed with skin from 1994-2006, and without skin in 2009.

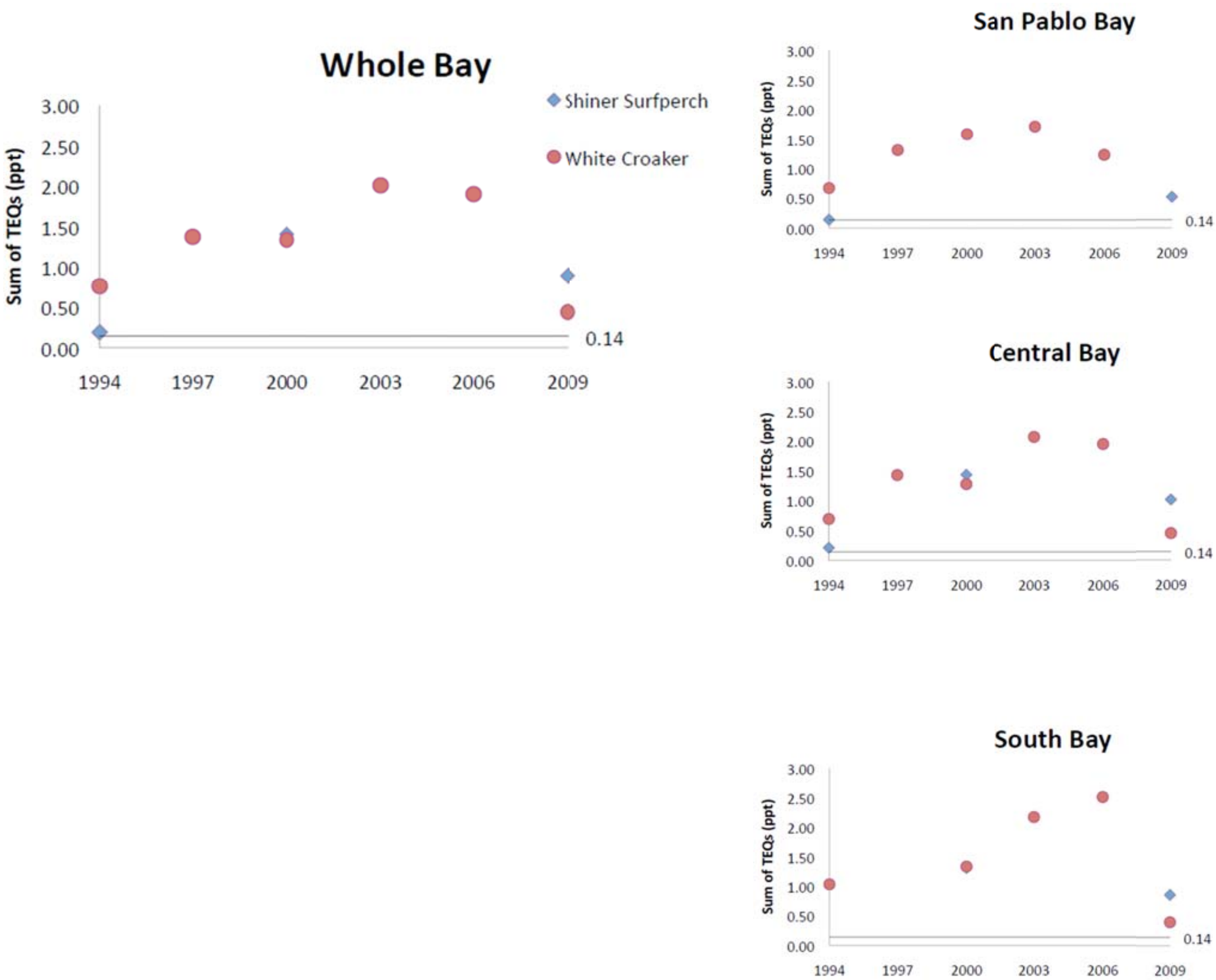


Figure 12. Average dieldrin concentrations in sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for dieldrin is 46 ppb, and the two serving advisory tissue level is 15 ppb. Average concentrations for both species in the most recent sampling were well below these thresholds. White croaker were analyzed with skin from 1994-2006, and without skin in 2009.

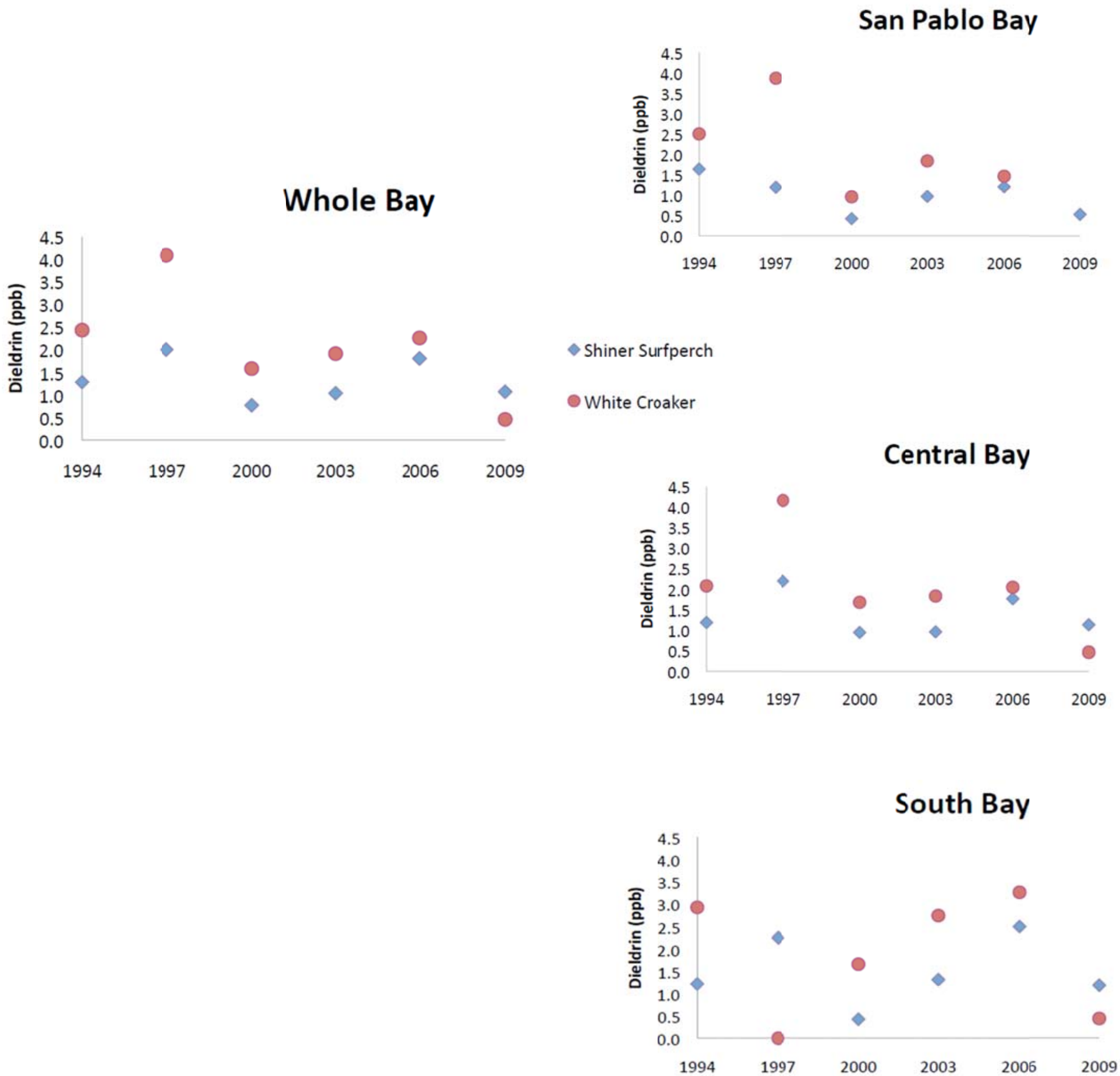


Figure 13. Average DDT concentrations in sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for DDT is 2100 ppb, and the two serving advisory tissue level is 520 ppb. Average concentrations for both species in the most recent sampling were well below these thresholds. White croaker were analyzed with skin from 1994-2006, and without skin in 2009.

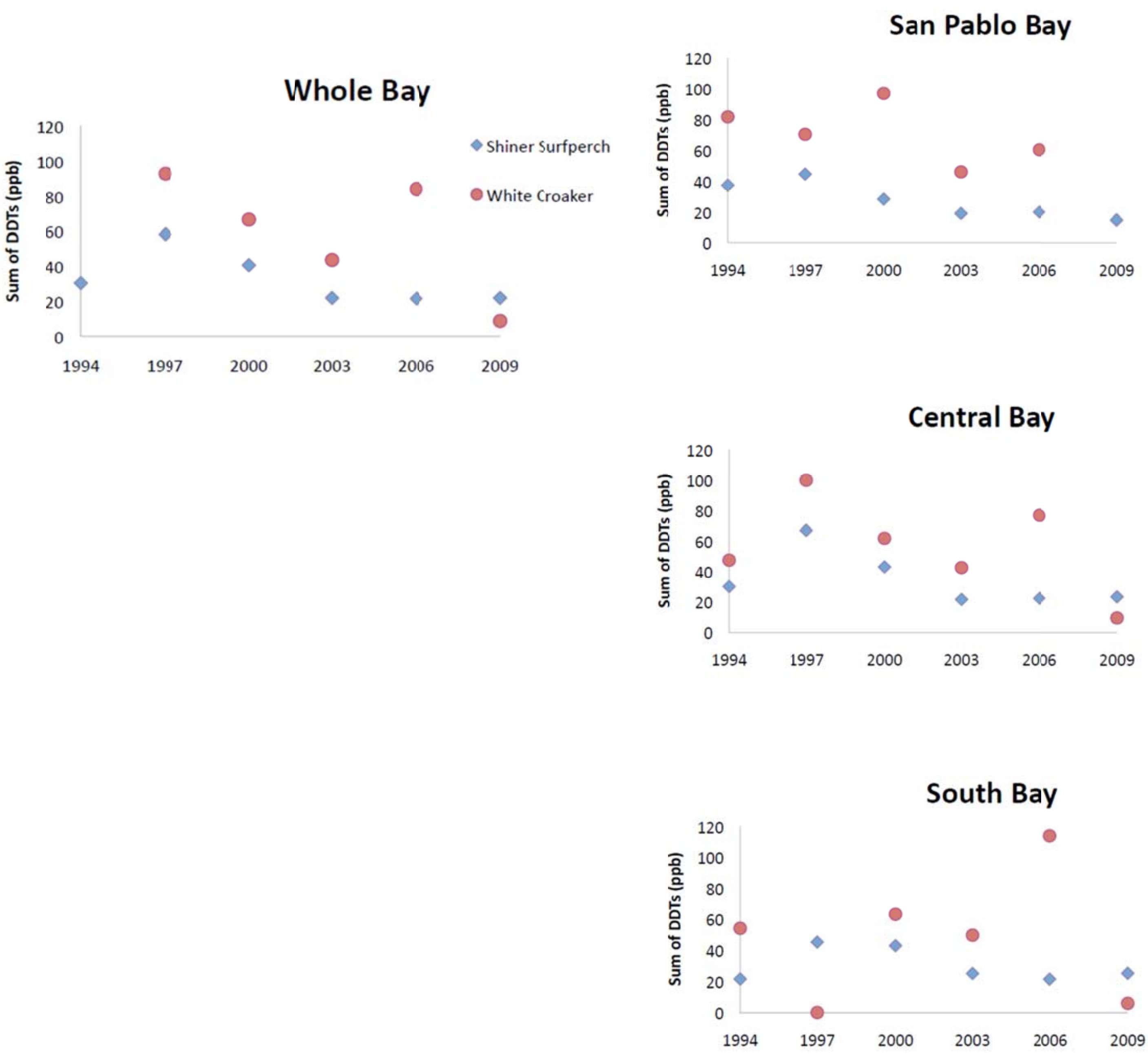


Figure 14. Average chlordane concentrations in sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for chlordane is 560 ppb, and the two serving advisory tissue level is 190 ppb. Average concentrations for both species in the most recent sampling were well below these thresholds. White croaker were analyzed with skin from 1994-2006, and without skin in 2009.

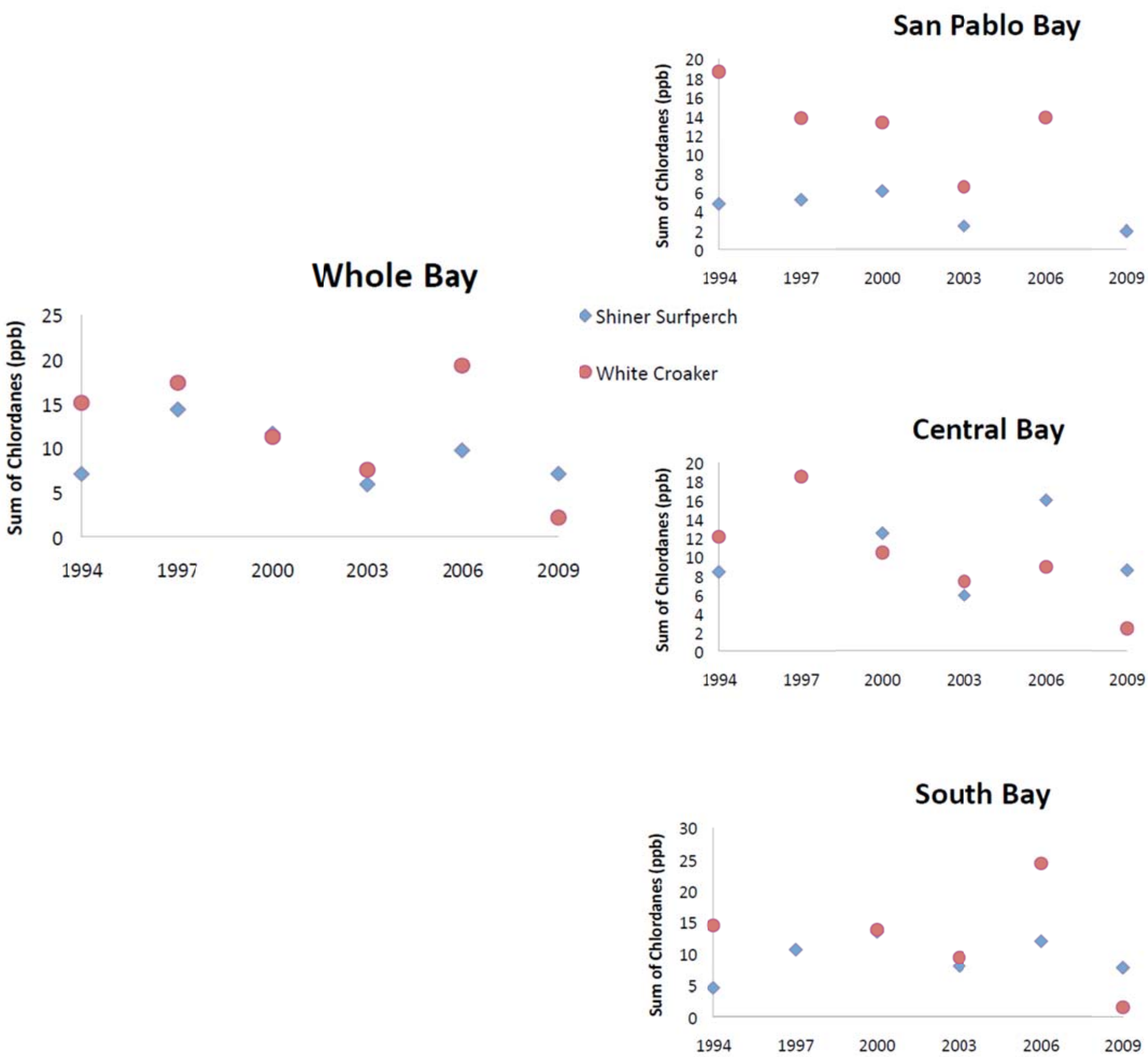


Figure 15. Average selenium concentrations in white sturgeon, the key sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for selenium is 15 ppm, and the two serving advisory tissue level is 2.5 ppm. Average concentrations for white sturgeon in the most recent sampling were well below these thresholds.

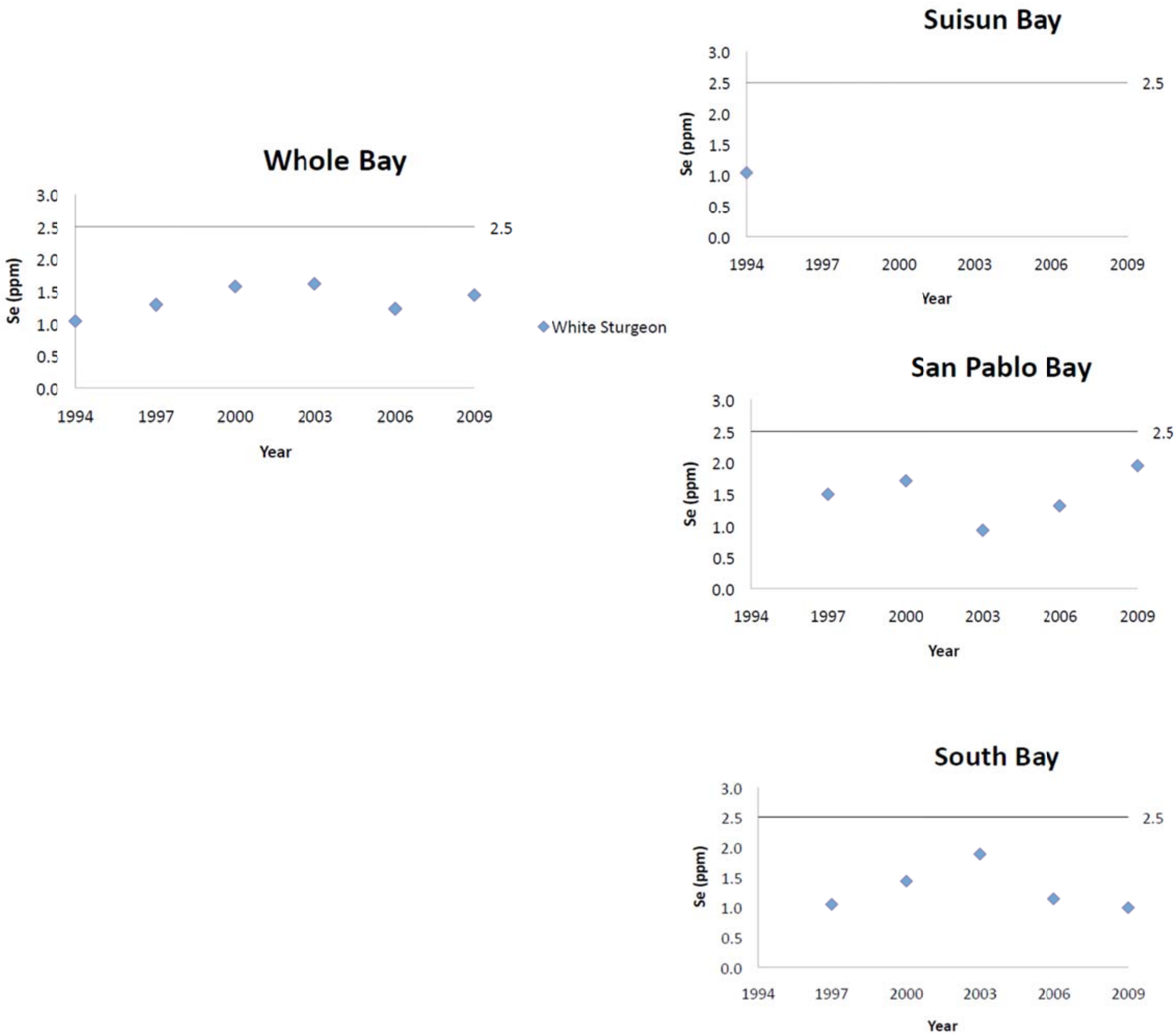
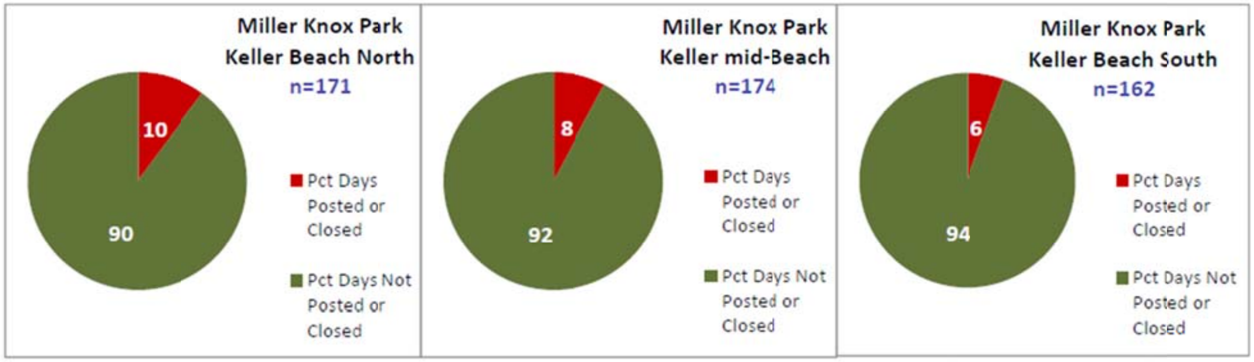


Figure 16. Frequency of beach closures, Alameda County.

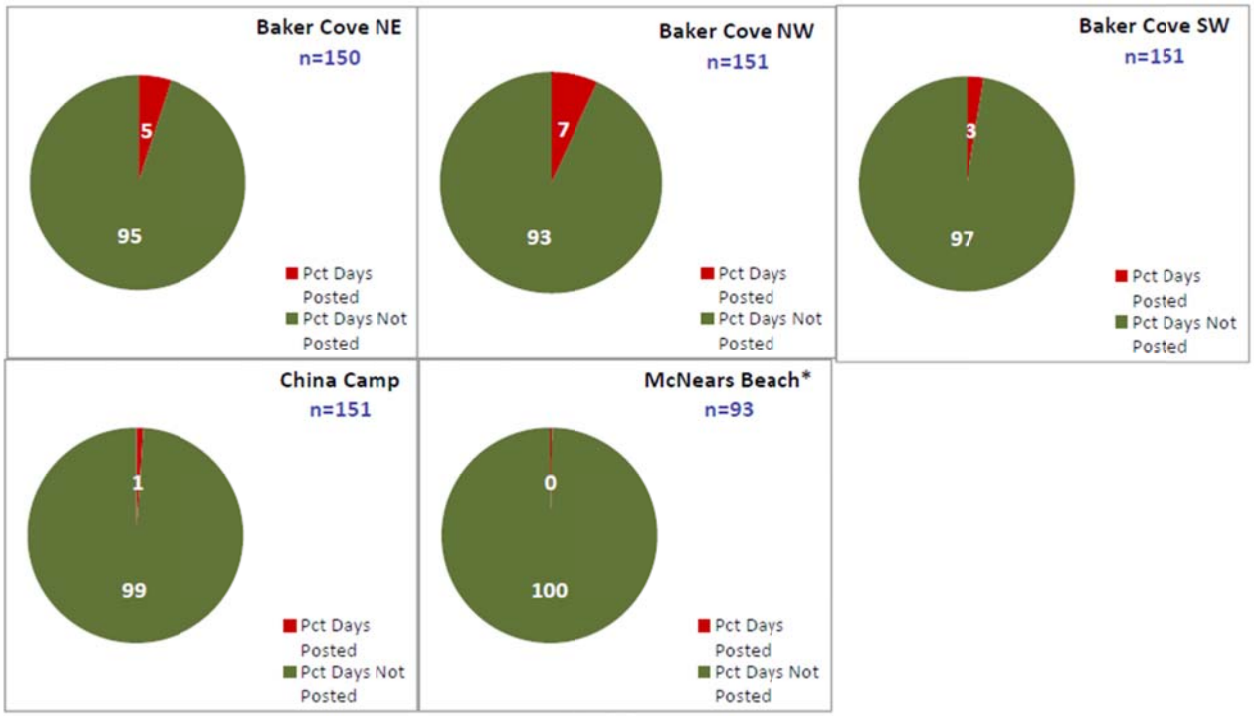
Alameda County
Percent of days during the prime beach season (April - October) that beaches were posted and not posted due to possible fecal contamination from 2006 through 2010 (n=number of samples)
*Alameda Point sampled 2008 - 2010

Figure 17. Frequency of beach closures, Contra Costa County.



Contra Costa County
Percent of days during the prime beach season (April - October) that beaches were posted and not posted due to possible fecal contamination from 2006 through 2010 (n=number of samples)

Figure 18. Frequency of beach closures, Marin County.



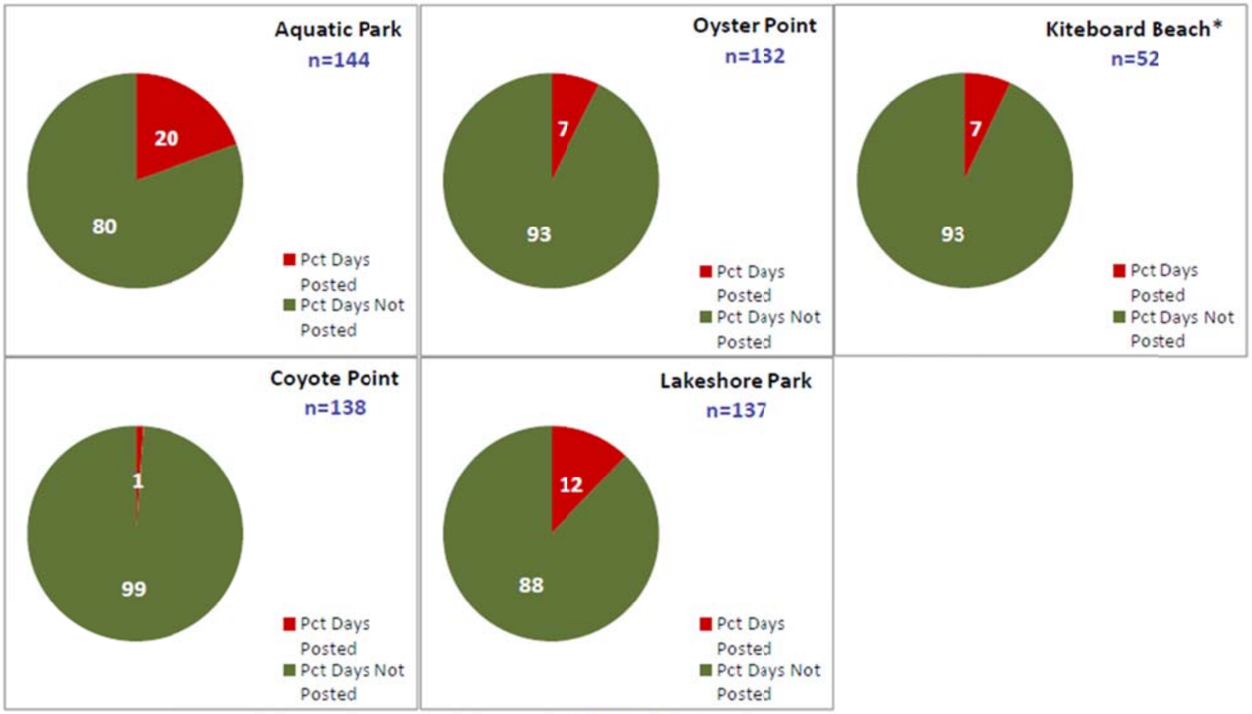
Marin County

Percent of days during the prime beach season (April - October) that beaches were posted and not posted due to possible fecal contamination from 2006 through 2010 (n=number of samples)

*McNears Beach sampled 2006 - 2008

Figure 19. Frequency of beach closures, San Francisco County.

Figure 20. Frequency of beach closures, Alameda County.



San Mateo County

Percent of days during the prime beach season (April - October) that beaches were posted and not posted due to possible fecal contamination from 2006 through 2010 (n=number of samples)

*Kiteboard Beach sampled 2008 - 2010

Figure 21. Percentage of samples from San Francisco beaches that exceeded the Enterococcus single sample maximum standard of 104 MPN/100 mL by month, percentage that exceeded the Enterocccus 30-day geometric mean standard of 35 MPN/100 mL by month, and average rainfall by month for the five-year period 2006 - 2010. The graph illustrates a pattern of higher incidence of fecal indicator bacteria in wet weather than in dry weather, a pattern common in San Francisco Bay and throughout coastal California. N=2,285

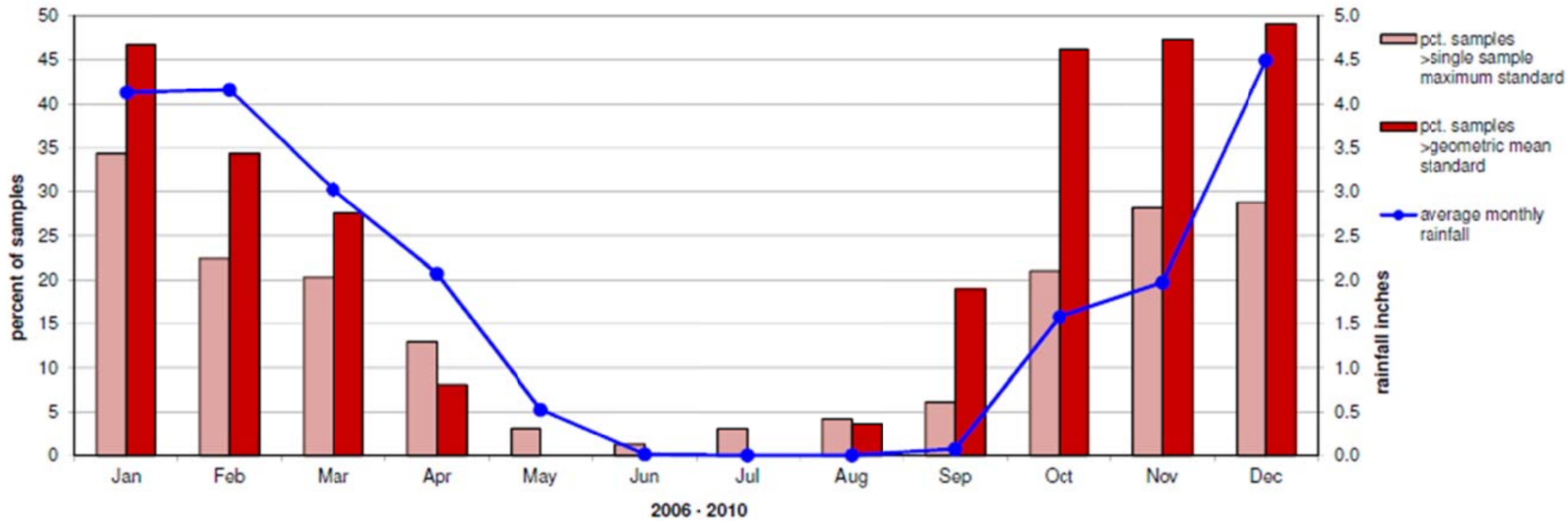


Table 1. Sources of Information on bacteria monitoring at Bay beaches.

Alameda County
website: www.ebparks.org/stewardship/water
hotline: 510-567-6706 (Crown Beach)

Contra Costa County
website: www.ebparks.org/stewardship/water

City and County of San Francisco
website: http://beaches.sfwater.org
hotline: 415-242-2214 or 1-877-SFBEACH (732-3224) toll free

San Mateo County
website: www.smhealth.org/environ/beaches
hotline: 650-599-1266

Heal the Bay Beach Report Cards
website: www.beachreportcard.org

California Safe to Swim Web Portal
website: www.waterboards.ca.gov/mywaterquality/safe_to_swim

California Beach Water Quality Information Page
website: www.swrcb.ca.gov/water_issues/programs/beaches/beach_water_quality/index.shtml

Table 2. California standards for fecal indicator bacteria.

Single Samples

Indicator	**Standard (colony forming units per 100 mL of water)**
Enterococcus	104
Fecal Coliform	400
Total Coliform	10,000
Total:Fecal Ratio (when Total is greater than or equal to 1,000)	10

Geometric Means

Indicator	**Standard (colony forming units per 100 mL of water)**
Enterococcus	35
Fecal Coliform	200
Total Coliform	1000

Table 3. Heal the Bay grades for San Francisco Bay Area beaches.

	Heal the Bay Annual Beach Report Card Grades (year-round = April 1 - March 31)														
	APRIL - OCTOBER					DRY YEAR-ROUND					WET YEAR-ROUND				
	2006	2007	2008	2009	2010	2006 - 2007	2007 - 2008	2008 - 2009	2009 - 2010	2010 - 2011	2006 - 2007	2007 - 2008	2008 - 2009	2009 - 2010	2010 - 2011
San Mateo County															
Oyster Point		A	A	B	A		A		A	A		C		F	D
Coyote Point		A	A+	A+	A+		A		A	A+		A		B	C
Aquatic Park		A	B	F	D		B		F	D		F		F	F
Lakeshore Park		A	D	D	D		C		D	D		F		F	F
Kiteboard Beach			B						A					F	
Alameda Couty															
Alameda Point North			A	A+	A			A	A+	A			A+	A	C
Alameda Point South			A	A	A+			A	A	A			A+	A	A
Crown Beach Bath House		A	A	B	A+		A	C	B	A+		C	A+	A	A
Crown Beach Windsurf Corner		A	A	A	A+		A	A	A	A+		A	A+	B	B
Crown Beach Sunset Road		A	A+	A	A+		A	A	A	A+		F	A	B	B
Crown Beach Shoreline Drive		A	A	A+	A+		A	A	A	A		F	A+	C	B
Crown Beach Bird Sanctuary		A	A	B	A		C	A	B	A		F	B	D	C
Contra Costa County															
Keller Beach North		B	F	D	F		B	D	D	F		A	A	B	A
Keller Beach Mid-Beach		B	C	D	F		B	C	D	F		B	B	B	A
Keller Beach South		A	C	D	D		A	C	D	D		A	B	C	B
San Francisco County															
Crissy Field Beach West			A+	A+	A+			A+	A+	A			A	C	B
Crissy Field mid-Beach	A	A+				A	A+				B	A			
Crissy field Beach East	A	A	A	A	A+	C	A	B	A	B	D	A	B	B	C
Aquatic Park Beach	A	B	A	A	A	A	C	B	A	B	B	A	C	A	B
Hyde Street Pier	A	A	A	A+	A+	A	A	A	A	A	A	A	A+	A	A
Jackrabbit Beach	A	A	A	A	A	A	A	A	A	A	A	F	D	C	B
CPSRA Windsurfer Circle	A	A	A	A	D	A	A	B	A	F	F	F	F	F	F
Sunnydale Cove	A	A	A	B	D	A	C	A	C	C	F	F	F	F	F
Marin County															
Horseshoe Cove NE	A	A	A	A+	A										
Horseshoe Cove NW	A	B	A	A	A										
Horseshoe Cove SW	A	A	A	A	A										
Schoonmaker Beach	A	A+	A+	A	A+										
Paradise Cove	A	A	A+												
China Camp	D	A+	A+	A	A										
McNears Beach	C	A	A												
Overall GPA	3.64	3.88	3.61	3.30	3.23	3.71	3.44	3.31	3.12	2.91	2.14	2.05	3.11	2.14	2.38
Overall Grade	B+	A-	B+	B	B	A-	B+	B+	B	B-	C	C	B	C	C+

State of San Francisco Bay 2011
Appendix B

WATER - Freshwater Inflow Indicators and Index
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background

San Francisco Bay, at the interface between California's largest rivers and the Pacific Ocean, is important spawning, nursery and rearing habitat for a host of fishes and invertebrates, a migration corridor for anadromous fishes like salmon, steelhead and sturgeon, and breeding and nesting habitat for waterfowl and shorebirds. The amounts, timing and patterns of freshwater inflow to the Bay define the quality and quantity of this estuarine habitat and drive key ecological processes (Jassby et al. 1995; Kimmerer 2002, 2004; Feyrer et al. 2008, 2010; Moyle and Bennett, 2008; Moyle et al., 2010; and see Estuarine Open Water Habitat indicator and Flood Events indicator). The mixing of inflowing fresh water and saltwater from the ocean creates low salinity, or "brackish" water habitat for estuary-dependent species. Changes in inflows trigger reproduction and migration, and high flows transport nutrients, sediments and organisms to and through the Bay, promote mixing and circulation within the estuary and flushing contaminants.

Most of the fresh water that flows into the San Francisco Bay comes from the Sacramento and San Joaquin Rivers, which provide >90% of total inflow in most years and have large impacts on salinity regimes in the estuary. Smaller streams around the estuary, like the Napa and Guadalupe rivers, Alameda, San Francisquito, Coyote, Sonoma creeks, and many smaller tributaries, contribute the balance and can have large environmental effects on a local level. All of these rivers have large seasonal and year-to-year variations in flow, reflecting California's seasonal rainfall and snowmelt patterns, and unpredictable times of floods and droughts that are often driven by ocean conditions like El Nino and La Nina.

In the Bay's Sacramento-San Joaquin watershed, several factors have had and are having substantial impacts on the amounts, timing and patterns of freshwater inflows to the estuary (Figure 1). First, flows in most of the Bay's largest tributary rivers have been greatly altered by dams. Many of these dams were built for the purpose of reducing flood events and to store the mountain runoff for later use and export to other regions in the state. Second, large amounts of water are extracted from the rivers and the Delta upstream of the Bay. Collectively, these diversions can remove large percentages of the total flow, even during of relatively high flow. This reduces the amounts of fresh water that flow into the estuary and can decrease inflows to levels below important threshold for habitat creation and sediment transport. Finally, the effects of climate change on flows in the watershed are already detectable and are predicted to increase. With warmer temperatures, increasing proportions of the precipitation in the watershed come as rain, which runs off immediately, rather than snow, which melts and flows into the rivers later in the season. In the rivers and the Bay, the result is more frequent but shorter duration high flow events earlier in the year driven by rain runoff rather than the long duration spring snowmelt flood flows of the past.

The Freshwater Inflow Index uses six indicators to evaluate the amounts, timing, and patterns of freshwater inflow to the estuary from the Sacramento-San Joaquin watershed. In order to account for the system's natural seasonal and year-to-year variability, each of the indicator measurements is made in comparison to what the freshwater inflow conditions would have been if there were no dams or water diversions, referred to as "unimpaired" conditions. The first two of the indicators measure how much water flows into the estuary each year and during the ecologically important spring period. Two other indicators measure the variability of freshwater inflows, both between years and the seasonal variability within each year. The fifth indicator measures how frequently the estuary receives high inflows, which are usually driven by flood conditions in the estuary's watershed. The final indicator measures how frequently the estuary experiences inflow conditions similar to what would have occurred in "critically dry" years, the driest 20% of years, under unimpaired conditions. For each year, the Freshwater Inflow Index is calculated by combining the results of the six indicators into a single number.

II. Data Sources and Definitions

A. Data Sources

Most of the fresh water that flows into the San Francisco Estuary comes from the Sacramento and San Joaquin River basins, which provide >90% of total inflow in most years.[1] Smaller streams around the estuary, principally the Napa and Guadalupe Rivers, Alameda, San Francisquito, Coyote, Sonoma Creeks, and many smaller tributaries, contribute the balance but flow data for these streams is very limited. Therefore, all of the Freshwater Inflow indicators were calculated using flow data from the Sacramento-San Joaquin Delta and watershed only.

The indicators and Index were calculated for each year[2] using data from the California Department of Water Resources (CDWR) DAYFLOW model (for "actual flows," termed "Delta outflow" in the dataset) and CDWR's Central Valley Streams Unimpaired Flows and Full Natural Flows datasets (for "unimpaired flows," and see below).[3] DAYFLOW is a computer model developed in 1978 as an accounting tool for calculating historical Delta outflow and other internal Delta flows.[4] DAYFLOW output is used extensively in studies by State and federal agencies, universities, and consultants. DAYFLOW output is available for the period 1930-2010. Annual and monthly unimpaired flow data for total Delta outflow were from the CDWR California Central Valley Unimpaired Flow dataset (1921-2003; using the "Delta Unimpaired Total Outflow" dataset).[5] For 2004-2010, annual and monthly unimpaired flows were calculated by a regressions developed from the Central Valley unimpaired flow data (using the 1930-1994 period) and the corresponding unimpaired runoff estimates from the "Full Natural Flows" (FNF) dataset[6] for the ten largest rivers in the watershed.[7]

[1] The Sacramento River provides 69-95% (median=85%) and the San Joaquin River provides 4-25% (median=11%) of total freshwater inflow to the San Francisco Bay (Kimmerer, 2002).
[2] Flow indicators were calculated for each water year. The water year is from October 1-September 30.
[3] For both the DAYFLOW and Central Valley Streams Unimpaired Flows datasets, total freshwater inflow to the San Francisco Estuary from the Sacramento-San Joaquin watershed is referred to as "net Delta outflow".
[4] More information about DAYFLOW is available at http://www.water.ca.gov/dayflow/.
[5] California Central Valley Unimpaired Flow dataset and report is available at: http://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/bay_delta_plan/water_quality_control_planning/docs/sjrf_spprtinfo/dwr_2007a.pdf
[6] Full Natural Flows datasets are available at: http://cdec.water.ca.gov/cgi-progs/previous/FNF
[7] The ten rivers are the Sacramento, Feather, Yuba, American, Cosumnes, Mokelumne, Stanislaus, Merced and San Joaquin Rivers. The regression for annual flow is: Unimpaired Delta outflow = -3692.54 + 1.31(10-river unimpaired runoff); $n=65$, $r^2=0.998$, $p<0.001$.

B. Definitions

Water Year Type: Runoff from the Sacramento-San Joaquin watershed can vary dramatically from year to year, a function of California's temperate climate and unpredictable occurrences of droughts and floods. To categorize these large year-to-year variations in flow, we used annual unimpaired inflows to classify each year as one of five water year types: wet, above normal, below normal, dry, and critical.[8] Year types were established based on frequency of occurrence during the period of 1930-2009, with each year type comprising 20% of all years.[9] Figure 2 shows annual unimpaired inflows with year type classification shown by the different colors of the bars.

Unimpaired Inflow: Unimpaired inflow is the freshwater inflow that, under the same hydrological conditions but without the effects of dams and diversions in the Sacramento-San Joaquin watershed, would have flowed into the estuary (see Figure 1).

Pre-dam Inflow: The period prior to the completion of major dams in the watershed, from 1930-1943, is referred to as the “pre-dam” period. During this period, actual flows were somewhat similar to unimpaired flows.

III. Indicator Evaluation

The San Francisco Estuary Partnership’s Comprehensive Conservation and Management Plan (CCMP) calls for “increase[ing] freshwater availability to the estuary”, “restor[ing] healthy estuarine habitat” and “promot[ing] restoration and enhancement of stream and wetland functions to enhance resiliency and reduce pollution in the Estuary” are non-quantitative. However, California’s State Water Resources Control Board (SWRCB) recently determined that, in order to protect public trust resources in the Sacramento-San Joaquin Delta and San Francisco Estuary, 75% of unimpaired runoff from the Sacramento-San Joaquin watershed should flow out of the Delta and into the estuary (SWRCB 2010).[10] Therefore, the “primary” reference conditions for most of the Freshwater Inflow indicators were developed based this recommendation.

A primary reference condition was established for each indicator. For most of the indicators, this reference condition was developed based on the SWRCB’s recommendation for 75% of unimpaired flow to the estuary. Pre-dam flows were also used as the basis for some indicators. Measured indicator values that were higher than the primary reference condition were interpreted to mean the indicator results met the CCMP goals and corresponded to "good" ecological conditions.

---

[8] Despite use of the term "below normal", this year type includes the median, with half of all years receiving more runoff and the other half of years receiving less runoff.

[9] Terminology for the five year types follows that used by state and federal water management agencies although, for water management purposes in the Sacramento and San Joaquin basins, water year types are determined using other factors, such as the previous year's precipitation, as well as than frequency of occurrence.

[10] The SWRCB recommendation was for the winter-spring period (January-June) and it was expressed as the 14-day running average of estimated unimpaired runoff, rather than as an annual or seasonal total.

In addition to the primary reference condition, information on the range and trends of indicator results, results from other watersheds, and known relationships between freshwater inflow conditions and physical and ecological conditions in estuaries was used to develop several intermediate reference conditions, creating a five-point scale for a range of evaluation results from “excellent,” “good, “fair,” “poor” to “very poor”. The size of the increments between the different evaluation levels was, where possible, based on observed levels of variation in the measured indicator values (e.g., standard deviations) in order to ensure that the different levels represented meaningful differences in the measured indicator values. Each of the evaluation levels was assigned a quantitative value, or score, from “4” points for “excellent” to “0” points for “very poor.” For each year, the Freshwater Inflow Index was calculated as the average of these six scores. Specific information on the primary and intermediate reference conditions is provided in the following sections describing each of the indicators.

Indicator and Index results were analyzed using analysis of variance and simple linear regression to identify differences among different time periods and trends with time.

IV. Indicators

A. Annual Inflow

1. Rationale

The amount of freshwater inflow to an estuary is a physical and ecological driver that defines the quality and quantity of estuarine habitat (Jassby et al. 1995; Kimmerer 2002; 2004 Feyrer et al. 2008, 2010; Moyle and Bennett, 2008; Moyle et al., 2010). In the Sacramento-San Joaquin watershed, annual runoff varies substantially for year-to-year, but during the past century, freshwater inflows into the estuary have been greatly altered by upstream dams and water diversions. Nine of the ten largest rivers that flow in the estuary’s Sacramento-San Joaquin watershed have large storage dams, where runoff is captured, stored and diverted. Additional water diversions are located along the rivers downstream of the dams and, in the Delta where the river flow into the estuary, local, state and federal water diversions extract more water for local and distant urban and agricultural. The resultant changes in the amount of freshwater flow that actually reaches the estuary have affected the estuarine ecosystem and the plants and animals that depend on it.

2. Methods and Calculations

The Annual Inflow indicator measures the amount of fresh water from the Sacramento-San Joaquin watershed that flows into San Francisco Estuary each year compared to the amount that would have flowed into the estuary under unimpaired conditions. The indicator was calculated for each year (1930-2010) using data for total annual actual freshwater inflow and estimated total annual unimpaired inflow.

The indicator is calculated as:

Annual Inflow (% of unimpaired) = [(actual inflow/unimpaired inflow)*100].

By incorporating unimpaired inflow as a component of the indicator calculation, the Annual Inflow indicator has been normalized to account for natural year-to-year variations in precipitation.

3. Reference Conditions

The primary reference condition for the Annual Inflow indicator was established as 75%, a level based on the SWRCB's recommendation for freshwater inflows needed to support public trust resources in the estuary.   Annual inflows that were greater than 75% of unimpaired inflows were considered to reflect "good" conditions. Additional information from trends and variability in annual inflows and from other estuaries was used to develop several other intermediate reference conditions. Table 1 below shows the quantitative reference conditions that were used to evaluate the results of the Annual Inflow indicator.

4. Results

Results of the Annual Inflow indicator are show in Figure 3.

The amount of fresh water flowing into the San Francisco Estuary each year has been reduced.
On an annual basis, the percentage of the freshwater runoff from estuary's largest watershed that flows into the estuary has been significantly reduced. For the most recent 10-year period (2001-2010), on average only 52% of unimpaired inflow actually flowed into the estuary. In 2010, only 39% of unimpaired inflow reached the estuary. In 2009, annual inflow only 32% of unimpaired, the third lowest percentage of freshwater inflow in the 81-year data record. In ten of the past 20 years (50% of years), the percentage of unimpaired flow that flowed into the estuary was less than 50%.

The proportional alteration in annual freshwater inflow to the estuary differs by water year type.
The greatest alterations to freshwater inflows (expressed as a percentage of estimated unimpaired inflow) occur in dry years. Since the 1950s, the percentages of unimpaired flow that reached the estuary averaged 43% in critically dry years, 53% in dry years, 62% in below normal years, 68% in above normal years and 73% in wet years.

Freshwater flow into the San Francisco Estuary, as a percentage of unimpaired flow, has declined over time.
The percentage of unimpaired flow that actually flowed into the estuary has declined significantly over the past several decades (regression, $p<0.001$). Significant declines in the percentage of unimpaired inflow reaching the estuary have occurred in all water years types (regression, all test, $p<0.05$). Before construction of most of the major dams on the estuary's tributary rivers (1930-1943, the "pre-dam" period), an average of 82% of estimated unimpaired flow actually reached the estuary. By the 1980s, the percentage had decreased significantly to

just 60% (1980-1989 average; Mann-Whitney, $p<0.01$). The average for the most recent 10-year period, 49%, is somewhat lower but, due to the large inter-annual variability associated with hydrology, not significantly different than flows during the 1980s.

Based on annual inflows, CCMP goals to increase fresh water availability to the estuary have not been met.
Current freshwater inflows to the estuary are well below the 75% level identified by the SWRCB as necessary to protect public trust resources and estuarine health. Current inflows are also somewhat lower than those measured in the 1980s, the period during which the CCMP was developed and established.

B. Spring Inflow

1. Rationale

Freshwater inflows during the spring provide important spawning and rearing habitat for many estuarine fishes and invertebrates (Jassby et al., 1995; Kimmerer, 2002; 2004; see also Estuarine Open Water habitat indicator). For a number of species, population abundance and/or survival are strongly correlated with the amounts of inflow the estuary receives during the spring and the location of low salinity, brackish water habitat, where fresh water from the rivers meets saltwater from the Pacific Ocean. Abundance and/or survival are higher when spring inflows are high and low salinity habitat is located downstream in the estuary, closer to the Golden Gate compared to years in which it is located further upstream (Jassby et al, 1995; Kimmerer 2002; 2004; Kimmerer et al., 2008).

2. Methods and Calculations

The Spring Inflow indicator measures the amount of fresh water from the Sacramento-San Joaquin watershed that flows into San Francisco Estuary during the spring, February-June, compared to the amount that would have flowed into the estuary during that season under unimpaired conditions. The indicator was calculated for each year (1930-2010) using data for February-June actual freshwater inflow and estimated total annual unimpaired inflow.

The indicator is calculated as:

Spring Inflow (% of unimpaired)
= [(actual inflow, Feb-June/unimpaired inflow, Feb-June)*100].

By incorporating unimpaired inflow as a component of the indicator calculation, the Spring Inflow indicator has been normalized to account for natural variations in precipitation.

3. Reference Conditions

The primary reference condition for the Spring Inflow indicator was established as 75%, a level based on the SWRCB's recommendation for freshwater inflows needed to support public trust resources in the estuary. Spring inflows that were greater than 75% of unimpaired inflows were considered to reflect "good" conditions. Additional information from trends and variability in spring inflows and from other estuaries was used to develop several other intermediate reference conditions. Table 2 below shows the quantitative reference conditions that were used to evaluate the results of the Spring Inflow indicator.

4. Results

Results of the Spring Inflow indicator are show in Figure 4.

The amount of fresh water flowing in the San Francisco Estuary during the spring has been reduced.
The percentage of the springtime runoff from estuary's largest watershed that flows into the estuary has been significantly reduced. For the most recent 10-year period (2001-2010), on average only 42% of unimpaired inflow actually flowed into the estuary. In 2010, only 32% of unimpaired inflow reached the estuary. In 2009, annual inflow only 27% of unimpaired, the seventh lowest percentage of freshwater inflow in the 81-year data record. In 12 of the past 20 years (60% of years), the percentage of unimpaired flow that flowed into the estuary was less than 50%.

The proportional alteration in spring inflow to the estuary differs by water year type.
The greatest alterations to freshwater inflows (expressed as a percentage of estimated unimpaired inflow) occur in dry years. Since the 1950s, the percentages of unimpaired flow that reached the estuary averaged 32% in critically dry years, 41% in dry years, 52% in below normal years, 60% in above normal years and 76% in wet years.

Spring flow into the San Francisco Estuary, as a percentage of unimpaired flow, has declined over time.
The percentage of unimpaired flow that actually flowed into the estuary during the spring has declined significantly over the past several decades (regression, $p<0.001$). Significant declines in the percentage of unimpaired inflow reaching the estuary have occurred in all water years types except wet years (regression, all tests, $p<0.05$). Before construction of most of the major dams on the estuary's tributary rivers (1930-1943, the "pre-dam" period), an average of 79% of estimated unimpaired flow actually reached the estuary. By the 1980s, the percentage had decreased significantly to just 49% (1980-1989 average; t-test, $p<0.001$). The average for the most recent 10-year period, 42%, is somewhat lower but, due to the large inter-annual variability associated with hydrology, not significantly different than flows during the 1980s.

Based on spring inflows, CCMP goals to increase fresh water availability to the estuary have not been met.
Current spring inflows to the estuary are well below the 75% level identified by the SWRCB as necessary to protect public trust resources and estuarine health. Current inflows are also somewhat lower than those measured in the 1980s, the period during which the CCMP was developed and established.

C. Inter-annual Variation in Inflow

1. **Rationale**

Runoff from the Sacramento-San Joaquin watershed, which provides >90% of the total freshwater inflow to the San Francisco Estuary, varies dramatically from year to year, a function of California's temperate climate and unpredictable occurrence of droughts and floods. Just as the amount of freshwater inflow into an estuary is a physical and ecological driver that defines the quality and quantity of estuarine habitat (Jassby et al. 1995; Kimmerer 2002, 2004; Feyrer et al. 2008, 2010; Moyle and Bennett, 2008; Moyle et al., 2010), the inter-annual variability of freshwater inflows, a key feature of estuaries, drives spatial and temporal variability in the ecosystem and creates the dynamic habitat conditions upon which native fish and invertebrate species depend.

2. **Methods and Calculations**

The Inter-annual Variation in Inflow indicator measures the difference between the inter-annual variation in actual annual inflow to San Francisco Estuary and that of unimpaired annual inflow for the same period. For the two annual inflow measures, variation was measured as the standard deviation (expressed in units of thousands of acre-feet, TAF) for prior ten-year period that ended in the measured year. The indicator was calculated for each year (1939-2010) as the difference between the standard deviations (SD).

The indicator is calculated as:

Inter-annual Variation in Inflow (TAF)
= (SD in actual inflow for year$_{(0 \text{ to } -9)}$) – (SD in unimpaired inflow for year$_{(0 \text{ to } -9)}$).

By incorporating unimpaired inflow as a component of the indicator calculation, the Inter-annual Variation in Inflow indicator has been normalized to account for natural year-to-year variations in precipitation.

3. **Reference Conditions**

The primary reference condition for the Inter-annual Variation in Inflow indicator was established by calculating the difference in inter-annual variation of unimpaired annual inflows and unimpaired inflows that had been reduced by 15-25% (depending on water year type)[11] for the same period. Based on this calculation, the reference condition was set at -1700 TAF. Differences between inter-annual variation of actual and unimpaired inflows that were less than this (i.e., less negative) were considered to reflect “good” conditions. Additional information from trends and variability in inter-annual variability was used to develop several other intermediate reference conditions. Table 3 below shows the quantitative reference conditions that were used to evaluate the results of the Inter-annual Variation in Inflow indicator.

4. **Results**

---

[11] For calculation of the reference condition, unimpaired inflows<29,500 TAF (60% of years) were reduced by 25%, unimpaired inflows between 29,500 and 42,000 TAF were reduced 20%, and unimpaired inflows >42,000 TAF were reduced by 15%.

Results of the Inter-annual Variation in Inflow indicator are show in Figures 5 and 6.

Inter-annual variability in inflows to the San Francisco Estuary has varied substantially over time.

The magnitude of inter-annual variability of unimpaired and actual freshwater inflows to the San Francisco Estuary is itself highly variable, reflecting unpredictable periodic differences in total annual flows that can vary by an order of magnitude (i.e., high inter-annual variation and large standard deviation) as well as periodic sequences of years with relatively comparable annual flows (i.e., low inter-annual variation and low small standard deviation) (Figure 5). Over the 81-year data record, unimpaired annual flows since the early 1980s have been substantially more variable (1980-2010 average variability: 17,042 TAF) than annual unimpaired flows during the earlier 40 years (1939-1979 average variability: 12,908 TAF). Inter-annual variation in actual annual flows showed a similar pattern (1939-1980 average: 12,082 TAF compared to the 1981-2009 average: 14,900 TAF).

Inter-annual variability in inflows to the San Francisco Estuary has been reduced.

Since the late 1960s, when large storage dams on most the estuary's large tributary rivers were completed (i.e., the "post-dam" period), there has been a significant decrease in the inter-annual variability of actual inflows to the estuary compared to the inter-annual variability of unimpaired flows measured for the same 10-year periods (t-test, $p<0.001$) (Figure 6). For the 1939-1967 period, the average difference in variability between actual and unimpaired flows was -256 TAF compared to the average difference in variability for the 1968-2010 period of -2158 TAF. Since the 1980s, inter-annual variation in annual freshwater inflows has varied but not changed significantly: the difference between actual and unimpaired variation in the 1980s (1980-1989), -2315 TAF, is not significantly different than that measured in the 2000s (2000-2010), -1573 TAF (t-test, $p>0.05$).

Based on recent inter-annual variation of inflows to the estuary, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have been met in some years.

Since 2005, inter-annual variation in annual freshwater inflow to the estuary conditions have been above the reference condition developed based on the SWRCB flow criteria. However, inter-annual variation conditions were well below this reference condition for the decade prior to this and for 19 of the past 30 years. This most recent five-year period also coincides with a period of relatively low inter-annual variation in annual flows (see Figure 5).

D. Seasonal Variation in Inflow

1. Rationale

Freshwater inflow to the San Francisco Estuary varies dramatically within the year, reflecting both California's Mediterranean climate with its wet and dry seasons as well as the high elevations in estuary's Sacramento-San Joaquin watershed in which large proportions of precipitation fall as snow that melts and runs off to the rivers later in the spring and early summer (see Figure 1). These seasonal variations in inflow create different kinds of habitat, for example, large areas of low salinity open water habitat in the estuary (Kimmerer 2002, 2004; Moyle et al. 2010). They drive important ecological processes such as flooding, which transports sediment, nutrients and organisms downstream and promotes mixing and circulation of estuary waters. And they trigger and facilitate key life history stages of both plants and animals, including reproduction, dispersal and migration.

2. Methods and Calculations

The Seasonal Variation in Inflow indicator measures the difference between the seasonal (or intra-annual) variation in actual monthly average inflow to San Francisco Estuary and that of unimpaired monthly inflow for the same year. For the two monthly inflow measures, variation was measured as the standard deviation (expressed in units of cubic feet per second, cfs). The indicator was calculated for each year (1930-2010) as the difference between the standard deviations.

The indicator is calculated as:

> Seasonal Variation in Inflow (cfs)
> = (SD in actual monthly average inflow) – (SD in unimpaired monthly average inflow).

By incorporating unimpaired inflow as a component of the indicator calculation, the Seasonal Variation in Inflow indicator has been normalized to account for natural year-to-year variations in precipitation.

3. Reference Conditions

The primary reference condition for the Seasonal Variation in Inflow indicator was established by calculating the difference in seasonal variation of unimpaired monthly inflows and unimpaired inflows that had been reduced by 15-25% (depending on water year type)[12] for the same period. Based on this calculation, the reference condition was set at -6700 cfs. Differences between inter-annual variation of actual and unimpaired inflows that were less than this (i.e., less negative) were considered to reflect "good" conditions. Additional information from trends and variability in seasonal variability was used to develop several other intermediate reference conditions. Table 4 below shows the quantitative reference conditions that were used to evaluate the results of the Seasonal Variation in Inflow indicator.

4. Results

Results of the Seasonal Variation in Inflow indicator are show in Figures 7 and 8.

Seasonal variability in inflows to the San Francisco Estuary is directly related to hydrology.

---

[12] For calculation of the reference condition, unimpaired inflows<29,500 TAF (60% of years) were reduced by 25%, unimpaired inflows between 29,500 and 42,000 TAF were reduced 20%, and unimpaired inflows >42,000 TAF were reduced by 15%.

The magnitude of seasonal variation in unimpaired and actual freshwater inflows to the San Francisco Estuary varies directly with hydrology, as measured by unimpaired inflows: variability is high in wet years and low in dry years (regression, both tests, $p<0.001$) (Figure 7).

Seasonal variability in inflows to the San Francisco Estuary has been reduced.
Seasonal variability of freshwater inflows to the estuary has declined significantly (regression, $p<0.001$) (Figure 8). The decline began in the mid-1940s, when the first of large storage dams in the estuary's watershed were completed. In the "pre-dam" period (1930-1943), the difference between actual and unimpaired seasonal variability was -2200 cfs; by the 1980s the difference between actual and unimpaired seasonal variation was significantly larger, -9069 cfs (t-test, $p<0.05$). Since the 1980s, seasonal variation has continued to decline: in the 1990s, the average difference between actual and unimpaired variation fell to -9723 cfs and in the 2000s it averaged -11,644 cfs. In 2010, it was -19,587 cfs, the sixth greatest difference between actual and unimpaired seasonal variability in the 81-year record.

Based on recent seasonal variations of inflows to the estuary, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have not been met in most years.
Since the 1980s, the seasonal variability of freshwater inflows to the estuary have been below the reference conditions and not met the CCMP goals in 70% of years. In the most recent decade, the CCMP goal was met only once, in 2006.

E. Peak Flow

1. Rationale

High, or "peak", freshwater inflows to the San Francisco Estuary occur following winter rainstorms and during the spring snowmelt. High inflows transport sediment and nutrients to the estuary, increase mixing of estuarine waters, and create low salinity habitat in Suisun and San Pablo Bays (the upstream reaches of the estuary), conditions favorable for many estuary-dependent fish and invertebrate species. In rivers and estuaries, peak flows and the flood events they typically produce are also a form of "natural disturbance" (Kimmerer 2002, 2004; Moyle et al., 2010).

2. Methods and Calculations

The Peak Flow indicator measures the frequency, as number of days per year, of peak flows into the San Francisco Estuary, compared to the number of days that would be expected based on unimpaired runoff from the estuary's watershed. Peak flow was defined as the 5-day running average of actual freshwater inflow>50,000 cfs. Selection of this threshold value was based on two rationales: 1) flows of this magnitude shift the location of low salinity habitat[13] downstream to 50-60 km (depending on antecedent conditions), providing favorable conditions for many estuarine invertebrate and fish species; and 2) examination of DAYFLOW data suggested that flows above this threshold corresponded to winter rainfall events as well as some periods during

[13] The location of low salinity habitat in the San Francisco Estuary is often expressed in terms of X2, the distance in km from the Golden Gate to the 2 ppt isohaline.

the more prolonged spring snowmelt, therefore this indicator evaluated the estuary's responses to a key aspect of seasonal flow variation in its watershed.

The indicator is calculated as the difference between the actual number of days of peak flow per year and the expected number of days of peak flow per year:

Peak flow (days) = # days peak flow (actual) – # days peak flow (predicted).

Daily unimpaired flow data are available for only a few recent years therefore, to predict the number of days of peak flow per year under unimpaired conditions, a polynomial regression was developed based on actual flows from the 1930-1943 "pre-dam" period, before major storage dams were constructed on the watershed's large rivers (Figure 7). Water Year 1983, the year with the highest annual unimpaired inflow on record and during which flows were minimally affected by water management operations, was also included in this regression analysis to provide a high inflow value and anchor the regression. The regression equation is shown in Figure 9. For years in which the polynomial regression predicted a number of days of peak that was less than zero and in which the actual number of days of peak flows was zero, the indicator value (the difference between actual and predicted) was set to zero.[14] By incorporating peak flow frequency predictions based on pre-dam conditions as an estimate of unimpaired inflow as a component of the indicator calculation, the Peak Flow indicator has been normalized to account for natural year-to-year variations in precipitation.

3. Reference Conditions

The reference condition was established based on the 95% confidence interval for the polynomial regression developed from pre-dam and 1983 data (see Figure 9 above). Over most of the range of unimpaired inflows, the maximum value for the 95% confidence interval was 15 days. Therefore the reference condition was set at twice this value, or -30 days (i.e., 30 fewer days of peak flow compared to the number predicted based on unimpaired inflow). Differences between actual and predicted number of days of peak flow that were less than this (i.e., less negative) were considered to reflect "good" conditions. Additional information from the polynomial regression and trends and variability in peak flows was used to develop several other intermediate reference conditions. Table 5 below shows the quantitative reference conditions that were used to evaluate the results of the Peak Flow indicator

4. Results

Results of the Peak Flow indicator are show in Figure 10.

The frequency of peak flows into the San Francisco Estuary varies with water year type. Actual peak flow frequency (as number of days per year) is highest in wet years, when there are of 144 days of peak flow per year on average for the 80 year data record, lowest in critically dry years (<2 days/year). Dry years have an average of 13 days/years, below normal years an average of 48 days/year and above normal years an average of 85 days. Since 1944, after dams on most the estuary's large tributary rivers were completed, actual peak flow frequency is

[14] This occurred in only three years: 1931, 1976 and 1977.

significantly lower that would be predicted based on estimated unimpaired flow conditions (Mann-Whitney, $p<0.001$). There are an average of 12 fewer days of peak flows in critically dry years, 31 fewer days in dry years, 42 fewer days in below normal years, 54 fewer days in above normal years and 41 fewer days in wet year.

Peak flow frequency has declined over time.

Peak flow frequency, expressed as the difference between actual peak flow frequency and predicted peak flow frequency under estimated unimpaired flow conditions, is highly variable but has declined significantly over the 81-year period of record (regression, $p<0.001$). Most of the decline occurred after 1943, immediately following completion of most of the large dams on the estuary's large tributaries. However, since 1944, peak flow frequency has continued to decline over time in dry, above normal and wet years (regression, $p<0.05$; regression for critically dry years, $p=0.052$; regression for below normal years, $p=0.08$). On average, there are 36 fewer days of peak flows per year since the mid-1940s than during the 1930-1943 period. In the 1980s, peak flows were reduced by an average of 39 days. In the 2000s, there was an average of 48 fewer days of peak flows.

Based on recent peak flow frequency, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have been met in less than 50% of years.

In the most recent decade (as well as for the most recent 5-year period), the reduction in peak flow frequency has been greater (i.e., more negative) than the reference condition (-30 days) in 60% of years. Since 1980, the reference condition for peak flow frequency has not been met in 58% of years.

F. Dry Year Frequency

1. Rationale

California's Mediterranean climate is typified by unpredictable cycles of droughts and floods. Runoff from the Sacramento-San Joaquin watershed, which provides >90% of the total freshwater inflow to the San Francisco Estuary, can vary dramatically from year to year, and freshwater inflow to the San Francisco Estuary is a key physical and ecological driver that defines the quality and quantity of estuarine habitat (Jassby et al. 1995; Kimmerer 2002, 2004; Feyrer et al. 2008, 2010; Moyle and Bennett, 2008; Moyle et al., 2010). Water storage and diversions in the estuary's watershed reduce the amounts of fresh water that reach the estuary and can result in inflow conditions comparable to dry hydrological conditions in years when actual hydrological conditions in the watershed are not dry. In dry years, total annual freshwater inflow, seasonal variations in inflow and the quantity and quality of low-salinity estuarine habitat are all reduced, resulting in stressful conditions for native resident and migratory species that rely on the estuary. Multi-year sequences of dry years, or droughts, exacerbate these stressful conditions and often correspond to population declines and shifts and/or decreases in species' distributions.

2. Methods and Calculations

The Dry Year Frequency indicator measures the difference between the frequency of critically dry years based on estimated unimpaired freshwater inflows to the estuary (and actual hydrological conditions in the Sacramento-San Joaquin watershed) and the frequency of critically dry years experienced by the estuary based on actual freshwater inflows. Critically dry (CD) years were defined as the driest 20% of years in the 80-year estimated unimpaired Delta outflows dataset, with total annual inflows to the estuary of less than 15,000 thousand acre-feet (see Table 6).

For the indicator, actual total annual freshwater inflows to the estuary for each year were categorized using this water year type classification scale. For each year (1939-2010), the number of CD years that occurred for the prior ten-year period that ended in the measured year was calculated for both unimpaired flows and actual flows. The indicator measured the difference between the number of CD years that occurred under unimpaired conditions and the number that occurred in actual conditions.

The indicator is calculated as:

Dry Year Frequency
= (# CD years, unimpaired inflow conditions for year$_{(0 \text{ to } -9)}$) – (# CD years, actual inflow conditions for year$_{(0 \text{ to } -9)}$).

By incorporating unimpaired inflow as a component of the indicator calculation, the Dry Year Frequency indicator has been normalized to account for natural year-to-year variations in precipitation.

3. Reference Conditions

The reference condition for the Dry Year Frequency indicator was established by calculating the average difference between CD frequency in unimpaired inflows and for unimpaired inflows that had been reduced by 15-25% (depending on water year type).[15] Based on this calculation, the reference condition was set at 1.5 years. Differences in the numbers of CD years between 10-year sequences of actual and unimpaired flows that were less than this were considered to reflect "good" conditions. Additional information from trends and variability in CD year frequency was used to develop several other intermediate reference conditions. Table 7 below shows the quantitative reference conditions that were used to evaluate the results of the Dry Year Frequency indicator.

4. Results

Results of the Dry Year Frequency indicator are show in Figures 11 and 12.

The frequency of critically dry inflows to the San Francisco Estuary has varied over time.

[15] For calculation of the reference condition, unimpaired inflows<29,500 TAF (60% of years) were reduced by 25%, unimpaired inflows between 29,500 and 42,000 TAF were reduced 20%, and unimpaired inflows >42,000 TAF were reduced by 15%.

While the classification of critically dry (CD) year inflows is based on the bottom quintile from the 80-year unimpaired dataset, the frequency of critically dry hydrological conditions (i.e., hydrological conditions that result in CD freshwater inflow to the estuary) has been more variable over that period (Figure 11, upper panel). The number of CD years per 10 year period for unimpaired conditions ranged from zero, during the 1950s and 1960s, to as high as six out of ten years, during the late 1980s and early 1990s. For actual conditions, which were affected by the amounts of water stored and diverted from the estuary's watershed, the frequency of freshwater inflows in amounts comparable to what the estuary would experience in CD years under unimpaired conditions, was higher (Figure 11, bottom panel, and Figure12). The largest increases in CD year frequency occurred in the 1960s, a period during which there were no CD years based on hydrological conditions in the estuary's watershed, but during which the estuary received freshwater inflows comparable to CD conditions in an average of six out of 10 years. In the 1980s, an average of 1.8 years were critically dry in the watershed but in the estuary an average of 4.4 years were critically dry (i.e., there were an average of 2.6 more CD years out of 10 years than there were based on hydrological conditions in the estuary's watershed). Conditions during the most recent decade (2001-2010) were similar, with an average of 4.8 CD out of 10 years for the estuary compared to just 2.7 CD years based on unimpaired conditions in the estuary's watershed.

The frequency of freshwater inflow conditions in the San Francisco Estuary that are comparable to critically dry years has increased.
Since 1944, when major dams on the estuary's tributary rivers were completed, the frequency of freshwater inflow conditions that correspond to CD years has increased significantly (Wilcoxon Signed Rank test, $p<0.001$) (Figure 12). On average, the estuary experienced 2.8 more CD years per 10-year period than it would have based on estimated unimpaired inflows and actual hydrological conditions in its largest watershed. On the basis of actual freshwater inflows, the estuary is experiencing chronic drought conditions, particularly during the 1960s and 2000s when conditions in the estuary's watershed were not chronically dry.

Based on recent critically dry year frequencies in the estuary, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have not been met in most years.
Since 2003, the estuary has experienced two to five more years per 10-year period of CD freshwater inflow conditions that it would have under unimpaired conditions. As of 2010, eight of the past 10 years, or 80% of years, were, for the estuary, critically dry (compared to just three critically dry years, 30% of years, in estuary's watershed based on actual hydrological conditions). During the past 60 years, the frequency of critically dry conditions in the estuary has been 50% (30 of 60 years had actual total annual freshwater inflows <15,000 TAF) and the running 10-year frequency of CD conditions was greater than the reference condition in 50 years (83% of years).

V. Freshwater Inflow Index

The Freshwater Inflow Index aggregates the results of the six indicators (Annual Inflow, Spring Inflow, Inter-annual Variation in Inflow, Seasonal Variation in Inflow, Peak Flow and Dry Year Frequency) into a single number.

A. Index Calculation

For each year, the Freshwater Inflow Index is calculated by averaging the quantitative scores of the six indicators. Each indicator is weighted equally. An index score greater than 3 was interpreted to represent "good" conditions and an index score less than 1 was interpreted to represent "very poor" conditions.

B. Results

Results of the Freshwater Inflow Index are shown in Figure 13.

Freshwater inflow conditions in the estuary have declined.
Based on the Freshwater Inflow Index, freshwater inflow conditions to the estuary have declined significantly (regression, $p<0.001$). The decline in the Index is driven by declines in all six indicators of freshwater inflow conditions. Most of the decline occurred during the 1950s and 1960s, the period after and during which major dams on the majority of the estuary's largest tributary rives were completed. The Index fell from an average of 3.2 in the 1940s (1939-1949 average), to 2.7 in the 1950s and 1.8 in the 1960s. The Index was relatively stable during the 1970s and 1980s (1970-1979 average: 1.7; 1980-1989: 1.85), improved slightly in the 1990s, concurrent with an unusually wet sequence of years (1990-1999: 2.0) and then declined somewhat in the 2000s, falling to 1.65 (2000-2009 average). The 2010 Freshwater Inflow Index (0.67, or "very poor") was the lowest for the 72-year record, and the 2009 Index (0.83) was the second lowest.[16]

Based on the Freshwater Inflow Index, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have not been met.
Based on the Freshwater Inflow Index, freshwater inflow conditions in the San Francisco Estuary are "fair" in some years, "poor" in most years and "very poor" in the two most recent years (2009 and 2010). Degraded inflow conditions reflect severe reductions in the amounts of freshwater inflow in most years, substantial reductions in both year-to-year and seasonal variability of inflows, severe reductions in the frequency of peak flows and high frequencies of inflows comparable to critically dry conditions, in effect, chronic drought conditions. y of "very poor" conditions for Annual Inflow, Spring Inflow, Peak Flow and Dry Year Frequency, and "poor" conditions for Seasonal Variation.

C. Summary and Conclusions

Collectively the six indicators of the Freshwater Inflow Index provide a comprehensive assessment of the status and trends for freshwater inflow conditions to the San Francisco Estuary from it largest watershed. Each of the indicators shows significant alterations to inflows to the

[16] The 1993 Freshwater Inflow Index was also 0.83.

estuary, including reductions in the amounts of inflows, reductions in inter-annual and seasonal variability, reduced frequency of peak flows and increased frequency of annual inflows to the estuary that are comparable to the relatively rare critically dry hydrological conditions in the watershed. Table 8 summarizes the indicator results relative to the CCMP goals (as they are expressed by the reference conditions).

VI. Peer Review

The Freshwater Inflow indicators and index build upon the methods and indicators developed by The Bay Institute for the 2003 and 2005 Ecological Scorecard San Francisco Bay Index and for the San Francisco Estuary Partnership Indicators Consortium. The Bay Institute's Ecological Scorecard was developed with input and review by an expert panel that included Bruce Herbold (US EPA), James Karr (University of Washington, Seattle), Matt Kondolf (University of California, Berkeley), Pater Moyle (University of California, Davis), Fred Nichols (US Geological Survey, ret.), and Phillip Williams (Phillip B. Williams and Assoc.). The versions of the indicators and index presented in this report were also reviewed and revised according to the comments of Bruce Herbold and Luisa Valiela (US EPA).

VII. References

Feyrer, F., M. L. Nobriga and T. R. Sommer. 2008. Multidecadal trends for three declining fish species: habitat patterns and mechanisms in the San Francisco Estuary, California, USA. Can J. Fish. Aquat. Sci. 64:723-734.

Feyrer, F., K. Newman, M. Nobriga and T. Sommer. 2010. Modeling the Effects of Future Outflow on the Abiotic Habitat of an Imperiled Estuarine Fish. Estuaries and Coasts DOI 10.1007/s12237-010-9343-9.

Jassby, A.D., W. J. Kimmerer, S. G. Monismith, C. Armour J. E. Cloern, T. M. Powell, J. R. Schubel and T. J. Vendlinski. 1995. Isohaline Position as a Habitat Indicator for Estuarine Populations. Ecological Applications 5:272-289.

Kimmerer, W. J. 2002. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25:1275-1290.

Kimmerer, W. J. 2004. Open-Water Processes of the San Francisco Estuary: from physical forcing to biological responses. San Francisco Estuary and Watershed Science [online serial]. Vol. 2, Issue 1 (February 2004), Article 1.

Kimmerer, W. J., E. S. Gross and M. L. MacWilliams. 2008. Is the Response of Estuarine Nekton to Freshwater Flow in the San Francisco Estuary Explained by Variation in Habitat Volume? Estuaries and Coasts 32:375-389. Available at: http://bestscience.org/docs/seminar/KimmererEtAl2009EstuariesCoasts%5B1%5D.pdf

Moyle, P. B. and W.A. Bennett. 2008. The future of the Delta ecosystem and its fish. Technical Appendix D. Comparing Futures for the Sacramento-San Joaquin Delta.

Public Policy Institute of California. San Francisco, CA. 1-38.

Moyle, P.B., W.A. Bennett, W.E. Fleenor, and J.R. Lund. 2010. Habitat variability and complexity in the Upper San Francisco Estuary. Working Paper, Delta Solutions, Center for Watershed Sciences, University of California, Davis.

SWRCB (2010) Development of Flow Criteria for the Sacramento-San Joaquin Delta Ecosystem. State Water Resources Control Board report prepared pursuant to the Sacramento-San Joaquin Delta Reform Act of 2009, August 3, 2010. Available at: http://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/deltaflow/final_rpt.shtml.

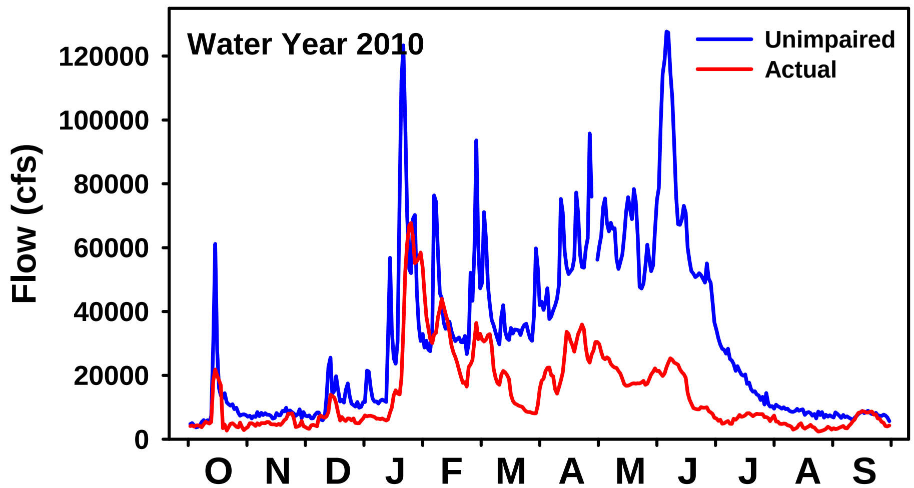


Figure 1. The amounts and timing of freshwater inflows to the San Francisco Estuary have been altered by dams and water diversions in the estuary's watershed. For Water Year 2010, this graph compares freshwater inflow conditions that would have occurred if there were no dams and water diversions, referred to as "unimpaired" conditions (blue line), with actual freshwater inflows (red line). Data sources: California Department of Water Resources and California Data Exchange Center.

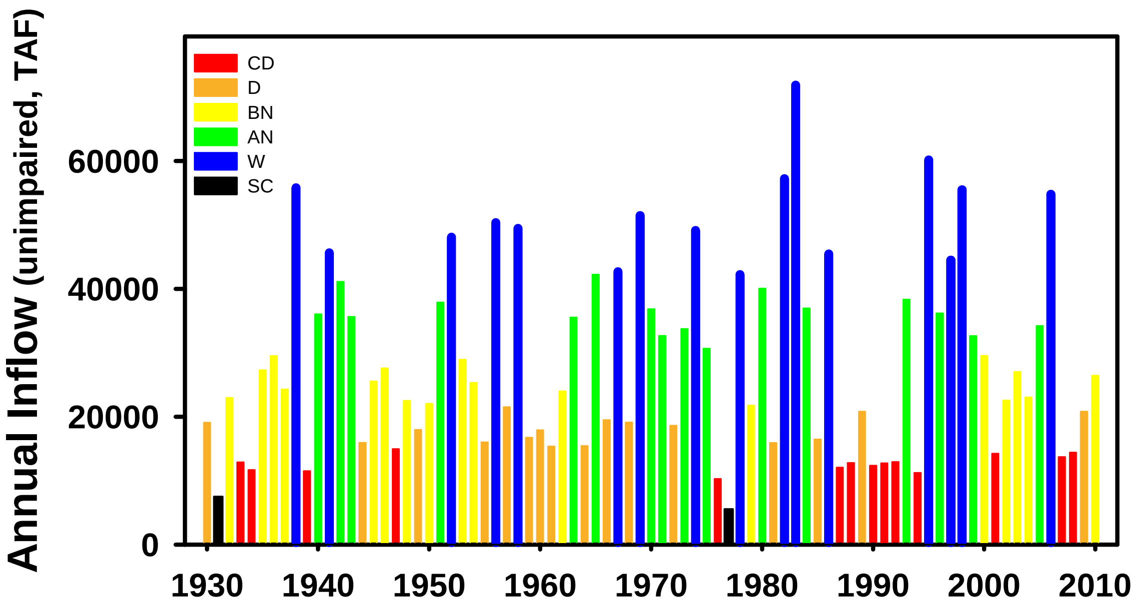


Figure 2. Freshwater inflow to the San Francisco Estuary varies from year-to-year. This graph shows the variations in unimpaired inflow for the 1930-2010, with the different water year types (critically dry, dry, below normal, above normal and wet) shown by the colors of the bars. In this graph, the driest 2.5% of years, or "super-critically dry" years, are also shown with the black bars.

Table 1. Quantitative reference conditions and associated interpretations for results of the Annual Inflow indicator. The primary reference condition, which corresponds to “good” conditions, is in bold.

Annual Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>87.5% of unimpaired	“Excellent”	4
>75% of unimpaired	**“Good”**	**3**
>62.5% of unimpaired	“Fair”	2
>50% of unimpaired	“Poor”	1
≤50% of unimpaired	“Very Poor”	0

Figure 3. Changes in the Annual Inflow indicator for the San Francisco Estuary, expressed as the percentage of estimated unimpaired flow that reaches the estuary, from 1930-2010. The top panel shows the results as the decadal averages and the bottom panel shows results for each year. Horizontal dashed line shows the reference condition (75%).

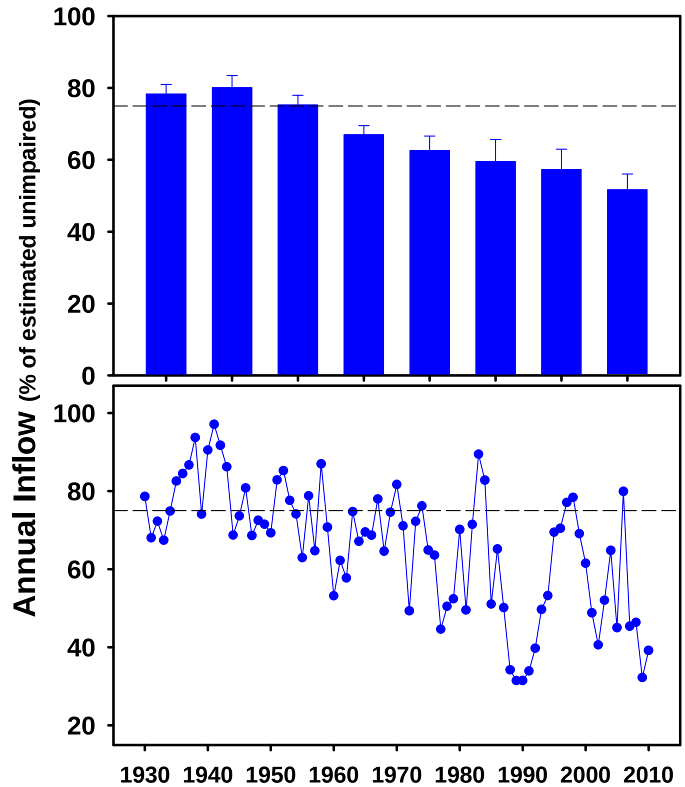


Table 2. Quantitative reference conditions and associated interpretations for results of the Spring Inflow indicator. The primary reference condition, which corresponds to "good" conditions, is in bold.

Spring Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>87.5% of unimpaired	"Excellent"	4
>75% of unimpaired	**"Good"**	**3**
>62.5% of unimpaired	"Fair"	2
>50% of unimpaired	"Poor"	1
≤50% of unimpaired	"Very Poor"	0

Figure 4. Changes in the Spring Inflow indicator for the San Francisco Estuary, expressed as the percentage of estimated unimpaired flow that reaches the estuary, from 1930-2010. The top panel shows the results as the decadal averages and the bottom panel shows results for each year. Horizontal dashed line shows the reference condition (75%).

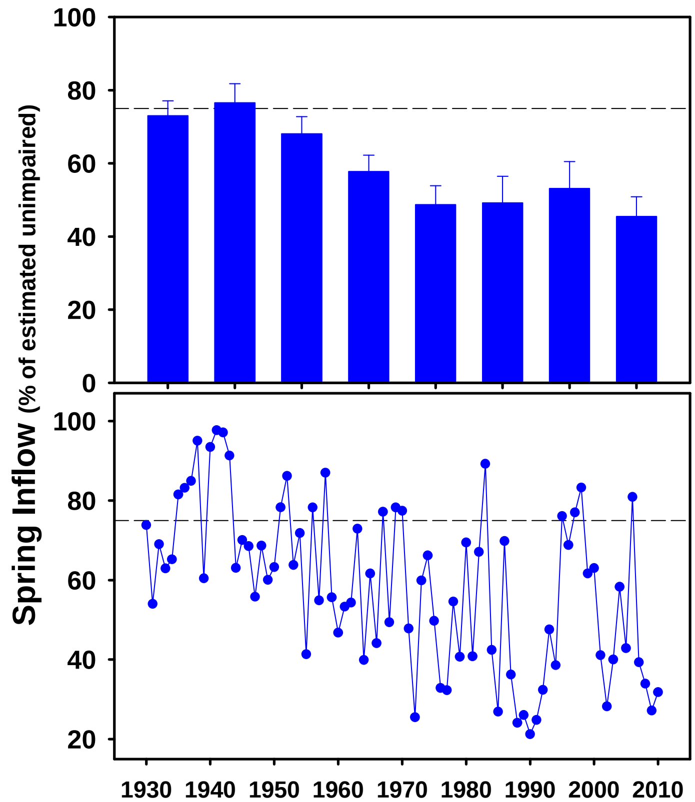
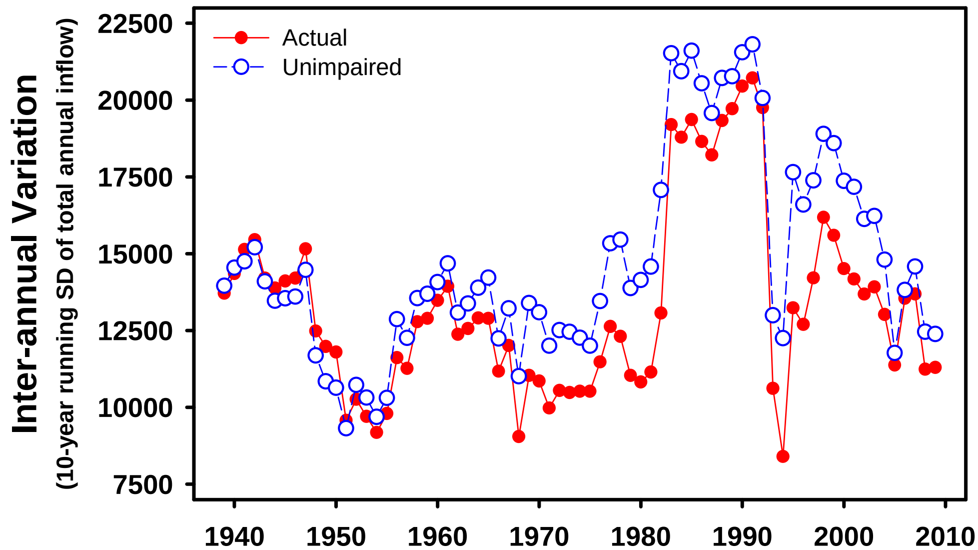


Table 3. Quantitative reference conditions and associated interpretations for results of the Inter-annual Variation in Inflow indicator. The primary reference condition, which corresponds to “good” conditions, is in bold.

Inter-annual Variation in Inflow		
Quantitative Reference Condition	Evaluation and Interpretation	Score
>-850 TAF	“Excellent”	4
>-1700 TAF	**“Good”**	**3**
>-2550 TAF	“Fair”	2
>-3400 TAF	“Poor”	1
≤-3400 TAF	“Very Poor”	0

Figure 5. Inter-annual variation in actual and unimpaired annual freshwater inflows to the San Francisco Estuary from 1939-2010 (each point is the standard deviation for running 10-year periods ending in that year).



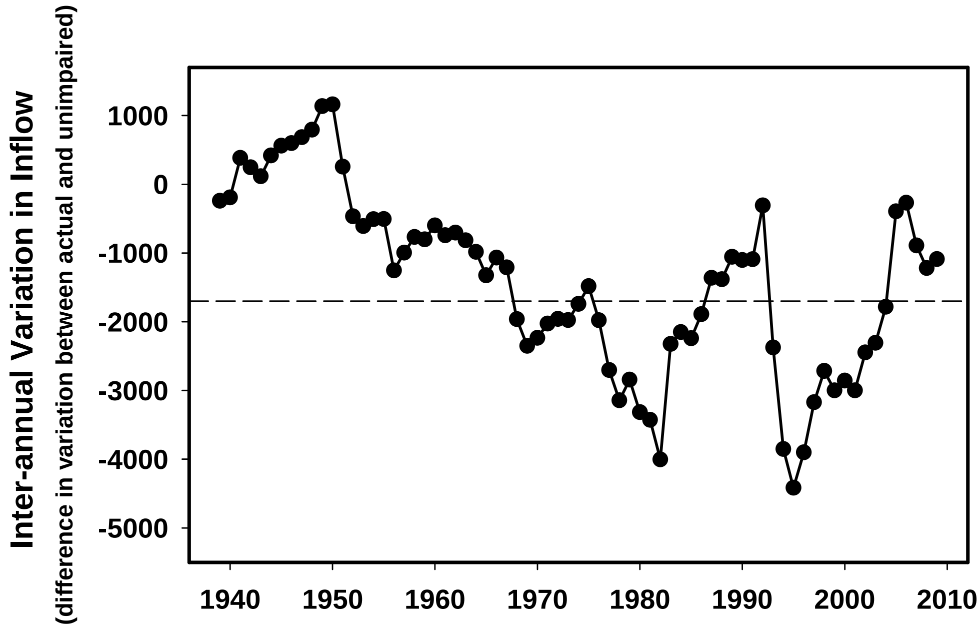


Figure 6. Changes in the Inter-annual Variation in Inflow indicator, expressed as the difference between actual and unimpaired inter-annual variations in inflow to the San Francisco Estuary, from 1939-2010. Horizontal dashed line shows the reference condition (-1700 TAF).

Table 4. Quantitative reference conditions and associated interpretations for results of the Seasonal Variation in Inflow indicator. The primary reference condition, which corresponds to “good” conditions, is in bold.

Seasonal Variation in Inflow		
Quantitative Reference Condition	Evaluation and Interpretation	Score
>0 cfs	“Excellent”	4
>-6700 cfs	**“Good”**	**3**
>-13,400 cfs	“Fair”	2
>-20,100 cfs	“Poor”	1
≤-20,100 cfs	“Very Poor”	0

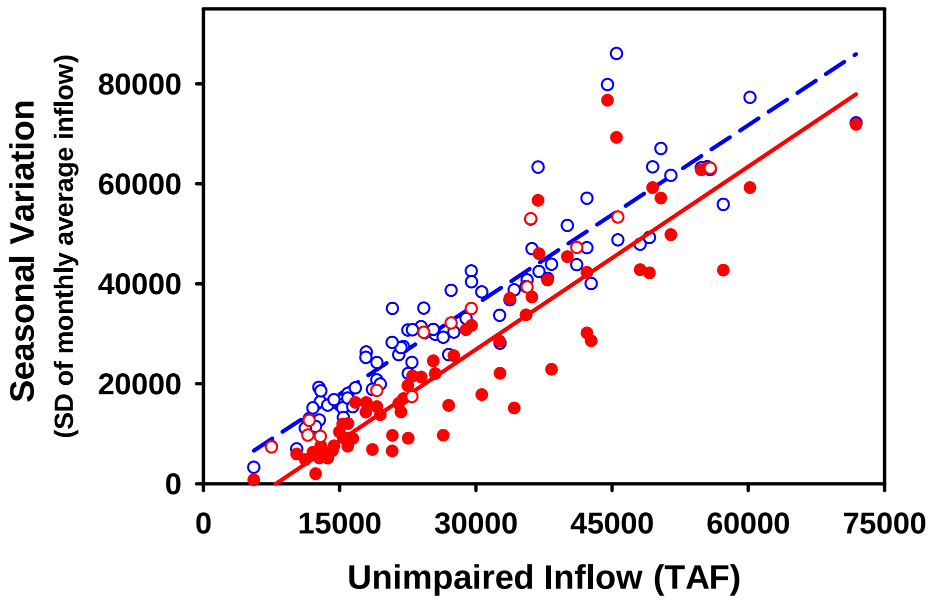


Fig 7. Seasonal variability in freshwater inflows (SD of average monthly flows, cfs; Y axis) is directly related to hydrology, as expressed by unimpaired inflow to the estuary (TAF; X axis). Seasonal variability of unimpaired inflows is shown in the blue open symbols, actual), seasonal variability of actual flows is shown in the open red symbols (1930-1943, the pre-dam period) and solid red circles (1944-2010).

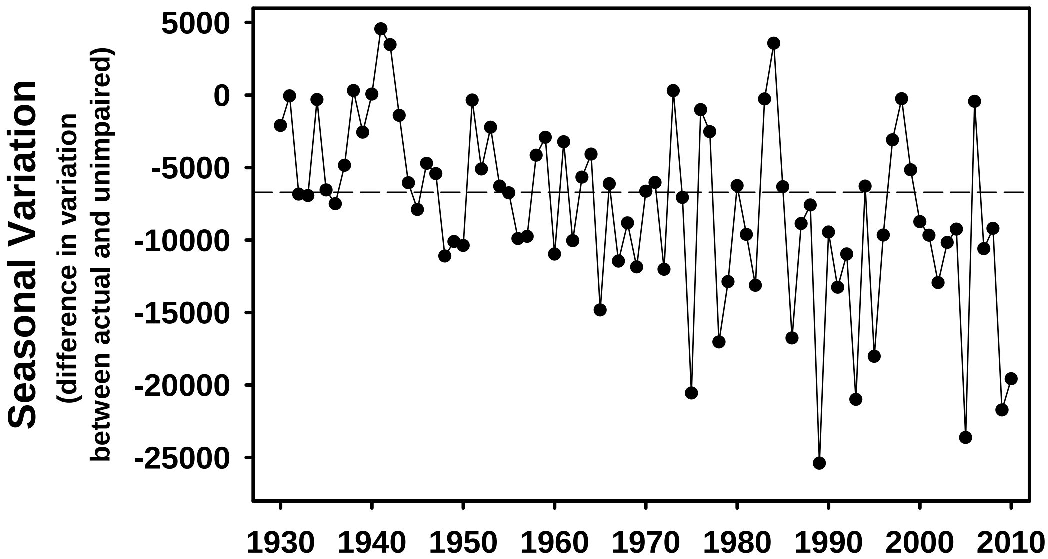


Figure 8. Changes in the Seasonal Variability in Inflows indicator, expressed as the difference between actual and unimpaired seasonal variations in inflow to the San Francisco Estuary, from 1939-2010. Horizontal dashed line shows the reference condition (-6700 cfs).

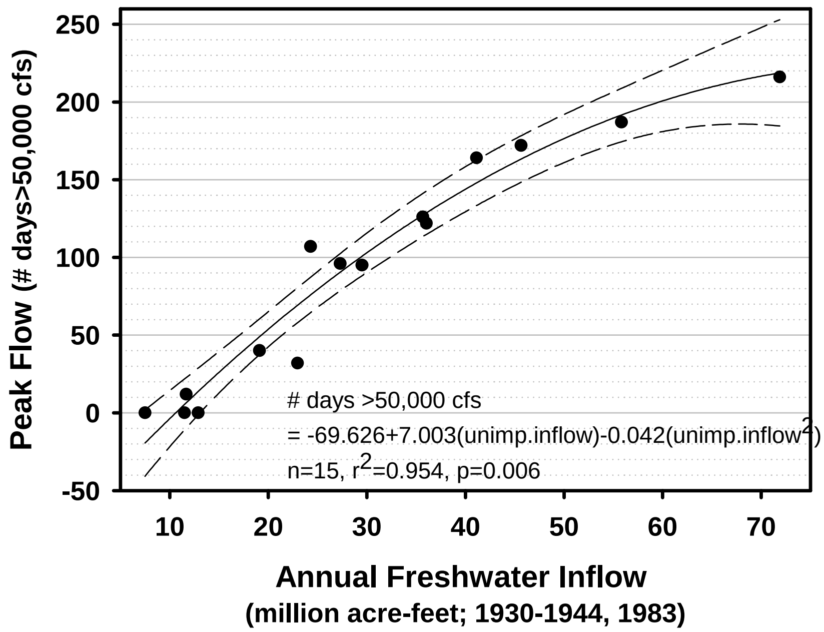


Figure 9. Actual (symbols) and predicted (regression with confidence limits) number of days of peak flow per year in relation to total annual inflow for 1930-1943 and 1983. This relationship was used to establish the reference condition for evaluation of the Peak Flow indicator.

Table 5. Quantitative reference conditions and associated interpretations for results of the Peak Flow indicator. The primary reference condition, which corresponds to “good” conditions, is in bold.

Peak Flow		
Quantitative Reference Condition	Evaluation and Interpretation	Score
>-15 days	“Excellent”	4
>-30 days	**“Good”**	**3**
>-45 days	“Fair”	2
>-60 days	“Poor”	1
≤-60 days	“Very Poor”	0

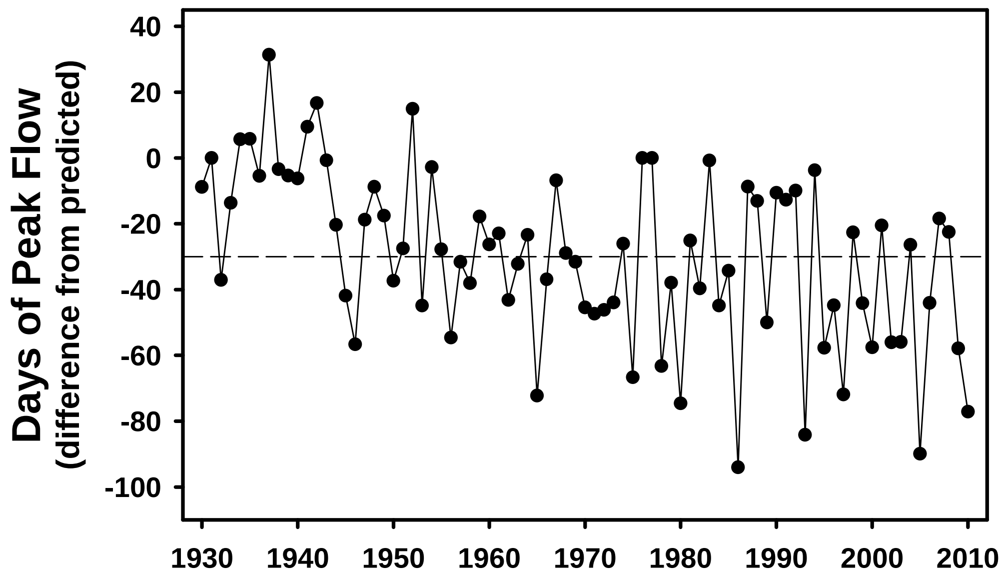


Figure 10. Changes in the Peak Flow indicator, expressed as the number of days different from predicted, for the San Francisco Estuary, from 1930-2010. Horizontal dashed line shows the reference condition (-30 days).

Table 6. Frequency-based classification of water years types based on estimated unimpaired annual inflow to the San Francisco Estuary.

Water year type	Estimated unimpaired inflow to the San Francisco Estuary (total annual, TAF)	Years (1930-2009)
Critically dry (driest 20% of years) NOTE: a "super-critical" category, corresponding to the driest 2.5% of years was also identified (see Figure 2)	<15,000 TAF (Super-critical: <8,000 TAF)	**1931**, 1933, 1934, 1939, 1947, 1976, **1977**, 1987, 1988, 1990, 1991, 1992, 1994, 2001, 2007, 2008 (n=16) Super-critical years are shown in **bold**.
Dry	15,000-21,500 TAF	1930, 1944, 1949, 1955, 1957, 1959, 1960, 1961, 1964, 1966, 1968, 1972, 1981, 1985, 1989, 2009 (n=16)
Below normal	21,500-29,500 TAF	1932, 1935, 1936, 1937, 1945, 1946, 1948, 1950, 1953, 1954, 1962, 1979, 2000, 2002, 2003, 2004 (n=16)
Above normal	29,500-42,000 TAF	1940, 1942, 1943, 1951, 1963, 1965, 1970, 1971, 1973, 1975, 1980, 1984, 1993, 1996, 1999, 2005 (n=16)
Wet (wettest 20% of years)	>42,000 TAF	1938, 1941, 1952, 1956, 1958, 1967, 1969, 1974, 1978, 1982, 1983, 1986, 1995, 1997, 1998, 2006 (n=16)

Table 7. Quantitative reference conditions and associated interpretations for results of the Dry Year Frequency indicator. The primary reference condition, which corresponds to “good” conditions, is in bold.

Dry Year Frequency		
Quantitative Reference Condition	Evaluation and Interpretation	Score
<1 year	“Excellent”	4
<2 years	**“Good”**	**3**
<3 years	“Fair”	2
<4 years	“Poor”	1
≥4 years	“Very Poor”	0

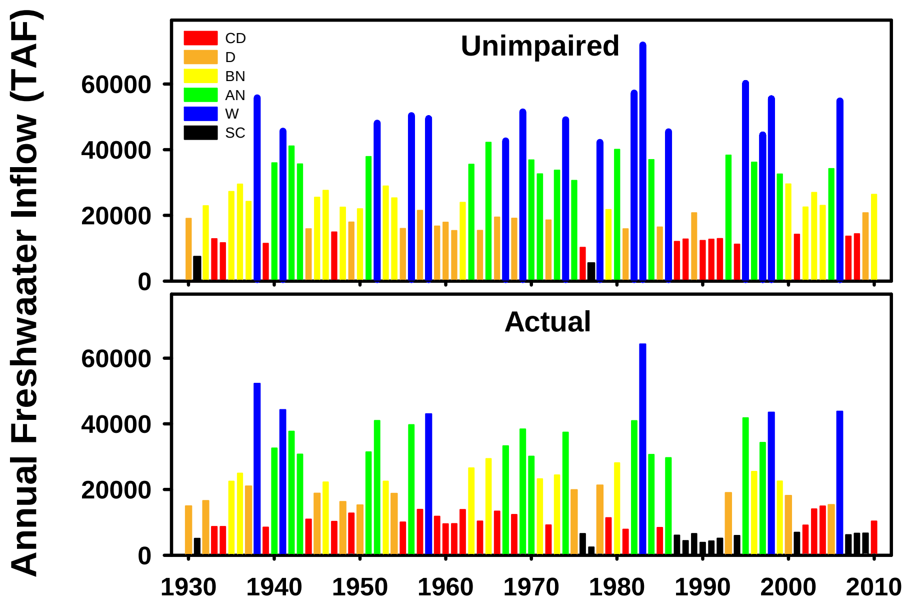


Figure 11. Freshwater inflows to the San Francisco Estuary under unimpaired conditions (top panel) and actual conditions (bottom panel). Each histogram bar has been colored to show the frequency-based water year type classification for unimpaired flows. The critically dry category was further partitioned to show "super-critical" years, comparable to the driest 2.5% of years of unimpaired flows.

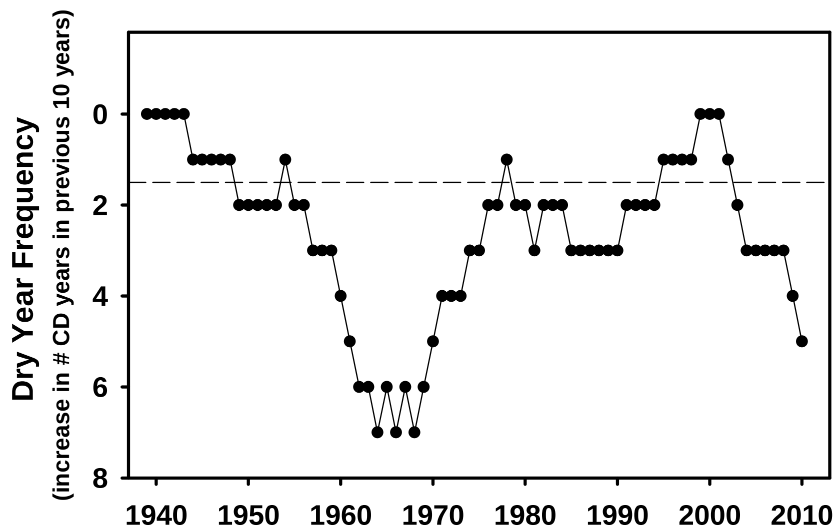


Figure 12. Changes in the Dry Year Frequency indicator, expressed as number of years more in a 10-year period)of critically dry freshwater inflow conditions to the San Francisco Estuary, from 1939-2010. Horizontal dashed line shows the reference condition (1.5 years).

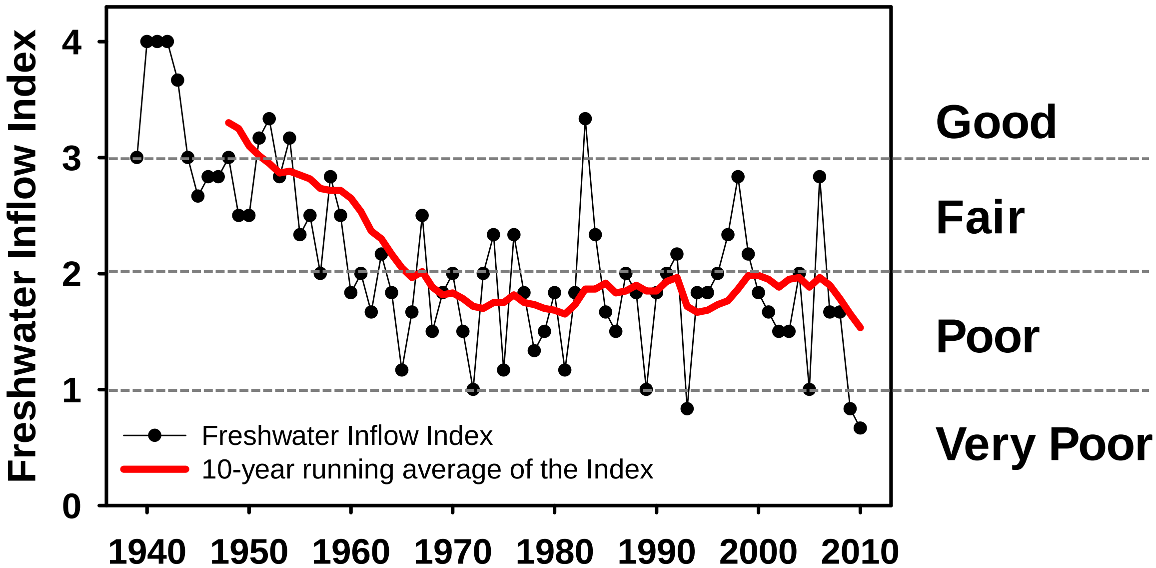


Figure 13. Changes in the Freshwater Inflow Index for the San Francisco Estuary from 1939-2010. Black lines and symbols show the annual Index values, solid red line shows the 10-year running average for the Index. Horizontal dashed lines shows the reference conditions and associated interpretations.

Table 8. Summary of results, relative to the CCMP goals to “increase freshwater availability to the estuary”, “restore healthy estuarine habitat” and “promote restoration and enhancement of stream and wetland functions to enhance resiliency and reduce pollution in the Estuary,” of the six freshwater inflow indicators for the San Francisco Estuary.

Indicator	CCMP Goal Met (yes, no or % met)	
	Past 10 years	Past 5 years
Annual Inflow	No (not met in 90% of years)	No (not met in 80% of years)
Spring Inflow	No (not met in 90% of years)	No (not met in 80% of years)
Inter-annual Variation in Inflow	Partially met (50% of years)	Yes
Seasonal Variation in Inflow	No (not met in 90% of years)	No (not met in 80% of years)
Peak Flow	Partially met (40% of years)	Partially met (40% of years)
Dry Year Frequency	Partially met (30% of years)	No

State of San Francisco Bay 2011
Appendix C

HABITAT - Estuarine Open Water Habitat Indicator
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background and Rationale

In an estuary, the place where fresh water from its tributary rivers begins to meet and mix with saltwater from the ocean is one of its most important habitats. The location, quantity and quality of this low-salinity habitat is largely determined by the amount of freshwater inflow. In the San Francisco Bay, the location of the low salinity zone and the associated amount and quality of this habitat is measured in terms of "X2," the point (in kilometers [km] upstream from the Golden Gate) where the salinity of the water near the bottom is 2 parts per thousand (approximately 6% seawater) (Jassby et al. 1995, Kimmerer 2002, 2004; Feyrer et al., 2007, 2010). During the spring, high freshwater inflows driven by rain and snowmelt in the Bay's watershed shift X2 and low salinity habitat downstream into the broad shallow reaches of Suisun Bay, creating a large expanse of estuarine open water habitat (Figure 1: X2 is low, closer to the Golden Gate). When springtime inflows are low, fresh and ocean waters mix farther upstream, X2 increases and the quality and quantity of the estuary's low salinity habitat is reduced. For a number of estuary-dependent fish and invertebrate species, each 10-kilometer upstream shift in average springtime X2 corresponds to a two- to five-fold decrease in abundance or survival (Kimmerer 2002, 2004; Kimmerer et al. 2009).

Springtime runoff from the Sacramento-San Joaquin watershed and freshwater inflow to the Bay varies dramatically from year to year, a function of California's Mediterranean climate and unpredictable occurrences of droughts and floods. However, since the 1960s, large dams on the Bay's major tributary rivers have captured and stored the majority of springtime snowmelt runoff in most years, with the result that less fresh water flows into the estuary during this ecologically sensitive period (see also Freshwater Inflow Index). Reduced spring inflows and more upstream locations of low salinity habitat affect the quality and quantity of the estuarine open water habitat and the plants and animals that use it.

It should be noted that the quantity and quality of low salinity open water habitat is important during all seasons, not just during the spring. For example, Feyrer et al. (2007, 2010) showed that the suitability of low salinity habitat during the fall (September-December) was important for two San Francisco Bay estuary-dependent fish species, delta smelt and striped bass, and that declines in fall habitat quality were significantly correlated with declines in delta smelt abundance. However, in the San Francisco Bay, the high magnitude freshwater inflows that create the largest amounts of low salinity open water habitat, the strongest relationships between low salinity habitat (and X2) and abundance and survival of estuarine species, and the greatest anthropogenic alteration in freshwater inflows all occur during the spring period (see also Freshwater Inflow Index). Therefore, this habitat indicator focuses on the springtime to evaluate the conditions and trends in the quantity and quality of this type of estuarine habitat.

The Estuarine Open Water Habitat indicator uses three measurements to assess the frequency ("how often?"), magnitude ("how much?") and duration ("how long?") of the occurrence of high quality estuarine open water habitat in the San Francisco Bay during the spring.

II. Data Source

The Estuarine Open Water Habitat indicator was calculated for each year using daily X2 data from the California Department of Water Resources (CDWR) DAYFLOW model. DAYFLOW is a computer model developed in 1978 as an accounting tool for calculating historical Delta outflow, X2 and other internal Delta flows.[1] DAYFLOW output is used extensively in studies by State and federal agencies, universities, and consultants. DAYFLOW output is available for the period 1930-2010.

III. Methods and Calculations

The Estuarine Open Water Habitat indicator uses three measurements to assess the frequency, magnitude and duration of the occurrence of high quality estuarine open water habitat in the San Francisco Estuary during the spring.

For each year, frequency was measured as:

> # of years in the past decade (i.e., ending with the measurement year) with X2<65 km for at least 100 days during the February-June period.

For each year, magnitude was measured as:

> average daily X2 during the February-June period.

For each year, duration was measured as:

> # of days with X2<65 km during the February-June period.

For each year, the Estuarine Open Water Habitat indicator was calculated by combining the results of the three measurements into a single number by calculating the average of the measurement "scores" described in the Indicator Evaluation and Reference Conditions section below.

VI. Indicator Evaluation and Reference Conditions

The San Francisco Estuary Partnership's Comprehensive Conservation and Management Plan's (CCMP) goal for "restor[ing] healthy estuarine habitat" is non-quantitative. However, based on the population and survival responses of a number of estuary-dependent species, estuarine open water habitat conditions with X2<65 km correspond to relatively good survival and abundance levels. In addition, based on review of X2 data from the "pre-dam" period (1930-1943, before large storage dams were constructed on most of the estuary's major Sacramento-San Joaquin watershed tributary rivers), open water habitat conditions with X2<65 km occurred for an average of 106 days during the February-June period and X2<65 km for more than 100 days in 71% of years. Therefore, the reference condition for high quality estuarine open water habitat conditions was set at X2<65 km for >100 days during the February-June period in seven out of ten years. Measured values that were above this reference condition were interpreted to correspond to "good" conditions. An additional "lower" reference condition was established to denote "poor" conditions. Measured values that were between the two reference conditions were

[1] More information about DAYFLOW is available at http://www.water.ca.gov/dayflow.

interpreted to correspond to “fair’ conditions. Table 1 shows the reference conditions and associated interpretations for the indicator metrics.

Results of the indicator and its component measurements were analyzed using analysis of variance and simple linear regression to identify differences among different time periods and trends with time.

V. Results

Results of the three component measurements of the Estuarine Open Water Habitat indicator are shown in Figure 2.

The frequency of occurrence of high quality estuarine open water habitat has declined (Figure 2, top panel).

Frequency of occurrence of high quality estuarine open water habitat during the spring has declined significantly (regression, $p<0.001$). The first decline occurred during the 1960s (when most of the large dams in the estuary’s main watershed were completed), with frequency falling from an average of 6.7 years out of 10 years in the 1940s and 1950s to an average of 4.6 years in the 1970s. Frequency declined again in the late 1980s and early 1990s during a severe multi-year drought, dropping to an average of just 1.9 years of good quality conditions per decade. Frequency increased during the late 1990s, concurrent with an unusually wet sequence of years, but then declined again in the 2000s. In the decade ending in 2010, the estuary experienced only two years (2005 and 2006) in which estuarine open water habitat conditions were “good.”

The quality and quantity of estuarine open water habitat has declined (Figure 2, middle panel).

As measured by average springtime X2 values, the quality and quantity of estuarine open water habitat has declined significantly (regression, $p<0.05$). Spring X2 conditions have degraded from an average of 62 km in the 1940s and 1950s to an average of 77 km in the late 1980s and early 1990s (1985-1994 average). In the 2000s, X2 averaged 69 km, significantly higher (i.e., poorer conditions) than during the 1940s and 1950s (t-test, $p<0.05$).

The duration of occurrence of high quality estuarine open water habitat has declined (Figure 2, bottom panel).

The number of days during the spring with “good” open water conditions and X2 downstream of 65 km has declined significantly (regression, $p<0.01$). Until the 1960s, X2 was downstream of 65 km for an average of 102 days during the February-June period. By the 1970s, the average had fallen to 69 days and, during the drought decade of the late 1980s and early 1990s, an average of only 22 days had “good” conditions. Conditions improved during the late 1990s but declined again in the 2000s. In the most recent ten years, X2 has been downstream of 65 km for an average of only 43 days during the spring and, in five of those years, daily X2 was never downstream of 65 km.

Results of the Estuarine Open Water Habitat indicator are shown in Figure 3.

Springtime estuarine open water habitat conditions have declined.

Results of the indicator reveal a steady and significant decline the springtime estuarine open water habitat (regression, p<0.001), from consistently “good” or “fair” conditions prior to the 1960s to mostly “poor” conditions by the 1990s. Conditions improved during the late 1990s, during a sequence of unusually wet years but declined again in the 2000s. Declining habitat conditions were driven by reductions in all three component measurements of the indicator. Frequency of occurrence of high quality open water habitat has been cut in half, from an average of seven out of ten years, or 70%, in the 1940s and 1950s to just 37% of years in the last decade. The location of springtime X2 has shifted nearly 7 kilometers upstream from an average of 62 kilometers to 69 kilometers in the 2000s. The number of days with “good” habitat conditions during the spring has declined by two thirds, from an average of more than 100 days per year in the 1940s and 1950s to just 43 days per year in the most recent decade.

Based on the Estuarine Open Water Habitat indicator, CCMP goals to restore healthy estuarine habitat and function have not been met.
The indicator shows that, for the past four decades, estuarine open water habitat conditions have been mostly “fair” or “poor.” Since the early 1990s, when the CCMP was implemented, open water habitat conditions in the estuary have been “good,” meeting the CCMP goal in just 19% of years. In the remaining 81% of years, open water habitat conditions have been “fair” (43% of years) or “poor” (38% of years).

VI. Peer Review

The Estuarine Open Water Habitat indicator builds upon the methods and indicators developed by The Bay Institute for the 2003 and 2005 Ecological Scorecard San Francisco Bay Index and for the San Francisco Estuary Partnership Indicators Consortium. The Bay Institute's Ecological Scorecard was developed with input and review by an expert panel that included Bruce Herbold (US EPA), James Karr (University of Washington, Seattle), Matt Kondolf (University of California, Berkeley), Pater Moyle (University of California, Davis), Fred Nichols (US Geological Survey, ret.), and Phillip Williams (Phillip B. Williams and Assoc.). The version of the indicator presented in this report was also reviewed and revised according to the comments of Bruce Herbold and Peter Vorster (The Bay Institute).

VII. References

Feyrer, F., M. L. Nobriga and T. R. Sommer. 2007. Multidecadal trends for three declining fish species: habitat patterns and mechanisms in the San Francisco Estuary, California, USA. Canadian Journal of Fisheries and Aquatic Sciences 64: 723-734.

Feyrer, F., K. Newman, M. Nobriga and T. Sommer. 2010. Modeling the Effects of Future Outflow on the Abiotic Habitat of an Imperiled Estuarine Fish. Estuaries and Coasts DOI 10.1007/s12237-010-9343-9.

Jassby, A.D., W. J. Kimmerer, S. G. Monismith, C. Armour J. E. Cloern, T. M. Powell, J. R. Schubel and T. J. Vendlinski. 1995. Isohaline Position as a Habitat Indicator for Estuarine Populations. Ecological Applications 5:272-289.

Kimmerer, W. J. 2002. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25:1275-1290.

Kimmerer, W. J. 2004. Open-Water Processes of the San Francisco Estuary: from physical forcing to biological responses. San Francisco Estuary and Watershed Science [online serial]. Vol. 2, Issue 1 (February 2004), Article 1.

Kimmerer, W.J., E.S. Gross, and M.L. MacWilliams. 2009. Is the Response of Estuarine Nekton to Freshwater Flow in the San Francisco Estuary Explained by Variation in Habitat Volume? Estuaries and Coasts 32:375-389.

The Bay Institute 2003. Ecological Scorecard: San Francisco Bay Index 2003. Available at: http://bay.org/assets/Scorecard_report.pdf.

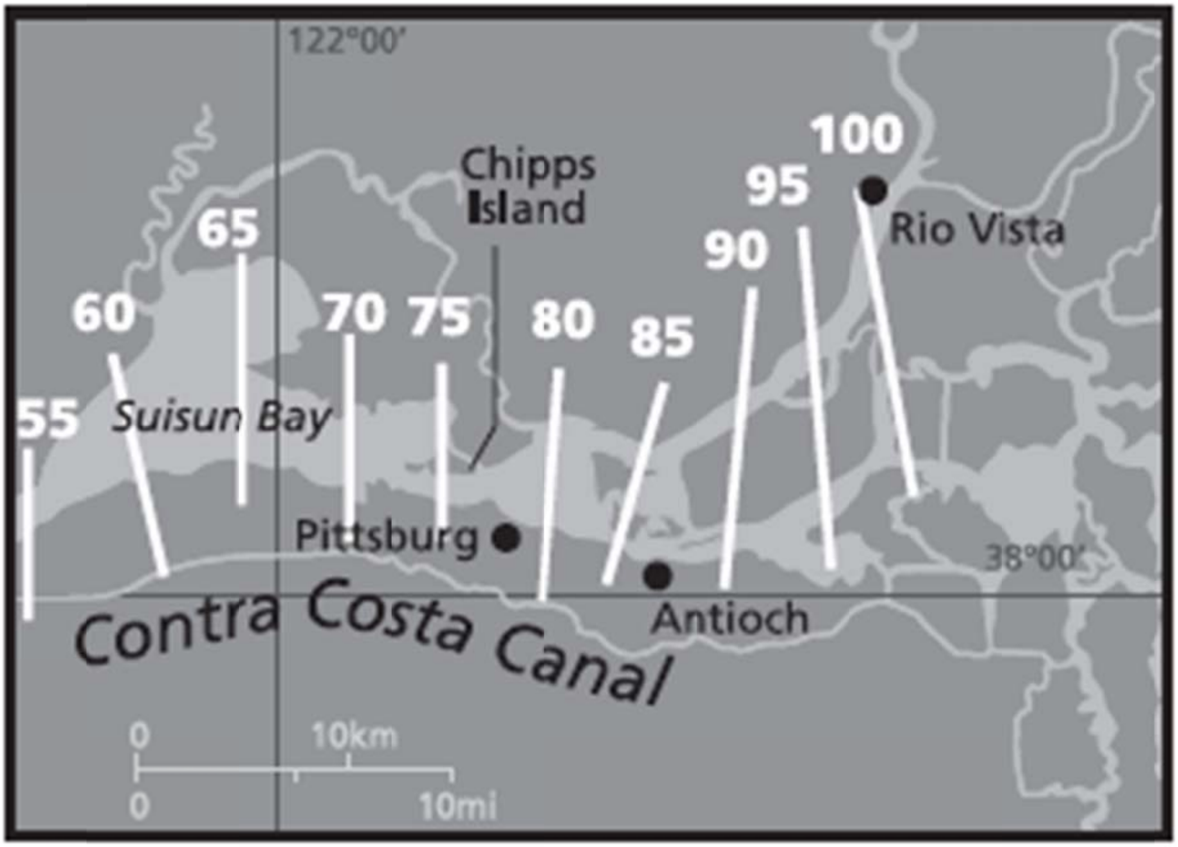


Figure 1. The location, quantity and quality of low salinity open water habitat is often measured in terms of "X2", the location in kilometers from the Golden Gate of the 2 parts per thousand isohaline. Based on survival and abundance of many estuary-dependent fish and invertebrate species, X2 locations at of downstream of 65 km provide good habitat conditions. Figure from: The Bay Institute, 2003.

Table 1. Quantitative reference conditions and associated interpretations for results of the three Estuarine Open Water Habitat indicator metrics. The primary reference condition, which corresponds to “good” conditions, is in bold.

Estuarine Open Water Habitat Indicator			
Metric	**“Good” Score=3**	“Fair” Score=2	“Poor” Score=1
Frequency	**≥7 years out of 10 years**	≥4 years out of 10 years	<4 years out of 10 years
Magnitude	**X2<65 km**	X2≥65 km and ≤75 km	X2>75 km
Duration	**>100 days**	>50 days	≤50 days

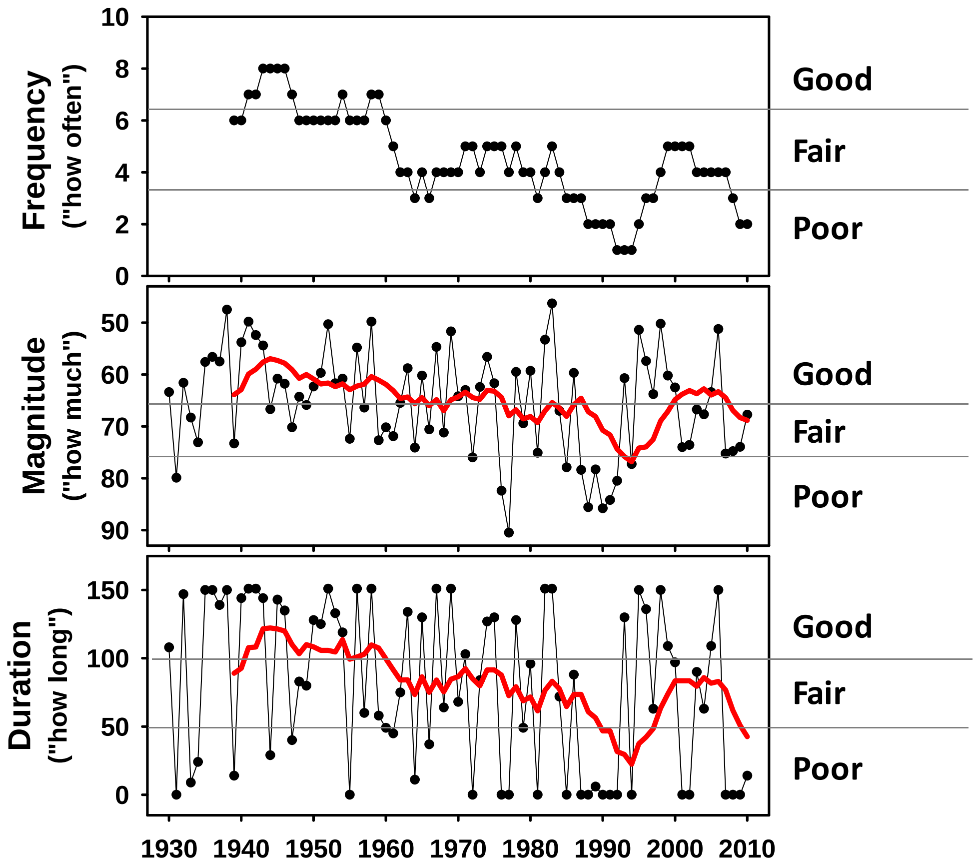
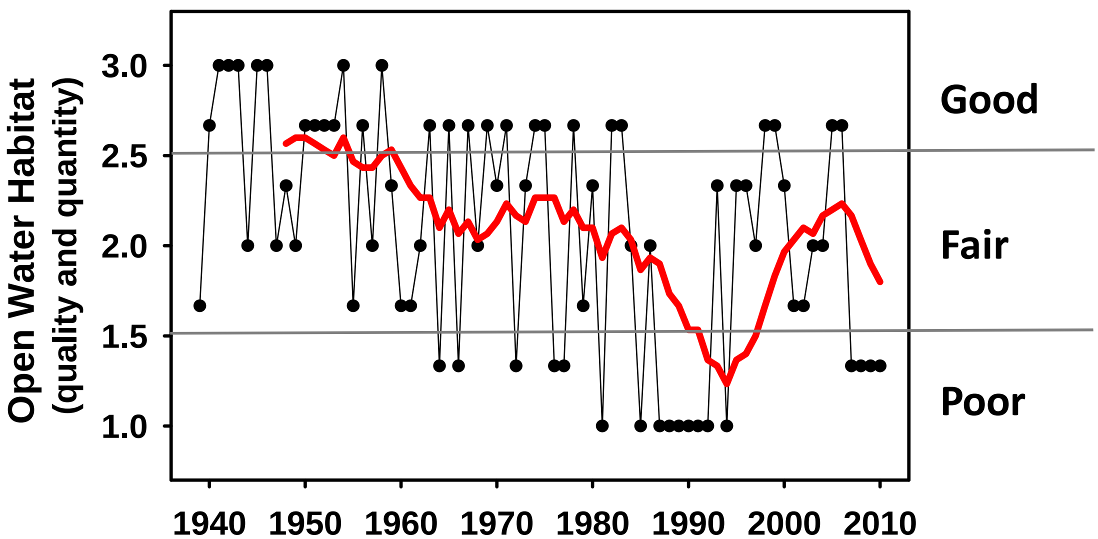


Figure 2. Changes in the frequency, magnitude and duration of occurrence of high quality estuarine open water habitat in the San Francisco Estuary during the spring, from 1930-2010. Black lines and symbols show the annual Index values, solid red line shows the 10-year running average for the Index. Horizontal dashed lines shows the reference conditions and associated interpretations.

Figure 3. Changes in the Estuarine Open Water Habitat indicator from 1939-2010. Black lines and symbols show the annual indicator values, solid red line shows the 10-year running average for the indicator. Horizontal dashed lines shows the reference conditions and associated interpretations.



State of San Francisco Bay 2011
Appendix D

HABITAT – Baylands and Watershed Indicators
Technical Appendix

Josh Collins, San Francisco Estuary Institute

Health Indicators of Intertidal and Watershed Wetlands

Background and Rationale

This report focuses on the health of Bay Area wetlands. The state is developing a wetland definition as part of its Wetland and Riparian Area Protection Policy. According to the proposed definition, wetlands are areas that, under normal circumstances, (1) are saturated by ground water or inundated by shallow surface water for a duration sufficient to cause anaerobic conditions within the upper substrate; (2) exhibit hydric substrate conditions indicative of such hydrology; and (3) either lack vegetation or the vegetation is dominated by hydrophytes (TAT 2009a). Shallow surface waters are less than 2 m deep. This is essentially the same definition used by the U.S. Army Corp of engineers (USACE) and U.S. environmental protection agency (USEPA) except that it explicitly includes areas that lack vegetation.

There are many kinds of wetlands, based on the recommended state definition. The main wetlands of the Estuary are the intertidal flats and marshes that adjoin the subtidal areas and open waters of the bays and straits. In the watersheds that drain to the Estuary, there are wetlands associated with lakes and ponds, rivers and streams, and springs and seeps. Some of these wetlands are perennial and others are seasonal. Seasonal wetlands can form along the beds of ephemeral or intermittent rivers and streams, and on hillsides and valley floors, where the root zone intercepts seasonally high groundwater. They also form in shallow depressions that fill with rain and slowly drain. Wetlands can be further classified according to differences in their sediment characteristics and plant community composition (Cowardin et al. 1979) and landscape position (Tiner 2003).

Wetlands provide many important services. They provide water and food, control pollution and flooding, and support wildlife and recreation. The types and level of services differ among the kinds of wetlands and their different settings. Some wetlands are managed to provide specific services, such as wastewater treatment or waterfowl hunting. Natural wetlands tend to provide broad suites of services. They are especially valued for their contribution to the native biological diversity of the region. Most of the region's rare and endangered plants and animals rely on wetlands for their survival. In the landscape context, wetlands are transitional between the wettest and the driest conditions. They tend to be very sensitive to changes in climate or land use that affect water supplies. In turn, wetlands influence water supplies by storing water and slowly releasing it downstream or into the ground. They increase the ability of landscapes to retain water and thus increase opportunities for land planners and managers to conserve water and use it wisely. The protection and restoration of wetlands can be significant aspects of our preparation and response to climate change, and of our overall approach to environmental health care. The public invests large sums of money each year to protect and restore wetlands through local, state, and federal policies and programs. The progress and success of these efforts can be assessed as the ability of wetlands to provide high levels of their desired or needed services.

There is an adequate foundation of scientific understanding to support separate assessments of many wetland services (e.g., Bedford and Preston 1988, Johnston et al. 1990, Brinson 1993, Carter 1997, Bayley 1999, MEA 2005). It is very unlikely, however, that funds and human resources will ever be adequate to assess every service for any kind of wetland. The assessments will need to focus on the services for which goals or benchmarks are set and for which adequate data can be assembled. The benchmarks are needed to define good health, or adequate service. The data are needed to track progress, improve our understanding, and to know when good health is achieved.

For this report, wetland assessment relies on a few metrics that integrate across many services. This approach is necessitated by a lack of benchmarks and data for all but a few wetland services, some of which are incorporated into separate assessments of water quality and wildlife support. The integrative metrics are useful for assessing the overall condition or health of wetlands.

Data Sources

Extent of Tidal Marshes and Tidal Flats

Definitions

For the purposes of this report, tidal marshes are intertidal areas at least 0.25 acres in size that support at least 5% cover of vascular vegetation, when viewed at a scale of 1:2,500. Tidal flats are similarly defined except that they lack at least 5% cover of vascular vegetation. For any given surface water salinity regime, tidal flats tend to occur lower in the intertidal zone.

The extent of marshes and flats is defined as their total acreage within the Estuary downstream of Broad Slough, which delimits the western boundary of the Delta. Measures of extent are based on maps. Expert mapping of marshes and flats requires defining their spatial limits using field surveys and remote data, such as aerial imagery. The upper (landward) margin of a marsh is recognized as a visually distinct shift from marsh vegetation to upland vegetation (e.g., Bodnar et al 1975, Culberson 2001). The lower limit is recognized as the shift to less than 5% cover of vegetation, which is generally abrupt. The upper limit of the flat is the same as the lower limit of the marsh. The lower limit of the flat is more difficult to discern. For the purposes of this report, the lower limit of tidal flat was taken directly from federal navigational charts, and represents an estimation of the mean lower-low tide datum, also referred to as the zero-tide height, based on depth soundings. The zero-tide contour is the conventional lower limit of the intertidal zone for the west coast of the U.S. (NOS 2000). The actual contour varies from year to year and the amount of this variability is generally unknown. The other boundaries of tidal flats and marshes are readily viewable in aerial imagery and have been verified in the field.

Tidal channels that are bounded by tidal marsh, almost entirely dewater at low tide, and are on average less than 200 ft wide are considered tidal marsh features, and are included in estimates of tidal marsh extent (see section below on marsh size). Marsh ponds, pannes, potholes, and other non-vegetated features of tidal marshes are also included in measures of marsh size. Portion of channels that are at least 200 ft wide and dewater at low tide are considered tidal flats features. Portions of channels that do not dewater at low tide are considered subtidal.

Historical Maps

There are multiple sources of maps of the tidal marshes and flats of the Estuary (U.S. Coast Survey 1857, U.S. Coast Survey 1897, Jones and Stokes Associates et al 1979, Dedrick 1989, Dedrick and Chu 1993, SFEI 2000, SFEI 2010). In order to describe historical and recent changes in the regional extent of marshes and flats, three strictly comparable maps are readily available. The earliest is the map of aquatic areas ca 1800 produced by SFEI in the 1990's as part of the Bay Area EcoAtlas (SFEI 2000, Goals project 1999). For the intertidal areas, the main source of data was the Topographic Sheets of the US Coast Survey that date from the mid nineteenth century (US Coast Survey 1857). For a thorough discussion of the early Coast Survey maps, go to the T-sheet User Guide (http://www.sfei.org/sites/default/files/T_sheet_user_guide_SFEI_highres_0.pdf).

The map of historical extent represents the expected average arrangement of wetlands and related habitats over the 400 years preceding Euro-American contact (i.e., 1400 to 1800 AD). For this region, the timeframe is characterized by moderate climatic variability with multiple droughts and wet periods lasting less than a decade each century. The period prior to 1400 AD is characterized by multi-decadal periods of persistent low and high rainfall and stream flow. The historical map also reflects indigenous land use practices, to the extent that they affected the distribution and shape of intertidal habitats. For example, there is anecdotal evidence that local tribes may have modified some tidal marsh pannes for salt production and waterfowl hunting.

Many additional sources of information were used to augment the Coast Survey maps. This included other eighteenth- and nineteenth-century maps, sketches, paintings, engineering reports, oral histories, explorers' journals, missionary texts, hunting magazines, oblique (land-based) photography, and interviews with living elders. Aerial photography did not exist at the time of the earliest maps. As part of the process of integrating these various data to create the best possible historical map, the relative certainty of each feature (e.g. a channel, marsh, or panne) was assessed, using a weight-of-evidence approach. In all regards, the historical map of tidal marshes and flats has a high level of certainty.

Modern Maps

Maps that are strictly comparable to the earliest historical compilation were made by SFEI during the late 1990s (SFEI 2000) and in 2010 (SFEI 2011). Other maps that pertain to the 1970s-80s (Jones and Stokes Associates et al 1979, Dedrick 1989) are not strictly comparable to each other or to the oldest and newest maps. They provide, however, a general picture of the declining extent of tidal marshes and little change in the extent of tidal flats during the latter half of the last century.

The standards used by SFEI to produce maps of marshes and flats published in 2000 and 2010 are essentially the same. The ca 2000 map integrates data collected from 1996-1999, and was revised in 2000 based on aerial imagery and an exhaustive account of the status of marsh restoration and mitigation projects to support the Wetland Tracker information system (CWQMC 2011). The ca 2010 map is an update of the ca 2000 map using the same kinds of data sources. The standards (http://www.wrmp.org/docs/SFEI%20MAPPING%20STANDARDS_01062011_v3.pdf) are being reviewed by state and federal interests as the basis for a statewide inventory of aquatic resources that would intensify the National Hydrographic Dataset (NHD) of USGS and the National Wetland Inventory (NWI) of USFWS. The regional pilot is called the Bay Area Aquatic Resource Inventory (BAARI; SFEI 2010) and serves as the base map for the Bay Area Wetland Tracker (http://www.californiawetlands.net/tracker/). The reader is referred to the BAARI standards as the source of detailed information about the methods and data sources used to produce the ca 2000 and ca 2010 regional maps of tidal habitats.

Methods and Calculations

The total acreage of tidal marshes and flats was measured separately for each of the three regional maps (ca 1800, 2000, and 2010) by compiling the acreage measures for each separate marsh and flat using the Geographic Information Systems (GIS) at SFEI. The separate marshes and flats were identified based on the definitions provided above (see also the section below on the sizes of marshes and flats).

Forecasts of the net change in extent for 2100 were derived by adding expected acres of projects for creating or restoring tidal marsh or flats to the ca 2010 map. Projects were included in the forecasts if they are represented by Notices of Intent, Environmental Impact Reports or Statements, environmental permit applications, existing permits, or strategic planning documents that are available to the public and well supported in concept by the community of agencies responsible for intertidal habitat protection. Much of information about the status of projects is available online as project-specific web sites and information linked to the Wetland Tracker, plus a backlog of project information that has been submitted to the Wetland Tracker but is not yet available online. Planned enhancements of the quality of existing marshes or flats were disregarded because they won't affect a net change in extent. Projects that are required as mitigation for unavoidable losses of marshes or flats due to land use were considered only in terms of their net effect. This necessitated accounting for both the acres of habitat mitigation and the associated acres of habitat loss. Such data are not always available. It is expected that the missing data are insignificant relative to the overall uncertainty of the forecasts. Some projects are represented by multiple plans that have accumulated over many years due to waxes and wanes in funding. In these cases only the most recent plans were considered. Some projects replicate information from over-arching, long-range strategic plans that they will help implement. Care was taken in these cases not to double-count expected changes in extent. Some agencies and interests have different names for the same project, necessitating a careful cross-reference between plans and places.

Despite these efforts to assure an acceptable level of quality for the data sources, the forecasts are very uncertain. The large-scale strategic plans, such as the Goals Project (Goals Project 1999), South Bay Salt Pond Restoration Project (Coastal Conservancy 2006) and the Suisun Restoration Plan (USBR et al. 2010) involve adaptively adjusting their habitat goals as experience is gained. They explicitly or implicitly provide a range of possible future changes in habitat extent. In these cases, the forecasts relied upon the more conservative expectations for change. Some projects that are just entering environmental review may not be implemented as currently designed, or at all. It was assumed, however, that they would be implemented as currently planned. Only the intended end points of projects were considered in the tallies of possible future net change, although it was recognized that some projects will initially provide shallow subtidal habitats or tidal flats before they evolve into tidal marsh.

Climate change and economic change will probably affect the outcome of every project. Perhaps most of the uncertainty relates to climate change. While the restoration of intertidal habitats is not inexpensive (Goals Project 1999, USFWS 2009), the public has supported initial implementation of large-scale projects in recent years. To some degree public support can be improved with education and outreach that highlights the success to-date. Climate change, however, can have a direct and lasting effect on project outcomes and it cannot be managed over the time span of these forecasts. The central question is whether or not newly created or restored marshes and flats, as well as those that have persisted until now, will survive increased rates of sea level rise, as affected by global climate warming. The question can be re-stated as whether or not supplies of new intertidal sediment will be adequate for flats and marshes to build upwards apace with sea level rise. Many factors and processes complicate the possible answers (e.g., French 1993, Orr et al. 2003, Callaway et al. 2007, Glick et al. 2007, Craft et al. 2009, Stevenson and Kearney 2009, US Climate Change Science Program 2009). The further into the future the forecasts are extended, the less certain and meaningful they are. The year 2100 is probably near the limit of reasonable forecasting (IPCC 2001, 2007). Despite these uncertainties, the forecasts in this report are based on the assumption that the planned projects will achieve their currently envisioned endpoints by about 2050, and that they will survive as envisioned to 2100.

Tidal Marsh and Tidal Flat Size

There are multiple approaches to assessing the size of tidal marshes and flats. Size can be measured based on extent, as explained above, and it can be measured based on function. For example, a marsh that is too small to support a viable population of one species of wildlife might be large enough for another species. An area of marsh that might be large enough to enhance flood control for one place in the Estuary might be too small for another place. In a general sense, whether a marsh or flat is large or small varies with the functions for which its size is being measured. Furthermore, the rules for deciding how to define the boundaries of a marsh or flat also vary with their functions. For example, whether or not a marsh is large enough to benefit a particular animal species depends on how much of the marsh it can safely access. In fact, most of the interest and concern about tidal marshes and flats relate to their function as habitat for native fishes, animals, and plants (USFWS 2010, BCDC 2008, SFBRWQCB 2010). Therefore, one useful way to look at marshes and flats is as habitat.

In 2002, SFEI began a study of intertidal habitat fragmentation in the Estuary as part of a west coast survey of estuary condition that was sponsored by the Environmental Monitoring and Assessment

Program (EMAP) of USEPA. A regional team of experts was assembled to recommend rules for using GIS to delineate patches of tidal marsh and tidal flat as habitat for different species of mammals and birds (Table 1). Geographic features, such as broad areas of pen water, major roads, and levees that tend to block or otherwise influence the dispersal or daily movements of these species were identified as patch boundaries. Different species required different rules, based on their different responses to the geographic features. The alternative sets of rules were applied to the historical map (ca 1800) and the most current map at that time (ca 2000). The results illustrate that the marshes and flats have become more fragmented for some species than for others (Collins et al. 2005). It is important to note that the default rules for defining separate tidal marshes for these two maps and for the more recent map (ca 2010) follow "Alternative 1" for tidal marshes as described in Table 1 below. In essence, the marshes and flats depicted in these three maps are bounded by geographic features that tend to inhibit the dispersal or daily movements of resident small mammals, such as the endangered Salt Marsh Harvest Mouse, and resident rails, including especially the endangered California Clapper Rail. These maps therefore generally represent the distribution and abundance of habitat patches for these species.

Methods and Calculations

The maps of the past and present distributions of patches of tidal marsh and tidal flats were used to assess changes in patch size. There are alternative approaches to such assessment (Forman and Godron, 1986). The simplest approach is to calculate the change in average patch size. However, the same differences in average size can result from a large variety of changes in the distribution and abundance of patches. For example, there might be a change in the number of large patches, or in the number of small patches, or there might be a change in the maximum or minimum patch sizes. These are ecologically important aspects of patch size that are not evident in simple measures of average size.

Another, more informative approach is to calculate the change in patch size-frequency. Size-frequency is the number of patches per category of size, when the categories together represent the complete size range. This approach reveals the change in abundance for each size category as well as the change in average size. It involves no assumptions about the importance of any particular patch size. This is an important consideration when the patches represent a variety of ecological functions for which optimal size might differ. Size-frequency analyses can help address a variety of concerns about habitat conservation including habitat fragmentation (e.g., Dorp and Opdam, 1987. Andrén1994, Dickson 2001), ecological connectectivity (e.g., Diamond1975, Brown and Dinsmore 1986, Lindenmayer and Nix 1993, Rosenburg et al. 1997,), and risks of local or regional extinction or recovery of wildlife species (e.g., Fahrig and Merriam 1985, Wilcox and Murphy 1985, Soulé 1987, Lindenmayer and Fischer 2006). The challenge is to identify meaningful patch size categories.

Table 1. Alternative rules for analyzing habitat fragmentation for tidal marshes and flats.

Patch Type	Patch Definition	Reference Species

Patch Type	Patch Definition		Reference Species
Tidal Flat	Alternative 1	Patch boundaries are any or all of the following: (A) the foreshore of adjacent marsh, (B) any non-tidal area at least 200 ft wide, (C) any area of open water at least 200 ft wide at low tide, (D) any man-made levee as shown on 1:24k scale USGS topographic quadrangles, (E) any "large channel" (i.e., tidal marsh channel or tidal reach of river or stream that is at least 200 ft wide from bank-to-bank for most of its length, or that receives perennial freshwater discharge, or that extends across the tidal flat to the subtidal zone), (F) any roads. Having considered all rules above, two patches that come together at a point are considered two separate patches because the point of intersection creates a place of such high risk of predation that two patches are ecologically separate.	Resident infauna, and vertebrate fauna resident in adjacent tidal marsh
	Alternative 2	Same as Alternative 1 above except disregard large channels (i.e., tidal flat Alternative 1 boundary type "E" above).	Shorebirds, large wading birds, intertidal fishes
Tidal Marsh	Alternative 1	Patch boundaries are any or all of the following: (A) the foreshore, (B) any non-tidal area at least 200 ft wide, (C) any area of open water at least 200 ft wide at low tide, (D) any man-made levee as shown on 1:24k scale USGS topographic quadrangles, (E) any roads (4 lane or larger), (F) any "large channel" (i.e., tidal marsh channel or tidal reach of river or stream that is at least 200 ft wide in cross-section from bank-top to bank-top at most points along the channel length or that receives perennial freshwater discharge). Having considered all rules above, two patches that come together at a point are considered two separate patches because the point of intersection creates a place of such high risk of predation that two patches are ecologically separate.	Resident intertidal rails (this rule set also defines marsh patches that are separate contributors to the tidal prism of a large channel or the Bay).
	Alternative 2	Same as Alternative 1 except disregard any man-made levees from rule D that partition or separate tidal marsh or muted tidal marsh.	Resident intertidal passerine birds (especially intertidal song sparrows)
	Alternative 3	Same as Alternative 2 except also disregard any man-made levees from rule D that partition or separate abandoned salt ponds (except where flooded) and diked managed marsh.	Resident intertidal small mammals, intertidal amphibians and reptiles
	Alternative 4	Same as Alternative 3 except include low-salinity and medium-salinity salt ponds, treatment ponds and mudflats, upland fill less than 60 meters wide, and disregard rules E and F and all channels regardless of their width as barriers.	Waterfowl and shorebirds
	Alternative 5	Same as Alternative 4 except include farmed baylands. (This patch represents partial habitat within the tidal area).	Raptors and medium to large mammalian predators

For this report, alternative size categories were tested relative to a set of three basic criteria: (1) does every category contain patches for each period (ca 1800, ca 2000, ca 2010); (2) does the number of patches in most categories change from one period to the next; and (3) are their separate categories for the large, medium-sized, and small restoration projects. The latter criterion was needed to make sure the analyses were sensitive to restoration efforts. The same categories were used for all three time periods, but different categories were selected for marshes and flats. Changes in size-frequency for tidal flats were insignificant because the patches of flats have remained very large. The analysis of patch size therefore focused on tidal marshes.

Tidal Marsh and Wadeable Stream Condition

Definition

Tidal marshes are defined above (see the sections on extent and size for tidal marshes). A wadeable stream is a natural or artificial channel that can be safely crossed on the ground during low flow.

The condition of a marsh or stream is its existing potential or capacity to provide high levels of one or more of its needed ecosystem services. Ecosystem services are consequences of natural processes, functions, and management actions that benefit society (MEA 2005). In California, there are multiple, overlapping, and incompletely coordinated processes for to identify the kinds and levels of service that a marsh or stream should provide (see section below on benchmarks). For the purposes of this report, the conditions of marshes and streams are assessed relative to the conditions that generally correspond to high levels of a broad suite of services.

Estuarine Wetland CRAM

The data source for assessing the overall condition of tidal marshes is the statewide ambient survey of marshes conducted in 2007 (Sutula et al. 2008). The survey results are available online (http://www.cramwetlands.org/cramdisplay/), and are summarized in the recent State of the State's Wetland Report (Natural Resources Agency 2010). The details of the ambient survey including the sampling plan, sample size, and sample precision are provided in the survey report (Sutula et al. 2008).

The method used in the ambient survey of marsh condition is the Estuarine Wetland Module of the California Rapid Assessment Method for wetlands and wadeable streams (CRAM; Collins et al. 2008). A detailed explanation of CRAM plus the Estuarine Wetland Module used to assess tidal marshes are available online (http://www.cramwetlands.org/). CRAM is a standardized method used in the field by teams of 2-3 practitioners to assess the overall conditions of wetlands and wadeable streams relative to statewide networks of reference sites that represent excellent condition. The method assumes that, for any given kind of wetland or stream, the more structurally complex sites that are surrounded by more natural buffers and landscapes are likely to provide higher levels of their expected ecosystem services (Collins et al. 2008).

Riverine CRAM

Ambient surveys of the overall condition of wadeable streams have recently been conducted for the Napa River Watershed in Napa County and for the Coyote Creek Watershed in Santa Clara County. The survey results are available online (http://www.cramwetlands.org/cramdisplay/). The details of these ambient surveys of wadeable streams are provided in their separate survey reports (Sutula et al. 2008, SCVWD 2011).

Methods and Calculations

CRAM provides numerical scores for metrics that represent four basic attributes of condition: biological structure, physical structure, hydrology, and buffer-landscape context. Each metric can have one of four alternative scores that together represent the full range of possible conditions (Sutula et al. 2006, Stein et al. 2009). The metric scores for each attribute are summed to produce an attribute score for each site, and the attribute scores are summed to produce a site score. The attribute scores and the site scores are percentages of their maximum possible scores. It is assumed that every site has some ecological value, and therefore no site can have a zero condition score.

The ambient CRAM assessments of tidal marsh and stream condition were summarized as relative Cumulative Frequency Distributions (CFDs; NIST/SEMATECH 2001). CFDs were developed for the scores of each metric, each attribute, and all sites in each survey. This supported easy determinations of median and quartile scores. Since the surveys were based on probabilistic sample designs (Stevens and Olsen 2004), the CDFs could be used to estimate the percentage of marsh acreage or stream miles having scores above or below a particular score, or between any two scores, given the confidence limits of the CFDs.

Riparian Width

Definitions

The National Research Council of the National Academies has defined riparian areas as integral components of landscapes through which surface and subsurface movements of water interconnect aquatic areas and connect them to their adjacent uplands (Brinson et al. 2002). Riparian Areas are distinguished by gradients in biophysical conditions, ecological processes, and biota. They can include wetlands and portions of uplands that significantly influence the conditions or processes of aquatic areas. Based on this definition, every aquatic area including wetlands can be bounded by riparian areas. There is no minimum amount of plant cover, no requirement for particular kinds of cover, and the areas do not have to be natural.

Riparian areas have their own intrinsic ecosystem services (Gregory et al. 1991, Naiman et al. 2005). There are, for example, species of plants and animals that are largely restricted to riparian areas (e.g., Conard et al. 1977, Reed 1988, Fischer 2000, Bryce et al. 2002, RHJV 2004, white 2011), and riparian areas can serve as corridors for the dispersal, migration, and daily movements of terrestrial animals (Naiman et al. 1993, Fischer et al. 2000).

However, with regard to wetland and streams, riparian areas are generally regarded as buffers against external stressors, or as sources of materials that enhance wetland and stream services (e.g., Wenger 1999, Johnson and Buffler 2008, Ellis 2008.). In this regard, the kinds and levels of riparian services vary with riparian width (Wenger 1999, Polyakov et al 2005, Collins et al. 2006 and references therein). For example, the riparian area that stabilizes the banks of a stream tends to be narrower that the area that shades the same stream or supplies it with woody debris; the riparian area defined by hillslope processes that supply the stream with sediment tends to be wider than the area that provides woody debris. The broadest riparian areas tend to be defined by the spatial limits of effective habitats for riparian wildlife, especially riparian birds. This is the riparian concept that the SWRCB is considering while developing its Wetland and Riparian Area Protection Policy (http://www.swrcb.ca.gov/water_issues/programs/cwa401/docs/wrapp/tatmemo3_061610.pdf).

Riparian Buffer Decision Tool

Based on the riparian definition provided above, the USEPA, SWRCB, and the California Riparian Habitat Joint Venture have been sponsoring the development of a GIS-based tool for estimating functional riparian widths. The tool maps the riparian areas that correspond to bank or shoreline stability, shading, allochthonous input, sediment input, and runoff filtration, based on reported relationships among these services and topography, land use, and vegetation height (Collins et al. 2006). Pilot applications of the tool are occurring in Southern California (http://www.csun.edu/~centergs/data/SGR_FINAL_REPORT.pdf), the San Francisco Bay Area, (http://www.wrmp.org/docs/No569_WRMP_BasemapFactsheet_finalMay09.pdf) and the Tahoe Basin (http://tahoemonitoring.org/trt-charter.html).

Methods and Calculations

For this report, the Riparian Buffer Decision Tool was used to map the maximum extent of riparian areas for all the channels evident in the Bay Area Aquatic Resource Inventory (BAARI) for the two pilot watersheds. Riparian width was determined for both sides of each channel, beginning at the channel bank. Riparian width was mapped for all non-tidal channels longer that 30m. The minimum riparian width calculated by the tool is 1m. The maps are not constrained by any maximum riparian width. The riparian areas were classified as natural or unnatural, based on the degree to which the plan-form and/or structure of the associated channels had been modified. Each width class corresponds to a unique set of the riparian services listed above. It was assumed that, for any given location, the number of services that a riparian area is likely to provide increases with its width.

Stream Biological Integrity

Definitions

Biological integrity is a term that first appeared in the federal Clean Water Act in 1972. A variety of definitions have been developed since then (Cairns 1975, Karr and Dudley 1981, Hughes et al. 1982, Karr et al. 1986). It is commonly defined by USEPA as the capability of an aquatic area to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of a region (USEPA, http://www.epa.gov/bioiweb1/html/biointeg.html.) This is a practical working definition that is still broadly used. In 1981 USEPA produced a framework for developing indices of biological integrity (IBIs) that reflects this working definition. The framework continues to be revised as experienced with IBIs is gained (e.g., Barbour et al. 1996).

Benthic Macroinvertebrate Index and Benthic Index of Biotic Integrity

The water quality of streams is typically assessed based on chemical data, whereas their habitat quality is commonly assessed based on their form and structure as physical systems. One common approach to assessing the biological integrity of streams is to characterize their fish communities or benthic macroinvertebrate communities. The latter is more commonly used because many more streams support macroinvertebrates than fish. Aquatic insects are the most common organisms used in such assessments. It is generally understood that changes in structure (species composition and the relative abundance of species] of benthic macroinvertebrate communities reflect changes in environmental conditions (e.g., Vannote et a 1980, Vinson and Hawkins 1998, Brown and May 2000, LaBonte et al. 2001, Griffith et al. 2003), and that benthic macroinvertebrates are essential components of stream food webs (Vannote et al. 1980, Wallace and Webster 1996, Harding et al. 1998). Benthic bioassessments can integrate over time to provide robust measures of ecological impairment and rehabilitation (Hellawell 1986, Rosenberg and Resh 1993). Many states have incorporated benthic bioassessment into their stream monitoring programs to improve pollution control, guide abatement, and track regulatory compliance (Davis et al. 1996). Volunteer citizen science groups are increasingly using benthic bioassessment in watershed-based stream health care (Barbour et al. 1999, Clean Water Team 2011).

The Surface Water Ambient Monitoring Program (SWAMP) of the SWRCB has been developing regional IBIs for benthic macroinvertebrates (e.g., Rhyen and Ode 2006, Ode 2007, Rheyn et al. 2008). The development process has relied on many collaborators in different eco-regions to collect benthic data using the Benthic Macroinvertebrate Index (BMI) (Harrington 1999) along stressor gradients. The BMI focuses on the relative abundances of different groups of benthic macroinvertebrates that differ in their sensitivities to common stream stressors. Within a given eco-region, the BMI data can be calibrated to sites that are independently assessed as minimally impacted to developed a Benthos IBI (B-IBI). Local agencies within the Bay Area have been collaborating to develop a B-IBI for this region (Buchan et al. 2009). Starting in 2001, SWAMP has been surveying the integrity of Bay Area streams using, among other tools, the BMI (Breaux et al. 2005, Taberski et al. 2010). These SWAMP data were used to assess stream integrity for this report.

Methods and Calculations

The details of the BMI are available online (http://www.dfg.ca.gov/abl/Field/professionals.PDF). The method has varied somewhat since the Bay Area surveys began. Beginning in 2007, the focus of the method was switched from stream riffles to entire stream reaches (Ode 2007, Rehn and Ode 2007). Data for this region have been compiled for both sampling approaches, notwithstanding there differences. While the B-IBI is likely to be a more resolute than the BMI for assessing the integrity of Bay Area streams, it has not yet been fully implemented. In the meantime, the BMI has been determined to be useful for detecting differences in stream integrity that correlate to major stressors (Breaux et al. 2005, Taberski et al. 2010). In practice, the BMI is used to classify the health status of each assessed stream reach as excellent, good, fair, or poor.

Benchmarks

Each metric that is used to assess the condition of the region's aquatic habitats should ideally be calibrated to a broadly accepted, regional, numeric benchmark that represents the desired or needed condition. The benchmarks could be water quality objectives, habitat goals, population sizes to recover endangered species, or any level of ecosystem service that indicates excellent condition. This is not generally the situation, however. Some metrics that are directly linked to benchmarks are not adequately supported with data. Other metrics are well supported with data but lack a clear relationship to any benchmark. In the future, there should be a concerted effort to (1) identify the most important metrics for assessing the health status of the region's aquatic habitats; (2) establish benchmarks for the selected metrics that represent good health; and (3) fund efforts to collect the data that are needed to calculate the metrics.

Tidal Marsh and Tidal Flat Extent

The basis for the recommended benchmarks for the extent of tidal marches and flats is the 1993 California Wetlands Conservation Policy (http://ceres.ca.gov/wetlands/policies/governor.html). It's first two objectives are to ensure no overall net loss and to achieve a long-term net gain in the quantity, quality, and permanence of wetlands acreage and values in California. The state's anti-degradation policy is also designed to prevent declines in the quality of state's aquatic areas, although it focuses on the areas that have better conditions than necessary to provide adequate levels of service (SWRCB 1968). Neither policy stipulates a baseline acreage that cannot be further reduced, nor do they set a numerical target for how much more wetland acreage is needed. Instead, the Wetland Conservation Policy calls for regional and statewide wetland goals. Not long after this policy was published, a broad coalition of wetland interests including federal and state agencies began developing numerical (acreage) goals for intertidal habitats in the Estuary (Collins 1993, Goals Project 1999). It is reasonable to conclude that the tidal marsh acreage goals that were set in 1999 are consistent with, and have helped to implement the 1993 policy.

The Goals Project recommended that the Estuary downstream of the Delta should have no fewer than 100,000 acres of tidal marshes. This goal represents about 50 percent of the total acreage of tidal marshes that existed historically. The Goals Project also recommended how the acreage should be allocated among major subregions of the Estuary. This report, however, only addresses the extent of marshes relative to the overall regional goal.

No quantitative goal or benchmark has been set for increasing the extent of tidal flats. However, the 1993 California Wetlands Conservation Policy in conjunction with the state's anti-degradation policy suggests that the amount of tidal flat that existed in 1993 is the minimum acceptable extent for the future. In other words, the future extent of tidal flat should not be less than what existed in 1993. As stated in this report, the 1993 extent of tidal flats is about half the historical extent, and is therefore commensurate with the goal set for tidal marsh.

This report assumes that the extent of tidal marshes and flats that existed in 1993 is represented well enough by the ca 2000 map (SFEI 2000). Since the ca 2000 map is a compilation of information spanning the period 1993-2000, the change in extent that occurred in this period can be estimated. The change was positive and very small, relative to the total amount tidal flats. It mostly resulted from levee breaches designed for tidal marsh restoration, and therefore represented tidal flats that are expected to evolve into tidal marsh.

<u>Tidal Marsh Size</u>

The benchmark for tidal marsh size is the historical size-frequency of marshes ca 1800. There are three main technical questions about this benchmark: what is the correct set of rules for mapping individual patches; what is the correct set of patch size categories; and why is the historical size-frequency a reasonable template for the future.

Correct Patch Mapping Rules

Patches are defined by geographic features or changes in land cover that delimit selected functions or ecosystem services. For this report, patches were defined as habitat for resident wildlife, especially rails and small mammals, based on the best professional judgment about the kinds of features and land cover that inhibit their dispersal and daily movements (see Alternative 1 of Table 1 above). These rules can be refined as information about the behavior of these species increases. Studies to-date of the behaviors, habitat preferences, and movements of these species do not refute the mapping rules used in this report (Shellhammer et al. 1982, Geissel et al. 1988, Shellhammer 1989, Albertson 1995, Foin et al. 1997, Bias 1999, Albertson and Evens 2000, Hulst 2000, Shellhammer 2000, Schwarzbach et al. 2006, Overton 2007, Casazza et al. 2008). It is important to note, however, that these rules are general and not absolute. Individuals within these species may not follow the rules. For example, most resident populations of animals contain a small number of individuals that tend to disperse much further than the rest (Murray 1967, Koenig et al. 1996).

Correct Size Categories

The historical and modern maps of tidal marshes and flats are exhaustive. The sum totals of all the individual patches of marshes and flats are equal to their total regional acreages. No areas large enough to map are left out, based on the mapping standards. This is an essential requirement of any effort to assess extent.

There are many different sets of size categories that meet the same selection criteria. And, given that the maps are exhaustive and follow the same mapping standards, different sets of criteria can be applied to them. One of the basic advantages of standardized mapping procedures that generate exhaustive maps is that many different patch definitions and size categories can be applied to them to answer different questions. The criteria used in this report (see *Methods and Calculation* in section above addressing tidal marsh and flat size) support an analysis of the effects of restoration and mitigation projects on the overall extent and size-frequency of tidal marshes and flats. A variety of qualified size categories were tested and the one selected was most sensitive to the effect of past and proposed projects. Given a different set of criteria, a different set of size categories might be optimal.

Historical Size-frequency

Three basic assumptions underlay the decision to use the historical (ca 1800) patch size-frequency of tidal marshes as the model for their future patchiness. First, it is assumed that the current size-frequency distribution, which reflects almost two centuries of tidal marsh fragmentation, is not an appropriate benchmark or goal for the future. The patchiness that existed at the starting dates of the State Wetland Conservation Policy of 1993 and the anti-degradation policy of 1968 might indicate the maximum acceptable amounts of fragmentation, but they do not represent the needed deceases in fragmentation. Second, it is assumed the historical size-frequency sustained the native species that are currently threatened or endangered. Although the increased fragmentation of their habitats is only one factor in the declining abundance of these species, it has likely increased the negative effects of other factors. For example, as the marsh patches have gotten smaller, the ratio of their edge length to their surface area has increased, as has the distance between patches (Collins et al 2005), which in theory has increased the risk of predation, exposure to external stressors, and failure to disperse (Troll 1971, Forman 1995, Turner 1989, 2005, Fahrig 2002). It should be noted however, that declines in the total quantity of habitat and in its quality can out-weight the effect of fragmentation (Harrison and Bruna 1999). Third, larger habitat patches are usually better than smaller patches for sustaining local animal populations (e.g., Andrén 1994, Kolozsvary and Swihart 1999, Lindenmayer and Fischer 2006). The historical landscape included much larger tidal marsh patches than exist today.

The vertebrate communities of tidal marshes exhibit a high degree of endemism. Many species are entirely restricted to tidal marshes, and some are restricted to marshes of one or a few estuaries (Greenberg and Maldonado 2006, Greenberg et al. 2006, SBSPRP 2007). A reasonable assumption is that these species have adapted to the particular characteristics of the marshes they inhabit, including their hydrology, salinity regimes, vegetation, predators, as swell as the natural patchiness of their habitats.

This emphasis on categorical environmental patchiness as a determinant of community structure is common but not without controversy. The central concern is that the patch-based approach to the analyses of the distribution and abundance of plants and animals disregards the interactions between individuals or populations and gradients in their key resources and limiting factors (e.g., Cushman et al. 2010a,b). There are, however, gradients in habitat patch size within the geographic distribution of a species, and, for animals, these gradients usually include patches that are too small to support viable populations. In other words, patch size can be limiting for animals in highly fragmented habitats (Wilcox and Murphy 1985, Fahrig and Merriam 1985, Fahrig 2002).

There are numerous studies of tidal marsh animals in the Estuary that clearly indicate their distributions vary along environmental gradients independent of patch size (e.g., Atwater and Hedel 1976, Shellhammer 2000, Albertson and Evens 2000, Watson and Byrne 2009). This is not unusual for estuaries that are characterized by strong gradients in salinity and other physical factors. It does not necessarily mean, however, that patch size is not important. It means that patch size is one of many inter-relating factors that together affect the distribution and abundance of tidal marsh species over time. In the absence of any known optimal patch sizes for tidal marsh species in the Estuary, and given the negative effect of past habitat fragmentation on the prospects for their survival, setting an initial benchmark for future patch sizes that reflect the historical, natural patch size-frequency seems reasonable.

Tidal Marsh and Stream Condition

Careful analyses of the CRAM scores for both tidal marshes and wadeable streams revealed that the lower site scores were generally due to low scores for the physical structure attribute. Mean scores for this attribute were lower for Bay Area marshes than for marshes in other regions of the state. Mean scores for Bay Area streams were lower than for stream along the north coast.

For tidal marshes, the low scores for physical structure were mainly due to low scores for the metrics for topographic complexity and physical patch richness. This can be explained in part by the early stages of evolution of many of the marshes that were assessed. Tidal marshes gain physical complexity as they naturally evolve upwards through the intertidal zone (Redfield 1972, Orson et al. 1987, Kirwan and Murray 2007). Most of the Bay Area marshes have developed rapidly due to excessive sediment supplies resulting from an influx of hydraulic mining debris during the late 1800s (Atwater et al. 1979, Nichols et al. 1986, Dedrick and Chu 1993), and increased erosion in local watersheds due to nineteenth and twentieth century land use changes (Collins 2006, McKee and Lewicki 2009). These relatively young, rapidly accreted marshes lack the physical structural complexity of the remnant, higher, ancient marshes of the Estuary, which tended to get higher CRAM scores for physical structure. Higher scores for physical structure were also obtained for older marshes along the north coast. These findings support the recommendation in this report that the benchmark for future marsh condition should focus on CRAM scores for physical structure that are comparable to the natural, older marshes of this region and the north coast. Natural evolution of existing newly restored low-elevation marshes should eventually achieve this benchmark.

For wadeable streams, the low scores for physical structure were mainly due to the entrenched state of most of the assessed stream reaches. Entrenchment is caused by an increase in flows, relative to the size of the sediment loads that the stream must transport, or a decrease in sediment loads relative to the flow, or both (Lane 1955, 1957, Schumm 1969). It greatly increases the range of flows that are contained within the stream channel, which in turn increases the tendency of the stream to incise its bed (Schumm et al. 1984, Rosgen 1996). This in turn increases the degree of entrenchment until the bed encounters material that resists erosion, or the channel reaches a new equilibrium between the flow, the sediment load, and the channel form. The result of such chronic entrenchment is a loss of floodplains, riparian vegetation, large woody debris, persistent pools, and other features that together support many of the ecological services expected of rivers and streams. In many cases, managers must intervene to engineer a stable channel and/or to adjust upstream inputs of water and sediment. Both approaches are expensive, and they are not mutually exclusive. The former approach usually involves restoring floodplains, which in the Bay Area usually involves purchasing expensive lands. The latter approach usually involves changes in land use that can be politically challenging. It is unlikely, however, that the needed services of Bay Area streams can be attained and sustained unless their entrenched state is corrected. It seems appropriate therefore to set a benchmark for steam health that focuses on restoring the natural complexity of the streams, which will require long-term reductions in stream entrenchment.

Riparian Width

For the purposes of this report, which is focused on wetlands and streams, riparian areas are primarily regarded as buffers that protect wetlands and streams from external stressors. Riparian areas can provide one or more buffer services. In other words, they can buffer against multiple kinds of stress. Riparian areas can also provide their own ecological and social services, such as riparian wildlife support and recreation.

For any given topographic side slope and vegetation community, each riparian service tends to require a certain range in riparian width. The functional riparian widths can overlap, and greater widths tend to provide more kinds and higher levels of service. Setting benchmarks for riparian buffers therefore requires knowing what services are needed, and knowing how the existing riparian structure and setting must be modified, if at all, to provide the needed services. Benchmarks can therefore vary from place to place, depending on the stressors involved and what other services, besides buffering, are needed.

The approach to setting riparian benchmarks that was adopted for this report emphasizes the relationship between riparian width and riparian service, and recognizes that different stream reaches may have different benchmarks. The approach is similar to that used for tidal marsh size. According to this approach, future riparian areas should be distributed among categories of width according to their historical distributions. The rationale for this approach is the same for riparian areas and tidal marshes (see section above on Tidal Marsh Size). There are two main technical questions about this approach as it pertains to riparian areas: what is the correct set of riparian width categories, and why is the historical distribution of riparian areas among these categories a reasonable template for the future.

Correct Width Categories

The western literature concerning the relationships between riparian buffer width, structure, and buffer functions or services have been summarized multiple times since the 1980s (e.g., Clinnick 1985, Phillips 1989, Barling and Moore 1994, Desbonnet et al. 1994, Collier et al. 1995, Mander et al. 1997, Wenger 1999, Collins et al. 2006). These summaries provide a basis for recommending width categories. The correlations are not precise, however, and the categories tend to get wider (i.e., inclusive of larger areas of a landscape), with distance away from the waterbody. This is because the services that extend furthest from the waterbody are mostly about the support of riparian wildlife species that sometimes have large home ranges. Each of the more physical services, such as bank stability, shading, and allochthonous input do not extend as far as the wildlife support functions, are their extents are less variable. The categories of riparian width devised for this report are remarkably applicable to historical maps of riparian areas. This is further evidence of the general usefulness of the categories.

Historical width-frequency

The argument in favor of using the historical riparian width frequencies as a model for the future parallels that for tidal marsh patch size (se section above on Tidal Marsh Size). First, it is assumed that the current width-frequency distribution, which reflects almost two centuries of increasingly intensive landscape modification, is not an appropriate benchmark or goal for the future. The riparian widths that existed when the State Wetland Conservation Policy and the anti-degradation policy were enacted might indicate the minimum acceptable widths, but they do not represent the needed increases in width. Second, it is assumed the historical width-frequency sustained the native species that are currently threatened or endangered, including steelhead and salmon. Although the loss of riparian areas is only one factor in the declining abundance of these species (see section above on Stream Condition), it has likely increased the negative effects of other factors. For example, the loss of riparian forests along the Napa River and elsewhere in the region has decreased shading and the input of woody debris, which in turn have caused an increase in stream temperature and a decrease in stream channel complexity (e.g., Napolitano et al. 2003, Stillwater Sciences and Dietrich 2002). Third, as sated above, wider riparian areas tend to provide higher levels of more kinds of riparian services (Collins et al 2006). The historical landscape included much wider riparian areas than exist today.

<u>Stream Biological integrity</u>

Stream integrity can be defined in terms of abiotic as well as biotic factors and processes. It might be defined as the absence hydromodification (USEPA 2007, Mohamoud et al. 2009), the persistence of geomorphic stability (Leopold et al. 1964, Heede 1980, Rosgen 1994, Trush et al. 2000), or the support of reference communities of plants and animals (KARR 1999).

These various bases for defining stream integrity are broadly covered by the beneficial uses of waterbodies that are defined and designated by the SWRCB and its Regional Water Boards under the Porter-Cologne Water Quality Control Act Beneficial uses are the needed ecosystem services of a water body. (http://www.waterboards.ca.gov/laws_regulations/docs/portercologne.pdf). For example, water filtration is a marsh process that functions to improve water quality, which is a service incorporated into the beneficial use called "Estuarine Habitat;" the support of biological diversity is one of many services of marshes and streams that are incorporated in the beneficial use called "Wildlife" (http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/planningtmdls/basinplan/web/bp_ch2.shtml).

Beneficial uses indicate the kinds, but not the levels, of services that a marsh or stream should provide. The needed levels of service are represented by water quality objectives. These are numerical or narrative descriptions of minimum conditions or levels of service that must be sustained to ensure that the waters of the state can support their designated beneficial uses. The SWRCB has initiated a process to set water quality objectives for streams (http://www.swrcb.ca.gov/plans_policies/biological_objective.shtml), and intends to set them for wetlands (http://www.swrcb.ca.gov/water_issues/programs/cwa401/wrapp.shtml).

As part of the state's effort to set water quality objectives for streams, the Surface Water Ambient Monitoring Program (SWAMP) of the SWRCB has been developing regional IBIs for benthic macroinvertebrates (B-IBIs) (e.g., Rhyen and Ode 2006, Ode 2007, Rheyn et al. 2008). The development process has relied on many collaborators in different eco-regions to collect benthic data using the Benthic Macroinvertebrate Index (BMI) (Harrington 1999) along stressor gradients. The BMI focuses on the relative abundances of different taxonomic groups of benthic macroinvertebrates that differ in their sensitivities to common stream stressors. Within a given eco-region, the BMI data can be calibrated to sites that are independently assessed as minimally impacted to developed a B-IBI. Local agencies within the Bay Area have been collaborating to develop a B-IBI for this region (Buchan et al. 2009).

The regional B-IBI will provide one method for scoring the biological integrity of streams relative to a regional standard. The SWRCB is encouraging practitioners to classify the level of integrity of each assessed stream reach as excellent, good, fair, or poor. There is no established benchmark, however, for the proportion of assessed reaches that should exist in any class. Based on the state's policies for wetland conservation and anti-degradation, it seems reasonable that the conditions existing at the time of the policies represent the minimum proportion of assessed reaches that have good or excellent levels of integrity. The benchmark for the future should indicate improved stream integrity region-wide. This means that the proportion of stream reaches classified as having good or excellent levels of integrity should increase. Given that only about 60% of the assessed reaches have these high levels of integrity at this time, an increase of 15% to reach a regional benchmark of 75% seems appropriate. As with any of the benchmarks, if monitoring shows that this one is not likely to be obtained, it can be revised downward. If it is likely to be surpassed, it can be revised upward. If additional indicators of integrity such as the algal IBI (Fetscher and McLaughlin 2008) are implemented in the region, they can be integrated together with the B-IBI to produce a more robust assessment of stream integrity.

Next Steps

The reported assessments of wetland and stream health are rudimentary. While the approach of assessing health status relative to established health goals or benchmarks is useful and workable, few benchmarks have been set and the data needed to set benchmarks or track progress towards them are scarce. One large step moving forward toward a more comprehensive assessment will be for the regional community of wetland and stream interests to prioritize the aspects of health (i.e., the wetland and stream services) that must be assessed, and then to develop benchmarks that define their ideal state. These decisions should be made with an aim to track the performance of wetland and stream protection policies and programs in the context of climate change.

The assessments of aquatic habitats in watersheds are especially weak because so few watersheds have been sampled. Although the assessments include some of the larger watersheds in the region, they do not represent the full regional range in watershed size, geology, dominant land use, or climate. The ambient assessments of watershed health should be extended throughout the region.

The capacity to track changes in habitat extent and overall condition using new maps and rapid assessment is increasing. The Wetland Tracker information system is being expanded to cover more wetland types with additional functionality including automated watershed delineation, on-screen mapping, and automated data summaries at user-defined scales. These new functions are being developed by SFEI with local partners. The intent is to develop local data stewards who can revise and update BAARI as needed and build CRAM into their local monitoring efforts. CRAM trainings continue to gain popularity as CRAM is incorporated into state and federal regulatory and management programs. CRAM and Wetland Tracker are being merged to enable the public to view and summarize CRAM results and mapping results for watersheds and for the region as a whole. These developments should improve the ease of assessment and the sharing of information about aquatic resources.

Maps and rapid assessment will not be able to track conditions for all the needed services of wetlands and streams. Field-based, quantitative measurements will be needed is some cases. For example, maps and rapid assessment by themselves cannot assess changes in the size of key wildlife populations or in the levels of contaminants. Standardized, quantitative methods of assessment already exist for some of these concerns, but others will be needed after the services and concerns are reviewed and prioritized.

In the context of setting benchmarks, tracking progress toward them, and reporting the results to the public, the importance of standardizing the data collection methods, providing adequate data quality assurance and control, and maintaining a common data library with broad accessibility cannot be overemphasized. These are essential elements of a comprehensive regional environmental monitoring program that can exist. The State of the Estuary reports can continued to foster public understanding and political will to improve the health of the Estuary and its watersheds. This should catalyze the support that is needed from the regional community of wetland and stream scientists.

Citations

Albertson, J.D. 1995. Ecology of the California Clapper Rail in south San Francisco Bay. Unpublished Master's Thesis, San Francisco State University, San Francisco CA.

Albertson, J.D., and J.G. Evens. 2000. California clapper rail, in baylands ecosystem species and community profiles: life histories and environmental requirements of key plants, fish and wildlife. P.R. Olofson (ed.). San Francisco Bay Regional Quality Control Board, Oakland CA.

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71(3): 355-366.

Atwater, B.F., and C.W. Hedel. 1976. Distribution of seed plants with respect to tide levels and water salinity in the natural tidal marshes of the northern San Francisco Bay estuary, CA. USGS open-file report 76-389. Menlo Park CA.

Atwater, B.F., S.G. Conard, J.N. Dowden, C.W. Hedel, R.L. Macdonald, and W. Savage. 1979. History, landforms, and vegetation of the estuary's tidal marshes. In San Francisco Bay: the urbanized estuary: investigations into the natural history of San Francisco Bay and Delta with reference to the influence of man: fifty-eighth annual meeting of the Pacific Division/American Association for the Advancement of Science held at San Francisco State University, San Francisco, California, June 12-16, 1977, ed. T. J. Conomos, 493 p. San Francisco, CA.: AAAS, Pacific Division.

Barbour M.T., J. Gerritsen, G.E. Griffith, R. Frydenborg, E. McCarron, J.S. White, and L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society 15:185-211.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington DC.

Barling, R.D. and I.D. Moore. 1994. Role of buffer strips in management of waterway pollution: a review. Environmental Management 18(4):543-558.

Bayley, P. (1999). Understanding Large River-Floodplain Ecosystems. BioScience, 45: 153-158.

BCDC. 2008. San Francisco Bay Plan. San Francisco Bay Conservation and Development Commission. San Francisco CA.

Bedford, B.L. and E.M. Preston. 1988. Developing the Scientific Basis for Assessing Cumulative Effects of Wetland Loss and Degradation of Landscape Functions: Status, Perspectives and Prospects. Environmental Management 12(5): 751-771.

Bias, M.A. 1999. Movements and home range of salt marsh harvest mice. Southwestern Naturalist 44: 348-353.

Bodnar et al. 1975. The relationship between the upper limit of coastal marshes and tidal datums. National Ocean Survey, preliminary report prepared for the Environmental Protection Agency, Washington DC.

Borcard, D., P. Legendre, and . P. Drapeau. 1992. Partialling out the spatial component of ecological variation. Ecology 73:1045–1055.

Breaux A. M. Born, L. Suer, S. Cochran, and R. Looker. 2005. Benthic macroinvertebrates as water quality indicators in highly urbanized streams in the San Francisco Bay Region, California. Final Report. San Francisco bay Area Regional Water Quality Control Board, Oakland, CA.

Brinson, M.M. 1993. A hydrogeomorphic classification for wetlands. Technical Report WRP-DE-4. U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS.

Brinson, M.M., L.J. MacDonnell, D.J. Austen, R.L. Beschta, T.A. Dillaha, D.A. Donahue, S.V. Gregory, J.W. Harvey, M.C. Molles Jr, E.I. Rogers, J.A. Stanford, and L.J. Ehlers. 2002. Riparian areas: functions and strategies for management. National Academy Press, Washington, DC.

Brown, L. R. and J.T. May 2000. Benthic macroinvertebrate assemblages and their relations with environmental variables in Sacrament and San Joaquin river drainages, California, 1993-1997. U.S. Geological Survey Water-Resources Investigations Report 00-4125. Sacramento, CA.

Brown, M., and J.J. Dinsmore. 1986. Implications of marsh size and isolation for marsh bird management. Journal of Wildlife Management 50:392-97.

Bryce, S.A., R.M. Hughes, and P.R. Kauffmann. 2002. Development of a bird integrity index: using bird assemblages as indicators of riparian condition. Environmental Management 30(2): 294-310.

Buchan, L., K. Lunde, and C Somers. 2009. Benthic index of biotic integrity for San Francisco Bay Area Creeks. Project update. Macroinvertebrate Bioassessment Information Network (*BAMBI*). http://www.scvurppp-w2k.com/pdfs/0809/IBI_Development_Update_Final_1.28.09%20V2.pdf.

Cairns, J. Jr. 1975. Quantification of biological integrity. Pages 171-85 in R.K. Ballentine and L.J. Guarrie (eds). The integrity of water, a symposium. USEPA, Washington, DC.

Callaway, J.C., V.T. Parker, and M.C. Vasey. 2007. Emerging issues for the restoration of tidal marsh ecosystems in the context of climate change. Madono 54(3): 234–248, 2007.

Carter, V. Technical aspects of wetlands: wetland hydrology, water quality, and associated functions. 1997. Geological Survey Water Supply Paper 2425. US Geological Survey, Washington DC.

Casazza, M.L., C.T. Overton, J. Y. Takekawa, T. Rohmer,and K. Navarre. 2008. Breeding behavior and dispersal of radio-marked California clapper rails. Western Birds 39:101–106.

Clean Water Team. 2011. California citizen monitoring programs 2010. State Water Resources Control Board, Sacramento CA.

Clinnick, P.F. 1985. Buffer strip management in forest operations: a review. Australian Forestry 48(1): 34-45.

Coastal Conservancy. 2006. South Bay Salt Pond Restoration Project final alternatives analysis report. California Coastal Conservancy, Oakland Ca.

Collier, K.J., A.B. Cooper, R.J. Davies-Colley, J.C. Rutherford, C.M. Smith, and R.B. Williamson. 1995. Managing riparian zones: a contribution to protecting New Zealand's rivers and streams. Vol. 2: Guidelines. Department of Conservation, Wellington NZ.

Collins, J.C., C. Grosso, M. Sutula, E. Stein, E. Fetscher, R. Clark, and B.J. Close. 2005. Understanding tidal marsh fragmentation. Abstracts, 6th Biennial Bay-Delta Science Conference, Sacramento CA. CalFed Science Program, Sacramento Ca.

Collins, J.N. 1986. Wetland hydrology and functional assessment: a Pacific Coast regional perspective. In: Proceedings of the Pacific Coast Wetlands Workshop, J.S. Larson (ed.). National Wetlands Technical Council, The Environmental Institute, Amherst MA.

Collins, J.N., E.D. Stein, M. Sutula, R. Clark, A.E. Fetscher, L. Grenier, C. Grosso, and A. Wiskind. 2008. California Rapid Assessment Method (CRAM) for Wetlands. Version 5.0.2. San Francisco Estuary Institute, Oakland, CA.

Collins, J.N., M. Sutula, E. Stein, M. Odaya, E. Zhang and K. Larned. 2006. Comparison of methods to map California riparian areas. Prepared for the California Riparian Habitat Joint Venture by the San Francisco Estuary Institute, Oakland CA.

Collins, L.M. 2006. Estimating sediment supplies from local watersheds to intertidal habitats of San Francisco Bay. In: 2006 South Bay Science Symposium for the South Bay Salt Pond Restoration Project: Presentation Synopses. L. Trulio (ed). California Coastal Conservancy, Oakland CA.

Conard S, R. McDonald, and R. Holland. 1977. Riparian vegetation and flora of the Sacramento Valley. In: A. Sands (ed.) Riparian forests in California: their ecology and conservation. University of California at Davis, Institute of Ecology Publication No. 15. Davis CA.

Cowardin, L.M., V. Carter, F.C. Golet, and ET. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U. S. Department of the Interior, Fish and Wildlife Service, Washington DC. 131pp.

Craft C, C. J. Ehman , H. Guo, S. Joye, M. Machmuller, R. Park, and S. Pennings. 2009. Effects of accelerated sea level rise on delivery of ecosystem services provided by tidal marshes: a simulation of the Georgia (USA) Coast. Frontiers in Ecology and the Environment 7(2): 73–78.

Craft, C., J. Clough, J. Ehman, S. Joye, R. Park, S. Pennings, H. Guo, and M. Machmuller. 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. Frontiers in Ecology and the Environment **7**: 73–78.

Culberson, S.D. 2001. The interaction of physical and biological determinants producing vegetation zonation in tidal marshes of the San Francisco Estuary, California USA. Doctor dissertation, Ecology, University of California at Davis, Davis CA.

Cushman, S. A.; McGarigal, K.; Gutzwiller, K.; Evans, J. 2010b. The gradient paradigm: a conceptual and analytical framework for landscape ecology. In: S.A. Cushman and F. Huettman (eds). Spatial complexity, informatics and wildlife conservation. Springer, Tokyo JP: 83-110.

Cushman, S.A., J.S. Evans, K. McGarigal, J.M. Kiesecker, Joseph M. 2010a. Toward Gleasonian landscape ecology: from communities to species, from patches to pixels. Research Paper RMRS-RP-84. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins CO.

CWQMC. 2011. Wetland tracker information system of the Wetland Portal. California Water Quality Monitoring Council (http://www.californiawetlands.net/tracker/).

Davis, W.S., B.D. Snyder, J.B. Stribling, and C. Stoughton. 1996. Summary of state biological assessment programs for streams and rivers. US Environmental Protection Agency, Office of Policy, Planning and Evaluation, Washington DC.

Dedrick. K.G. 1989. San Francisco bay tidal marshland acreages: recent and historical values. In: Proceedings of the sixth symposium on coastal and ocean management, Coastal Zone 89. ASCE, New York.

Dedrick. K.G. and L.T. Chu. 1993. Historical atlas of tidal creeks San Francisco Bay, California. Proceedings of the eighth symposium on coastal and ocean management (Coastal Zone 93). American Society of Engineers, New York, NY.

Dedrick. K.G. and L.T. Chu. 1993. Historical atlas of tidal creeks San Francisco Bay, California. Proceedings of the eighth symposium on coastal and ocean management (Coastal Zone 93). American Society of Engineers, New York NY.

Desbonnet, A., P. Pogue, V. Lee and N. Wolf. 1994. Vegetated buffers in the coastal zone: a summary review and bibliography. University of Rhode Island, Providence RI.

Diamond, J.M. 1975. The island dilemma: Lessons of modern biogeographic studies for the design of natural reserves. *Biological Conservation* 7:129-46.

Dickson, J. G. 2001. Early History. Pages 20-30 In J. Dickson, Editor. Wildlife of the Southern forests: Habitat and management. Blaine, WA: Hancock House Publishing.

Dorp, D. van, and P. Opdam. 1987. Effects of patch size, isolation and regional abundance on forest bird communities. Landscape Ecology 1:59-73.

Ellis, J.H., 2008. Scientific recommendations on the size of stream vegetated buffers needed to protect fish and aquatic habitat. Report to Montana Department of Environmental Quality, EPA/DEQ Wetland Development Grant. Montana Audubon, Helena MT.

Fahrig, L. 2002. Effect of habitat fragmentation on the extinction threshold: a synthesis. Ecological Applications 12:346–353.

Fahrig, L. and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology 66: 1762-1768.

Fetscher, A.E. and K. McLaughlin. 2008. Incorporating bioassessment using freshwater algae into California's Surface Water Ambient Monitoring Program (SWAMP). Technical Report 563. Southern California Coastal Water Research Project. Costa Mesa CA.

Fischer, R.A., 2000. Width of riparian zones for birds. EMRRP Technical Notes Collection (TN EMRRP-SI-09), U.S. Army Corps of Engineer Research and Development Center, Vicksburg, MS.

Fischer, R.A., C.O. Martin, and J.C. Fischenich, 2000. Improving riparian buffer strips and corridors for water quality and wildlife. International Conference on Riparian Ecology and Management in Multi-Land Use Watersheds. American Water Resources Association. August 2000.

Foin, T.C., E.J Garcia, R.E. Gill, R. E., S.D. Culberson, and J.N. Collins. 1997. Recovery strategies for the California Clapper Rail (*Rallus longirostris obsoletus*) in the heavily-urbanized San Francisco estuarine ecosystem. Landscape and Urban Planning 38:229–243.

Forman, R. T. T. and Godron, M. 1986. Landscape Ecology. New York, John Wiley & Sons.

Forman, R.T.T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge: Cambridge University Press. Cambridge UK.

French, J.R. 1993. Numerical simulation of vertical marsh growth and adjustment to accelerated sea-level rise, North Norfolk, U.K. Earth Surface Processes and Landforms. 18(1): 63–81.

Geissel, W.H., H.S. Shellhammer and H.T. Harvey. 1988. The ecology of the salt marsh harvest mouse (*Reithrodontomys raviventri*s) in a diked salt marsh. J. Mammalogy 69: 696-703.

Giller, Paul S., and Bjorn Malmqvist. The Biology of Streams and Rivers. New York: Oxford University Press, 1999.

Glick, P., J. Clough, and B. Nunley. 2007. Sea-level rise and coastal habitats in the Pacific Northwest, an analysis for Puget Sound, Southwestern Washington, and Northwestern Oregon. National Wildlife Federation.

Goals Project. 1999. Baylands Ecosystem Habitat Goals: A report of habitat recommendations. Prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. First Reprint. U.S. Environmental Protection Agency, San Francisco, California and San Francisco Bay Regional Water Quality Control Board, Oakland CA.

Greenberg, R. and J.E. Maldanado. 2006. Diversity and endemism in tidal marsh vertebrates. R. Greenberg, S. Droege, J. Maldonado, and M.V. McDonald (eds). Vertebrates of tidal marshes: ecology, evolution, and conservation. Studies in Avian Biology, Lawrence KS.

Greenberg, R., J. Maldonado, S. Droege , and M.V. McDonald M.V. 2006. Tidal marshes: a global perspective on the evolution and conservation of their terrestrial vertebrates. BioScience 56: 675–685.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. BioScience 41:540-551.

Griffith, M.B., P. Husby, R.K. Hall, P.R. Kaufmann and B.H. Hill. 2003. Analysis of macroinvertebrate assemblages in relation to environmental gradients among lotic habitats of California's Central Valley. Environmental Monitoring and Assessment 82:281-309.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman and E.B D. Jones III. 1998. Stream biodiversity: the ghost of landuse 9ast. Proceedings of the National Academy of Sciences 95: 14843-14847.

Harrington, J.M. 1999. California stream bioassessment procedures. California Department of Fish and Game, Water Pollution Control Laboratory. Rancho Cordova, CA.

Harrison S, and E. Bruna. 1999. Habitat fragmentation and large-scale conservation: What do we know for sure? Ecography 22:225–32

Heede, B. H. 1980. Stream dynamics: an overview for land managers. US Forest Service, General Technical Report RM-72.Rocky Mountain Forest and Range Experiment Station,

Hellawell JM. 1986. Biological Indicators of Freshwater Pollution and Environmental Management. Elsevier, London UK.

HRJV. 2004. Version 2.0. The riparian bird conservation plan: a strategy for reversing the decline of riparian associated birds, version 2.0. Riparian Habitat Joint Venture, Sacramento Ca.

Hughes, R.M., J.H. Gakstatter, M.A. Shirazi.and J.M. Omernik. 1982. An approach for determining biological integrity in flowing waters. Pages 877-888 in T.B. Brown (ed). In place resource inventories: principles and practices, a national workshop. Paper presented at the workshop August 9-14, 1981. Society of American Foresters, Bethesda, MD.

Hulst, M.D. 2000. Salt marsh harvest mouse habitat use and habitat quality at San Pablo Bay, California. Master's Thesis, California State University, Sacramento CA.

IPCC. 2001: Climate change 2001: the scientific basis. Contribution of Working Group I to the third assessment report of the intergovernmental panel on climate change. Houghton, J.T.,Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K.Maskell, and C.A. Johnson (eds.). Cambridge University Press, Cambridge UK.

IPCC. 2007: Climate change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E Hanson (eds.). Cambridge University Press, Cambridge, UK.

Johnson, C.W. and S.Buffler. 2008. Riparian buffer design guidelines for water quality and wildlife habitat functions on agricultural landscapes in the intermountain west. USDA Forest Service Rocky Mountain Research Station, Ft. Collins CO.

Johnston, C.A., N.E. Detenbeck, and G.J. Niemi. 1990. The cumulative effect of wetlands on stream water quality and quantity - a landscape approach: Biogeochemistry, 10: 105-141.

Jones and Stokes Associates, H.T. Associates, and John Blayney Associates. 1979. Protection and restoration of San Francisco Bay fish and wildlife habitat. U.S. Fish and Wildlife Service and CA Department of Fish and Game I and II, U.S. Fish and Wildlife Service, Sacramento CA.

Karr, J.R. 1999. Defining and measuring river health. Freshwater Biology 41-221-234.

Karr, J.R. and D.R. Dudley. (1981). Ecological Perspectives on Water Quality Goals. Environ. Manage. 5: 55-68.

Karr, J.R., K.D. Fausch, P.L. Angermier, P.R. Yant, and I.J. Schlosser. (1986). Assessing biological integrity in running waters: a method and its rationale. Spec. Publ. 5. Ill. Nat. Hist. Surv.

Kirwan, M.L. and A.B. Murray. 2007. A coupled geomorphic and ecological model of tidal marsh evolution. Proceedings of the National Academy of Sciences of the United States of America. 104(15): 6118-6122.

Koenig, W.D., D. Van Vuren, and P.N. Hooge. 1996. Detectability, philopatry, and the distribution of dispersal distances in vertebrates. Trends in Ecological and Evolution. 11:514–517.

Kolozsvary M.B. and R.K. Swihart. 1999. Habitat fragmentation and the distribution of amphibians: patch and landscape correlates in farmland. Can. J. Zool. 77:1288–99.

LaBonte, J. R., D. W. Scott, J. D. McIver, J. L. Hayes. 2001. Threatened, endangered and sensitive insects in Eastern Oregon and Washington forests and adjacent lands. Northwest Science. 75:185-198.

Lane, E.W. 1955. The importance of fluvial morphology in hydraulic engineering. Proceedings 81:1-17.

Lane, E.W. 1957. A study of the shape of channels formed by natural streams flowing in erodible material. Series Number 9. US Army Engineer Division, Missouri River Corps of Engineers, Missouri River Division Sediment. Omaha, NE.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. Fluvial processes in geomorphology. W.H. Freeman and Company, San Francisco.

Lindenmayer, D.B. and J. Fischer. 2006. Habitat fragmentation and landscape change: an ecological and conservation synthesis. Island Press, Washington, DC.

Lindenmayer, D.B., and H.A. Nix. 1993. Ecological Principles for the Design of Wildlife Corridors. Conservation Biology 7:627-630.

Mander, U., V. Kuusemets, K. Lohmus, and T. Mauring. 1997. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. Ecological Engineering 8: 299-324.

McGarigal, K., S. Tagil, and S.A. Cushman. 2009. Surface metrics: an alternative to patch metrics for the quantification of landscape structure. Landscape Ecology. 24: 433-450.

McKee, LJ and M. Lewicki. 2009. Watershed specific and regional scale suspended sediment load estimates for Bay Area small tributaries. A technical report for the Sources Pathways and Loading Workgroup of the Regional Monitoring Program for Water Quality. San Francisco Estuary Institute, Contribute 566, Oakland CA.

MEA. 2005. Ecosystems and human well-being: synthesis. Millennium Ecosystem Assessment (MEA), Island Press, Washington DC. 155pp.

Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: Wetlands and water synthesis . World Resources Institute, Washington, DC.

Mohamoud, Y.M., A. Sigleo and R. Parmar. 2009. Modeling the impacts of hydromodification on water quantity and quality. USEPA Publication No. EPA/600/R-09/116. U.S. Environmental Protection Agency, Athens GA.

Murray, B.C., Jr (1967) Dispersal in vertebrates, Ecology 48,975-978.

Naiman, R.J., H. DeCamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3:209-212.

Naiman, R.J., H. Decamps, and M.E. McClain. 2005. Riparia: ecology, conservation, and management of streamside communities. Academic Press, New York NY.

Natural Resources Agency. 2010. State of the state's wetlands: ten years of challenges and progress. Natural Resources Agency, State of California, Sacramento CA.

Nichols, F.H., J.E. Cloern, S.N. Luoma, and D.H. Peterson. 1986. The Modification of an Estuary. Science. 231: 567-573.

NIST/SEMATECH.2001. e-handbook of statistical methods, http://www.itl.nist.gov/div898/handbook/, June 2011.

NOS. 2000. Tidal datums and their applications. NOAA Technical Report NOS CO-OPS 1, Center For Operational Oceanographic Products and Services, Silver Spring MD.

Ode, P.R. 2007. Standard operating procedures for collecting macroinvertebrate samples and associated physical and chemical data for ambient bioassessments in California. California State Water Resources Control Board Surface Water Ambient Monitoring Program (SWAMP), Bioassessment SOP 001, Sacramento CA.

Orr, M., S. Crooks, and P.B. Williams. 2003. Will restored tidal marshes be sustainable? In: L.R. Brown (ed.). Issues in San Francisco Estuary Tidal Wetlands Restoration. San Francisco Estuary and Watershed Science. 1(1): Article 5.

Orson, R.A., R.S. Warren and W.A. Niering. 1987. Development of a tidal marsh in a New England river valley. Estuaries. 10(1): 20-27.

Overton, C.T. 2007. Data summary: a review of literature regarding California clapper rail (*Rallus longirostris obsoletus*) demographics, habitat use, home range, movements, and effects of disturbance. Prepared For California Coastal Conservancy, San Francisco Estuary Invasive Spartina Project, Berkeley CA.

Phillips, J.D. 1989. An evaluation of the factors determining the effectiveness of water quality buffer zones. Journal of Hydrology 107:133-145.

Polyakov, V., A. Fares, and M.H. Ryder. 2005. Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: a review. Environmental Reviews 13:129-144.

Redfield, A.C. 1972. Development of a New England salt marsh. Ecological Monographs. 42:201–237.

Reed, P.B. Jr. 1988. National list of plant species that occur in wetlands: 1988 National summary. US. Fish and Wildlife Service. Biological Report 88(24):1-244.

Rehn, A.C. and P.R. Ode. 2007. Comparisons of targeted-riffle and reach-wide benthic macroinvertebrate samples: implications for data sharing in stream-condition assessments, 26(2): 332-348.

Rehn, A.C., J.T. May, and P.R. Ode. 2008. An Index of Biotic Integrity (IBI) for perennial streams in California's Central Valley. Surface Water Ambient Monitoring Program, State Water Resources Control Board, Sacramento Ca.

Rhen, A.C. and P. Ode. 2006. Development of a Benthic Index of Biotic Integrity (B-IBI) for Wadeable Streams in Northern Coastal California and its Application to Regional 305(b) Assessment. Draft Report. Prepared by the California Dept. of Fish and Game, Aquatic Bioassessment Laboratory. Rancho Cordova, CA.

Rosenberg D.M. and V.H. Resh. 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. New York: Chapman and Hall, New York NY.

Rosenburg, D. K., B. R. Noon, and E. C. Meslow. 1997. Biological corridors: Forms, function, and efficacy. Bioscience. 47(10): 677-687.

Rosgen, D. L. 1994. A classification of natural rivers. *Catena* 22:169–199.

Rosgen, D.L. 1996. Applied River Morphology. Wildland Hydrology. Pagosa Springs CO.

SBSPRP. 2007. South Bay Salt Pond Restoration Project final environmental impact statement/environmental impact report. Prepared for the U.S. Fish and Wildlife Service and Ca Department of Fish and Game, Sacramento CA.

Schumm, S.A. 1969. River metamorphosis. Journal of the Hydraulics Division 95:255-273.

Schumm, S.A., M.D. Harvey and C.C. Watson. 1984. Incised Channels: Morphology, Dynamics, and Control. Water Resources Publications. Littleton, CO.

Schwarzbach, S.E., J.D. Albertson, and C.M. Thomas. 2006. Effects of predation, flooding, and contamination on reproductive success of California clapper rails (*Rallus longirostris obsoletus*) in San Francisco Bay. The Auk 123: 1-16.

SFBRWQCB. 2010. Basin Plan, Chapter 2: Beneficial Uses. San Francisco Bay Regional Water Quality Control Board, Oakland CA. http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/planningtmdls/basinplan/web/bp_ch2.shtml.

SFEI 2000. Ecoatlas: Historical and modern views of San Francisco Bay Baylands (http://www.sfei.org/ecoatlas).

SFEI. 2010. BAARI standards for mapping surface aquatic resources. (http://www.wrmp.org/docs/SFEI%20MAPPING%20STANDARDS_01062011_v3.pdfrandards).

Shellhammer, H.S. 1989. Salt marsh harvest mice, urban development and rising sea levels. Conservation Biology 3: 59-65.

Shellhammer, H.S. 2000. Salt marsh harvest mouse. *in:* Olofson, P.R. (ed.). Baylands Ecosystem Species and Community Profiles: life histories and environmental requirements of key plants, fish, and wildlife. Goals Project (Baylands Ecosystem Habitat Goals), San Francisco Bay Regional Water Quality Control Board, Oakland CA.

Shellhammer, H.S., R. Jackson, W. Davilla, A.M. Gilroy, H.T. Harvey and L. Simons. 1982. Habitat preferences of salt marsh harvest mice (*Reithrodontomys raviventris*). The Wasmann Journal of Biology 40(1-2):102-114.

Soulé, M. E., editor. 1987. Viable populations for conservation. Cambridge, UK: Cambridge University Press

Stein E.D. , A.E. Fetscher, R.P. Clark, A. Wiskind, J.L. Grenier, M. Sutula, J.N. Collins, C. Grosso. 2009. Validation of a Wetland Rapid Assessment Method: Use of EPA's Level 1-2-3 Framework for Method Testing and Refinement. Wetlands 29(2):648-665. 2009.

Stevens, D.L., Jr. and A.R. Olsen. 2004. Spatially-balanced sampling of natural resources. Journal of American Statistical Association 99(465): 262-278.

Stevenson, J. C. and M. S. Kearney. 2009. Impacts of global climate change on tidal wetlands. In: M. Bertness, B. Silliman, and D. Strong (eds.): Human impacts on salt marshes: a global perspective. University of California Press, Berkeley CA.

Sutula, M., J.N. Collins, A. Wiskind, C. Roberts, C. Solek, S. Pearce, R. Clark, A.E. Fetscher, C. Grosso, K. O'Connor, A. Robinson, C. Clark, K. Rey, S. Morrissette, A. Eicher, R. Pasquinelli, M. May, K. Ritter. 2008. Status of perennial estuarine wetlands in the state of California: Final Report to the Surface Water Ambient Monitoring Program and the State Water Resources Control Board. Technical Report 571. Costa Mesa CA.

Sutula, M.A., E.D. Stein, J.N. Collins, A.E. Fetscher, R. Clark. 2006. A practical guide for the development of a wetland assessment method: the California experience. Journal of the American Water Resources Association 42(1):157-175.

SWRCB. 1968. Resolution no. 68-16: statement of policy with respect to maintaining high quality waters in California. State Water Resources Control Board, Sacramento CA.

Taberski, K. K. Rissa-Anserson, O. Otis, M. Cover, N. White, and R. Katznelson. 2010. The Surface water ambient monitoring program: a view from the SWAMP. In: The Pulse of the Estuary: Linking the Watersheds and the Bay. SFEI Contribution 613. San Francisco Estuary Institute, Oakland, CA.

Tiner, RW. 2003. Dichotomous keys and mapping codes for wetland landscape position, landform, water flow path, and waterbody type descriptors. U.S. Fish and Wildlife Service, National Wetlands Inventory Program, Northeast Region, Hadley, MA.

Troll, C. 1971. Landscape ecology (geoecology) and biogeocenology - a terminology study. Geoforum 8(71): 43-46.

Trush, W.J., S.M. McBain and L.B. Leopold, 2000. Attributes of an alluvial river and their relation to water policy and management, Proceedings of the National Academy of Sciences, 97(22):1158-1163.

Turner, M.G. 1989. Landscape ecology: the effect of pattern on process. Annual Review of Ecology and Systematics.20: 171-197.

Turner, M.G. 2005. Landscape ecology: what is the state of the science? Annual Review of Ecology, Evolution, and Systematics. 36: 319-344.

Turner, M.G., R.H. Gardner, and R.V. O'Neill. 2001. Landscape ecology in theory and practice. Springer-Verlag New York NY.

U.S. Climate Change Science Program. 2009. Synthesis and assessment product 4.1, coastal sensitivity to sea level rise: a focus on the Mid-Atlantic Region. U.S. Climate Change Science Program and the Subcommittee on Global Change Research, U.S. Environmental Protection Agency, Washington DC.

U.S. Coast and Geodetic Survey. 1897. Report of the Superintendent of the Coast Survey, Showing the Progress of the Survey During the Year 1897. Washington, DC.

U.S. Coast Survey. 1857. The Report of the Superintendent of the Coast Survey, Showing the Progress of that Work During the Year Ending November 1st, 1857. Washington, DC.

U.S. Fish and Wildlife Service. 2009. Draft Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California. Sacramento, California.

USBR, USFWS, and CDFG. 2010. Suisun Marsh habitat management, preservation, and restoration plan. Draft Environmental Impact Statement/Environmental Impact Report. US Bureau of Reclamation, Sacramento CA.

USEPA. 2007. National management measures guidance to control nonpoint source pollution from hydromodification. EPA 841-B-07-002. U.S. Environmental Protection Agency, Washington DC.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Vinson, M.R., C.P. Hawkins. 1998. Biodiversity of stream insects: variation at local, basin and regional scales. Annual Review of Entomology 43:271-193.

Wallace, J.B., and J.R. Webster. 1996. The role of macroinvertebrates in stream ecosystem function. Annual Review of Entomology 41:115-139

Watson, E.B. and R. Byrne. 2009. Abundance and diversity of tidal marsh plants along the salinity gradient of the San Francisco Estuary: implications for global change ecology. Plant Ecology 205(1): 113-128.

Wenger, S. 1999. A review of the scientific literature on riparian buffer width, extent, and vegetation. Office of Public Service and Outreach, Institute of Ecology, University of Georgia, Athens GA.

White, H.M. 2011. Riparian bird-habitat association models: a framework for informing management and developing restoration guidelines in Utah. Unpublished Master's Thesis in Ecology, Utah State University, Logan UT.

Wilcox, B.A., and D.D. Murphy. 1985. Conservation strategy: The effects of fragmentation on extinction. American Naturalist 125:879-87.

Napolitano, N., S. Potter and D. Whyte. 2003. Conceptual approach for developing sediment TMDLs for San Francisco Bay Area streams. San Francisco bay Regional Water Quality Control Board, Oakland CA.

Stillwater Sciences and W. Dietrich, 2002. Napa river basin limiting factors analysis - final technical report. Prepared for California Coastal Conservancy and San Francisco bay Regional Water Quality Control Board, Oakland CA.

State of San Francisco Bay 2011
Appendix E

LIVING RESOURCES - Shrimp and Crab Indicators
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background and Rationale

The San Francisco Estuary is important habitat for several shrimp and crab species, including Bay shrimp, which once supported an extensive commercial fishery in the Bay, and Dungeness crab, an icon of San Francisco's Fisherman's Wharf. Even today, California's commercial crab fishery relies heavily on crabs that rear in the Bay, feeding and growing in the estuary's brackish waters and tidal marshes for the first year or two of their lives before migrating to the ocean to mature and breed.

Abundance and distribution of shrimp and crabs in the Bay is affected by environmental conditions in the estuary and in the nearby ocean, and different species use different regions of the estuary. Estuarine species like the Bay shrimp, which prefer low salinity waters, are strongly influenced by the amounts and timing of freshwater inflows (Kimmerer 2002). Other species restricted to higher salinity habitats closer to the Golden Gate may be more affected by environmental conditions in the nearby ocean. Thus, while measures of shrimp and crab abundance, distribution and species composition within the Bay can be useful biological indicators for environmental conditions in the estuary, they must be interpreted carefully because they may also be affected by the ocean conditions outside the Bay (Cloern et al., 2007, 2010).

The State of San Francisco Bay 2011 report uses several indicators to assess the condition of the shrimp and crab communities in the San Francisco Estuary. The simplest ones measure abundance, or "how many?" shrimp and crabs does the estuary support. For shrimp, this measurement is also made for the different regions of estuary, from Central Bay near the Golden Gate, which is essentially a marine environment, to Suisun Bay, which is strongly affected by freshwater inflows from the Sacramento and San Joaquin Rivers. Another indicator for shrimp compares the abundance and distribution of species that prefer low salinity waters to those that prefer saltier waters. For both shrimp and crabs, other indicators measure species composition and the prevalence of non-native species in the estuary.

II. Data Source

All of the indicators were calculated using data from the California Department of Fish and Game (CDFG) Bay Study surveys, conducted every year since 1980.[1] The Bay Study collects crabs and shrimp using an otter trawl, which is towed near the bottom and selectively captures shrimp, crabs and demersal fishes that utilize bottom and near-bottom habitats. Each year, the survey samples the same 35 fixed stations in the estuary. These stations are relatively evenly distributed throughout the estuary, among the four sub-regions of the estuary (South, Central, San Pablo and Suisun Bays), and among channel and shoal habitats, and they are sampled once per month for most months of the year.[2] Information on sampling stations, locations and total number of surveys conducted each year in each of the four sub-regions is shown in Figure 1 and Table 1. The Bay Study survey collects and identifies seven species of shrimp (five native

[1] Information on the CDFG Bay Study is available at http://www.dfg.ca.gov/delta/projects.asp?ProjectID=BAYSTUDY

[2] The Bay Study samples more than four dozen stations but the 35 sampling stations used to calculate the indicators are the original sampling sites for which data are available for the entire 1980-2008 period.

native species) (Table 2). It should be noted that, although the Bay Study Otter Trawl survey samples the Bay's open water benthic habitats reasonably comprehensively, it does not survey historic or restored tidal marsh or tidal flat habitats where these 12 species, as well as other crustacean species, may also be found. Therefore, results of the Bay Study and of these indicators should not be interpreted to mean that these are the only species of crustaceans found in the Bay or that these species are found in only these regions of the estuary.

III. Methods and Calculations

Three indicators were developed to assess conditions and trends in the shrimp community in the San Francisco Estuary. For each year, native shrimp abundance in the estuary was measured as:

$$\text{\# native shrimp}/10{,}000\ m^2 = [(\text{\# of native shrimp})/(\text{\# of trawls x av. trawl area, } m^2)] \times (10{,}000)$$

Native shrimp abundance was also measured using this equation for each sub-region of the estuary (i.e., South, Central, San Pablo and Suisun Bay/western Delta).

A second indicator compared the abundance and distribution of native estuarine shrimp that prefer low salinity waters (i.e., Bay shrimp, *C. franciscorum*) with that of native shrimp that prefer saltier waters (all other native shrimp species) and with the two non-native shrimp species (both of which prefer brackish waters). Abundance was calculated as above. Distribution was calculated by comparing the relative abundance of the shrimp populations in each of the four sub-regions of the estuary using the following equation:

$$\text{Distribution = SD of: } \frac{(\text{\#shrimp}/10{,}000m^2_{\text{(South Bay)}})}{(\text{\#shrimp}/10{,}000\ m^2_{\text{(SF estuary)}})}, \frac{(\text{\#shrimp}/10{,}000m^2_{\text{(Central Bay)}})}{(\text{\#shrimp}/10{,}000\ m^2_{\text{(SF estuary)}})}, (_{\text{(other sub-regions)}})$$

A large standard deviation (SD) indicated that the shrimp population was concentrated in one or two sub-regions of the estuary while a small SD indicated that the population was more evenly among the four sub-regions.

A third indicator assessed the species composition of the shrimp community in the four sub-regions by measuring the percentage of the total shrimp community was comprised of native shrimp.

$$\text{\% native shrimp} = [\text{\# native shrimp}/(\text{\# native shrimp + \# non-native shrimp})] \times 100$$

Three indicators were used to assess conditions and trends in the crab community. For each year, native crab abundance in the estuary was measured as:

$$\text{\# native crabs}/10{,}000\ m^2 = [(\text{\# of crabs})/(\text{\# of trawls x av. trawl area, } m^2)] \times (10{,}000)$$

The second indicator measured and compared the abundance of Dungeness crabs (*Cancer magister*) and of Rock crabs (*C. antennarius*, *C. gracilis*, and *C. productus*). Abundance for each was calculated as above.

The final indicator assessed species composition of the crab community by measuring the percentage of the total crab community was comprised of native crabs.

% native crabs = [# native crabs/(# native crabs + # non-native crabs)] x 100

VI. Indicator Evaluation and Reference Conditions

The San Francisco Estuary Partnership's Comprehensive Conservation and Management Plan (CCMP) calls for "recovery" and "reversing declines" of estuarine fish and wildlife but does not provide quantitative targets or goals. The length of the available data records allows for use of historical data to establish "reference conditions." However, there is also good evidence that characteristics of the shrimp and crab communities in the Bay, in particular abundance, are influenced by environmental and ecological conditions in the nearby Pacific Ocean, outside of the estuary and not directly linked to local estuarine or marsh habitat, freshwater inflow or pollution conditions (e.g., Cloern et al., 2007; 2010). Therefore, evaluation of indicators based on indicator levels measured in the estuary in the past may not reflect changes in the estuary's health but rather unrelated changes in ocean conditions and must be interpreted cautiously. With this caveat, the reference condition for the abundance indicators was established as the average abundance for the first ten years of the Bay Study, 1980-1989. Abundance levels that were greater than the 1980-1989 average were considered to reflect "good" conditions.

There is an extensive scientific literature on the relationship between the presence and abundance of non-native species and ecosystem conditions. In general, ecosystems with high percentages of non-natives (e.g., >50%) are considered to be seriously degraded while high percentages of native species (e.g., >85-95%) are indicative of less impacted ecosystems. San Francisco Estuary is known to be heavily invaded with non-native species (Cohen and Carlton, 1998), with some non-native species present in the Bay for more than 100 years and new species being introduced every year. Therefore, the reference condition was established at 85% native species.[3] Percentages that were greater than this value were considered to reflect "good" conditions.

The distribution indicator for shrimp was analyzed and interpreted but not compared to a quantitative reference condition.

For all the indicators, differences among sub-regions and different time periods, and trends with time were evaluated using analysis of variance and simple linear regression.

V. Results

Results of the estuary-wide native shrimp and crab abundance indicators are shown in Figures 2-8.

Abundance of native shrimp and crabs in the San Francisco Estuary has increased.

[3] This is the same reference level used in the species composition indicators for the estuary's fish community.

Since the 1980s, abundance of both shrimp and crabs in the San Francisco Bay has increased significantly (regression, $p<0.001$, both tests) (Figure 2). From the 1980s to the most recent decade, shrimp abundance doubled, from 3,645 shrimp/10,000m^2 to 7,745 shrimp/10,000m^2. Crab abundance increased by more than 400%, from just 9.6 crabs/10,000m^2 to 44.7 crabs/10,000m^2. For both groups, the increase occurred in the late 1990s and coincided with a shift in the Pacific Decadal Oscillation (PDO) from its "warm regime" (mid-1970s to the later 1990s) to its "cool regime" (NFSC, 2010), as well as an unusually wet sequence of years and high freshwater outflow conditions (see Freshwater Inflow Index). The short duration decline in crab abundance in the mid-2000s also coincided with a short duration reversal in the PDO to a warm regime

Abundance and trends in abundance of native shrimp differ among the four sub-regions of the estuary.

When the Bay Study survey began, native shrimp were significantly more abundant in Suisun and San Pablo Bays than in Central and South Bays (Kruskal-Wallis One Way Analysis of Variance for 1980-1989; Suisun>Central and South, San Pablo>South; $p<0.05$, all listed comparisons) (Figure 3). During the past three decades, shrimp abundance has significantly increased in all sub-regions of the estuary except Suisun Bay (regression, $p<0.05$, all tests). The magnitudes of the population increases differed substantially: shrimp abundance increased more than ten-fold in Central Bay, doubled in South Bay but only increased by 45% in San Pablo Bay. Now, in the most recent decade, native shrimp are significantly more abundant in Central Bay than most other sub-regions of the estuary (Kruskal-Wallis One Way Analysis of Variance for 1999-2008; Central>South and Suisun; $p<0.05$, all listed comparisons).

Increased shrimp abundance is attributable to increased abundance of "coastal" shrimp species.

The increase in shrimp abundance in the San Francisco Bay was driven by substantial increases in the abundance of shrimp species that prefer saltier water (referred to as "coastal" species in Figure 4, middle panel) and which are distributed in the downstream regions of the estuary (see also Figure 3). Population increases in "coastal" shrimp occurred throughout the three decades of the Bay Study survey but tended to coincide with periods of dry hydrological conditions, when freshwater inflows to the estuary were low (i.e., late 1980s to early 19902; early 2000s, and 2007-2008). In contrast, during those periods, abundance of Bay shrimp, which prefer low salinity water, tended to decline (Figure 4, top panel). The distributions of these two types of shrimp also responded to hydrological conditions. In "wet" years, the distribution of Bay shrimp broadened as the shrimp were able to occupy more downstream regions of the estuary (i.e., the distribution metric decreased). During dry periods, Bay shrimp distribution became more concentrated in the upstream region of the estuary (i.e., the distribution metric increased). Coastal shrimp species exhibited the opposite pattern, with broader distributions during dry periods than during wet periods. Non-native shrimp abundance and distribution have fluctuated throughout the past three decades, with no significant trends or no clear patterns relative to either hydrological or ocean conditions (Figure 4, bottom panel). However, the distribution of non-native shrimp in the estuary has broadened (regression, $p<0.05$) as the more abundant Oriental shrimp, which was concentrated in Suisun and San Pablo Bays during 1980s and 1990s, has become established in South Bay.

Variations in the crab abundance largely reflect changes in ocean conditions.
Both Dungeness crab and the other rock crab species exhibited significant increases in abundance in the late 1990s (Figure 5), coincident with a regime shift in the Pacific Decadal Oscillation, from a "warm regime" (~1975-1997) to a "cool regime" (~1998-2002) (Figure 6). For both groups, abundance declined sharply during the short-duration shift back to a "warm regime" in the mid-2000s and then rebounded with the return of "cool regime" conditions in 2008.

The San Francisco Estuary's shrimp and crab communities are dominated by native species.
Native species dominate the San Francisco Estuary's shrimp (Figure 7) and crab communities (Figure 8). The two non-native shrimp species (one, a recent arrival first reported in the 2001) prefer brackish, low salinity waters but even in Suisun Bay, the most upstream region of the estuary, they comprise an average of only 5% of the shrimp population. In all other regions of the estuary, the shrimp community is >98% native. As a percentage of the shrimp community, the prevalence of non-native shrimp species in the estuary has not changed in the past three decades. Only one non-native crab has been reported collected by the Bay Study, the Chinese mitten crab. This species first appeared in the mid-1990s and by 1999 it comprised more than 25% of the crab community. After this, its numbers dropped and, since the mid-2000s, mitten crabs have not been collected by Bay Study surveys.

VI. Summary and Conclusions

Collectively, the abundance, distribution and species composition indicators provide a reasonably comprehensive picture of the shrimp and crab communities in the open water benthic habitats of San Francisco Bay. The results illustrate the influences of environmental conditions both within and outside of the estuary on the shrimp and crab communities in the Bay. With the notable exception of the estuary-dependent Bay shrimp, shrimp and crab abundance appears to be largely driven by ocean conditions, which influence reproduction, recruitment and larval survival for most of these coastal spawning species. In contrast, freshwater inflow to the estuary appears to influence the distribution of shrimp species within the estuary. In dry years, marine-type shrimp species extend their range upstream in the estuary while the abundance of the estuarine Bay shrimp declines and its distribution becomes more restricted. Based on the recent substantial increases in shrimp and crab abundance, as well as the relatively stable Bay shrimp populations, overall ecological conditions in the estuary appear to be "good" for shrimp and crab.

VII. Peer Review

The Shrimp and Crab indicators build upon the methods and indicators developed by The Bay Institute for the 2003 and 2005 Ecological Scorecard San Francisco Bay Index and for the San Francisco Estuary Partnership Indicators Consortium. The Bay Institute's Ecological Scorecard was developed with input and review by an expert panel that included Bruce Herbold (US EPA), James Karr (University of Washington, Seattle), Matt Kondolf (University of California, Berkeley), Peter Moyle (University of California, Davis), Fred Nichols (US Geological Survey, ret.), and Phillip Williams (Phillip B. Williams and Associates).

VIII. References

Cloern, J.E., A.D. Jassby, J.K. Thompson, K. Hieb. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. *Proceedings of the National Academy of Sciences of the United States of America* 104(47):18561-18656

Cloern, J. E., K. A. Hieb, T. Jacobson, B. Sansó, E. Di Lorenzo, M. T. Stacey, J. L. Largier, W. Meiring, W. T. Peterson, T. M. Powell, M. Winder, and A. D. Jassby. 2010. Biological communities in San Francisco Bay track large-scale climate forcing over the North Pacific. Geophysical Research Letters, Vol. 37, L21602, doi:10.1029/2010GL044774.

Cohen A.N. and J. T. Carlton. 1998. Accelerating invasion rate in a highly invaded estuary. Science 279:555–558.

Kimmerer, W. J. 2002. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25:1275-1290.

Northwest Fisheries Science Center. 2011. National Oceanic and Atmospheric Administration Pacific Decadal Oscillation website, available at: http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/ca-pdo.cfm.

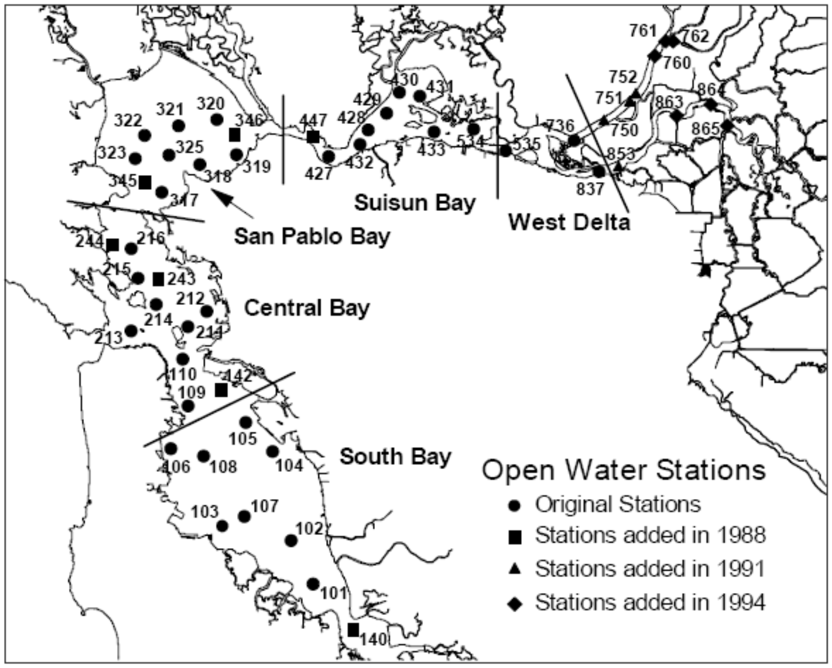


Figure 1. Locations of the sampling stations for the CDFG Bay Study Otter Trawl survey in different sub-regions of the San Francisco Bay. For the Crab and Shrimp indicators 2007 Fish Index, only data from the "original stations" (sampled continuously for 1980-2006 period) were used to calculated indicators for four sub-regions: South Bay, Central Bay, San Pablo Bay, and Suisun Bay (which for this study includes the West Delta sub-region).

Table 1. Sampling stations and total numbers of surveys conducted per year (range for the 1980-2009 period) by the CDFG Bay Study Otter Trawl survey in each of four sub-regions of San Francisco Bay. See Figure 1 for station locations.

Sub-region	Sampling stations	Number of surveys (range for 1980-2005 period)
South Bay	101, 102, 103, 104, 105, 106, 107, and 108	64-96
Central Bay	109, 110, 211, 212, 213, 214, 215, and 216	64-96
San Pablo Bay	317, 318, 319, 320, 321, 322, 323, and 325	64-96
Suisun Bay (includes West Delta sub-region shown in Figure 1)	425, 427, 428, 429, 430, 431, 432, 433, 534, 535, 736, and 837	88-132

Table 2. Shrimp and crab species collected by the CDFG Bay Study Otter Trawl survey, 1980-2009.

Common name	Scientific name	Native v Non-native
Shrimp		
Bay shrimp	*Crangon franciscorum*	Native
Blacktail bay shrimp	*C. nigricauda*	Native
Blackspotted bay shrimp	*C. nigromaculata*	Native
Stimpson coastal shrimp	*Heptacarpus stimpsoni*	Native
Smooth bay shrimp	*Lissocrangon stylirostric*	Native
Siberian prawn	*Exopalaemon modestus*	Non-native
Oriental shrimp	*Palaemon macrodactylus*	Non-native
Crabs		
Dungeness crab	*Cancer magister*	Native
Pacific rock crab	*C. antennarius*	Native
Graceful rock crab	*C. gracilis*	Native
Red rock crab	*C. productus*	Native
Chinese mitten crab	*Eriocheir sinensis*	Non-native

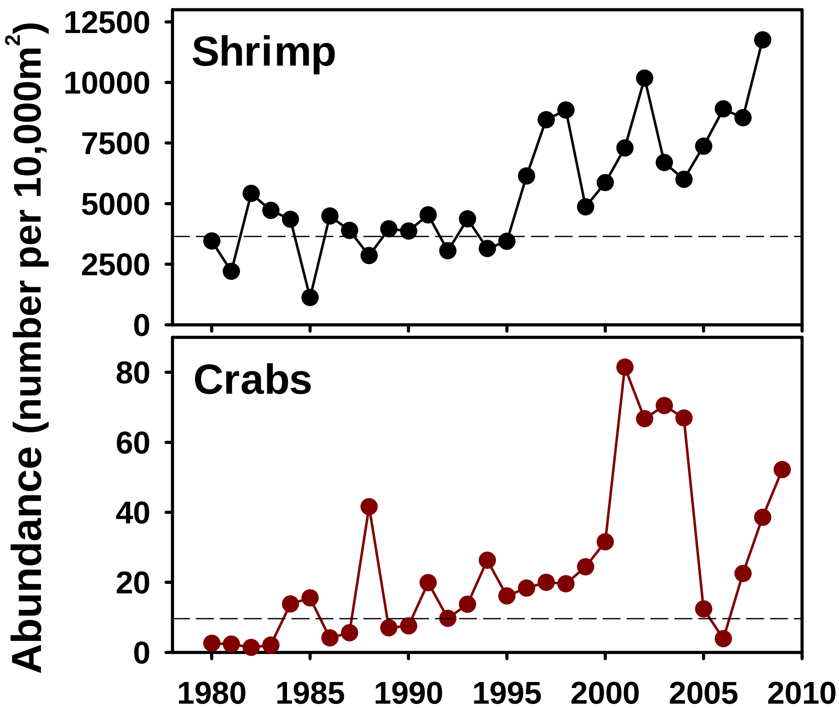


Figure 2. Changes in the Native Shrimp Abundance (top panel) and Native Crab Abundance indicators. Horizontal dashed line shows the reference condition (1980-1989 average).

Figure 3. Changes in the native shrimp abundance in each of the four sub-regions of the San Francisco Estuary, from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

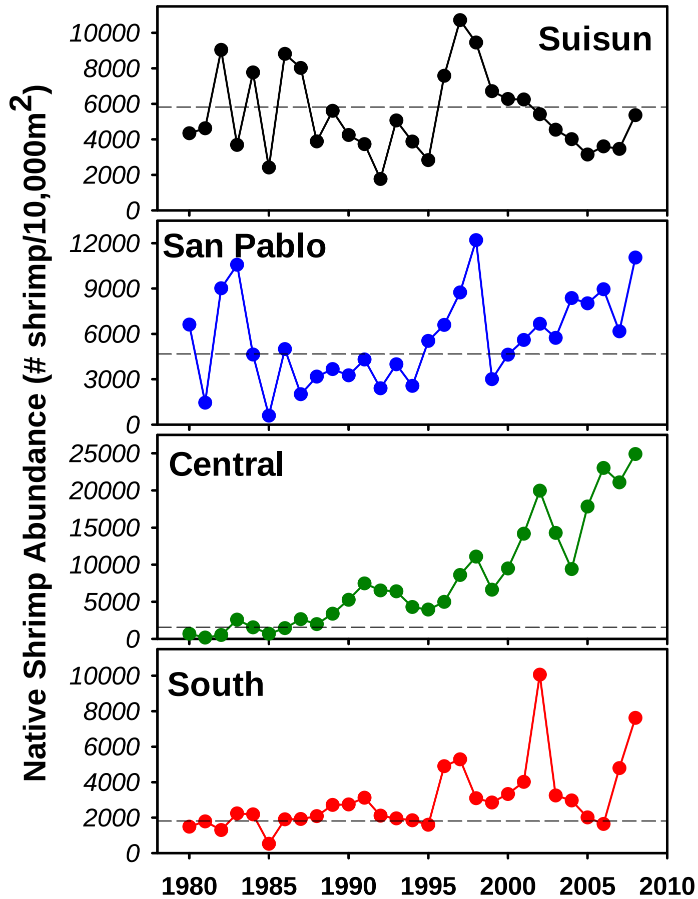
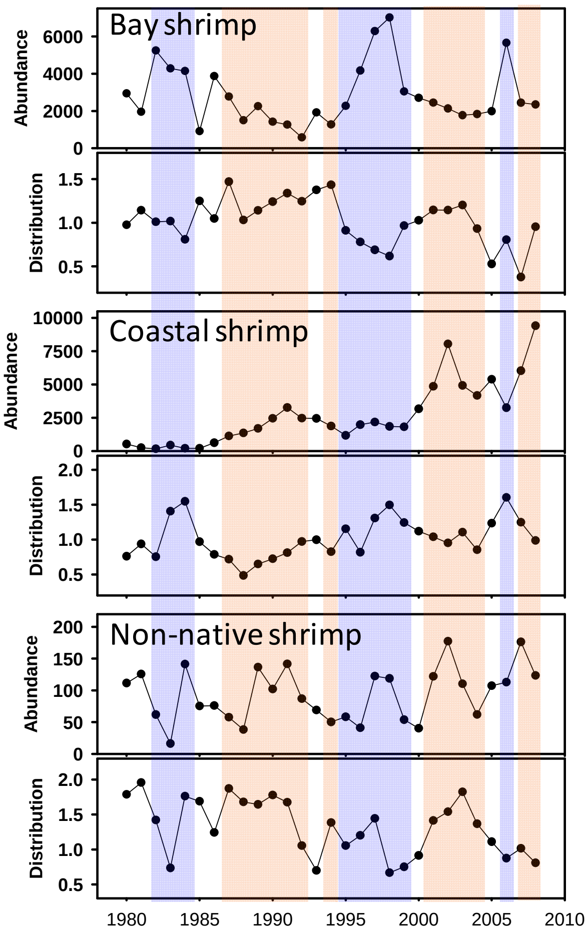


Figure 4. Abundance and distribution of Bay shrimp (top panels), which prefer low salinity waters, “coastal” shrimp (middle panels), which prefer saltier waters, and non-native shrimp (bottom panels). For distribution, low numbers (<1.0) indicate relatively broad and even distribution among the four sub-regions of the estuary, while larger numbers (>1.0) indicate narrower and more uneven distribution within the estuary. The pink bars indicate dry hydrological conditions (i.e., “dry” or “critically dry” freshwater outflow conditions) and the blue bars indicate “wet” hydrological conditions with high freshwater outflows..



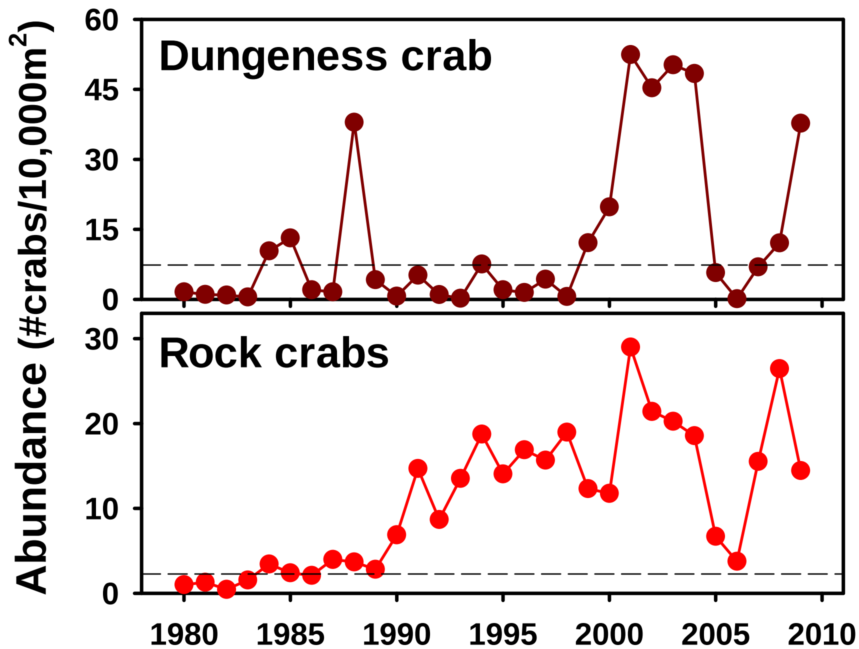


Figure 5. Changes in the Dungeness crab (top panel) and Rock crab (bottom panel) abundance indicators. Horizontal dashed line shows the reference condition (1980-1989 average).

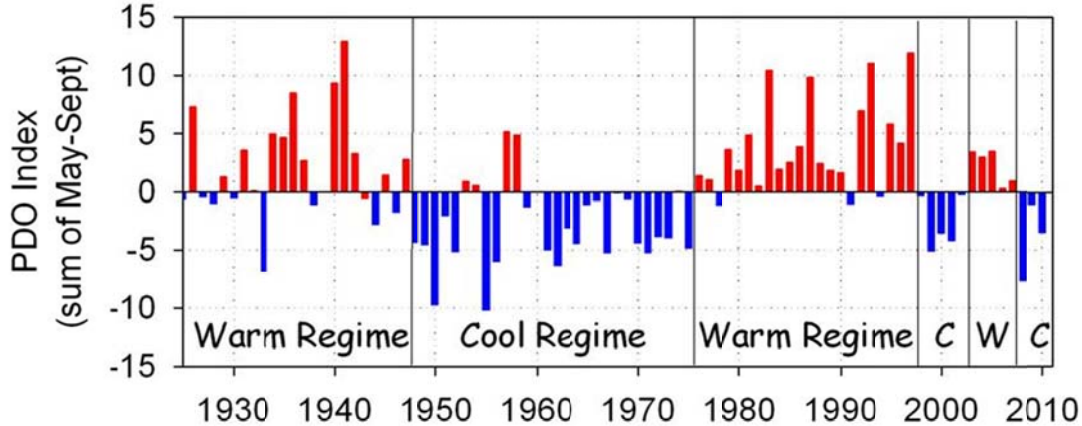
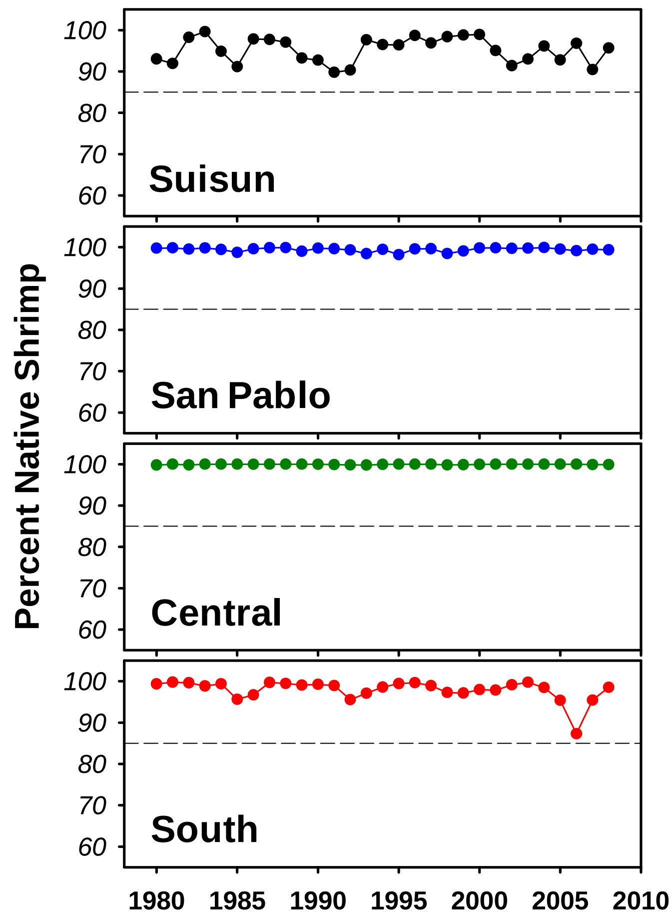


Figure 6. Time series of shifts in sign of the Pacific Decadal Oscillation (PDO), 1925 to 2010. Values are averaged over the months of May through September. Red bars indicate positive (warm) years; blue bars negative (cool) years. Note that 2008 was the most negative since 1956. From: Northwest Fisheries Science Center, National Oceanic and Atmospheric Administration, available at: http://www.nwfsc.noaa.gov/research/divisions/fed/oeip/ca-pdo.cfm

Figure 7. Changes in the Percent Native Shrimp indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (85% native).



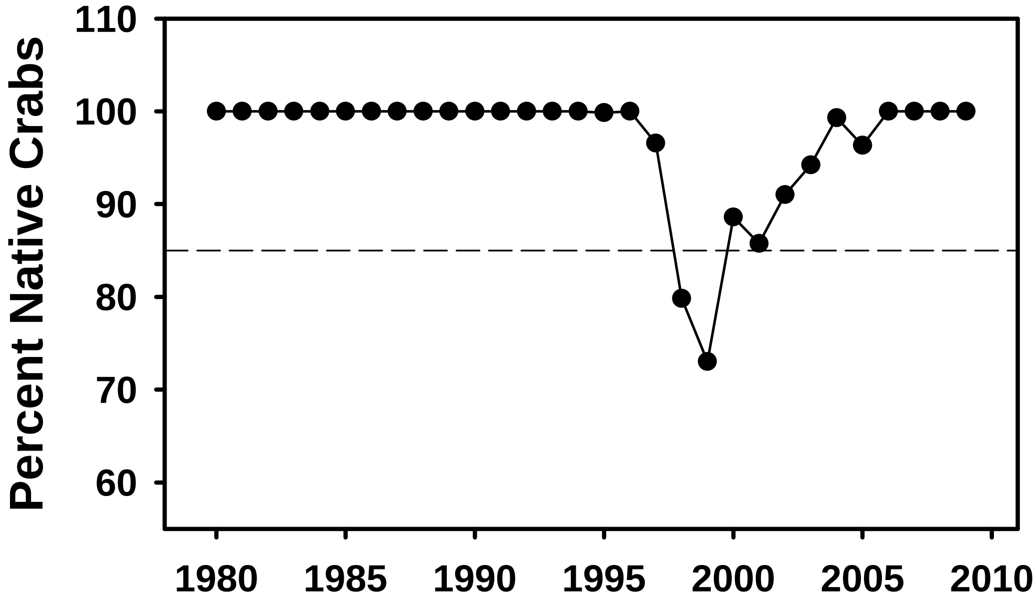


Figure 8. Changes in the Percent Native Crabs indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (85% native).

State of San Francisco Bay 2011
Appendix F

LIVING RESOURCES - Fish Indicators and Index
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background

San Francisco Bay is important habitat for more than 100 fish species, including commercially important Chinook salmon and Pacific herring, popular sport fishes like striped bass and sturgeon, and delicate estuary-dependent species like delta smelt. These fishes variously use the estuary for spawning, nursery and rearing habitat, and as a migration pathway between the Pacific Ocean and the rivers of the estuary's watersheds. Environmental conditions in the estuary – the amounts and timing of freshwater inflows, the extent of rich tidal marsh habitats, and pollution – affect the numbers and types of fish that the Bay can support. Thus, measures of fish abundance, diversity, species composition and distribution are useful biological gauges for environmental conditions in the estuary. A large, diverse fish community that is distributed broadly throughout the Bay and dominated by native species is a good indicator of a healthy estuary.

The Fish Index uses ten indicators to assess the condition of the fish community within the San Francisco Bay. Four of the indicators measure abundance, or "how many?" fish the estuary supports. Two indicators measure the diversity of the fish community, or "how many species?" are found in the Bay. Two indicators measure the species composition of the fish community, or "what kind of fish?," in terms of how many species and how many individual fish are native species rather than introduced non-natives.[1] The final two indicators assess the distribution of fish within the estuary, or "where are the fish?," measuring the percentage of sampling locations where native fishes are found. For each year, the Fish Index is calculated by combining the results of the ten indicators into a single number.

Because the estuary is so large and its environmental conditions so different in different areas – for example, Central Bay, near the Golden Gate is essentially a marine environment while Suisun Bay is dominated by freshwater inflows from the Sacramento and San Joaquin Rivers – the types of fishes found in each area differ. Therefore, each of the indicators and the index was calculated separately for four "sub-regions" in the estuary: South Bay, Central Bay, San Pablo Bay and Suisun Bay and the western Delta (Figure 1). For each year and for each sub-region, the Fish Index is calculated by combining the results of the ten indicators into a single number.

II. Data Source

All of the indicators were calculated using data from the California Department of Fish and Game (CDFG) Bay Study surveys, conducted every year since 1980.[2] The Bay Study uses two different types of sampling gear to collect fish from the estuary: a midwater trawl and an otter trawl. The midwater trawl is towed from the bottom to the top of the water column and

[1] Native species are those that have evolved in the Bay and/or adjacent coastal or upstream waters. Non-native species are those that have evolved in other geographically distant systems and have been subsequently transported to the Bay and established self-sustaining populations in the estuary.

[2] Information on the CDFG Bay Study is available at http://www.dfg.ca.gov/delta/projects.asp?ProjectID=BAYSTUDY

collect smaller and/or younger fish that are too slow to evade the net.[3] The otter trawl is towed near the bottom and captures demersal fishes that utilize bottom and near-bottom habitats and also tends to collect smaller and/or younger fish. Each year, the two survey sample the same 35 fixed stations in the estuary. These stations are distributed among the four sub-regions of the estuary and among channel and shoal habitats, once per month for most months of the year.[4] In one year, 1994, the Midwater Trawl survey was conducted during only two months, compared to the usual 8-12 months per year. Because the sampling period was limited, data from this year were not included in calculation of some indicators and of the Fish Index. Information on sampling stations, locations and total number of surveys conducted each year in each of the four sub-regions is shown in Figure 2 and Table 1.

It should be noted that, although the Bay Study midwater and otter trawl surveys sample the Bay's pelagic and open water benthic habitats reasonably comprehensively, they do not survey historic or restored tidal marsh or tidal flat habitats where many of the same fish species collected by the Bay Study, as well as other fish species, may also be found. Therefore, results of the Bay Study and of these indicators should not be interpreted to mean that these are the only fishes or fish communities found in the Bay or that these species are found in only these regions of the estuary.

III. Indicator Evaluation

The San Francisco Estuary Partnership's Comprehensive Conservation and Management Plan (CCMP) calls for "recovery" and "reversing declines" of estuarine fish and wildlife but does not provide quantitative targets or goals. However, the length of the available data records, which include the Bay Study surveys used for the indicator calculations here as well as several other surveys, allows for use of historical data to establish "reference conditions."[5] There is also an extensive scientific literature on development, use and evaluation of ecological indicators in aquatic systems and, because San Francisco Bay is among the best studied estuaries in the world, an extensive scientific literature on its ecology.

For each indicator, a "primary" reference condition was established. This reference condition was based on either measured values from the earliest years for which quantitative data were available (1980-1989 for the Bay Study survey), maximum measured values for the estuary or sub-regions, recognized and accepted interpretations of ecological conditions and ecosystem health (e.g., native v non-native species composition), and best professional judgment. Measured indicator values that were higher than the primary reference condition were interpreted to mean the indicator results met the CCMP goals and to correspond to "good" ecological conditions. For each of the four sub-regions, reference conditions were identically selected but

[3] The Bay Study primarily catches fishes that range in size from approximately 1-12 inches (3-30 cm). Other survey programs that monitor fishes in the estuary target smaller or larger fishes (e.g., CDFG 20-mm survey for small juvenile fishes or CDFG creel surveys for adult fishes).

[4] The Bay Study samples more than four dozen stations but the 35 sampling stations used to calculate the indicators are the original sampling sites for which data are available for the entire 1980-2006 period.

[5] For example, CDFG's Fall Midwater Trawl Survey, conducted in most years since 1967, and Summer Townet Survey, conducted since 1959. However, the geographic coverage of the Fall Midwater trawl and Summer Townet surveys is less extensive than that of the Bay Study and does not extent into all of the four sub-regions of the estuary. Therefore, data from these surveys were less suitable for developing indicators for the entire estuary.

for some indicators their absolute values were calibrated to account for differences among the sub-regions. For example, a reference condition based on historical abundance (i.e., average abundance during the first ten years of the survey) was used to evaluate the abundance indicators but, because overall fish abundance levels differed among the sub-regions, the actual reference abundance level differed among the four sub-regions. In contrast, because the reference condition for the species composition indicators was based the ecological relationship between the prevalence of non-native species and ecosystem and habitat condition, the value of the reference condition was set at the same level for each of the regions, despite the large differences in species composition that already existed between the four sub-regions.

In addition to the primary reference condition, information on the range and trends of indicator results, results from other surveys, and known relationships between fish community attributes and ecological conditions were used to develop several intermediate reference conditions, creating a five-point scale for a range of evaluation results from "excellent," "good, "fair," "poor" to "very poor".[6] The size of the increments between the different evaluation levels was, where possible, based on observed levels of variation in the measured indicator values (e.g., standard deviations) in order to ensure that the different levels represented meaningful differences in the measured indicator values. Each of the evaluation levels was assigned a quantitative value from "4" points for "excellent" to "0" points for "very poor." An average score was calculated for the indicators in each of the fish community attributes (i.e., abundance, diversity, species composition and distribution) and the Fish Index was calculated as the average of these four scores. Specific information on the primary and intermediate reference conditions is provided in the following sections describing each of the indicators.

Differences among sub-regions and different time periods, and trends with time in the indicators and the multi-metric index were evaluated using analysis of variance and simple linear regression. Comparisons among sub-regions were made using results from the entire 29-year period as well as for the earliest ten-year period (i.e., the reference period; 1980-1989) and the most recent five years (i.e., 2004-2008). Regression analyses were conducted using continuous results for the entire 29-year period for each sub-region.

IV. Indicators

A. Fish Community Attributes

The ten indicators used to calculate the Fish Index assess four different attributes of the San Francisco Estuary fish community: abundance, diversity, species composition and distribution (Table 2). Information on indicator rationale, calculation methods, units of measure, specific reference conditions and results is provided in the following sections.

[6] For example, data from the Fall Midwater trawl and Summer Townet surveys indicate that abundance of fish within the estuary was already in decline by the 1980s. Therefore, for indicator evaluation, abundance levels measured in the 1980s, which were already lower than they have been just ten years earlier, were interpreted to correspond to "good" conditions but not "excellent" conditions.

B. Abundance Indicators

1. Rationale

Abundance (or population size) of native fish species within an ecosystem can be a useful indicator of aquatic ecosystem health, particularly in urbanized watersheds (Wang and Lyons, 2003; Harrison and Whitfield, 2004). Native fishes are more abundant in a healthy aquatic ecosystem than in one impaired by altered flow regimes, toxic urban runoff and reduced nearshore habitat, the usual consequences of urbanization. In the San Francisco Bay, abundances of a number of fish (and invertebrate) species are strongly correlated with ocean conditions immediately outside of the estuary (Cloern et al., 2007; 2010) and freshwater inflow from the estuary's Sacramento and San Joaquin watersheds, which vary widely due to California's climate and but have been reduced and stabilized by water development, flood control efforts, agriculture and urbanization (Jassby et al., 1995; Kimmerer, 2002; and see Estuarine Open Water Habitat indicator, Freshwater Inflow Index and Flood Events indicator).

The Fish Index includes four different abundance indicators, each measuring different components of the native fish community within the estuary. The **Pelagic Fish Abundance** indicator measured how many native pelagic, or open water, fish are collected in the Midwater trawl survey. This indicator does not include data for Northern anchovy because, in most years and in most sub-regions of the estuary, northern anchovy comprised >80% of all fish collected in the Bay and obscured results for all other species. **Northern Anchovy Abundance** was measured as a separate indicator, using data from the Midwater trawl survey. Northern anchovy, the most abundant species collected in the Bay, is consistently collected in all sub-regions of the estuary in numbers that are often orders of magnitude greater than for all other species. The **Demersal Fish Abundance** indicator measured how many native demersal, or bottom-oriented, fish are collected by the Otter Trawl Survey. The **Sensitive Fish Species Abundance** indicator measured the abundance of four representative species – longfin smelt, Pacific herring, starry flounder and striped bass[7] – using data from both the Midwater and Otter trawl surveys. All of these species are broadly distributed throughout the Bay and rely on the estuary in different ways and at different times during their life cycle. Each is relatively common and consistently present in all four sub-regions of the estuary, and all except starry flounder are targets of environmental or fishery management in the estuary. In addition, the population abundance of each of these species is influenced by a key ecological driver for the estuary, seasonal freshwater inflows (Jassby et al. 1995; Kimmerer 2002). Key characteristics of each of the four species are briefly described below

- **Longfin smelt** are found in open waters of large estuaries on the west coast of North America.[8] The San Francisco Estuary population spawns in upper estuary (Suisun Bay and Marsh and the Delta) and rears downstream in brackish estuarine and, occasionally,

[7] Although striped bass is not native to the Pacific coast, the species was introduced to San Francisco Bay more than 100 years ago and, since then, has been an important component of the Bay fish community. On the North American west coast, the main breeding population of the species is in the San Francisco Bay (Moyle, 2002).
[8] In California, longfin smelt are found in San Francisco Bay, Humboldt Bay, and the estuaries of the Russian, Eel, and Klamath rivers.

coastal waters (Moyle, 2002). The species was listed as "threatened" under the California Endangered Species Act in 2008.

- **Pacific herring** is a coastal marine fish that uses large estuaries for spawning and early rearing habitat. The San Francisco Estuary is the most important spawning area for eastern Pacific populations of the species (CDFG, 2002). Pacific herring supports a commercial fishery, primarily for roe (herring eggs) but also for fresh fish, bait and pet food. In the San Francisco Estuary, the Pacific herring fishery is the last remaining commercial finfish fishery.

- **Starry flounder** is an estuary-dependent, demersal fish that can be found over sand, mud or gravel bottoms in coastal ocean areas, estuaries, sloughs and even fresh water. The species, whose eastern Pacific range extends from Santa Barbara to arctic Alaska, spawns near river mouths and sloughs; juveniles are found exclusively in estuaries. Starry flounder is one of the most consistently collected flatfishes in the San Francisco Estuary.

- **Striped bass** was introduced into San Francisco Bay in 1879 and by 1888 the population had grown large enough to support a commercial fishery (Moyle, 2002). That fishery was closed in 1935 in favor of the sport fishery, which remains popular today although at reduced levels. Striped bass are anadromous, spawning in large rivers and rearing in downstream estuarine and coastal waters. Declines in the striped bass population were the driving force for changes in water management operations in Sacramento and San Joaquin Rivers and the Delta in the 1980s. Until the mid-1990s, State Water Resources Control Board-mandated standards for the estuary were aimed at protecting larval and juvenile striped bass.

2. Methods and Calculations

The **Pelagic Fish Abundance** indicator was calculated for each year (1980-2008, excluding 1994) for each of four sub-regions of the estuary using catch data for all native species except northern anchovy from the Bay Study Midwater Trawl survey. The indicator was calculated as:

$$\text{\# fish/10,000 m}^3 = [(\text{\# of fish})/(\text{\# of trawls x av. trawl volume, m}^3)] \text{ x } (10{,}000)$$

The **Northern Anchovy Abundance** indicator was calculated for each year (1980-2008, excluding 1994) for each of four sub-regions of the estuary using catch data for northern anchovy from the Bay Study Midwater Trawl survey using the same equation as for pelagic abundance.

The **Demersal Fish Abundance** indicator was calculated for each year (1980-2008) for each of four sub-regions of the estuary using catch data for all native species from the Bay Study Otter Trawl survey. The indicator was calculated as:

$$\text{\# fish/10,000 m}^2 = [(\text{\# of fish})/(\text{\# of trawls x av. trawl volume, m}^2)] \text{ x } (10{,}000)$$

The **Sensitive Fish Species Abundance** indicator, the abundance of each of the four species was calculated for each year (1980-2008, excluding 1994) for each of four sub-regions of the estuary as the sum of the abundances from each of the two Bay Study surveys using the equations below.

$$\# \text{ fish}/10{,}000 \text{ m}^3 = [(\# \text{ of fish})/(\# \text{ of trawls x av. trawl volume, m}^3)] \text{ x } (10{,}000)$$
(for Midwater trawl)

$$\# \text{ fish}/10{,}000 \text{ m}^2 = [(\# \text{ of fish})/(\# \text{ of trawls x av. trawl area, m}^2)] \text{ x } (10{,}000)$$
(for Otter trawl)

The summed abundance for each species was then expressed as a percentage of the average 1980-1989 for that species. The indicator was calculated as the average of the percentages for the four species. Each species was given equal weight in this calculation.

3. Reference Conditions

For the four Abundance indicators, the primary reference condition was established as the average abundance for the first ten years of the Bay Study, 1980-1989. Abundance levels that were greater than the 1980-1989 average were considered to reflect "good" conditions. Additional information from other surveys and trends in fish abundance within the estuary was used to develop several other intermediate reference conditions. Table 3 below shows the quantitative reference conditions that were used to evaluate the results of the abundance indicators.

4. Results

Results of the **Pelagic Fish Abundance** indicator are shown in Figure 3.

Abundance of pelagic fishes differs among the estuary's sub-regions.
Pelagic fishes are significantly more abundant in Central Bay than in all other sub-regions of the estuary (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$). Abundance of pelagic fishes in South Bay is greater than that in Suisun Bay ($p<0.05$) but comparable to that in San Pablo Bay. In 2008, pelagic fishes were three times more abundant in Central Bay (89 fish/10,000m^3) than either South (30 fish/10,000m^3) or San Pablo Bays (32 fish/10,000m^3) and nearly 30 times more abundant than in Suisun Bay (3 fish/10,000m^3).

Abundance of pelagic fishes has declined in most sub-regions of the estuary.
Pelagic fish abundance declined significantly over time in all sub-regions of the estuary except Central Bay (regression: $p<0.05$ for South and San Pablo Bays, $p<0.001$ for Suisun Bay). Abundance of pelagic fishes in Central Bay showed no long-term trend and its high inter-annual variability reflects the periodic presence of large numbers of marine species such as Pacific sardine. However, for the most recent five years (2004-2008) compared to 1980-1989 levels, average abundance of native pelagic fishes was significantly lower in all regions: 55% lower in South Bay, 65% lower in Central Bay, 68% lower in San Pablo Bay and 88% lower in Suisun Bay.

Based on the abundance of pelagic fishes, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met.

Both current levels (expressed as the 2004-2008 average) and trends in pelagic fish abundance are below the 1980-1989 reference period for all sub-regions of the estuary (t-test or Mann-Whitney, $p<0.05$, all regions). However, in the most recent two years there is some evidence of increases in pelagic fish abundance in all sub-regions of the San Francisco Estuary except Suisun Bay.

Results of the **Northern Anchovy Abundance** indicator are shown in Figure 4.

Abundance of northern anchovy differs among the estuary's sub-regions.

Although northern anchovy are always found in all sub-regions of the estuary, their abundance differs markedly. For the past 29 years, northern anchovy have been more abundant in Central Bay (mean: 1000 fish/10,000m^3) than all other sub-regions, least abundant in Suisun Bay (18 fish/10,000m^3), and present at intermediate abundance levels in San Pablo (259 fish/10,000m^3) and South Bays (304 fish/10,000m^3) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

Trends in abundance of Northern anchovy differ in different sub-regions of the estuary.

During the past 29 years, abundance of northern anchovy has been variable but roughly stable in South and Central Bays although, in most recent years, Central Bay abundance has averaged about 45% lower than 1980-1989 levels. Northern anchovy abundance has steadily declined in San Pablo Bay (regression: $p<0.01$), falling to 41% of 1980-1989 levels during the most recent five years (2004-2008). The decline was more abrupt in Suisun Bay (regression: $p<0.05$), with northern anchovy virtually disappearing from this upstream portion of the estuary: since 1995, northern anchovy population levels in this region of the estuary averaged less than 6% of 1980-1989 levels and less than 2% of populations in adjacent San Pablo Bay. This decline is contemporaneous with the establishment of the non-native overbite clam (*Corbula amurensis*) at high densities, the general disappearance of phytoplankton blooms and substantial declines in the abundance of several previously abundant zooplankton species.

Based on the abundance of northern anchovy, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met in the upstream sub-regions of the estuary.

The abundance of northern anchovy, the most common fish in the San Francisco Estuary, has declined throughout the upstream regions of the estuary to levels that significantly below the 1980-1989 average reference conditions (t-test or Mann-Whitney, $p<0.05$ for San Pablo and Suisun Bays). In contrast, in Central and San Pablo Bays, recent northern anchovy abundance levels are comparable to levels measured in the 1980s (t-test or Mann-Whitney, $p>0.05$, both regions). As with demersal fishes, the markedly different trends between the upstream sub-regions (Suisun and San Pablo Bays) and downstream sub-regions (Central and South Bays) suggest that different environmental drivers are influencing northern anchovy in different sub-regions of the estuary: ocean conditions in the downstream sub-regions and watershed conditions, in particular hydrological conditions and planktonic food availability, in the upstream sub-regions.

Results of the **Demersal Fish Abundance** indicator are shown in Figure 5.

Abundance of demersal fish species differs among the estuary's sub-regions.
Demersal fishes are more abundant in Central Bay (942 fish/10,000m^2) than in all other sub-regions of the estuary and least abundant in Suisun Bay (50 fish/10,000m^2) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$). Demersal fish abundance in South (288 fish/10,000m^2) and San Pablo Bays (277 fish/10,000m^2) are comparable. In 2008, demersal fishes were nearly ten times more abundance in Central Bay (2093 fish/10,000m^2) than either South (231 fish/10,000m^2) or San Pablo Bays (335 fish/10,000m^2) and nearly 40 times more abundant than in Suisun Bay (54 fish/10,000m^2).

Abundance of demersal fishes has increased in Central Bay and declined in Suisun Bay.
During the past 29 years, abundance of native demersal fishes increased in Central Bay (regression: $p<0.05$) but declined in Suisun Bay (regression: $p<0.05$). In South and San Pablo Bays, demersal fish abundance has fluctuated widely. Compared to 1980-1989 levels, recent average abundances (2004-2008) were 56% and 51% lower in Suisun and San Pablo Bays, respectively, and 22% and 161% higher in South and Central Bays, respectively.

Increases in demersal fish abundance in Central and South Bays were driven by multiple species.
In South Bay, increases in demersal fish abundance were largely attributable to high catches of Bay goby, a Bay resident species. In contrast, demersal fish abundance increases in Central Bay in the late 1990s and early 2000s were largely driven by two species of flatfishes, seasonal species that use the estuary as nursery habitat but which maintain substantial populations outside the Golden Gate. It is likely that increases in the abundance of these species reflected improved ocean conditions.

Based on the abundance of demersal fishes, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions except Suisun Bay, the upstream reach of the estuary.
Both current levels (expressed as the 2004-2008 average) and trends in demersal fish abundance were comparable to the 1980-1989 reference period for all sub-regions of the estuary except Central Bay, where demersal fish abundance increased (t-test or Mann-Whitney, $p>0.05$, South, San Pablo and Suisun Bays; $p=0.012$ for Central Bay). However, demersal fish abundance fluctuates widely in all sub-regions of the San Francisco Estuary, suggesting that this indicator may be inadequately responsive to watershed conditions. In addition, the different trends between the upstream sub-regions (Suisun and San Pablo Bays) and downstream sub-regions (Central and South Bays) suggest that different environmental drivers are influencing demersal fish abundance in the different sub-regions of the estuary: ocean conditions in the downstream sub-regions and watershed conditions, in particular hydrological conditions, in the upstream sub-regions.

Results of the **Sensitive Fish Species Abundance** indicator are shown in Figure 6.

Abundances of longfin smelt, Pacific herring, starry flounder and striped bass differ among the different sub-regions of the estuary.

The Bay-wide abundance of the four species was roughly comparable (although starry flounder densities are generally lower than those of the pelagic species), but different species use different sub-regions within the estuary. Longfin smelt and starry flounder are most abundant in San Pablo, Suisun and Central Bays and rare in South Bay. Pacific herring are most commonly found in Central, South and San Pablo Bays and rarely collected in Suisun Bay. Striped bass are mostly collected in Suisun Bay and, to a lesser extent, San Pablo Bay and rarely found in Central and South Bays.

Abundance of sensitive fish species has declined in all sub-regions of the estuary.
During the past 29 years, combined abundance of the four sensitive fish species has declined in all sub-regions of the estuary (regression: $p<0.05$ all sub-regions). For the most recent five-year period (2004-2008), abundance of sensitive fish species abundance Central Bay is just 20% of that sub-region's 1980-1989 average, 32% in San Pablo Bay, 35% in South Bay and 51% in Suisun Bay. The higher abundances measured in Suisun Bay in 2008 reflect increases in Pacific herring and starry flounder, species that are relatively uncommon in that sub-region. In each sub-region, most of the decline occurred during the late 1980s and early 1990s and, with the exceptions of a few single years in different sub-regions, the abundance of the four sensitive fish species has remained below 50% of the 1980-1989 since then.

Abundance declines were measured for most of the species in most sub-regions of the estuary.
All of the species except Pacific herring declined significantly in the sub-region in which they were most prevalent (regression: $p<0.05$ for all species except Pacific herring in Central Bay). Longfin smelt declined in both San Pablo and Suisun Bays (regression: $p<0.05$ both tests), starry flounder declined in Central and San Pablo Bays (regression: $p<0.05$ both tests), striped bass declined in all sub-regions (regression: $p<0.05$ in all sub-regions except South Bay, where $p=0.051$), and Pacific herring declined in South Bay (regression: $p<0.05$).

Based on the abundance of sensitive fish species, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met in any sub-region of the estuary.
The combined abundance of the four estuary-dependent species assessed with this indicator have fallen to levels that are consistently 50% or less than the 1980-1989 average abundance reference condition. However, sensitive species abundance exhibited high variability during the 1980s, thus recent levels (2004-2008) were significantly lower in only South and Central Bay (t-test or Mann-Whitney, $p<0.05$). Although recent abundance levels in San Pablo and Suisun Bay were markedly lower than during the 1980-1989 reference, the differences were not statistically significant due to high variability during the 1980s. The significant declines measured for three of the four individual species indicates that population declines of estuary-dependent species span multiple species and all geographic regions of the estuary.

C. Diversity Indicators

1. Rationale

Diversity, or the number of species present in the native biota that inhabit the ecosystem, is one of the most commonly used indicators of ecological health of aquatic ecosystems (Karr et al., 2000; Wang and Lyons, 2003; Harrison and Whitfield, 2004). Diversity tends to be highest in healthy ecosystems and to decline in those impaired by urbanization, alteration of natural flow patterns, pollution, and loss of habitat area.

More than 100 native fish species have been collected in the San Francisco Bay by the Bay Study surveys. Some are transients, short-term visitors from nearby ocean or freshwater habitats where they spend the majority of their life cycles, or anadromous migrants, such as Chinook salmon and sturgeon, transiting the Bay between freshwater spawning grounds in the Bay's tributary rivers and the ocean. Other species are dependent on the Bay as critical habitat, using it for spawning and/or rearing, spending a large portion or all of their life cycles in Bay waters.

Of the more than 100 fish species collected by the Bay Study since 1980, 39 species can be considered "estuary-dependent" species (Table 4). These species may be resident species that spend their entire life-cycle in the estuary, marine or freshwater species that depend on the San Francisco Estuary for some key part of their life cycle (usually spawning or early rearing), or local species that spend a large portion of their life cycle in the San Francisco Estuary. Just as diversity, or species richness, of the native fish assemblage is a useful indicator of the ecological health of aquatic ecosystems, diversity of the estuary-dependent fish assemblage is a useful indicator for the ecological health of the San Francisco Estuary.

The Fish Index includes two different diversity indicators. The **Native Fish Species Diversity** indicator uses Midwater and Otter trawl survey data to measure how many of the estuary's native fish species are present in the Bay each year. The **Estuary-dependent Fish Species Diversity** indicator uses data from both surveys to measure how many estuary-dependent species are present each year.

2. Methods and Calculations

The **Native Fish Species Diversity** indicator was calculated for each year and for each of four sub-regions of the estuary as the number of species collected, expressed as the percentage of the maximum number of native species ever collected in that sub-region, using catch data from the Bay Study Midwater and Otter Trawl surveys. The indicator was calculated as:

% of species assemblage = (# native species/maximum # of native species reported) x 100

The **Estuary-dependent Fish Species Diversity** indicator was calculated for each year and for each of four sub-regions of the estuary as the number of estuary-dependent species collected (see Table 4), expressed as the percentage of the maximum number of estuary-dependent species ever collected in that sub-region, using catch data from the Bay Study Midwater and Otter Trawl surveys. The indicator was calculated as:

% of species assemblage =
(# estuary-dependent species/maximum # of estuary-dependent species reported) x 100

3. Reference Conditions:

For the two diversity indicators, the primary reference condition was based on the average diversity (expressed as % of the native fish assemblage present), measured for the first ten years of the Bay Study, 1980-1989, and for all four sub-regions combined. Diversity levels that were greater than the 1980-1989 average were considered to reflect "good" conditions. The average percentage of the native fish assemblage present during the 1980-1989 period diversity differed slightly among the four sub-regions for the Native Fish Species Diversity indicator (1980-1989 average: 49%; Suisun Bay diversity was lower than that in the other three sub-regions) and significantly for the Estuary-dependent Fish Species Diversity indicators (1980-1989 average: 72%; Suisun Bay was lowest and Central and South Bay were highest). This approach tended to reflect the relatively lower species diversity observed in Suisun Bay in the indicator results. Table 5 below shows the quantitative reference conditions that were used to evaluate the results of the two diversity indicators.

4. Results

Results of the **Native Fish Species Diversity** indicator are shown in Figure 7.

Maximum native species diversity differs among the four sub-regions of the estuary.
The greatest numbers of native fish species are found in Central Bay (94 species) and the fewest are in Suisun Bay (48 species). A maximum of 73 native species have been collected in South Bay and 66 native species have been found in San Pablo Bay.

The percentage of the native fish species assemblage present differs among the sub-regions.
In addition to having a smaller native fish species assemblage, Suisun Bay has a significantly lower percentage (44%) of that assemblage present each year compared to all other sub-regions (48% in Central Bay; 49% in South Bay and 51% in San Pablo Bay) (ANOVA: $p<0.001$, all pairwise comparisons: $p<0.01$). In recent years (2004-2008), native fish diversity has been highest in Central Bay (ANOVA: $p<0.05$ for Central Bay compared to Suisun Bay).

Trends in native species diversity differ among the sub-regions.
Native species diversity has increased significantly in Central Bay (regression: $p<0.01$) with an average of six more species in the most recent five-year period compared to the 1980-1989 reference period. Native fish species diversity decreased significantly in San Pablo Bay (regression: $p=0.05$), with an average of four fewer species in the 2005-2008 period compared to the 1980-1989 period. Native fish species diversity fluctuated in both South and Suisun bays.

Based on the diversity of the native fish community, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary.
Comparison of average native fish species diversity in the most recent five years (2004-2008) to that measured during the 1980-1989 period shows no significant differences except for Central Bay, where diversity is significantly higher (t-test: $p<0.05$).

Results of the **Estuary-dependent Fish Species Diversity** indicator are shown in Figure 8.

The diversity of estuary-dependent species is lower in Suisun Bay than in other sub-regions of the estuary.

Although roughly the same number of estuary-dependent species are found in each sub-region (38 species in San Pablo Bay; 36 species in Central and South Bays; and 31 species in Suisun Bay), a significantly smaller percentage of the estuary-dependent fish assemblage occurs in Suisun Bay (49% of the assemblage) than in all other regions of the San Francisco Estuary (84% in Central Bay; 80% in South Bay; and 69% in San Pablo Bay) (ANOVA: $p<0.001$, all pairwise comparisons, $p<0.05$).

Diversity of Bay-dependent species is generally stable in most sub-regions of the estuary.
Estuary-dependent species diversity has declined slightly in San Pablo Bay (regression: $p<0.05$, for a decrease of 2 species from the 1980-1989 period to the 2004-2008 period) and South Bay (regression: $p<0.05$, for an average decrease of 1.5 species). In all other regions, estuary-dependent diversity has fluctuated but remained relatively stable over the 29-year period.

Based on the diversity of the estuary-dependent fish community, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except South Bay.
Comparison of average estuary-dependent fish species diversity in the most recent five years (2004-2008) to that measured during the 1980-1989 period shows no significant differences, except for South Bay, where diversity of estuary-dependent fishes was significantly lower (Mann-Whitney Rank Sum test: $p<0.05$).

D. Species Composition Indicators

1. Rationale

The relative proportions of native and non-native species found in an ecosystem is an important indicator of ecosystem health (May and Brown, 2002; Meador et al., 2003). Non-native species are most prevalent in ecosystems that have been modified or degraded with resultant changes in environmental conditions (e.g., elevated temperature, reduced flood frequency), pollution, or reduction in area or access to key habitats (e.g., tidal marsh, seasonal floodplain). The San Francisco Estuary has been invaded by a number of non-native fish species. Some species, such as striped bass, were intentionally introduced into the estuary; others have arrived in ballast water or from upstream habitats, usually reservoirs.

The Fish Index includes two different indicators for species composition. The **Percent Native Species** indicator uses Midwater and Otter trawl survey data to measure what percentage of the fish species collected in each sub-region of the estuary are native species. The **Percent Native Fish** uses the survey data to measure what percentage of the individual fish collected in each sub-region of the estuary are native species.

2. Methods and Calculations

The **Percent Native Species** indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of fish species collected in the estuary that are native to the estuary and its adjacent ocean and upstream habitats using the equation below.

% native species = [# native species/(# native species + # non-native species)] x 100

The Percent Native Fish indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of fish collected in the estuary that are native to the estuary and its adjacent ocean and upstream habitats using the equation below.

% native fish = [# native fish/(# native fish + # non-native fish)] x 100

3. Reference Conditions:

There is an extensive scientific literature on the relationship between the presence and abundance of non-native species and ecosystem conditions and the length of the available data record for the San Francisco Estuary allows for establishment of "reference conditions". In general, ecosystems with high proportions of non-natives (e.g., >50%) are considered to be seriously degraded. Furthermore, non-native fish species have been present in the San Francisco Estuary Bay for more than 100 years; therefore, 100% native fish species is unrealistic. Among the four sub-regions, the 1980-1989 average percentage of native species was 87% and the average percentage of native fish was 90%. For both indicators, Suisun Bay values were lowest. Based on this information, the primary reference condition for both indicators was established at 85%. Percent Native Species levels that were greater than this value were considered to reflect "good" conditions. Table 6 below shows the quantitative reference conditions that were used to evaluate the results of the two species composition indicators.

4. Results

Results of the **Percent Native Species** indicator are shown in Figure 9.

The percentage of native species in the fish community differs among the four sub-regions of the estuary.
For the past 29 years, non-native species have been most prevalent in Suisun Bay, where in most years less than 75% of species are natives, intermediate in South and San Pablo Bays (88% and 86% native, respectively), and the least prevalent in Central Bay (92%) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

Trends in the percentage of native species differ among the sub-regions.
The percentage of native species is declining in all sub-region of the estuary except Central Bay. In San Pablo Bay, the percent native species declined significantly (regression: $p<0.001$) from 90% in the 1980-1989 period to 81% in the most recent five-year period. Percent native species declined in Suisun Bay from 77% to 69% (regression: $p<0.01$) and in South Bay the percentage of native species declined from 89% to 85% (regression: $p<0.05$).

Trends in the percentage of native species in Bay fish assemblages are driven by declines in the numbers of native species and increases in non-native species.
During the past 29 years, the number of native species in San Pablo Bay declined by three species and the number of non-native species increased by three, to an average of seven non-

native species of the 2004-2008 period. The number of non-native species collected in Suisun Bay increased by an average of three species, from six species in the 1980-1989 period to nine species in the most recent five years. In South Bay, native species declined by one and non-natives increased by one. In Central, the total number of native species collected increased by six species.

Based on fish species composition, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met in Suisun and San Pablo Bays.
Compared to the 1980-1989 period and the biologically based 85% native species reference condition, recent measurements (2004-2008) of the fish species composition indicate significantly poorer condition for San Pablo Bay (Mann-Whitney Rank Sum test: $p<0.01$) and Suisun Bay (t-test: $p<0.01$). Although both a long-term (1980-2008) and recent (2004-2008) decline were evident in South Bay, the average percentage of native species for the most recent five year period was not significantly different than that for the 1980-1989 reference period.

Results of the **Percent Native Fish** indicators are shown in Figure 10.

The percentage of native fish in the fish community differs among the four sub-regions of the estuary.
For the past 29 years, non-native fish have dominated the Suisun Bay sub-region, where in most years less than 50% of fish collected are natives (1980-2008 average: 49%). Non-native fish are rare in the other three sub-regions. Central Bay has the least (1980-2008 average: 0.1%), South Bay has just 1% non-native fishes and San Pablo Bay less than 3% non-native fishes (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of native fishes is declining in the Suisun and South Bay sub-region of the estuary but not in Central or San Pablo Bays. In Suisun Bay, the percent native fish declined significantly (regression: $p<0.001$) from 63% in the 1980-1989 period to just 37% in the most recent five-year period. Percent native fish declined in South Bay from more than 99% to 96%% (regression: $p<0.01$). The increases in the numbers of non-native fish in South Bay in 2007 and 2008 were largely attributable to higher catches of two non-natives, striped bass and chameleon goby.

Based on fish species composition, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native fish comprise the vast majority of the fish community, exceeding 95% of the total fish present in nearly all years. In Suisun Bay, the percentage of the fish community that is comprised of non-native fish is extremely high and increasing, indicating that the condition of this region of the estuary is poor and deteriorating.

E. Distribution Indicators

1. Rationale

The distribution of native fishes within a habitat is an important indicator of ecosystem condition (May and Brown, 2002; Whitfield and Elliott, 2002; Nobriga et al., 2005). Native fishes may be excluded or less abundant in degraded habitats with unsuitable environmental conditions and/or those in which more tolerant non-native species have become established. The Fish Index includes two indicators to assess the distribution of native fishes within the estuary. The **Pelagic Fish Distribution** indicator uses Midwater trawl survey data to measure the percentage of the survey's sampling stations at which native species were regularly collected. The **Demersal Fish Distribution** indicator uses Otter trawl survey data to make a similar measurement for bottom-oriented native fishes.

5. Methods and Calculations

The **Pelagic Fish Distribution** indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of Midwater trawl survey stations at which at least one native fish was collected in at least 60% of the surveys conducted in that year.

Pelagic Fish Distribution =
(# survey stations with native fish in 60% of surveys)/(# survey stations sampled) x 100

The **Demersal Fish Distribution** indicator was calculated identically using Otter trawl survey data.

6. Reference Conditions:

There is an extensive scientific literature on the relationship between the presence and abundance of non-native species and ecosystem conditions. The length of the available data record for the San Francisco Estuary allows for establishment of "reference conditions". For the two Distribution indicators, the primary reference condition was established based on the number of stations sampled by the Bay Study surveys (8-12 stations per sub-region; therefore the maximum resolution of this indicator is limited to 8-13% increments depending on sub-region) and the average percentage of stations with native species present for the first ten years of the Bay Study, 1980-1989 (~96%). Distribution levels that were greater than the reference condition were considered to reflect "good" conditions. Table 7 below shows the quantitative reference conditions that were used to evaluate distribution indicators.

7. Results

Results of the **Pelagic Fish Distribution** indicator are shown in Figure 11.

The percentage of Midwater trawl survey stations that regularly have native fish differs among the four sub-regions of the estuary.
For the past 29 years, native fish have been consistently present at nearly all Midwater trawl survey stations in all sub-regions of the estuary except Suisun Bay. During the 1980-2008 period, native fish were present at 98-99% of survey stations in South, Central and San Pablo Bays. In contrast, native fish were present in only an average of 81% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, Suisun v all other sub-regions; $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of survey stations with native fish was stable in all sub-regions of the estuary except Suisun Bay. In Suisun Bay, distribution of native fishes declined significantly from 88% of stations (1980-1989) to 63% in the most recent five years (2004-2008) (Mann-Whitney Rank Sum test; $p<0.01$; regression: $p<0.05$). This decline in distribution occurred abruptly in 2003 and is largely drive by low distribution in 2005, when native fish were collected in only five of 12 stations (42%). Prior to 2003, distribution of native pelagic fish in Suisun Bay was generally stable at 86% of stations (1980-2002 average) but since 2003 native pelagic fish were present at only 63% of Suisun Bay stations (2003-2008 average). Native fish were most frequently absent from survey stations located in the lower San Joaquin River and the western region of Suisun Bay.

Based on native pelagic fish distribution, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native pelagic fish are regularly collected at all Midwater trawl survey stations. In contrast, native fish are increasingly absent from the western region of Suisun Bay, the most upstream region of the estuary, suggesting that the condition of this region of the estuary is deteriorating.

Results of the **Demersal Fish Distribution** indicator are shown in Figure 12.)

The percentage of Otter trawl survey stations that regularly have native fish differs among the four sub-regions of the estuary.
For the past 29 years, native fish have been consistently present at nearly all Otter trawl survey stations in all sub-regions of the estuary except Suisun Bay. During the 1980-2008 period, native fish were present at 98-100% of survey stations in South, Central and San Pablo Bays. In contrast, native fish were present in only an average of 81% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, Suisun v all other sub-regions; $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of survey stations with native fish was stable in all sub-regions of the estuary except Suisun Bay. In Suisun Bay, distribution of native fishes declined briefly but significantly in the early 1990s, from 91% of stations (1980-1991) to just 64% of stations (1992-1994), and then recovered to 89% (1995-2000). In 2001, distribution declined significantly again, falling to 62% of stations (2001-2007) before returning to 91% in 2008 (Mann-Whitney Rank Sum test; $p<0.05$ both tests). For the most recent five years (2004-2008), native demersal fish have been present at 62% of stations. Similar to pelagic fish, native demersal fish were most frequently absent from survey stations located in the western region of Suisun Bay.

Based on native demersal fish distribution, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native demersal fish are regularly collected at all Otter trawl survey stations. In contrast, native fish are increasingly absent from the western region of Suisun Bay, the most upstream region of the estuary, suggesting that the condition of this region of the estuary is deteriorating.

V. Fish Index

The Fish Index aggregates the results of the four abundance indicators (Pelagic Species, Demersal Species, Northern Anchovy, and Sensitive Species), two diversity indicators (Native Species and Estuary-dependent Species), two species composition indicators (Percent Native Species and Percent Native Fish) and the two distribution indicators (Pelagic Fish and Demersal Fish Distribution).

A. Index Calculation

For each year and for each sub-region, the Fish Index is calculated by combining the results of the ten indicators into a single number. First, results of the indicators in each fish community attribute (i.e., abundance, diversity, species composition and distribution) were combined by averaging the quantitative scores of each of the component indicators. Within the fish community attribute, each indicator was equally weighted. Next the average scores for each fish community attribute were combined by averaging, with each fish community attribute equally weighted. An index score greater than 3 was interpreted to represent "good" conditions and an index score less than 1 was interpreted to represent "very poor" conditions.

B. Results

Results of the Fish Index for each sub-region~~s~~ are shown in Figure 13.

The Fish Index differs among the four sub-regions of the estuary.
For the 29-year survey period, the Fish Index was highest in the Central Bay (1980-2008 average: 3.14), lowest in Suisun Bay (1.77), and intermediate in South and San Pablo Bays (3.01 and 2.78, respectively) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.05$; Central>South and San Pablo>Suisun). For the most recent five years, the differences among the regions are even greater. The Fish Index was highest in Central (2004-2008 average: 3.025), lowest in Suisun (1.28) and intermediate in South and San Pablo Bays (2.84 and 2.56, respectively). Lower Fish Index values for Suisun Bay at the beginning of the survey period were attributable to lower diversity (i.e., smaller percentages of the sub-region's species assemblage were present) and species composition (i.e., high prevalence of non-native species and non-native fish).

Trends in the Fish Index differ among the sub-regions.
During the 29-year survey period, the Fish Index has declined significantly in Suisun, San Pablo and South Bays but not in Central Bay (regression 1980-2008: $p<0.005$ all sub-regions except Central Bay). The overall condition of the fish community in Suisun Bay has declined from "fair" in the early 1980s (1980-1989 average: 2.21) to consistent "poor" conditions throughout the 1990s and 2000s. In 2006, when diversity, species composition and distribution all dropped, condition of the fish community in Suisun Bay was "very poor." In San Pablo Bay, the Fish Index has declined steadily, from mostly "good" conditions in the early 1980s to "fair" conditions by the 1990s: since then, the San Pablo Bay Fish Index has not fallen to "poor" levels and has continued to decline. The decline in the Fish Index in South Bay, while significant, is

not as severe. In Central Bay, the Fish Index has been relatively stable with generally "good" fish community conditions.

Based on Fish Index, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in only the Central Bay sub-region.
The overall condition of the fish community is "good" in Central Bay, the most downstream region of the San Francisco Estuary. In all other sub-regions of the estuary, the condition of fish community is declining. In Suisun Bay, the most upstream region of the estuary most directly affected by watershed degradation, alteration of freshwater inflows and declines in the quality and quantity of low-salinity habitat, the fish community is in "poor" condition. These declines in the Fish Index are largely driven by declines in fish abundance (all three sub-regions), declining diversity (South and San Pablo Bays), increasing prevalence of non-native species (all three sub-regions), and declines in the distribution of native fish within the sub-region (Suisun Bay).

C. Summary and Conclusions

Collectively, the ten indicators and the Fish Index provide a reasonably comprehensive assessment of status and trends San Francisco Estuary fish community. The results show substantial geographic variation in both the composition and condition of the fish community within the estuary and in the response of specific indicators over time. Table 8 below summarizes the indicator and Index results by sub-region. In addition, the following general conclusions can be made:

1. The San Francisco Estuary fish community differs geographically within the estuary in fish community composition, fish abundance, and trends in various attributes of its condition over time.
2. Different indicators show different responses over time, some demonstrating clear declines in condition over time, others no change, and a few increases. In some cases, the same indicators measured in different sub-regions of the estuary show different responses over time. These results suggest that different physical, chemical or biological environmental variables (or combinations of these variables) influence the fish community response in different sub-regions.
3. Overall condition, as measured individually by the fish indicators and by the Fish Index for the community response, is poorest in upstream region of estuary, Suisun Bay; best in Central Bay, the region most strongly influenced by ocean conditions and with a predominantly marine fish fauna; and intermediate in San Pablo and South Bays. However, condition of the fish community in San Pablo and South Bays is declining and, for San Pablo Bay, could deteriorate to "poor" condition if the current rate of decline continues for the next two decades.
4. Even 30 years ago, the condition of the fish community in Suisun Bay was poorer than in all other sub-regions of the estuary. The fish community was less diverse with relatively lower percentages of the native fish assemblage present, and dominated by high percentages of non-native species.
4. The abundance of pelagic fishes in the estuary (which include Northern anchovy and most of the sensitive species measured in those two indicators) has shown the greatest changes over time, indicating this component of the fish community has low resilience and/or is tightly linked to just one or a few environmental drivers that have also experienced substantial change in conditions during the sampling period.

VI. Peer Review

The Fish indicators and index build upon the methods and indicators developed by The Bay Institute for the 2003 and 2005 Ecological Scorecard San Francisco Bay Index and for the San Francisco Estuary Partnership Indicators Consortium. The Bay Institute's Ecological Scorecard was developed with input and review by an expert panel that included Bruce Herbold (US EPA), James Karr (University of Washington, Seattle), Matt Kondolf (University of California, Berkeley), Pater Moyle (University of California, Davis), Fred Nichols (US Geological Survey, ret.), and Phillip Williams (Phillip B. Williams and Assoc.). These recent versions of the indicators and indices were also reviewed and revised according to the comments of Bruce Herbold and Luisa Valiela (US EPA).

VII. References

CDFG (2002) Pacific herring commercial fishing regulations. Final Supplemental Environmental Document. California Department of Fish and Game, SCH No. 98052052.

Cloern, J.E., A.D. Jassby, J.K. Thompson, K. Hieb. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. *Proceedings of the National Academy of Sciences of the United States of America* 104(47):18561-18656

Cloern, J. E., K. A. Hieb, T. Jacobson, B. Sansó, E.Di Lorenzo, M. T. Stacey,J. L. Largier, W. Meiring, W. T. Peterson, T. M. Powell, M. Winder, and A. D. Jassby. 2010. Biological communities in San Francisco Bay track large-scale climate forcing over the North Pacific. Geophysical Research Letters, Vol. 37, L21602, doi:10.1029/2010GL044774.

Harrison, T. D. and A. K. Whitfield (2004) A multi-metric fish index to assess the environmental condition of estuaries. J. Fish Biology. 65:283-710.

Nobriga, M., F. Feyrer, R. Baxter, and M. Chotkowski. 2005. Fish community ecology in an altered river delta: spatial patterns in species composition, life history strategies, and biomass. Estuaries 28:776-785.

Jassby, A. D., W. J. Kimmerer, S. G. Monismith, C. Armor, J. E. Cloern, T. M. Powell, J. R. Schubel, and T. J. Vendlinski (1995) Isohaline position as a habitat indicator for estuarine populations. Ecol. Appl. 5:272-280.

Karr, J. R. (1981). Assessment of biotic integrity using fish communities. Fisheries 6, 21–27.

Karr, J. R., J. D. Allan, and A. C. Benke (2000) River conservation in the United States and Canada: science, policy and practice. In: P. J. Boon, B. R. Davies, and G. E. Petts (eds.), *River Conservation: Science Policy and Practice*. J. Wiley & Sons, New York, pp. 502-528.

Kimmerer, W. J. (2002) Physical, biological, and management responses to variable freshwater flow into the San Francisco estuary. Estuaries 25:1275-1290.

May, J. T. and L. R. Brown (2002) Fish communities of the Sacramento River Basin: implications for conservation of native fishes in the Central Valley, California. Env. Biol. Fish. 63:373-388.

Meador, M. R., L. R. Brown, and T. Short (2003) Relations between introduced fish and environmental conditions at large geographic scales. Ecological Indicators 3:81-92.

Moyle, P. B. (2002) Inland Fishes of California. University of California Press, Berkeley. 502 pp.

SFEIT (2008) Assessment Framework as a tool for integrating and communication watershed health conditions for the San Francisco Estuary. Technical Report #1, submitted by the San Francisco Estuary Indicators Team to the California Department of Water Resources, September 30, 2008.

Wang, L. and J. Lyons (2003) Fish and benthic macroinvertebrate assemblages as indicators of stream degradation in urbanizing watersheds. In *Biological Response Signatures. Indicator Patterns Using Aquatic Communities*, (ed. T. P. Simon), pp. 227-249. CRC Press: New York.

Whitfield, A. K. & Elliott, M. (2002). Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. Journal of Fish Biology 61 (Suppl. A), 229–250. doi: 10.1006/jfbi.2002.2079.

Figure 1. Because the San Francisco Estuary is so large and its environmental conditions so different in different areas, the Fish Index and each of its component indicators were calculated separately for four "sub-regions" in the estuary: South Bay, Central Bay, San Pablo Bay and Suisun Bay and the western Delta.

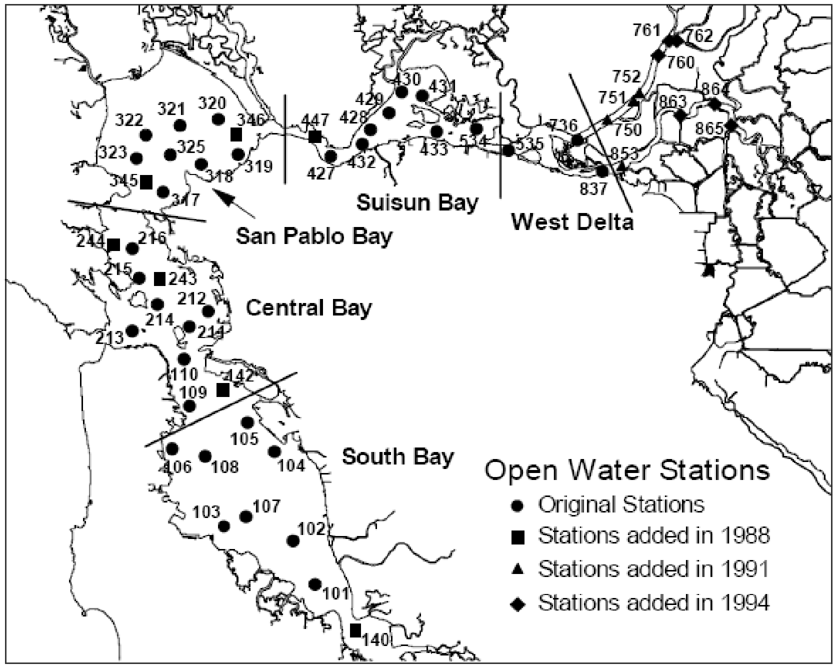


Figure 2. Locations of the sampling stations for the CDFG Bay Study Midwater Trawl and Otter Trawl surveys in different sub-regions of the San Francisco Bay. For the 2007 Fish Index, only data from the "original stations" (sampled continuously for 1980-2006 period) were used to calculated indicators for four sub-regions: South Bay, Central Bay, San Pablo Bay, and Suisun Bay (which for this study includes the West Delta sub-region).

Table 1. Sampling stations and total numbers of surveys conducted per year (range for the 1980-2006 period, excludes 1994) by the CDFG Bay Study Survey in each of four sub-regions of San Francisco Bay. MWT=Midwater Trawl survey; OT= Otter Trawl survey. See Figure 1 for station locations.

Sub-region	Sampling stations	Number of surveys (range for 1980-2005 period)
South Bay	101, 102, 103, 104, 105, 106, 107, and 108	64-96 (MWT) 64-96 (OT)
Central Bay	109, 110, 211, 212, 213, 214, 215, and 216	64-96 (MWT) 64-96 (OT)
San Pablo Bay	317, 318, 319, 320, 321, 322, 323, and 325	64-96 (MWT) 64-96 (OT)
Suisun Bay (includes West Delta sub-region shown in Figure 1)	425, 427, 428, 429, 430, 431, 432, 433, 534, 535, 736, and 837	87-132 (MWT) 88-132 (OT)

Table 2. Fish community characteristics and indicators used to calculate the Fish Index.

Fish Community Characteristic	Indicators
Abundance	• Pelagic Fish Abundance • Northern Anchovy Abundance • Demersal fish Abundance • Sensitive Species Abundance
Diversity	• Native Fish Diversity • Estuary-dependent Fish Diversity
Species Composition	• Percent Native Species • Percent Native Fish
Distribution	• Pelagic Fish Distribution • Demersal Fish Distribution

Table 3. Quantitative reference conditions and associated interpretations for the results of the fish abundance indicators. The primary reference condition, which corresponds to “good” conditions, is in bold.

Abundance Indicators (Pelagic Fish, Northern Anchovy, Demersal Fish, Sensitive Species)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>150% of 1980-1989 average	“Excellent”	4
>100% of 1980-1989 average	**“Good”**	**3**
>50% of 1980-1989 average	“Fair”	2
>15% of 1980-1989 average	“Poor”	1
≤15% of 1980-1989 average	“Very Poor”	0

Figure 3. Changes in the Pelagic Fish Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

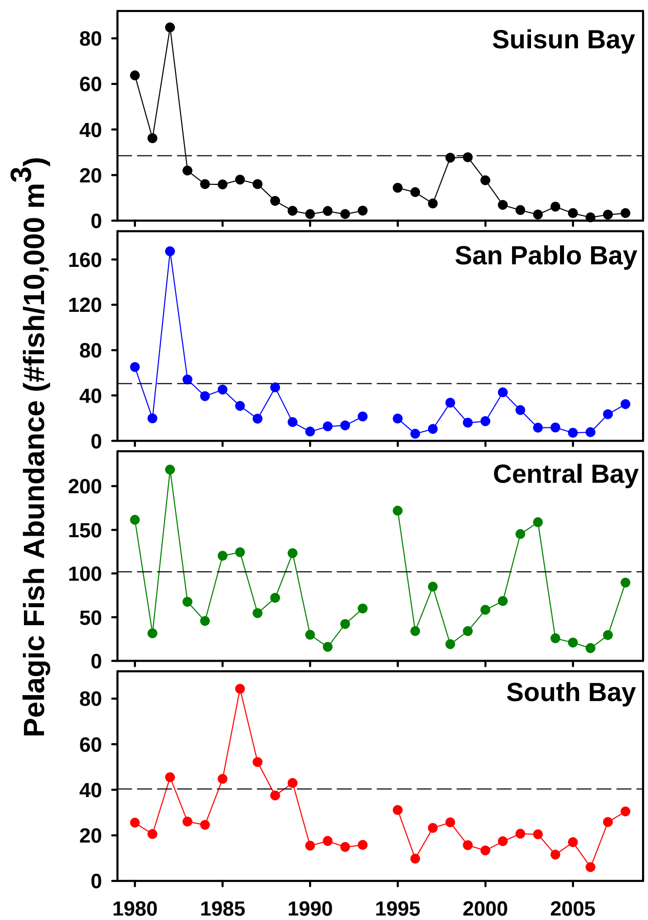


Figure 4. Changes in the Northern Anchovy Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

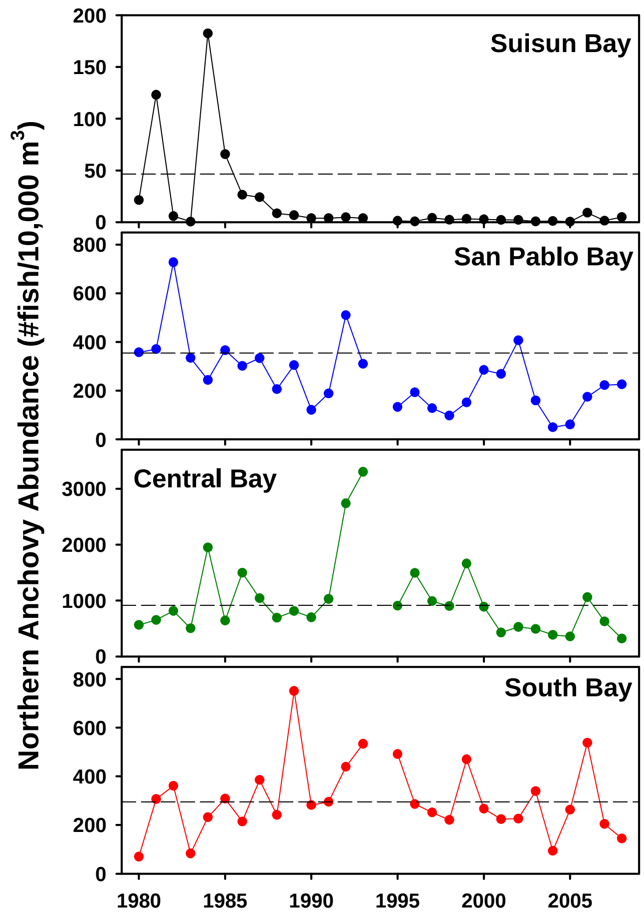


Figure 5. Changes in the Demersal Fish Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

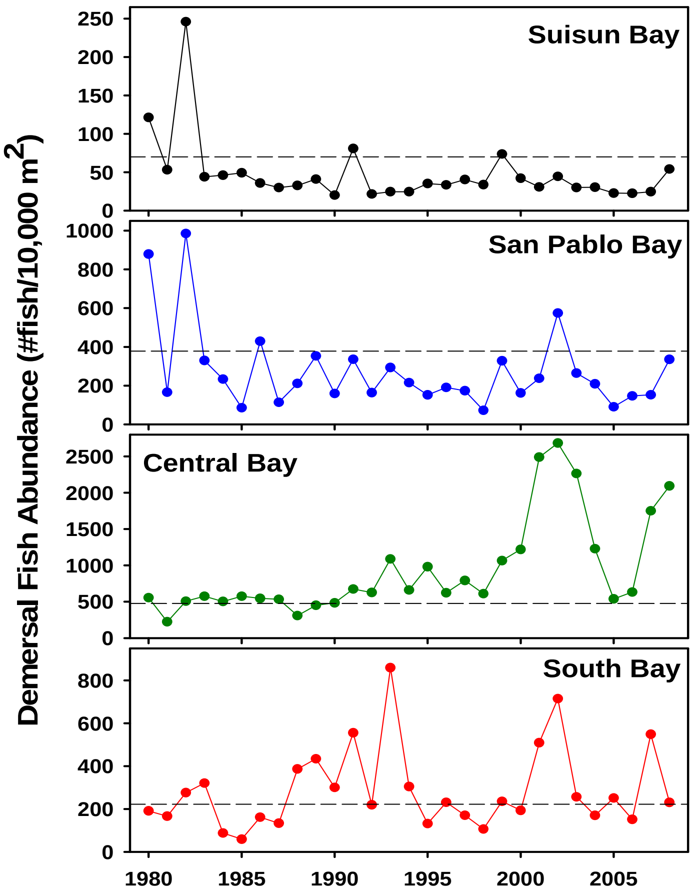


Figure 6. Changes in the Sensitive Fish Species Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

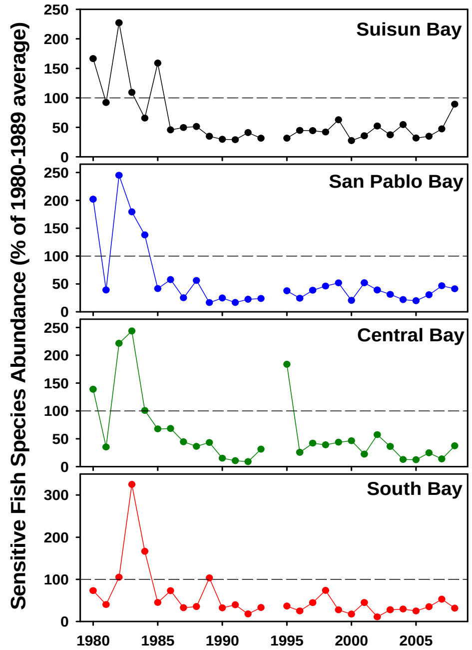


Table 4. San Francisco Estuary-dependent fish species collected in the CDFG Bay Study Midwater Trawl and Otter Trawl surveys.

Estuary-dependent fish species (common names)	
Estuary resident species Species with resident populations in the estuary and/or estuary-obligate species that use the estuary as nursery habitat	**Seasonal species** Species regularly use the estuary for part of their life cycle but also have substantial connected populations outside the estuary
Arrow goby	Barred surfperch
Bat ray	Black perch
Bay goby	Bonehead sculpin
Bay pipefish	California halibut
Brown rockfish	California tonguefish
Brown smoothhound	Diamond turbot
Cheekspot goby	English sole
Delta smelt	Northern anchovy
Dwarf surfperch	Pacific sandab
Jack smelt	Pacific tomcod
Leopard shark	Plainfin midshipman
Longfin smelt	Sand sole
Pacific herring	Speckled sanddab
Pacific staghorn sculpin	Spiny dogfish
Pile perch	Splittail
Shiner perch	Starry flounder
Threespine stickleback	Surfsmelt
Topsmelt,	Walleye surfperch
Tule perch	
White croaker	
White surfperch	

Table 5. Quantitative reference conditions and associated interpretations for the results of the diversity indicators. The primary reference condition, which corresponds to "good" conditions, is in bold.

Diversity Indicators		
Native Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>60%	"Excellent"	4
>50% (~1980-1989 average)	**"Good"**	**3**
>40%	"Fair"	2
>30%	"Poor"	1
≤30%	"Very Poor"	0
Estuary-dependent Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>85%	"Excellent"	4
>70% (~1980-1989 average)	**"Good"**	**3**
>55%	"Fair"	2
>40%	"Poor"	1
≤40%	"Very Poor"	0

Figure 7. Changes in the Native Fish Species Diversity indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

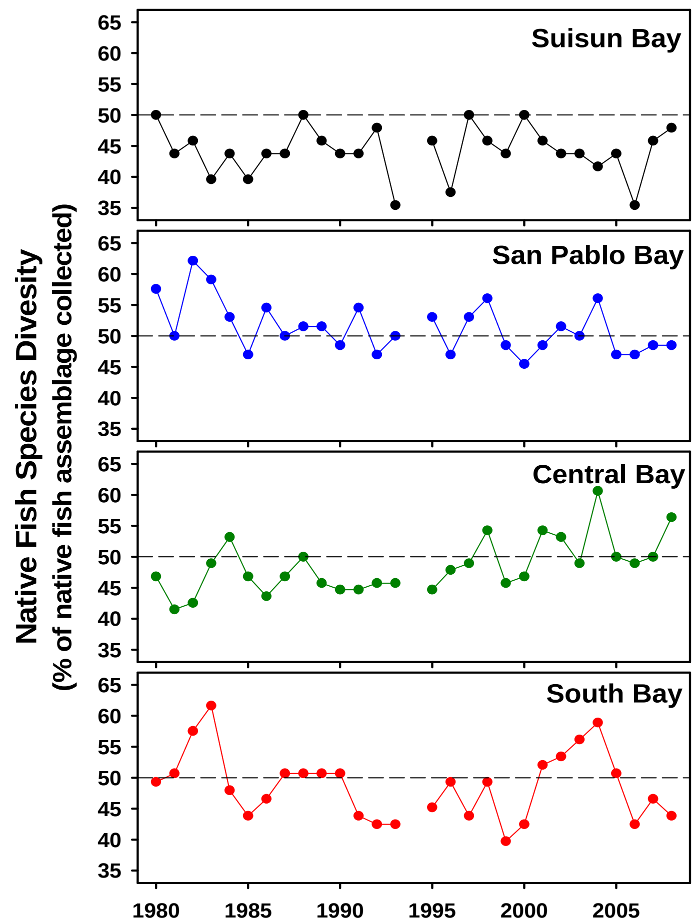


Figure 8. Changes in the Estuary-dependent Fish Species Diversity indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1989-1989 average).

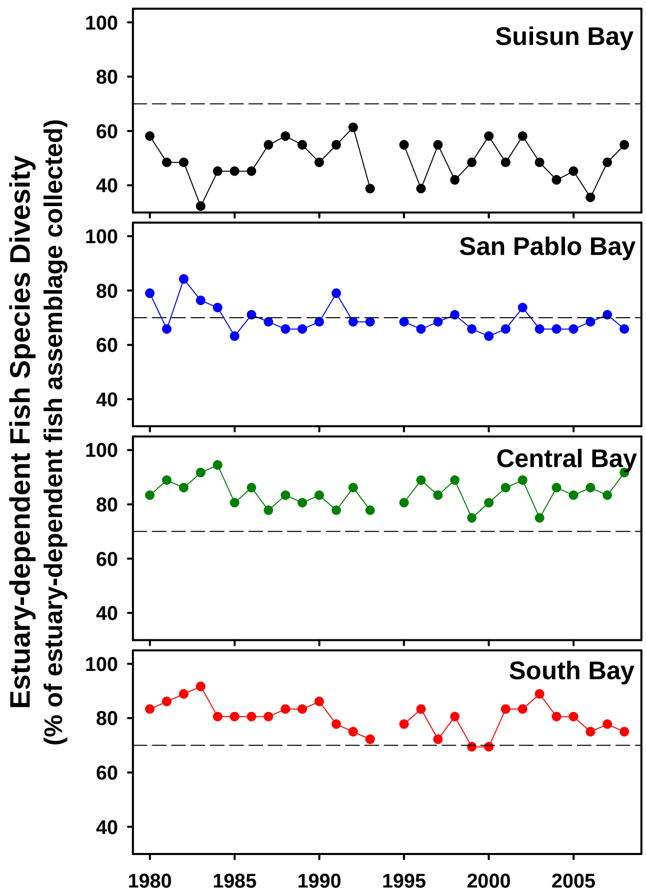


Table 6. Quantitative reference conditions and associated interpretations for the results of the species composition indicators. The primary reference condition, which corresponds to “good” conditions, is in bold.

Species Composition Indicators (Percent Native Species, Percent Native Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>95%	“Excellent”	4
>85% (~1980-1989 average)	**“Good”**	**3**
>70%	“Fair”	2
>50%	“Poor”	1
≤50%	“Very Poor”	0

Figure 9. Changes in the Percent Native Species indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

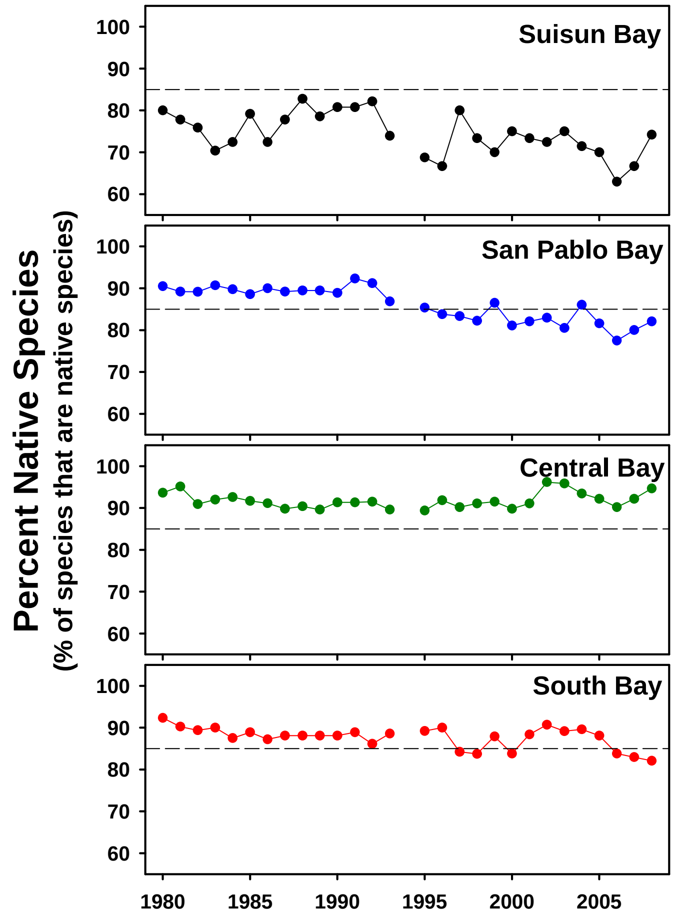


Figure 10. Changes in the Percent Native Fish indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

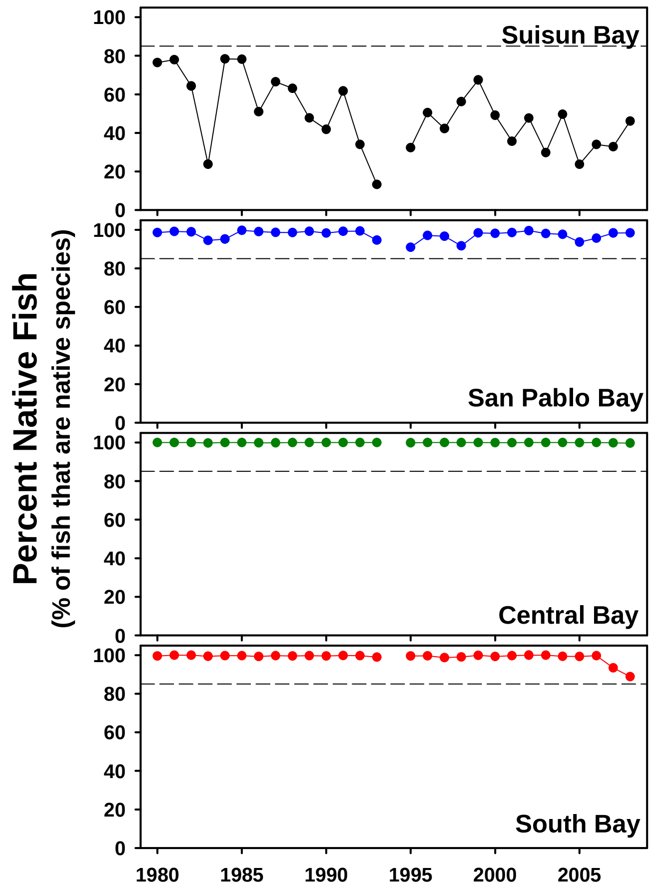


Table 7. Quantitative reference conditions and associated interpretations for the results of the distribution indicators. The primary reference condition, which corresponds to "good" conditions, is in bold.

Distribution Indicators (Pelagic Fish, Demersal Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
100%	"Excellent"	4
>80% (~1980-1989 average)	**"Good"**	**3**
>60%	"Fair"	2
>40%	"Poor"	1
≤40%	"Very Poor"	0

Figure 11. Changes in the Pelagic Fish Distribution indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

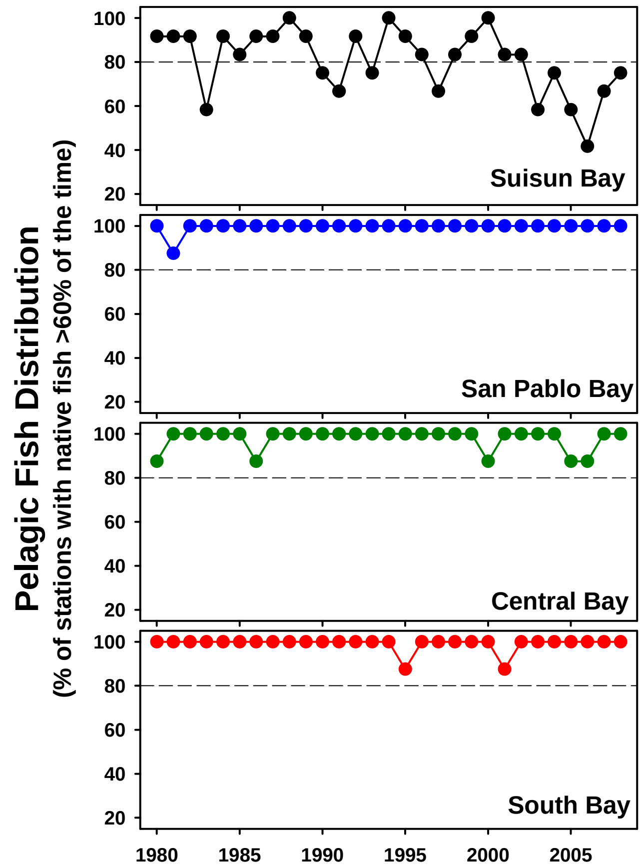


Figure 12. Changes in the Demersal Fish Distribution indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

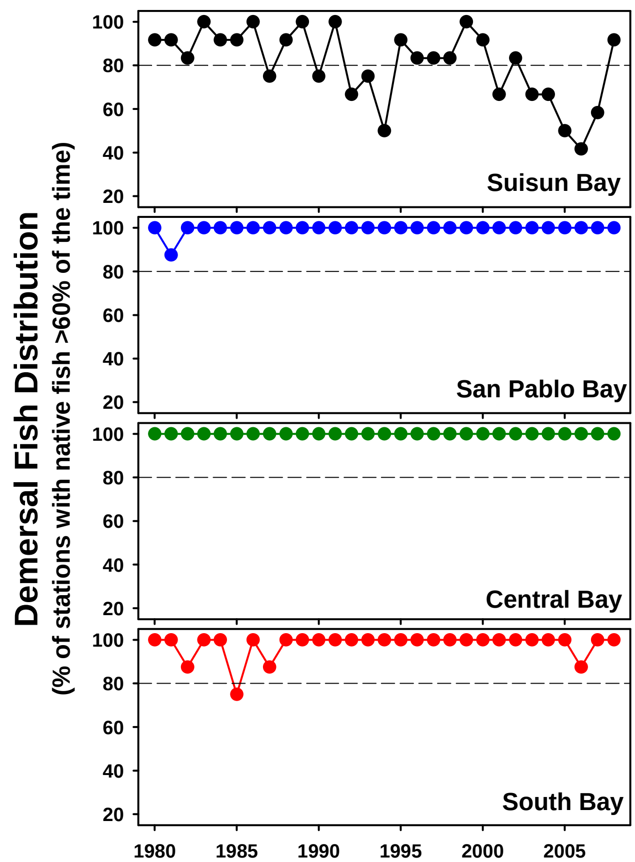


Figure 13. Changes in the Fish Index in each of four sub-regions of the San Francisco Estuary from 1980-2008. Black lines and symbols show the annual Index values, solid red line shows the linear regression for the 1980-2008 period. Horizontal dashed lines shows the reference conditions and associated interpretations.

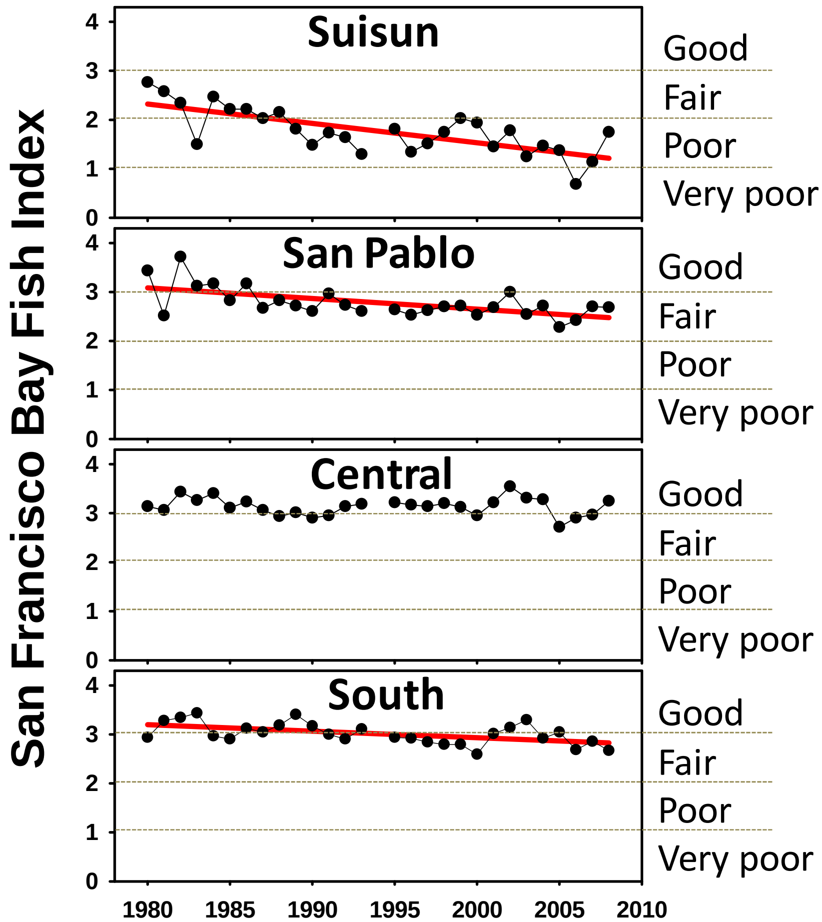


Table 8. Summary of results, relative to the CCMP goals to "recover" and "reverse declines" of estuarine fishes, of the seven fish indicators for each of the four sub-regions of the San Francisco Estuary.

Indicator or Index	Sub-region	CCMP Goal Met (yes or no)	Trend long-term (29 yrs)	Trend short-term (last 5 yrs)
Pelagic Fish Abundance	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Stable
	Central	No	Stable	Stable
	South	No	Decline	Stable
Northern Anchovy Abundance	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Increase
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Demersal Fish Abundance	Suisun	Yes	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Increase	Stable
	South	Yes	Stable	Stable
Sensitive Fish Species Abundance	Suisun	Yes	Decline	Stable
	San Pablo	Yes	Decline	Stable
	Central	No	Decline	Stable
	South	No	Decline	Stable
Native Fish Species Diversity	Suisun	Yes	Stable	Stable
	San Pablo	Yes	Decline	Stable
	Central	Yes	Increase	Stable
	South	Yes	Stable	Stable
Estuary-dependent Fish Species Diversity	Suisun	Yes	Stable	Stable
	San Pablo	Yes	Decline	Stable
	Central	Yes	Stable	Stable
	South	No	Decline	Stable
Percent Native Species	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Stable
	Central	Yes	Stable	Stable
	South	Yes	Decline	Decline
Percent Native Fish	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Decline*	Decline
Pelagic Fish Distribution	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Demersal Fish Distribution	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Fish Index	**Suisun**	**No**	**Decline**	**Stable**
	San Pablo	**No**	**Decline**	**Stable**
	Central	**Yes**	**Stable**	**Stable**
	South	**No**	**Decline**	**Stable**

State of San Francisco Bay 2011
Appendix G

LIVING RESOURCES – Birds
Technical Appendix

Prepared by Nadav Nur and John Kelly, PRBO Conservation Science

Overview
“Living Resources-Birds” are assessed using five distinct indicators. We made use of a five-fold approach in order to provide insights into the ecological functioning and biological status of the estuary in order to best reflect: (a) the taxonomic diversity of birds that rely on estuarine habitat (i.e., rails, songbirds, wading birds, and waterfowl are all included), (b) the different stages of the annual cycle that are important to birds (in particular, breeding and over-wintering), and (c) the ecological diversity of birds relying on estuarine habitat (including species that feed on fish, invertebrates, and/or plants). The first, third, and fifth indicators provide indices of population size or density for three important and indicative groups of birds: tidal marsh-dependent birds, herons and egrets, and ducks (both diving and dabbling ducks).

1. Tidal Marsh Bird Populations
Nadav Nur

Background and Rationale:
San Francisco Estuary tidal marsh habitat has been dramatically altered in the past one hundred and sixty years. Approximately 85% of the original tidal marsh habitat in the region has been lost due to creation of salt ponds, conversion to agricultural and industrial/urban use, and water diversion and management (Marshall & Dedrick 1994, Goals Project 1999). The reduction in area, fragmentation of remaining habitat, degradation in habitat quality, and spread of invasive species have all contributed to reductions in the population size and viability of tidal marsh obligate species (Takekawa et al. 2006). For these reasons, many of the species that depend on tidal marsh habitat are currently listed as Federally- or State- threatened or endangered, in particular Clapper Rail and Black Rail, or are of conservation concern (e.g., California Species of Special Concern, Shuford & Gardali 2008). It is for these reasons that the first-listed “Aquatic Resources Goal” of the CCMP is

- “Stem and reverse the decline in the health and abundance of estuarine biota (indigenous and desirable non-indigenous), restoring healthy natural reproduction.”

The indicator presented here, **Tidal Marsh Bird Population Indicator**, assesses abundance of target species of concern and provides information on health of these populations by determining changes in abundance metrics. This indicator also provides information regarding progress towards the second and third stated goals for Aquatic Resources, i.e.,

- “Restore healthy estuarine habitat to the Bay-Delta” and
- “Ensure the survival and recovery of listed (and candidate) threatened and endangered species, as well as other species in decline.”

This indicator does not assess healthy estuarine **habitat** directly, but instead allows for inference to be made, based on bird populations that depend on healthy estuarine habitat. The indicator also allows assessment of progress made with respect to the recovery of threatened and endangered species, as well as additional species that are known or presumed to have reduced abundance compared with earlier time periods, especially the period before 1800.

The proposed indicator draws primarily on PRBO’s tidal marsh bird monitoring project begun in 1996 (Nur et al. 1997, Spautz et al. 2006). This program has been studying tidal marsh-dependent species throughout the San Francisco Estuary, utilizing an extensive array of breeding-season point count surveys (about 10 point count locations per mash), conducted twice per breeding season, between 1996 and 2010. Until 2007, surveys were conducted at about 20 to 40 marshes per year; from 2008 to the present, surveys have been conducted at about 8 marshes per year. The indicator is calculated for three identified regions: Suisun Bay, San Pablo Bay, and San Francisco Bay. The San Francisco Bay region includes both Central and South San Francisco Bay, combined.

Three species are included in this indicator. Each is year-round resident (or primarily resident) and **is dependent on, or strongly associated with, tidal marsh habitat** (Goals Project 2000). One species is the **Black Rail** (family Rallidae); specifically, the California Black Rail subspecies (*Laterallus jamaicensis coturniculus*). The other two species are songbirds, **Song Sparrow** (*Melospiza melodia*) and **Common Yellowthroat** (*Geothlypis trichas*, a North American warbler). The proposed indicator (and data available) are specific to the tidal marsh-dependent subspecies of the Song Sparrow and Common Yellowthroat (Marshall and Dedrick 1994, Nur et al. 1997).

Data Sources and Species:
For the three species, data are from PRBO tidal marsh bird project (www.prbo.org/cms/135; Nur et al. 1997, Spautz et al. 2006). Survey results are available for 1996 to 2008, and presented here. Information from 2009 and 2010 will soon be available for inclusion in a subsequent iteration of the indicator, and field surveys were conducted in 2011 as well.

One important tidal marsh species is not included here: the California Clapper Rail (*Rallus longirostris obsoletus*). At the outset of this exercise, extensive data was available only for the period 2005-2008. Information prior to 2005 is less comprehensive (Albertson and Evens 2000). Data from 2009, 2010, and 2011 are being analyzed by a consortium of agencies and organization, including PRBO biologists and other investigators; these data were not available in time for the indicator analysis. As soon as data from 2009-2011 are available for analysis to add to the 2005-2008 data set (see Liu et al. 2009), we advocate that this indicator, Tidal Marsh Bird Populations Indicator, include the California Clapper Rail as well.

Methods and Calculations:
Abundance data were collected regarding Black Rails, Song Sparrows, and Common Yellowthroats, using point count surveys conducted at multiple marshes per region per year (usually 5 to 8 marshes per region per year) during the breeding season (March to end of May). Generally, 6 to 10 point count stations were established per marsh survey (Liu et al. 2007). For each species and each region, we estimated mean number of individuals detected per hectare of surveyed marsh per survey (usually, two surveys per year per marsh). These surveys did not use tape playback. Statistical analysis was conducted on densities per marsh per year, averaged over the number of survey visits. “Density” for this indicator refers to the number of birds detected per hectare surveyed, and is more properly termed “apparent density” since we did not correct for detectability (but see Nur et al. 1997; Thomas et al. 2010). All analyses were conducted on log-transformed values (with a constant added so that all densities were > 0; Nur et al. 1999).

Between 1996 and 2008, many marsh sites were surveyed, but the same sites were not surveyed in each year. To control for site-to-site differences in abundance, "site" was included as a categorical variable in the analyses. The statistical analysis was carried out separately for each region (SF Bay, San Pablo Bay, Suisun Bay), and for each species. Finally, a multiple-species metric was calculated based on the single species densities, while controlling for site differences. The multiple-species metric was calculated on log-densities, controlling for differences in apparent density among the three species. Note: (1) The statistical control for site effects was carried out separately for each species. (2) Black Rail density was not estimated for SF Bay region, due to lack of detections of individuals in that region (see Evens and Nur 2002).

In addition to presenting year-by-year results for 1996 to 2008, we calculated trends for two time periods: 1996 to 2008 (i.e., the most recent 13 years of survey data), and 2004 to 2008 (i.e., the most recent 5 years). We also compare the most recent three year-mean values (for 2006-2008) to the benchmark 5-year values (for 1996-2000). Trends were calculated for each of the three species and for all species combined in a multi-species statistical model that fit a single slope, common to the three species, but allowed species log(density) to differ among the three species. See Pyle et al. (1994) for similar example. Because these analyses were conducted on log-densities, the coefficients obtained (i.e., slopes) represent the constant proportional increase or decrease for each species or for the three combined species (Nur et al. 1999).

Goals, Targets, and Reference Conditions:
There are no agreed upon, explicitly stated goals, targets or reference conditions for any of the three focal species (Black Rail, Song Sparrow, and Common Yellowthroat). Because of loss of habitat, population size has been reduced from historical levels (e.g., since c. 1800). Therefore, one means of assessment is to evaluate trends since 1996 (the earliest year for which annual survey data are available for Black Rail, Song Sparrow, and Common Yellowthroat). To assist in evaluation of the "longer-term" trends (in this case, 1996 to 2008), we also consider more recent "short-term" trends (in this case from 2004 to 2008). Finally, we compare mean densities observed in 1996 to 2000 (best available 5-year benchmark) to the period 2006 to 2008 (most recent 3 years of data).

The goal (target) is for trends to be positive (indicating recovery of tidal marsh species), or at least to be non-negative. For all species considered, evaluations are carried out for each region within the Estuary.

Results:
For this indicator, results differed strikingly from one region of the SF Estuary to another. In addition, each species displayed a distinctive pattern.

For **Black Rail**, the trend in both San Pablo Bay and Suisun Bay has been positive (Figure T1 A, B). In San Pablo Bay the positive trend is exemplified in the longer-term (since 1996) and shorter-term (Table T1). In Suisun Bay, the positive trend is only evident in the last 5 years; in fact, the highest density values for Black Rails are all in the most recent 5 years of surveys (2004-2008; Table T1). The overall increase in density of Black Rails for San Pablo and Suisun is confirmed when one compares the most recent 3-year period with the earlier 5-year benchmark period (Table T2).

For **Common Yellowthroat**, there has been little increase in San Francisco Bay over the 13-year period, except that the most recent 10 years have higher densities than the first three years (Figure T1C). Nevertheless, the overall trends for the longer-time period and the shorter-time period are non-significant, nor does the most recent three-year period differ significantly from the five-year benchmark period (Table T1, T2). In contrast, in both San Pablo Bay and Suisun Bay, there have been significant increases over the long-term, but this trend has abated in recent years in San Pablo Bay (Figure T1 D, E). In Suisun Bay, it is less clear whether the increasing trend is evident, but the overall pattern is of higher densities in recent years compared to earlier years. Note that the density index for Suisun Bay Common Yellowthroats has remained about 10-fold greater than the comparable density index for San Francisco Bay or San Pablo Bay Common Yellowthroats (Figure T1 C, D, E). This consistent regional difference is likely due to habitat affinities: Common Yellowthroats prefer brackish marsh to saline marsh (Spautz et al. 2006, Stralberg et al. 2010).

For **Song Sparrows**, only the San Francisco Bay region shows an increase, and even then the increase has reversed, i.e., this region demonstrates a recent decline (Figure T1 F, Table T1). In contrast to the overall-increase for the San Francisco Bay region, Suisun and San Pablo Bay regions show overall decreases (Figure T1 G, H; Table T1). Moreover, all three regions demonstrate recent, short-term declines. As a result of these divergent trends, San Francisco Bay Song Sparrows no longer demonstrate the lowest density of the three regions, instead, Suisun Song Sparrows evidence the lowest density, and San Francisco Bay Song Sparrows the middle level of density. For this species, there are no significant differences between the 3 most recent years and the 5-year benchmark period for any of the three regions (Table T2).

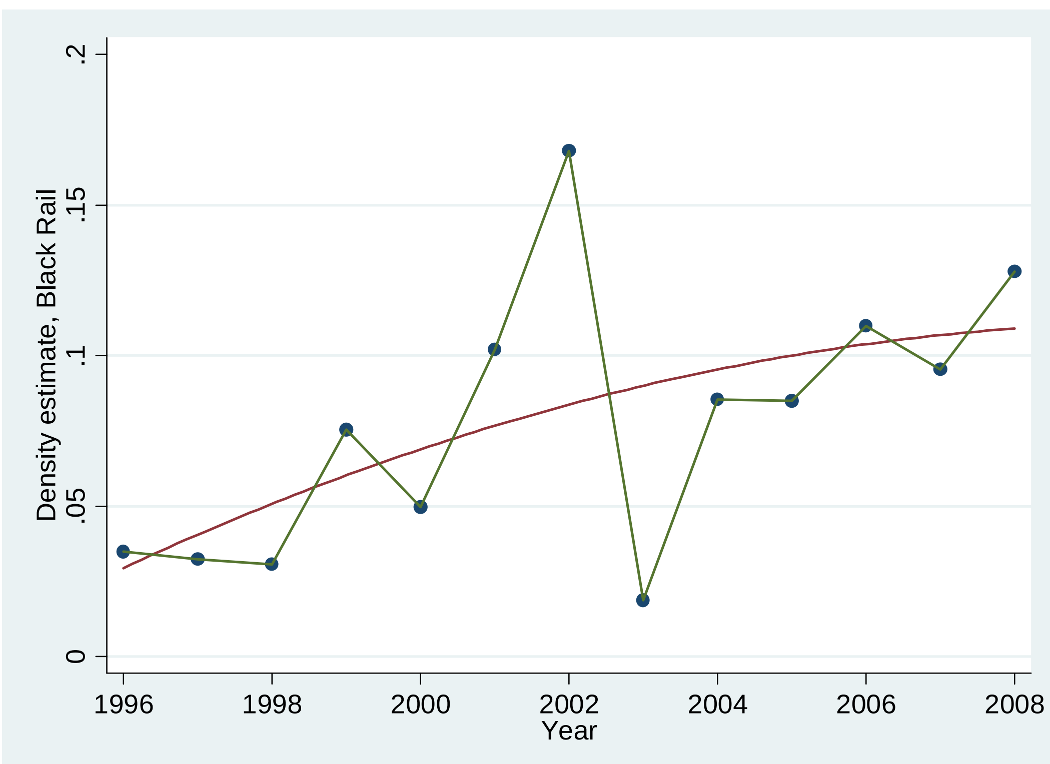
The **combined species** analysis demonstrates a different pattern for each region, though the overall-result is a net increase. In San Francisco Bay, the increase is evident earlier in the period but more recently demonstrates a decrease (Figure T1 I). In San Pablo Bay, the overall increase in density is evident during the entire period (Figure T1 J). In Suisun, an initial decrease has been followed by a more recent increase in density (Figure T1 K).

We conclude that the tidal marsh bird population indicator reveals a mixed picture: The combined species index shows overall increases in marsh bird population density since 1996, which **indicates some success in meeting the CCMP's first stated Aquatic Resources goal**: "Stem and reverse the decline in the health and abundance of estuarine biota." In San Pablo and San Francisco Bay regions, the increase for the combined species index is evident comparing 1999-2008 with 1996-1998, but recent years have not demonstrated further increases. For Suisun, the increase is evident comparing 2004-2008 to earlier years. Black Rails, a State-threatened species, clearly show a population-level increase which suggest that progress is also being made with regard to the third stated Aquatic Resources goal: "Ensure the survival and recovery of listed (and candidate) threatened and endangered species...." Song Sparrows reveal the other side of the story: this species demonstrates declines in density in San Pablo and Suisun Bays. Song Sparrows in San Francisco Bay show a recent decline (2002 to 2008) which partly counteracts the early improvements seen, from 1996 to 2002. The declines observed for this species are a cause for concern.

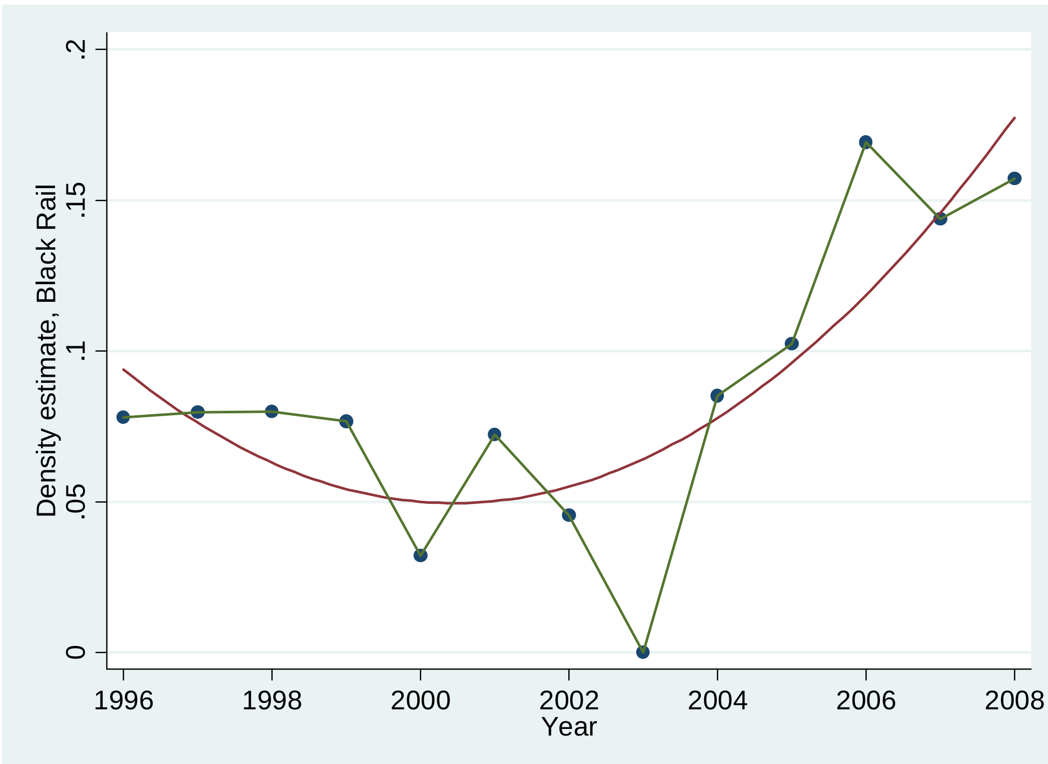
The overall declines in the Song Sparrow population index are consistent with the low levels of reproductive success that are apparent (see Living Resources - Birds Indicator 2. *Marsh bird reproductive success*). The increase in density seen since 1996 reflects an improvement in habitat quality, at minimum increased habitat quality in restored tidal marshes. It is less clear whether mature marshes (those over 100 years of age) are showing increases in habitat quality.

Figure T1. Population Trends for Three Tidal Marsh Species (Black Rail, Common Yellowthroat, and Song Sparrow) and Combined Trend for all 3 species. Shown is density index (birds detected per hectare per survey) by SF Estuary region, controlling for site-to-site differences in density within a region. Note: There are no breeding Black Rails in San Francisco Bay. Combined species trend depicts geometric mean across the three species (see text). Each species-region graph shows the best linear fit (Figures T1-E and T1-G) or quadratic fit (Figures T1-A to T1-D, T1-F, and T1-H to T1-K) as appropriate; choice of fit (linear vs. quadratic) determined by maximization of adjusted R^2 (Nur et al. 1999).

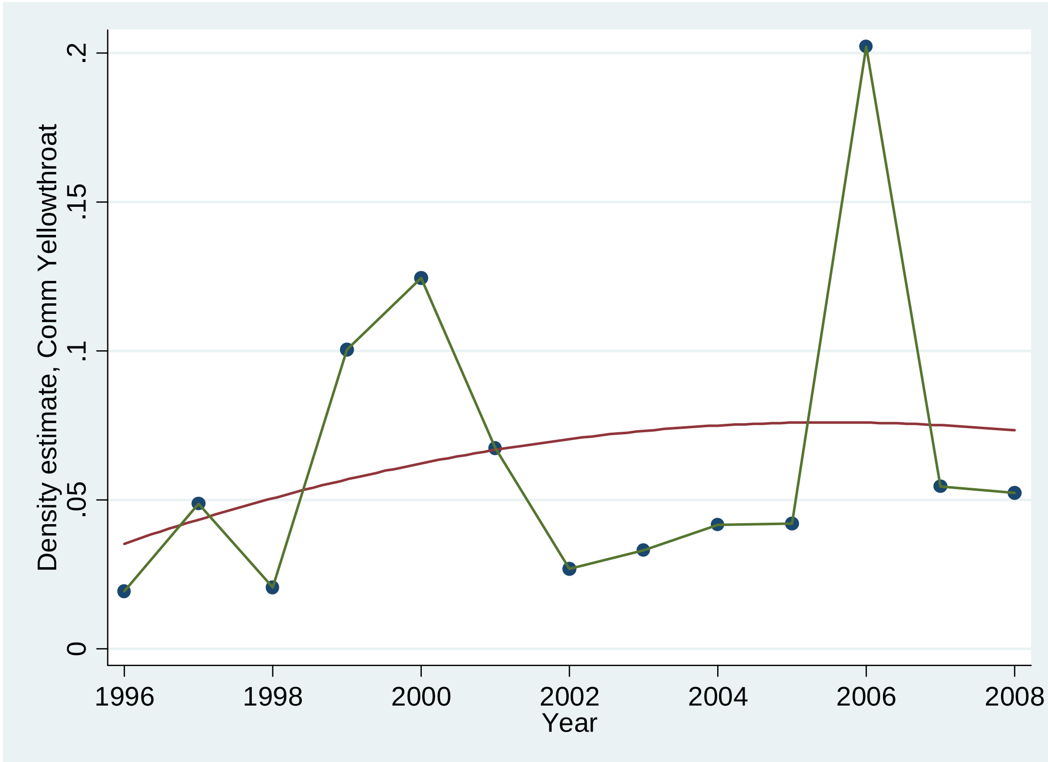
A) Black Rail, San Pablo Bay



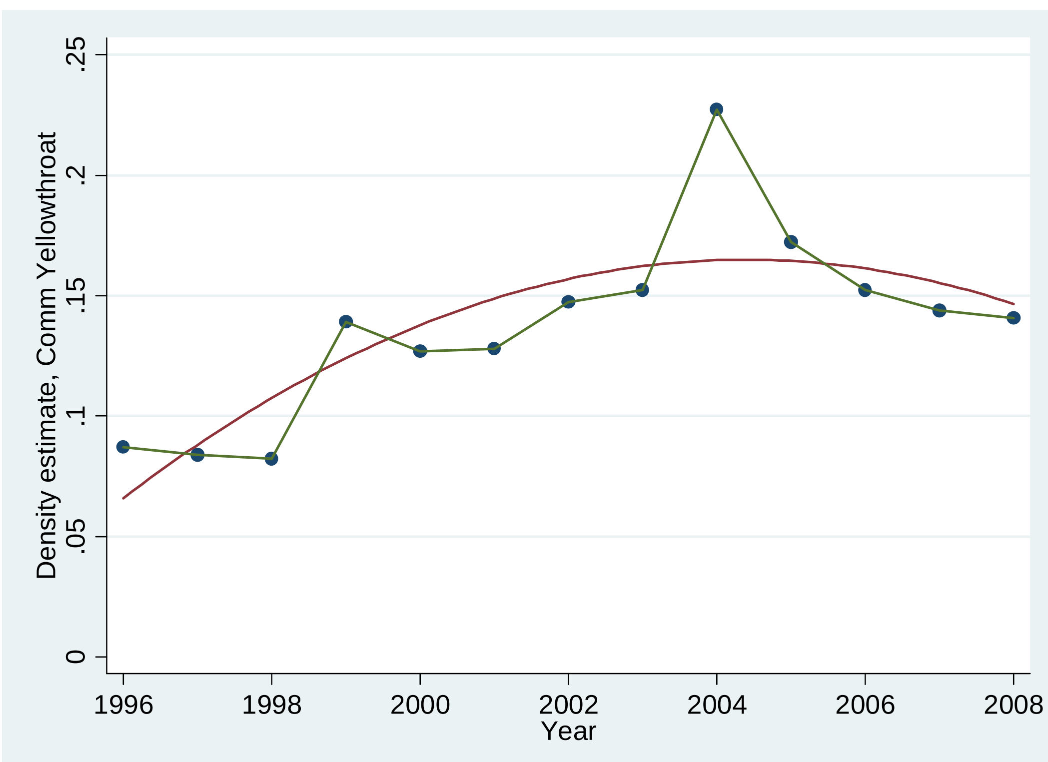
B) Black Rail, Suisun Bay



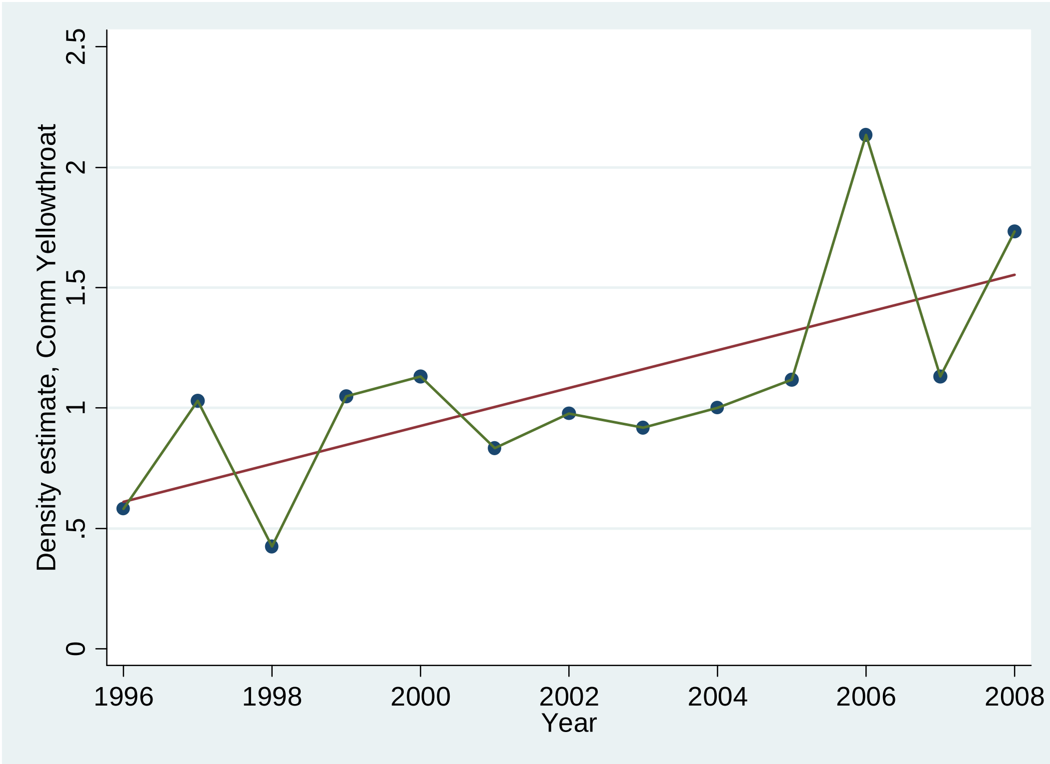
C) Common Yellowthroat, San Francisco Bay



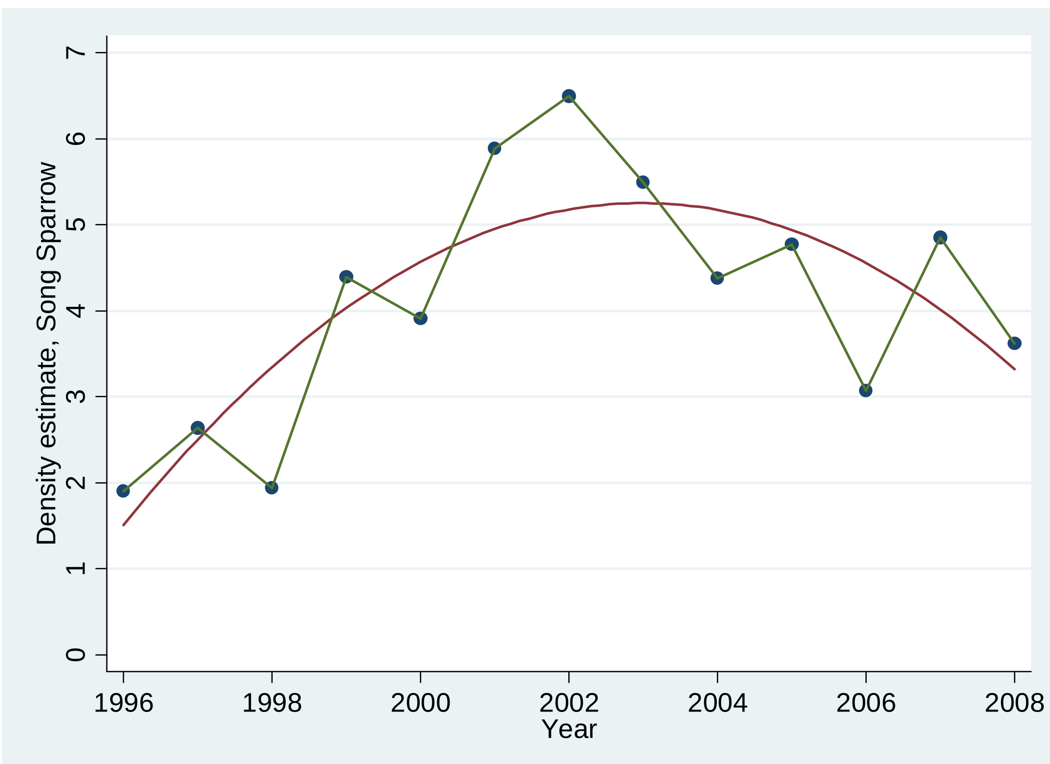
D) Common Yellowthroat, San Pablo Bay



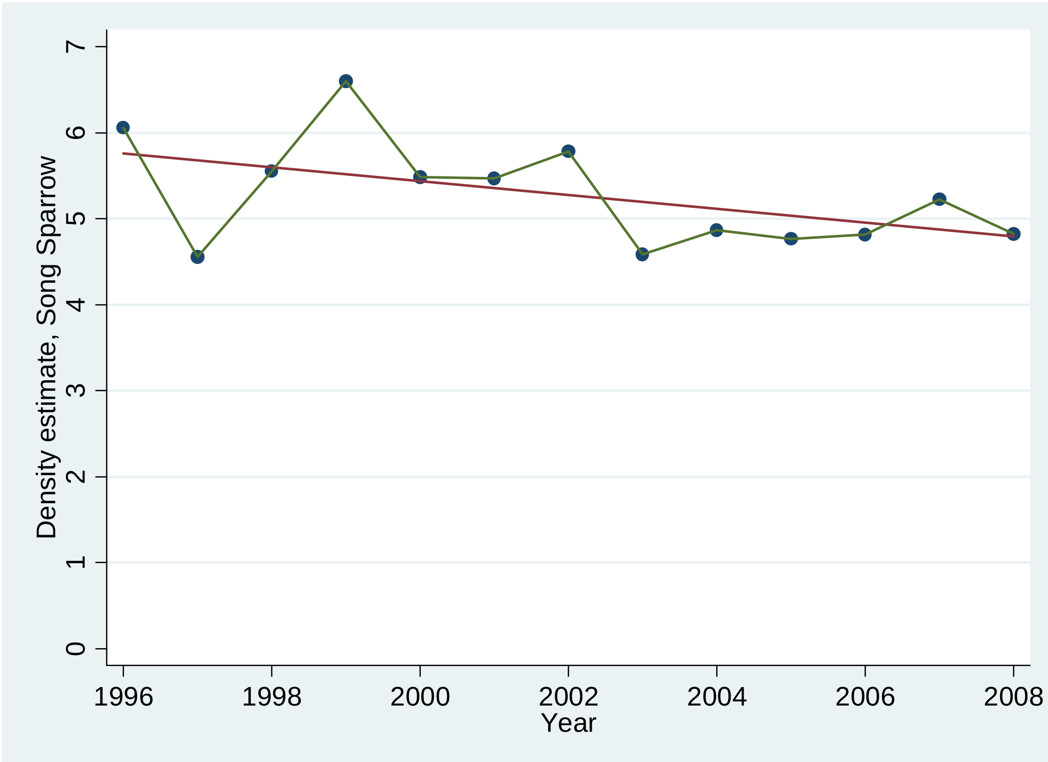
E) Common Yellowthroat, Suisun Bay



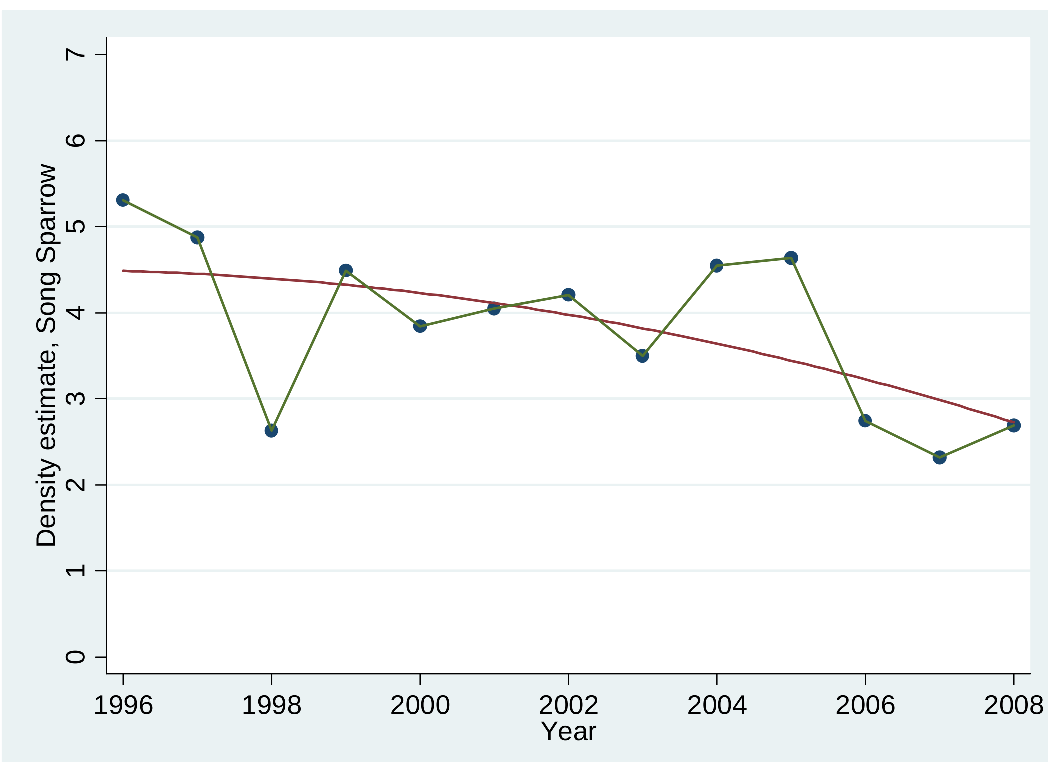
F) Song Sparrow, San Francisco Bay



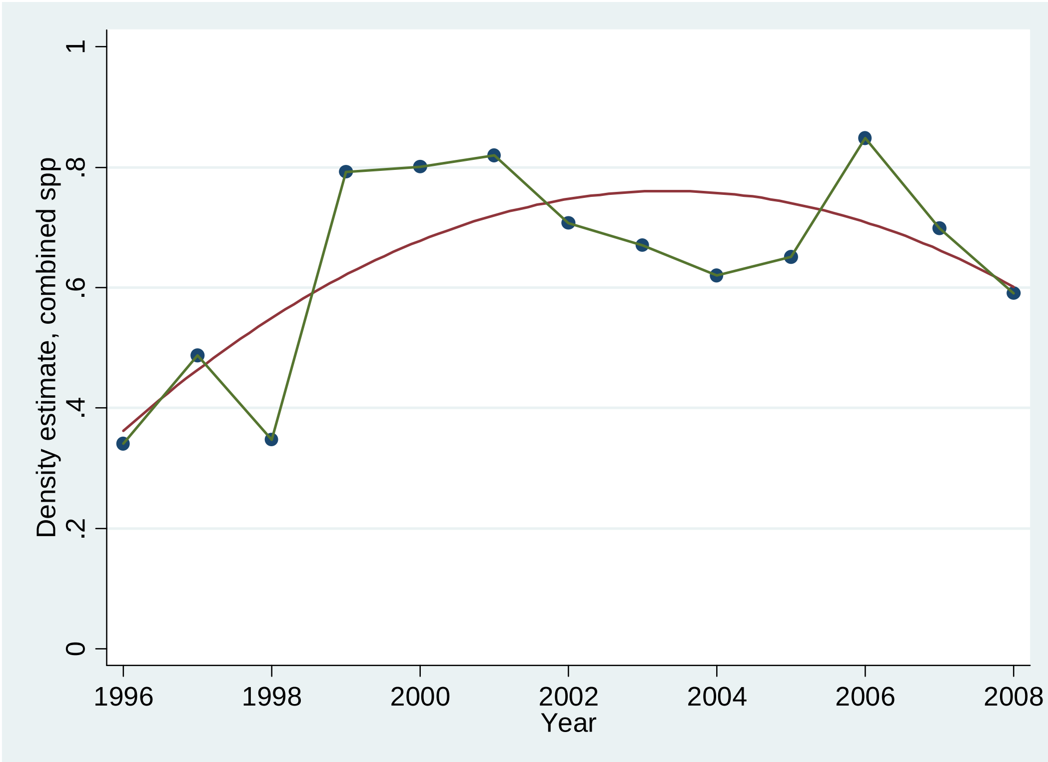
G) Song Sparrow, San Pablo Bay



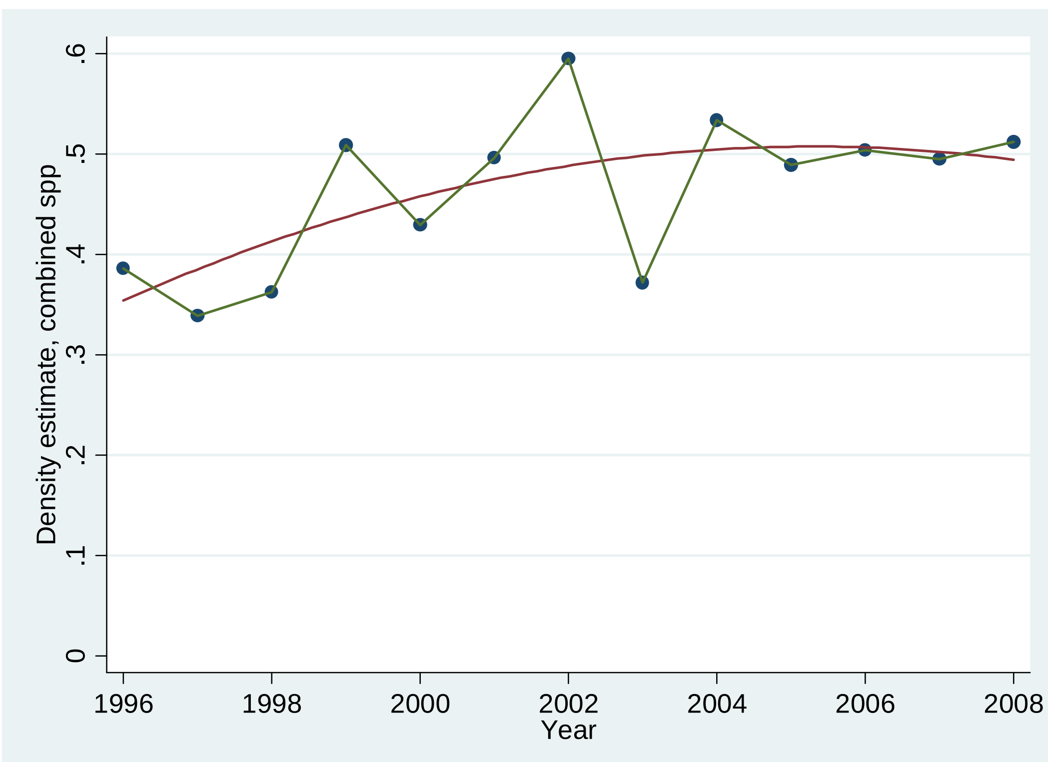
H) Song Sparrow, Suisun Bay



I) Combined species, San Francisco Bay



J) Combined species, San Pablo Bay



K) Cominbed species, Suisun Bay

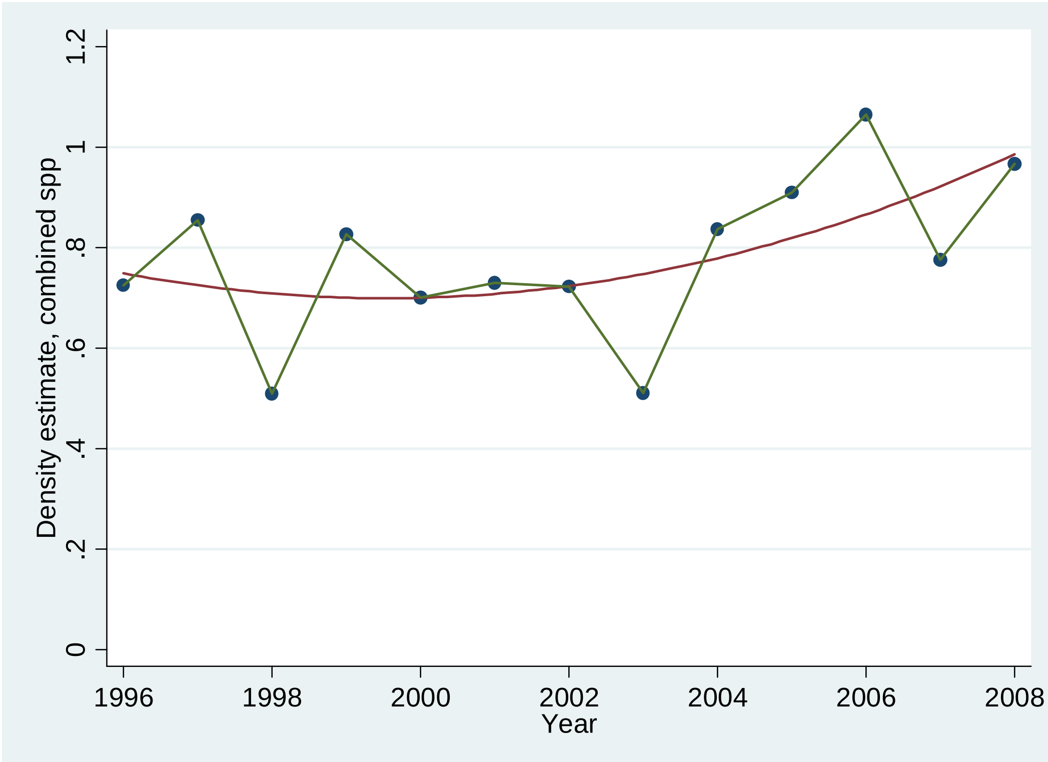


Table T1.

Long-term (1996 to 2008) and Short-term (2004 to 2008) trends for tidal marsh bird species

Shown are estimated annual percent changes per year in density index. Highlighting indicates significant differences (P < 0.05; bright yellow) or marginally significant (0.05 ≤ P < 0.10; pale yellow)

	San Francisco B		**San Pablo B**		**Suisun & W. Delta**	
Song Sparrow	Ann Pct	P-val	Ann Pct	P-val	Ann Pct	P-val
Long-term	5.77%	P = 0.008	-1.54%	P = 0.16	-2.63%	P > 0.2
Short-term	-0.67%	P > 0.9	-2.81%	P > 0.3	-14.7%	P = 0.19
Common Yellowthroat						
Long-term	-0.45%	P > 0.8	4.33%	P = 0.019	7.10%	P = 0.019
Short-term	1.37%	P > 0.8	-10.3%	P = 0.083	14.7%	P > 0.3
Black Rail						
Long-term	ND		4.08%	P = 0.034	2.18%	P > 0.4
Short-term	ND		5.19%	P > 0.5	7.37%	P > 0.4
Combined species						
Long-term	2.61%	P = 0.14	2.26%	P = 0.018	2.14%	P = 0.15
Short-term	0.34%	P > 0.9	-2.83%	P > 0.4	1.65%	P > 0.8

Table T2.

Comparison of 3-year Current (2006-2008) vs. 5-year Benchmark (1996 to 2000)
Shown are estimated percent differences in density index for two time periods. Highlighting indicates significant ($P < 0.05$) differences (bright yellow) or marginally significant ($0.05 \leq P < 0.10$)

	San Francisco Bay		San Pablo Bay		Suisun Bay	
	Percent	P-val	Percent	P-val	Percent	P-val
Song Sparrow						
Comparison	2.70%	P > 0.9	-11.7%	P > 0.2	-33.8%	P = 0.18
Common Yellowthroat						
Comparison	20.0%	P > 0.4	38.7%	P = 0.073	74.3%	P = 0.15
Black Rail						
Comparison	ND		49.5%	P = 0.034	83.0%	P = 0.041
Combined species						
Comparison	11.0%	P > 0.5	22.3%	P = 0.033	28.3%	P = 0.19

2. Marsh Bird Reproductive Success
Nadav Nur

Background and Rationale:
San Francisco Estuary tidal marsh habitat has been dramatically altered in the past one hundred and sixty years. Approximately 85% of the original tidal marsh habitat in the region has been lost due to creation of salt ponds, conversion to agricultural and industrial/urban use, and water diversion and management (Marshall & Dedrick 1994). The reduction in area, fragmentation of remaining habitat, degradation in habitat quality, and spread of invasive species have all contributed to reductions in the population size and viability of tidal marsh obligate species. Future threats such as climate change will also alter the area and distribution of marshes and may lead to increased risk of mortality due to flooding, as a result of sea level rise and increased frequency of storm surges (Takekawa et al. 2006). For these reasons, many of the species that depend on tidal marsh habitat are currently listed as Federally- or State- threatened or endangered, in particular Clapper Rail and Black Rail, or are of conservation concern (e.g., California Species of Special Concern, Shuford & Gardali 2008). It is for these reasons that the first-listed "Aquatic Resources Goal" of the CCMP is

- "Stem and reverse the decline in the health and abundance of estuarine biota (indigenous and desirable non-indigenous), restoring healthy natural reproduction."

The indicator presented here, **Marsh Bird Reproductive Success**, provides for informative assessment of progress in meeting this goal, as well as providing information regarding progress towards the second and third stated goals for Aquatic Resources, i.e.,

- "Restore healthy estuarine habitat to the Bay-Delta" and
- "Ensure the survival and recovery of listed (and candidate) threatened and endangered species, as well as other species in decline."

Successful reproduction involves several components, for which we focus on one, **nest survival**. Other components of reproductive success include number of young reared per successful breeding attempt and number of breeding attempts per breeding pair (Chase et al. 2005). Nest survival in avian species is a parameter that is monitored and evaluated on the national and international levels (Greenberg et al. 2006, Jones and Geupel 2007).

Nest survival refers to the probability that a nesting attempt survives to fledge one or more young. Nest survival of tidal marsh **Song Sparrows** reflects two principal mortality pressures: predation on nests and flooding of nests (Greenberg et al. 2006, Nordby et al. 2008). For tidal marsh Song Sparrows, this indicator reflects primarily nest-predation (either predation on eggs or nestlings). Principal predators are birds (especially corvids), mammals (especially raccoons), and snakes. Secondarily, the indicator reflects inundation, and thus flooding due to high tides. Flooding is the second-leading cause of nest failure for tidal marsh Song Sparrows (Greenberg et al. 2006, Nordby et al. 2009).

Between 1996 and 2006, PRBO conducted systematic nest monitoring at up to five sites per year for two regions: San Pablo Bay and Suisun Bay. In addition, there is partial information from San Francisco Bay for 2002 and 2003 (Nordby et al. 2009).

Data Source:

PRBO biologists conducted nest-monitoring in tidal marsh habitat for Song Sparrows at three to five sites in each year, distributed between San Pablo and Suisun Bays, between 1996 and 2006. In 9 out of 11 years, there were at least two sites monitored per bay per year.

Methods and Calculations:

Nest monitoring was conducted following methods outlined in Martin and Geupel (1993) and Liu et al. (2007). At each site, two to four study plots were established. For each breeding pair, nests were intensively searched for and then monitored, from nest discovery to the fledging or failure of a nesting attempt. Nests were usually visited every 2-4 days in order to accurately estimate dates of nest failure, dates of egg laying, hatching of eggs, and fledging of young. The ultimate outcome of each nest (success or failure) was determined based on nest condition and behavior of the breeding pair (Martin and Geupel 1993). For each breeding season, we calculated daily nest survival of a specific site using the Mayfield method (Mayfield 1975). We then converted daily nest survival (calculated separately for each stage of the nesting cycle) into overall survival, from laying of the first egg until fledging following Nur et al. (1999).

Not every site was monitored in every year. Therefore, in order to adjust for site-specific differences in nest survival, which may confound differences among years, we included "site" as a categorical variable to be controlled for, when analyzing sites and years. This "standardization" of nest survival was carried out separately for each region, i.e., for San Pablo Bay sites and Suisun Bay sites. The statistical analysis was similar to that presented for the Tidal Marsh Bird Population Indicator (above). Note: no PRBO monitoring was carried out in Central or South San Francisco Bay (but see Nordby et al. 2009 for two years of results for that region).

Goals, Targets, and Reference Conditions:

This indicator focuses on a single species, the Song Sparrow; specifically, the subspecies that are endemic to tidal marsh habitat (Spautz and Nur 2008a, 2008b).   For this indicator, it is possible and desirable to identify an absolute benchmark that will provide insight regarding success at meeting the first stated goal, "restoring healthy natural reproduction" for this species. On the basis of demographic modeling of this species, drawing on PRBO studies and the literature, it appears that a stable population of tidal marsh Song Sparrows requires nest survival probability of 20% or greater, and more likely 25% or greater, to achieve "source" status rather than "sink" status (Nur et al. 2007), where "source" refers to a population which can sustain itself without net immigration (Nur and Sydeman 1999). There is some uncertainty here, due to uncertainty with regard to other demographic parameter value. Our **best estimate** is 22 to 25%, but, we recognize that values as low as 20% may be sufficient.

Results:

Nest survival probabilities, standardized for site-to-site variation are shown for San Pablo Bay and Suisun Song Sparrows (Fig T-2). In 7 years out of 11, Suisun values were below 15%. This is a serious concern, given that at least 20% survival probability is needed for sustainability of the population. For San Pablo, the situation is less grave: only 3 out of 10 years were below

15%, but, nevertheless, in 7 years out of 11, nest survival was below 20%. A key point of this analysis is that absolute values are meaningful and not just the trend. The longer-term trend (1996 to 2006) is for nest survival to demonstrate a weak negative trend (5.5% decline per year, $P = 0.093$) for San Pablo Song Sparrows, and a slight increase (6.6% per year, $P > 0.1$) for Suisun Song Sparrows.

Reproductive Success in tidal marsh songbirds appears to be insufficient to maintain population levels. Substantial improvement is needed to meet the goal of "restoring healthy natural reproduction." Suisun Song Sparrows have shown a slight increase in nest survival, between 1996 and 2006, but nevertheless in every year except one, nest survival was below the 20% threshold. Low reproductive success may account for the decline in Suisun Song Sparrow population density observed since 2000 (see Biotic Condition 1, above). San Pablo Song Sparrow nest survival rates are closer to meeting the minimum threshold of 20%, but at the same time this subspecies has demonstrated an apparent decline in nest survival, especially since 2000. The Alameda subspecies of tidal-marsh Song Sparrow appears to have low nest survival rates as well (Nordby et al. 2009), though no trend information is available.

The causes of low nest survival probability are likely two-fold: high levels of predation on nests and nest failure due to flooding (i.e., tidal inundation; Greenberg et al. 2006, Nordby et al. 2009). Nest-predators are not well identified for tidal marsh Song Sparrows (Spautz and Nur 2008a, Spautz and Nur 2008b), but certainly include non-native predators, such as feral cats (*Felix catus*), red fox (*Vulpes fulva*), and Norway rats (*Rattus norvegicus*), as well as native predators, such as corvids (American Crow [*Corvus brachyrynchos*] and Common Raven [*Corvus corax*]) that thrive in proximity to humans.

Nordby et al. (2009) also identified a specific threat associated with the invasive cordgrass, *Spartina alterniflora* and its hybrids: nests in this type of plant were more likely to fail due to flooding, possibly because of the low elevation of the invasive *Spartina*, relative to high tides.

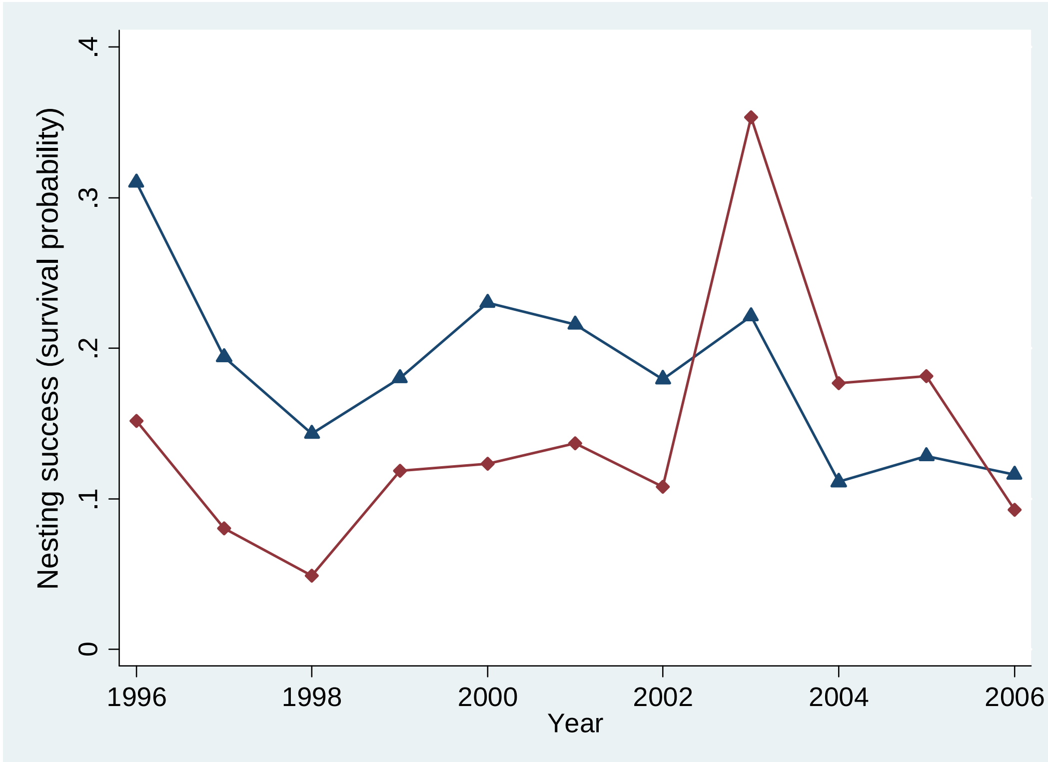


Figure T-2. Nesting success (standardized probability nesting attempt survives to fledge 1 or more young, see text) for San Pablo (blue triangle) and Suisun Song Sparrows (red diamond), based on PRBO (unpublished) and Liu et al. (2007).

3. Heron and Egret Nest Density

John Kelly and Nadav Nur

Background and Rationale:

Audubon Canyon Ranch has monitored Great Blue Heron (*Ardea herodias*) and Great Egret (*Ardea alba*) nest abundance at all known nesting colonies (40-50 sites) in the northern San Francisco Estuary, annually, since 1991. The conspicuousness of heron and egret nesting colonies facilitates the use of nest abundance as an effective index of breeding population abundance and distribution. Heron and egret nest abundance is recognized as a valuable metric for assessing biotic condition in estuarine and wetland ecosystems (Fasola et al. 2010, Kelly et al. 2008, Erwin and Custer 2000). Energetic limits on the foraging ranges of these species are associated with interannual shifts among nesting colony sites that in turn lead to dynamic variation in nest density which reflects suitability of surrounding feeding areas (Gibbs 1991, Wittenberger and Hunt 1985, Kelly et al. 2008). The two target species are used to indicate population responses to different habitat conditions: Great Egrets preferentially forage in small ponds in emergent wetlands and in areas with shallow, fluctuating water depths for foraging. In contrast, Great Blue Herons forage along the edges of larger bodies of water and creeks and are less sensitive to water depth (Custer and Galli 2002, Gawlik 2002). This indicator is sensitive to changes in land-use, hydrology (especially water circulation and depth), geomorphology, environmental contamination, vegetation characteristics, and the availability of suitable prey (Kushlan 2000).

Differences in breeding abundance reflect responses to habitat conditions within 30-300 km^2 (Custer et al. 2004, Kelly et al. 2008) and can be used to evaluate differences in habitat use between or across years at multiple spatial scales (colony sites, major wetland subregions, region-wide). Linkage between nest abundance and the landscape distribution of wetland habitat types is well-documented in the San Francisco Estuary (Kelly et al 2008) and in the Sacramento Valley (Elphick 2008). At the local scale of colony sites and adjacent marshes, changes in heron and egret nest abundance reflect variation in other factors, such as disease, nest predation, especially by human commensal species such as raccoons or ravens, and direct human disturbance to colony sites (Kelly et al. 2007).

Herons and egrets are frequently used as symbols of wetland conservation (Parnell et al. 1988, Kushlan and Hancock 2005) and are widely recognized as indicators of wetland health (Kushlan 1993, Erwin and Custer 2000). These values lead to compelling interest by policy makers, resource managers, and the public, in metrics related to the ecological status of herons and egrets.

Data Source:

The Heron and Egret Nest Density Indicator was calculated using data from ongoing regional heron and egret studies by Audubon Canyon Ranch (Kelly et al. 1993, 2007). The data, which reflect repeated annual nest counts at all known colony sites, provide intensive and extensive measurements of nest abundance and an effective index of regional breeding population sizes. Additional data on nest abundances in the southern San Francisco Bay (not presented here) are available from partners at the San Francisco Bay Bird Observatory.

Methods and Calculations:
The Heron and Egret Nest Density Indicator includes metrics calculated for Great Egrets and Great Blue Herons. Results are provided for each year (1991-2008; updated results to 2010 are pending), for each colony within each of three northern subregions (Central San Francisco Bay, San Pablo Bay, and Suisun Bay). Nest density estimates are based on the peak number of active nests among four (monthly) visits to at each colony site (total of 40-50 sites) within foraging range (10 km) of the historic tidal wetland boundary (ca.1770–1820; San Francisco Estuary Institute 1999; Figure H1), summed annually within and across subregions. Density is calculated based on the estimated peak nest abundance within 10 km of the historic tidal-marsh boundary of Suisun Bay, San Pablo Bay, and Central San Francisco Bay, and within the combined area of the three subregions, excluding the extensive open water areas of the San Francisco Estuary (Figure H1). The detection of new colony sites is achieved through ongoing communications with state, regional and local natural resource managers, county breeding bird atlas efforts, local birding networks, and occasional ground-based and aerial searches of the region. The Nest Density Indicator is calculated as the geometric mean percent change in nest density for the two species, in comparison to the 1991-1995 average (Great Blue Heron: 181±15 nests [S. E.], corresponding to 5.1±0.43 100 km^{-2}; Great Egret: 535±47 nests, corresponding to 15.1±1.33 100 km^{-2}). That is, the proportional change was calculated for each species for the specified time period and then the geometric mean was calculated; finally, the mean proportional change was converted to percent change.

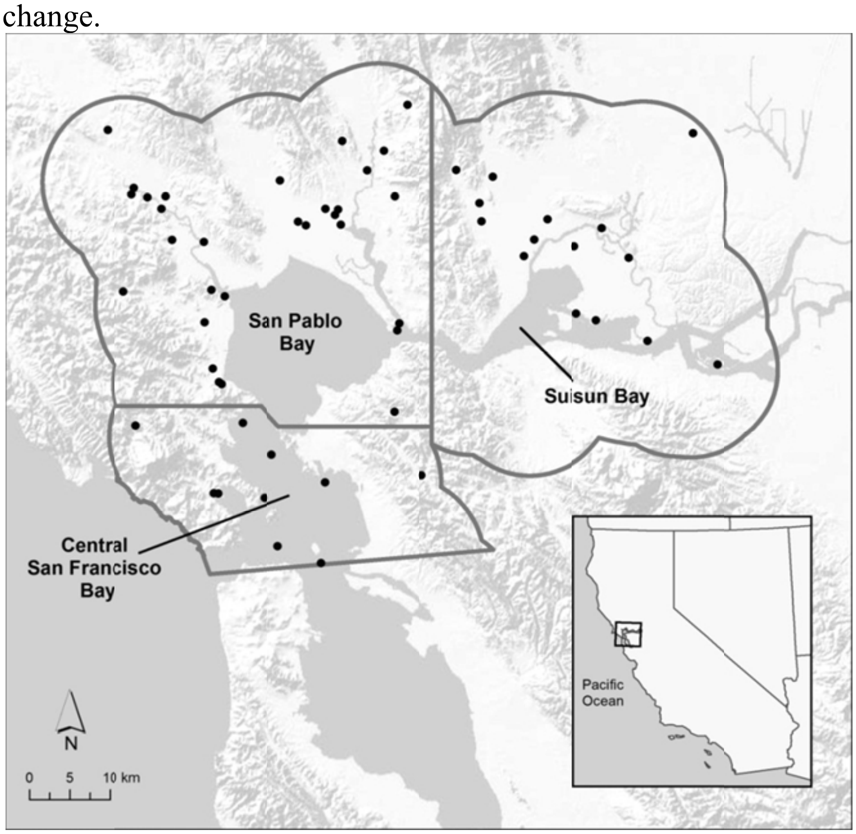


Figure H1. Heron and egret nesting colonies within 10 km of historic tidal wetlands in northern San Francisco Estuary, 1991-2008. Areas indicated by boundary lines, excluding the open waters of Suisun Bay, San Pablo Bay, and the Central Bay, were used to determine heron and egret nest density.

Goals, Targets and Reference Conditions:
CCMP goals to "restore" and "enhance" the ecological productivity and habitat values of wetlands are non-quantitative in nature. However, the use of time series back to 1991 allows the specification of appropriate quantitative reference conditions. Differences or trends in nest density can be quantified and used for assessment.

<u>Maintenance of current regional or subregional breeding densities</u>
- Target: current 5-year trend (linear) ≥ 0, i.e., stable or increasing
- Target: current 15-year trend ≥ 0, i.e., stable or increasing
- Target: current 3-year mean ≥ 5-year reference mean (1991-1995), i.e., current levels equal to or greater than reference.

<u>Enhancement of regional or subregional breeding densities with wetland restoration</u>
- Target: current 5-year trend (linear) ≥ current 15-year reference trend
- Target: current 3-year mean ≥ highest 5-year *subregional* reference mean (1991-1995)

Results: Annual results of the Heron and Egret Nest Density Indicator are shown in Figure H2.
Regional nest densities are stable for both species but 5-year trends provide evidence suggesting recent declines.
Recent (2006-2008), regional nest densities of herons and egrets did not differ significantly compared to 1991-1995 reference levels (t-tests, $P > 0.05$). Recent 15-year (1994-2008) linear trends in percent change in (log-transformed) nest density are > 0, but are marginally or not statistically significant for the combined species index (Indicator: $F_{1,13} = 3.3$, $0.05 < P < 0.10$) and for individual species (Great Blue Heron: $P < 0.08$; Great Egret: $P < 0.18$). In contrast, the recent 5-year regional trends (2004-2008) are declining, although not significantly ($P > 0.05$), for both species, and trends are significantly less than the *current* 15-year trends, for the Indicator ($t_{18} = 4.2$, $P < 0.001$, Figure H2) and for each species (Great Blue Heron: $P = 0.02$; Great Egret: $P < 0.01$). This suggests recent, relative regional declines in breeding densities. Trends within subregions were similar to regional trends, with one exception: trends in San Pablo Bay were dominated by a small but dramatic increase in Great Egrets nest abundance, from less than 5 nests, in the early 1990s, to 163 in 2008 (Figure H2).

Nest densities were lower in San Pablo Bay than in other subregions, with some evidence of relative increases and a reduced variation among subregions.
During the reference period (1991-1995), Great Egret nest density was significantly lower in San Pablo Bay than in both other subregions, for Great Egret and, marginally, for Great Blue Heron (multiple comparisons, $P < 0.001$ and $P \leq 0.08$, respectively). The nest density indicator revealed a dramatic percent increase in San Pablo Bay in recent years (2006-2008) relative to the reference period (981±51%), that which was significantly greater than in other subregions (multiple comparisons, $P = 0.001$). As a result, Great Egret nest density in San Pablo Bay during the response period (2006-2008) was significantly lower only in comparison with Suisun Bay (multiple comparisons, $P < 0.05$), and Great Blue Heron density did not differ significantly among subregions ($F_{4,4} = 2.4$, $P = 0.21$).

Based on nest densities of Great Blue Herons and Great Egrets, CCMP goals of restoring or enhancing wetland productivity and associated wetland habitat values have not been generally met, but possible responses to habitat enhancement are suggested.

Nest densities in most of northern San Francisco Bay are stable, with some suggestion of gradual, long-term, subregional increases in San Pablo Bay wetlands. In that subregion, evidence of increasing nest density, especially for Great Egrets, suggests possible responses to continuing habitat restoration and enhancement. However, regional trends in recent years provide evidence of possible declines across northern San Francisco Bay.

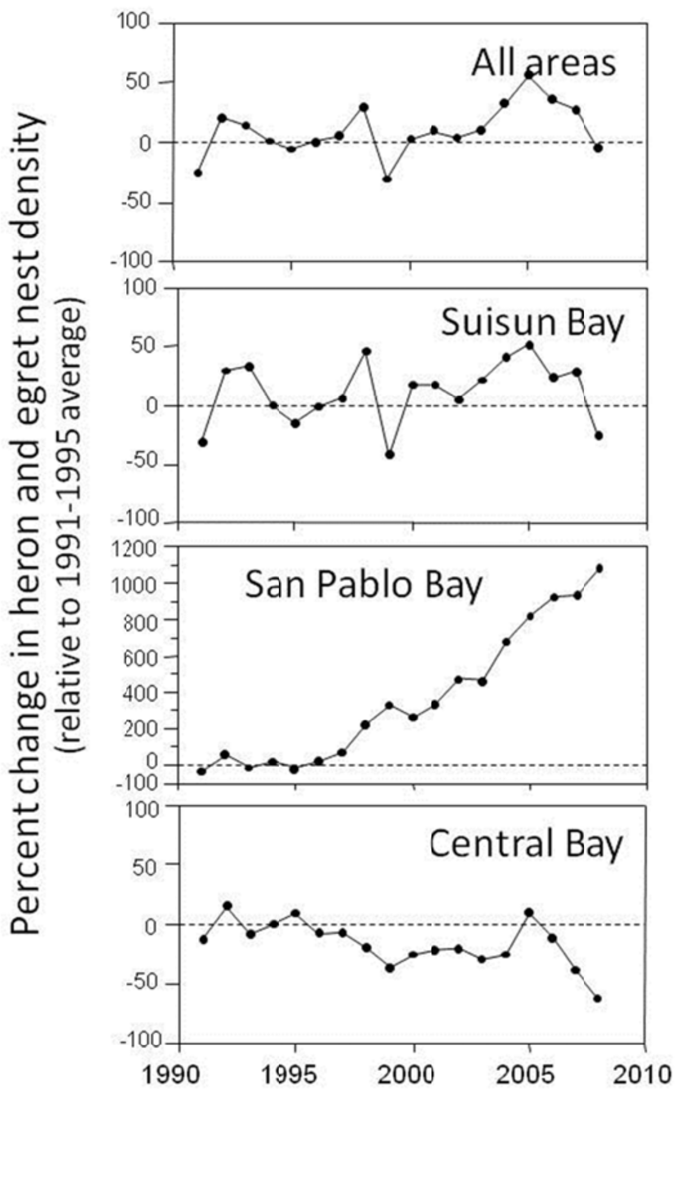


Figure H2. Annual percent change in heron and egret nest density, 1991-2008, relative to the average nest density (dashed line), 1991-1995, in the northern San Francisco Estuary.

4. Heron and Egret Nest Survival

John Kelly and Nadav Nur

Background and Rationale:
Audubon Canyon Ranch has monitored the survival of focal Great Blue Heron (*Ardea herodias*) and Great Egret (*Ardea alba*) nests across breeding colonies throughout the northern San Francisco Estuary, annually, since 1994 (Kelly et al. 2007). Here we use "nest survival" as a term that also encompasses "nest survivorship"; the latter refers to the proportion of nests that survive from initiation to a specified point in time, whereas the former can refer to the probability of survival during any relevant time period. An extensive literature has developed regarding nest survival and its analysis, see Jones and Geupel (2007). Another commonly used name for the same parameter is "nesting success". In all cases, what is estimated is the probability that an individual nesting attempt will survive to successfully produce one or more fledged (or independent) young.

The conspicuousness of heron and egret nesting colonies and the visibility of nests facilitates the monitoring of nesting activity and the use of nest survival as an effective index of overall nest success. This indicator is sensitive to nest predation and colony disturbance by native and introduced nest predators (especially by human commensal species such as raccoons and ravens), land development and human activity near heronries, and severe weather (Pratt and Winkler 1985, Frederick and Spalding 1994, Kelly et al. 2005 and 2007). Such ecological processes can vary over space and time in response to landscape patterns of habitat change, dynamics of predator populations, and changes in human land use, and are therefore likely to differentially affect nesting colonies of herons and egrets. Note that heron and egret nest survival is not a particularly strong indicator of food availability. Rather, food availability (and more generally, the food web) for piscivorous birds is reflected in the "Heron and Egret Brood Size Indicator", see Ecological Processes, Food web Indicator.

Data Source:
The Heron and Egret Nest Survival Indicator was calculated using data from ongoing regional heron and egret studies by Audubon Canyon Ranch (Kelly 1993, 2007). The data, which reflect the survival of focal nests followed through the entire nesting cycle on repeated visits to colony sites throughout the northern San Francisco Estuary, provide an effective index of regional and subregional nest success.

Methods and Calculations:
The Heron and Egret Nest Survival Indicator, calculated as the apparent nest success of Great Egrets and Great Blue Herons, is based on the proportion of focal nests that remain active through the nesting cycle, from nest initiation or early in the incubation period, at 40-50 colony sites within 10 km of the historic tidal wetland boundary (ca.1770–1820; San Francisco Estuary Institute 1999; see Figure H1). Great Egret and Great Blue Heron nests are considered successful if at least one young survives to minimum fledging age of seven or eight weeks, respectively (Pratt 1970, Pratt and Winkler 1985). Nest are sampled I approximate proportion to colony size. In colonies with fewer than 15 active nests, all nests initiated before the colony reaches peak nest

abundance are treated as focal nests. At larger colonies, random samples of at least 10-15 focal nests are selected. The nest survival indicator is calculated by (1) comparing observed nest survival by subregion, year, and species to average nest survival during the five year reference period (1994-1998) for the appropriate region and species, (2) determining the proportional difference between observed and reference period, (3) calculating the geometric mean across the two species, and (4) converting this to percent change. The 5-year reference value, averaged across the two species was 0.765, 0.880, and 0.823, for Central SF Bay, San Pablo Bay, and Suisun Bay, respectively.

Goals, Targets and Reference Conditions:
CCMP goals to "restore" and "enhance" the ecological productivity and habitat values of wetlands are non-quantitative. However, the use of time series back to 1994, allows the specification of appropriate quantitative reference conditions. Differences or trends in nest survival can be quantified and used for assessment.

<u>Maintenance of current resource levels</u>
- Target: current 3-year mean (2006-2008) ≥ 5-year reference mean (1994-1998)

<u>Enhancement of resources with wetland restoration</u>
- Target: *current* 3-year mean (2006-2008) ≥ highest 5-year *subregional* reference mean (1994-1998).

Results: Results of the Heron and Egret Nest Survival are shown in Figure H3.
Recent rates of nest survival (2006-2008) were generally lower than reference levels (1994-1998).
A marginally significant regional decline in nest survival (12.7%, $t_{272} = 1.97$, $P = 0.05$) reflected primarily a 16.8% decline in the survival of Great Egret nests (not shown). Within subregions, nest survival for both species combined was significantly lower than the 1994-1998 regional level only in San Pablo Bay, which was lower, primarily because of an 18.1% decline in the survival in Great Egret nests ($t_{445} = 2.3$, $P < 0.02$; Figure H3). However, in the Central Bay, Great Blue Heron nest survival was 21.6% lower ($t_{75} = 2.6$, $P < 0.05$) than in reference period and, in Suisun Bay, survival of Great Egret nests was 27.5% lower ($t_{146} = 4.1$, $P < 0.001$). Species-specific reference nest survival probability (1994-1998) are 0.805 for Great Blue Heron and 0.818 for Great Egret.

Nest Survival differs among subregions, with differential ranking between species.
The Nest Survival Indicator differed significantly among subregions ($F_{2, 817} = 3.6$, $P < 0.05$). Suisun Bay exhibited significantly higher Great Blue Heron nest survival (10.0% increase over the regional reference level) and significantly lower Great Egret nest survival (32.0% decline) than other subregions (multiple comparisons, $P < 0.05$).

Based on the survival of Great Blue Heron and Great Egret nests, CCMP goals of restoring or enhancing wetland productivity and associated wetland habitat values have not been met in the region, although evidence suggests some subregional enhancement in nest survival.
Recent survival rates of Great Blue Heron and Great Egret nests are generally lower than rates measured during the 1994-1998 reference period. However, possible enhancement of Great Blue Heron nest success was suggested by the results for Suisun Bay. Differences in the survival of

heron and egret nests among the subregions suggest that breeding performance in these species may contribute to informed comparisons of biotic condition among regions within the San Francisco Estuary.

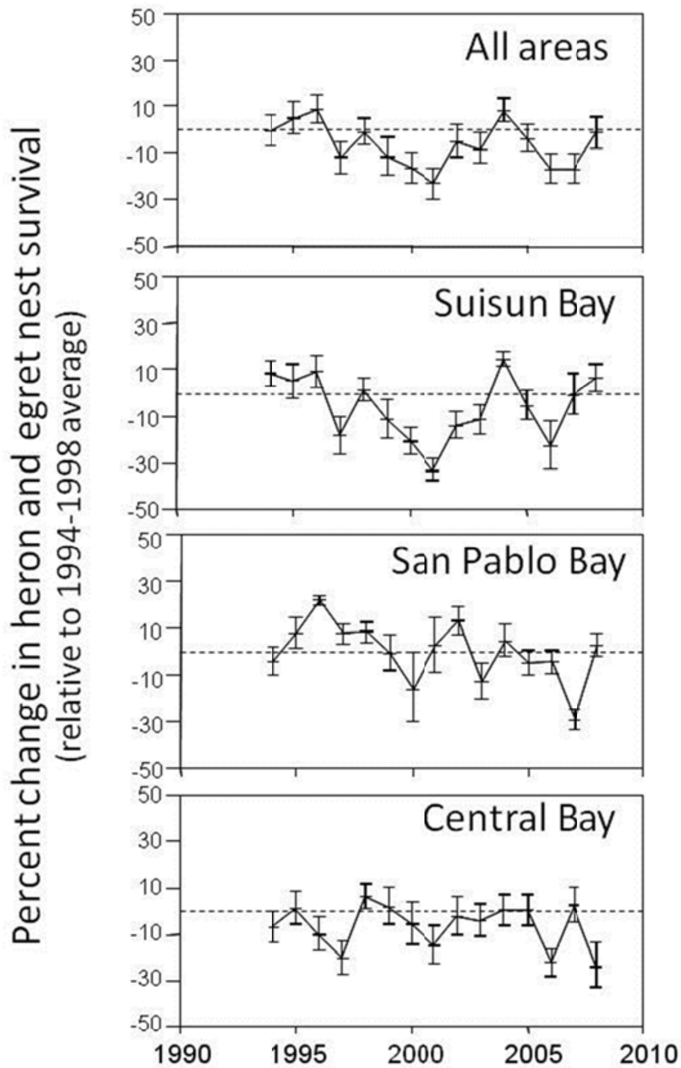


Figure H3. Annual percent change in heron and egret nest survival (i.e., nesting success), 1994-2008, relative to the reference period (dashed line), 1994-1998, in the northern San Francisco Estuary; separate results shown for the three subregions.

5. Wintering Waterfowl Populations
Nadav Nur

Background and Rationale:
San Francisco Estuary provides important wintering habitat for waterfowl (Goals Project 2000, Steere & Schaefer 2001), one of the most important such areas in North America. For some species, during the winter, San Francisco Estuary hosts a majority of the entire Pacific Flyway population (Steere & Schaefer 2001). This is in addition to the estuary's value to waterfowl during the breeding season (especially in Suisun Bay region) and during the spring and fall migratory periods. More than 30 species of waterfowl are commonly observed in the San Francisco Bay region (Goals Project 2000).

The importance of the estuary for waterfowl has long been recognized. The San Francisco Bay region is identified as a waterfowl habitat area of major concern in the North American Waterfowl Management Plan (NAWMP; U.S. Fish and Wildlife Service 1998). NAWMP is implemented and financed through joint venture partnerships involving federal and state agencies, along with non-government organizations, and the private sector. The San Francisco Bay Joint Venture is one such partnership, playing an active role in conservation throughout the Bay area (Steere and Schaefer 2001).

Because of the long-recognized importance of waterfowl to the mission of the U. S. Fish and Wildlife Service, the "Mid-Winter Waterfowl Surveys" have been conducted by this agency, throughout the United States since 1955, in cooperation with state agencies (Eggeman and Johnson 1989). The biotic indicator used here for the San Francisco Estuary, therefore, is just a subset of the nation-wide effort. The survey attempts to enumerate all waterfowl, by species, for the entire estuary. Survey efforts target three habitats or areas: open bay throughout the estuary; salt ponds in the estuary (San Pablo Bay and South San Francisco Bay); and Suisun Marsh (including Grizzly Island Wildlife Area). The principal objective of the MWW Surveys is to provide information on population trends.

Waterfowl include **dabbling ducks**, which feed at the surface or in shallow water, **diving ducks**, which forage underwater, **swans**, and **geese**, which feed on plants in wetlands and fields. For the "Wintering Waterfowl Populations Indicator" we focus just on the two most abundant (and species-rich) groups of waterfowl, **dabbling ducks** and **diving ducks**. Swans and geese are not currently a primary component of San Francisco Bay waterfowl, with the exception of the Canada Goose which has become a pest species recently. In addition to the four waterfowl groups listed above (dabbling ducks, diving ducks, swans, and geese) the Mid-Winter Waterfowl surveys identify a fifth group: **sea ducks**. We have chosen not to include in this indicator the sea ducks, which are considered a distinct group of waterfowl and have their own joint venture (www.seaduckjv.org). Sea ducks are most commonly found in coastal and off-shore areas of the Bay region. In San Francisco Bay, sea ducks are almost entirely represented by scoters (*Melanitta* spp.; Surf Scoter, Black Scoter, and White-winged Scoter). The indicator presented here could be re-calculated to include scoter species as well, but we have not done so.

Data Source:
USFWS and CDFG jointly conduct surveys in the San Francisco Estuary in January of each year. Joelle Buffa (USFWS) and Michael Wolder (USWFS) kindly provided the data used here. Data are summarized by survey area and then compiled into regional summaries.

Methods and Calculations:
Surveys are conducted on a single day per survey area per year; sometimes several areas are surveyed in a single day. Surveys are conducted mainly from fixed-wing aircraft, but sometimes from the ground or by boat. Open bay and salt ponds are the target of surveys by USFWS observers throughout the estuary. The Mid-Winter Waterfowl Survey summarizes counts by bay region: Suisun Bay, North Bay (i.e., San Pablo Bay and the northern portion of San Francisco Bay), Central San Francisco Bay, and South San Francisco Bay. Suisun Marsh, including Grizzly Island Wildlife Area, is the target of surveys by CDFG, which also surveys the Delta. Thus, bayland habitat in the estuary is surveyed in the Suisun region but elsewhere the focus is on open water and salt ponds (Takekawa 2002).

As noted on the USFWS website for Mid-Winter Waterfowl Surveys, "[S]pecific sampling procedures are not defined. Instead, an aerial crew determines the best and most practical means to conduct a complete count of all waterfowl within a predefined unit area." Surveys are not standardized with respect to tide. Weather and other physical conditions during the survey period are noted but analyses do not statistically adjust for weather conditions. Survey effort may be noted, but numbers are not adjusted by effort. In theory, one could convert counts into densities by dividing by the area surveyed, but this has not yet been implemented.

The analysis presented here uses the regional totals in each year, broken out by species, where region is Suisun Bay, North Bay, Central San Francisco Bay, and South San Francisco Bay. Since "North Bay" is predominantly San Pablo Bay, and since many of the other indicators refer to San Pablo Bay, we use the latter term here, but note that the surveyed area extends beyond San Pablo Bay proper. In addition, "Suisun Bay" refers to the open water of the bay. Suisun Marsh is not currently included in the indicator results, but we are working to include these counts in the metric in the future. The indicator will be re-analyzed when such data are available.

We analyzed changes in the natural log-transformed counts per region and per species in a statistical model, analyzed separately for diving ducks and dabbling ducks. The modeling was similar to the analysis used for combined species Tidal Marsh Bird Populations indicator (Living Resources – Birds, Indicator 1). Individual dabbling and diving duck species were excluded if the majority of years had zero counts for that species. This left twelve species for analysis: six species of dabblers (American Wigeon, Gadwall, Green-winged Teal, Mallard, Northern Pintail, Northern Shoveler) and six species of diving ducks (Bufflehead, Canvasback, Goldeneye, Redhead, Ruddy Duck, Scaup). Statistical models were fit separately for each waterfowl group (dabbling or diving ducks) and for each bay region (four regions). Each model had data from six species and included a species "main effect." Thus, we allowed for differences in the overall abundance of the six species (dabbler or diving ducks) while estimating the trend over time common to the six (dabbler or diving duck) species for the specific bay region. The approach used was similar to that used for the "combined-species" index of tidal marsh bird populations (described above, Living Resources – Birds, Indicator 1; Nur et al. 1999).

For this exercise, we analyzed data from the period 1988 (South San Francisco Bay) or 1989 (rest of the estuary) to 2006. We propose that, in the future, analyses be conducted incorporating more recent data (e.g., through 2011), once these data are available. We used three methods to evaluate change over time: (1) long-term trends over time, for the period 1989 to 2006 (except 1988 to 2006 for South San Francisco Bay), (2) short-term trends over time, for the most recent 5-year period (i.e., 5 winters), which was 2002-2006 (except 2001-2006 for Suisun because surveys were not conducted there in 2005), and (3) comparison of the period 2004-2006 with the 5-year benchmark period, 1989 to 1993 (except 1988 to 1992 for South San Francisco Bay; only South San Francisco Bay had data available for 1988).

Goals, Targets, and Reference Conditions:
The San Francisco Bay Joint Venture (Steere and Schaefer 2001) has determined that values for estimated waterfowl abundance in the period 1988 to 1990 should be used as a baseline for comparison, and furthermore these estimated abundances should also provide goals for individual species. Table W1 provides estimates of the "long-term trends" by waterfowl group since 1988 or 1989 up until 2006, which allows preliminary evaluation of trends. In addition, survey estimates of recent numbers can be compared directly with the earlier, benchmark period (in this case, 1989 to 1993 or 1988 to 1992). Because of the high year to year variation in number, primarily due to the fact that the survey is conducted only on a single day, and that some important influences on counts are not statistically controlled for, we feel that all comparisons must be based on data from multiple years, and no comparisons should be based on any single year. In our case, we compared the most recent three-year period (2004-2006) with the five-year period that includes 1988 to 1990, but with two additional years included, so as to provide a robust benchmark value.

Results:
In **Suisun Bay**, **dabbling ducks** demonstrate an increase, but only in recent years (since 2001; Figure W1 A; Table W1). As a result, their numbers show a significant increase in recent years, compared with the 5-year benchmark period (Table W2). **Diving ducks** in Suisun Bay demonstrate a weak (non-significant) decline, with an estimated decline of 18.4%/year in the most recent 5 years (Figure W1 B; Table W1). Note that these population changes only refer to numbers as assessed in open water of the Bay. We intend to add Suisun Marsh data at a later time.

In **San Pablo Bay** (i.e., North Bay**)**, **dabbling ducks** also demonstrate an increase, over a sustained period of time, 1995 to 2006 (Figure W1 C). However, the most recent 5-year period evidences a decrease, not an increase, for this group. Nevertheless, the result, when comparing the most recent 3-years with the 5-year benchmark is a significant increase (Table W2). **Diving ducks**, in contrast, have shown an overall decrease, and in the most recent years, this decline is significant (Figure W1 D, Table W1). The result is that counts for the most recent 3-year period are significantly lower for diving ducks in the North Bay compared to the 5-year benchmark period (Table W2).

In the **Central San Francisco Bay**, **dabbling duck** numbers have been low in every year except 1999. As a result, there are no significant trends or differences for this group, though the tendency has been for decreases in number (Figure W1 E, Tables W1 and W2). Compared to historical numbers, there is likely cause for concern. **Diving duck**s in this region have demonstrated no significant trends for the long-term or short-term, though since 1999 the trend has clearly been negative (Figure W1 F). That is, a decrease from 1989 to 1997 was followed by an increase from 1997 to 2001, followed by another drop.

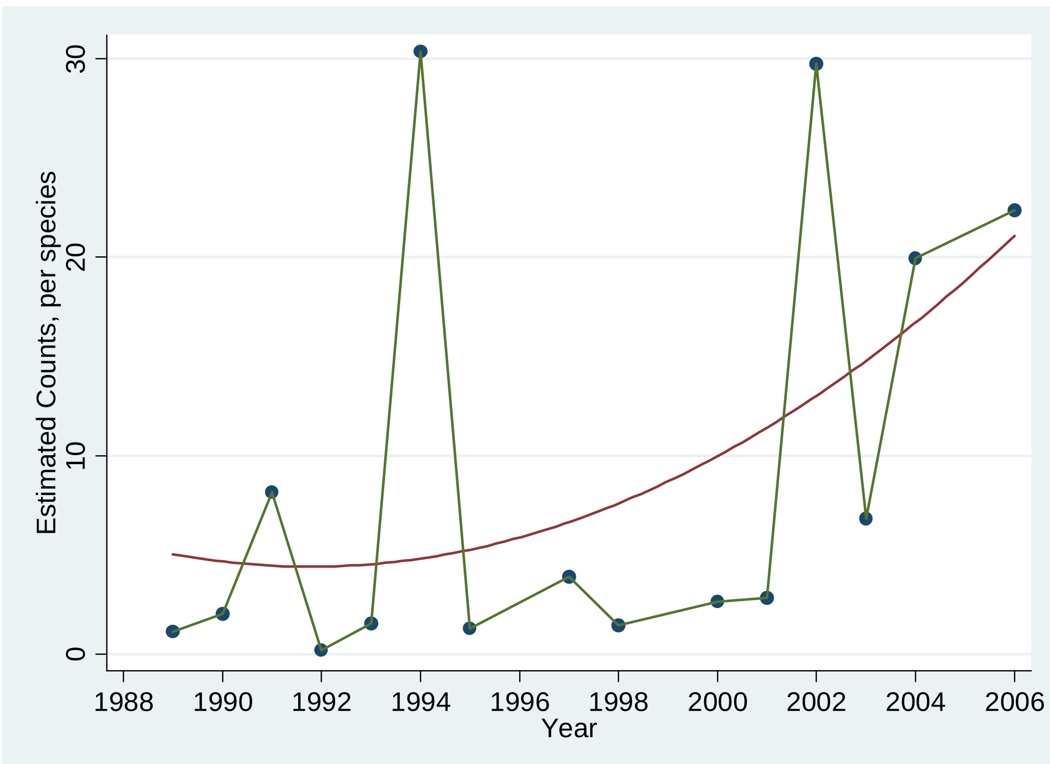
In **South San Francisco Bay**, **dabbling ducks** demonstrate a slight increase overall (Figure W1 G). Numbers in the most recent 3-year period are marginally significantly greater in the most recent 3 years compared to the 5-year benchmark period (P = 0.096, Table W2). **Diving ducks** show an increase from 1988 to 2001, resulting in a significant increase over the long-term (Figure W1 H), with a non-significantly higher numbers in the most recent 3 years compared to the benchmark period (Table W2). However, since the peak in 2001-2002, there has been an overall decline, which during the last 5 years (2002-2006) is marginally significant (P = 0.083).

Scoters, since they are usually considered sea ducks were not included, but it is interesting to compare their trends to dabblers and divers. There were no significant (P > 0.1) long-term or short-term trends evident for scoters in any region. However, numbers in the most recent 3-year period were marginally significantly lower in San Pablo Bay (North Bay) than they were in the 5-year benchmark period (P = 0.072).

To summarize, the patterns are very different comparing dabbling ducks to diving ducks: Dabbling ducks have increased in Suisun and San Pablo Bay, and there is the suggestion of an increase in the South Bay, too. Diving ducks have decreased in San Pablo Bay and they demonstrate recent, short-term declines in all bay regions, though the declines are not significant in every case. Still, **the magnitude of decline for diving ducks is of concern:** for each bay region, recent declines exceeded 18% per year during the period 2002 to 2006. Thus, **CCMP Aquatic Resources Goal 1, to stem and reverse the decline in abundance of estuarine biota, has not been met for diving ducks, but the situation is encouraging for dabbling ducks.** Furthermore, current tidal marsh habitat restoration efforts are likely benefitting dabbling ducks, but not diving ducks, since the former utilize the shallow water habitat found in tidal marshes, but the latter group does not (Stralberg et al. 2009). The discrepancy for the two groups of waterfowl will only be enhanced in the future as more restoration projects come to fruition.

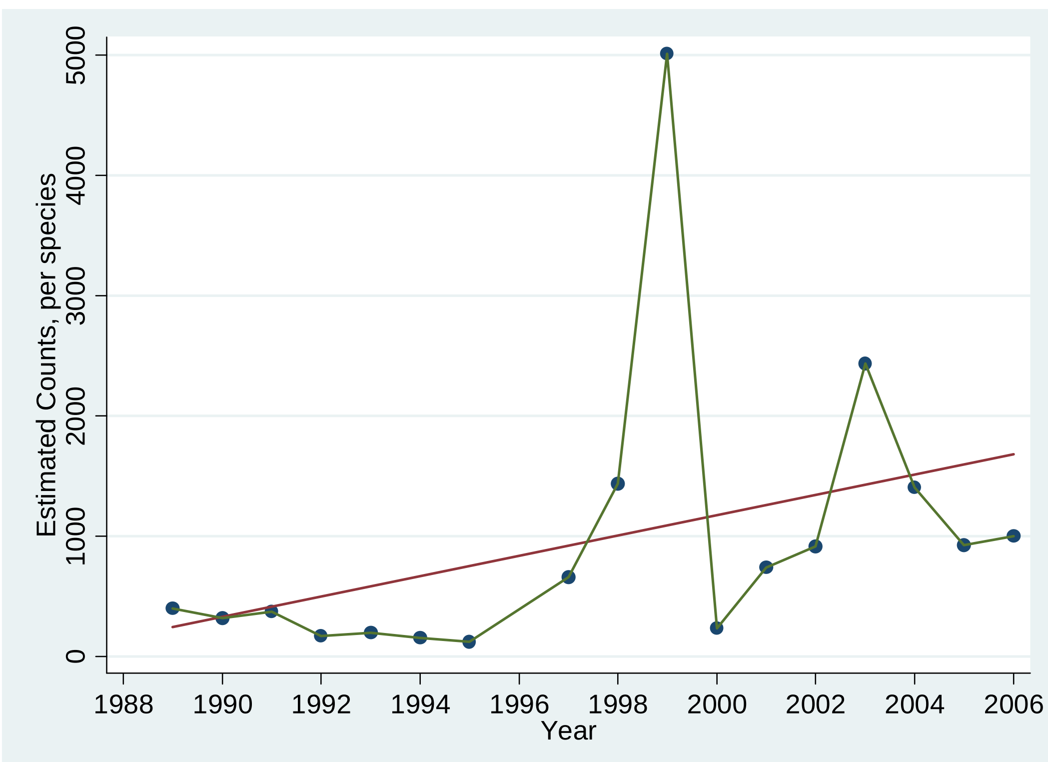
Figure W1. Population Trends for Waterfowl in San Francisco Estuary, 1988 to 2006. Results are from USFWS Midwinter Waterfowl Surveys. Shown are statistically estimated mean counts **per species** per year for two groups of waterfowl: Dabbling ducks (6 species included: American Wigeon, Gadwall, Green-winged Teal, Mallard, Northern Pintail, Northern Shoveler) and Diving ducks (6 species included: Bufflehead, Canvasback, Goldeneye, Redhead, Ruddy Duck, Scaup). Results shown for Suisun Bay, San Pablo Bay (i.e., North Bay), Central San Francisco Bay, and South San Francisco Bay. Analyses controlled for species differences in log-transformed counts. Trend lines are shown as quadratic fits (Figure W1-A) or linear fits (Figure W1-B to W1-H); choice of fit (linear vs. quadratic) determined by maximization of adjusted R^2 (Nur et al. 1999).

A) Dabbling ducks, Suisun Bay



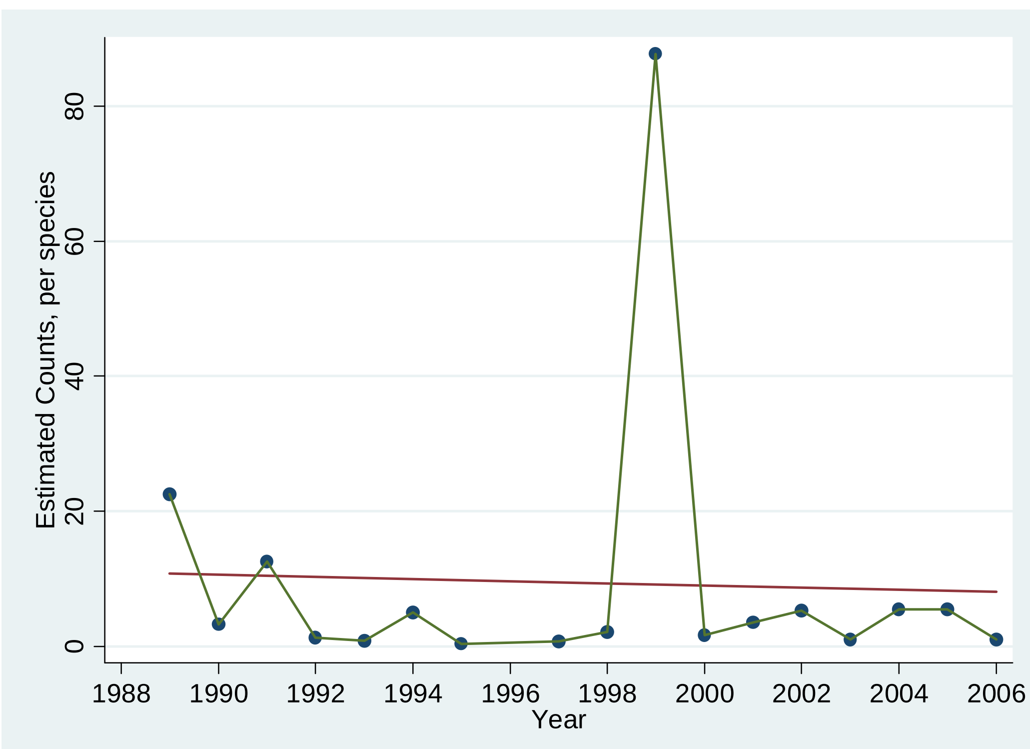
B) Diving ducks, Suisun Bay

C) Dabbling Ducks, San Pablo Bay (North Bay)



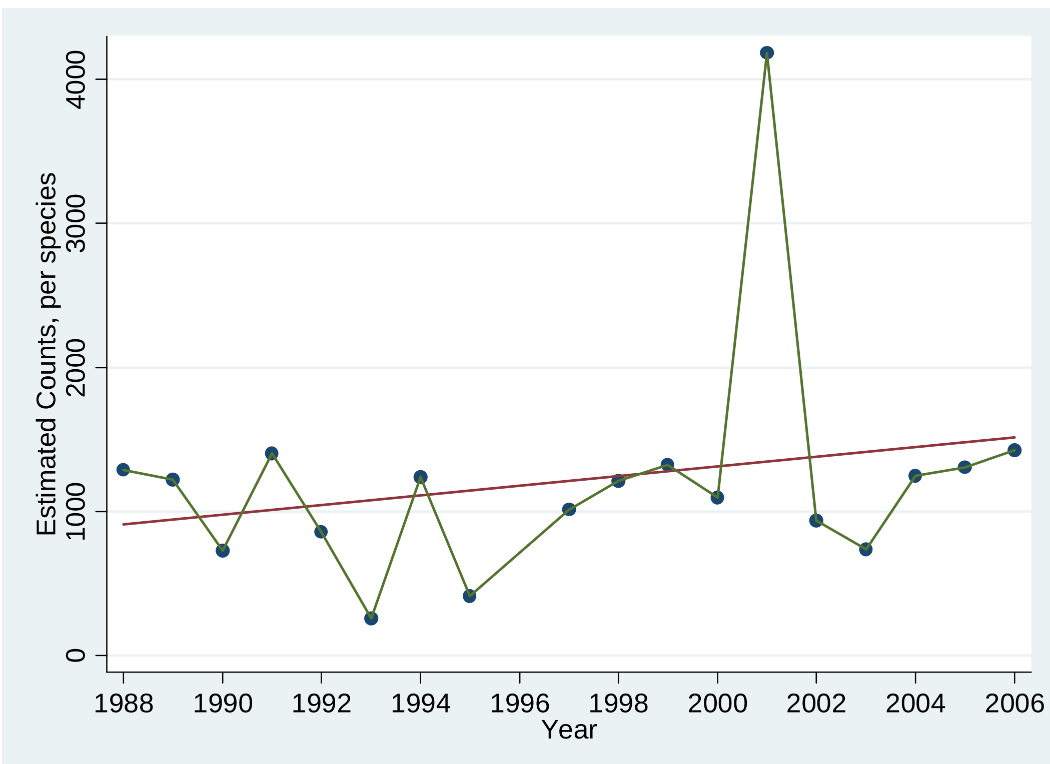
D) Diving Ducks, San Pablo Bay (North Bay)

E) Dabbling Ducks, Central San Francisco Bay



F) Diving Ducks, Central San Francisco Bay

G) Dabbling Ducks, South San Francisco Bay



H) Diving Ducks, South San Francisco Bay

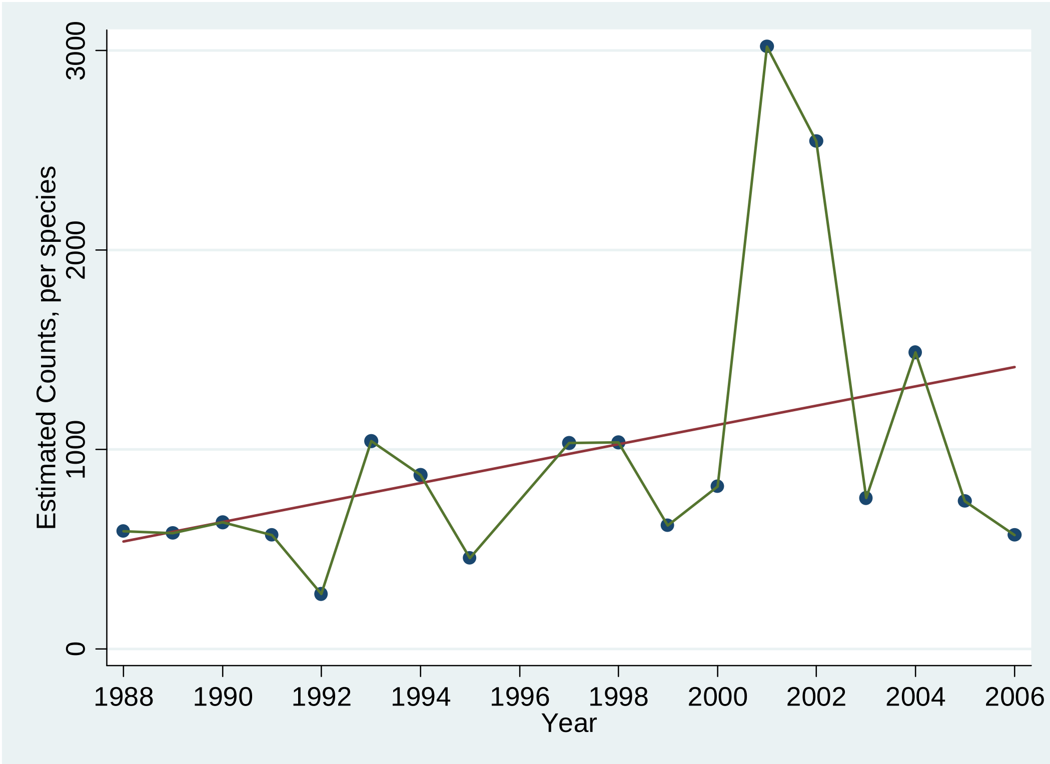


Table W1. San Francisco Estuary Waterfowl: Long-term (from 1988 or 1989 to 2006) and Short-term (2002 to 2006) trends for two groups of waterfowl. Shown are estimated annual percent changes per year in population index, as estimated by statistical modeling of log-transformed counts. (Data from mid-winter waterfowl surveys, USFWS). Highlighting indicates significant ($P < 0.05$) differences (bright yellow) or marginally significant (light yellow; $0.05 \leq P < 0.10$)

		Dabbling Ducks		Diving Ducks	
	number of years	Ann Pct	P-val	Ann Pct	P-val
Suisun Bay					
Long-term	15	11.5%	P = 0.001	-2.04%	P > 0.5
Short-term	5	29.0%	P = 0.19	-18.4%	P > 0.3
San Pablo Bay					
Long-term	18	12.5%	P < 0.001	-1.91%	P > 0.3
Short-term	5	-7.63%	P > 0.5	-26.5%	P = 0.033
Central SF Bay					
Long-term	18	-2.44%	P > 0.4	-0.09%	P > 0.9
Short-term	5	-10.3%	P > 0.5	-18.8%	P > 0.2
South SF Bay					
Long-term	19	2.70%	P = 0.14	4.54%	P = 0.037
Short-term	5	15.1%	P > 0.2	-26.0%	P = 0.083

Notes: Suisun, San Pablo Bay, Central SF Bay long-term is for 1989 to 2006; South Bay long-term is for 1988 to 2006.

Short-term is 2002 to 2006, except for Suisun Bay, which is 2001 to 2006 (no survey data in 2005)

Table W2. San Francisco Estuary Waterfowl: Comparison of 3-year recent period (2004-2006) vs. 5-year Benchmark (1989 to 1993). Shown are percent differences in standardized count index for the two time periods (Mid-winter waterfowl surveys, USFWS). Highlighting indicates significant ($P < 0.05$) differences (bright yellow) or marginally significant (light yellow; $0.05 \leq P < 0.10$)

	Dabbling Ducks		**Diving Ducks**	
	Percent	P-value	Percent	P-value
Suisun				
Comparison	683%	P < 0.001	-20%	P > 0.5
North Bay				
Comparison	295%	P < 0.001	-41%	P = 0.021
Central SF Bay				
Comparison	-21%	P > 0.6	-17%	P > 0.6
South SF Bay				
Comparison	58%	P = 0.096	49%	P = 0.17

Note: South SF Bay benchmark is for 1988 to 1992

Peer Review: The above indicators were evaluated using methods and analysis described in the following peer-reviewed publications: Kelly *et al*. (2007), Nur *et al.* (1999), and Spautz *et al.* (2006).

References

Albertson, J. D. and J. Evens. 2000. California Clapper Rail. Pp. 332-341 **in**: Baylands Ecosystem Species and Community Profiles, Olofson, P. R., Ed. Goals Project, San Francisco Bay Regional Water Quality Control Board, Oakland, California.

Chase, M. K., N. Nur, and G. R. Geupel. 2005. Effects of weather and population density on reproductive success and population dynamics in a Song Sparrow (*Melospiza melodia*) population: A long-term study. Auk 122:571-592.

Custer, C. M., and J. Galli. 2002. Feeding habitat selection by Great Blue Herons and Great Egrets nesting in east central Minnesota. Waterbirds 25:115–124.

Custer, C. M., S. A. Suarez, and D. A. Olsen. 2004. Feeding habitat characteristics of the Great Blue Heron and Great Egret nesting along the upper Mississippi River, 1995-1998. Waterbirds 27:254-268.

Eggeman, D. R. and F. A. Johnson. 1989. Variation in effort and methodology for the midwinter waterfowl inventory in the Atlantic Flyway. Wildlife Society Bulletin 17:227-233.

Elphick, C. 2008. Landscape effects on waterbird densities in California rice fields: Taxonomic differences, scale-dependence, and conservation implications. Waterbirds 31:62-69.

Erwin, R. M., and T. W. Custer. 2000. Herons as indicators. Pp. 311-330, in J. A. Kushlan and H. Hafner (eds.), Heron Conservation. Academic Press, San Diego, CA, USA.

Fasola, M., D. Rubolini, E. Merli, E. Boncompagni, and U. Bressan. 2010. Long-term trends of heron and egret populations in Italy, and the effects of climate, human-induced mortality, and habitat on population dynamics. Population Ecology 52: 59-72.

Frederick, P. C. 2002. Wading birds in the marine environment. Pp. 617–55. In E. A. Schreiber and J. Burger (eds.) Biology of Marine Birds. CRC Press, Boca Raton, FL, USA.

Frederick, P. C., and M. G. Spalding. 1994. Factors affecting reproductive success of wading birds (Ciconiiformes) in the Everglades ecosystem. Pp. 659-691, in S. Davis and J. C, Ogden (eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, FL.

Gawlik, D. E. 2002. The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72:329–346.

Gibbs, J. P. 1991. Spatial relationships between nesting colonies and foraging areas of Great Blue Herons. Auk 108:764–770.

Goals Project. 1999. Baylands ecosystem habitat goals. A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystems Goals Project. Joint publication of the U. S. Environmental Protection Agency, San Francisco, CA, and San Francisco Bay Regional Water Quality Control Board, Oakland, CA.

Goals Project. 2000. *Baylands ecosystem species and community profiles: Life histories and environmental requirements of key plants, fish, and wildlife.* Prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. P.R. Olofson,ed. San Francisco Bay Regional Water Quality Control Board, Oakland, Calif.

Greenberg, R., C. Elphick, J. C. Nordby, C. Gjerdrum, H. Spautz, G. Shriver, B. Schmeling, B. Olsen, P. Marra, N. Nur, and M. Winter. 2006. Flooding and predation: trade-offs in the nesting ecology of tidal-marsh sparrows. Studies in Avian Biology 32:96-109.

Jones, S. L. and G. R. Geupel, Eds. 2007. Beyond Mayfield: Measurements of nest-survival data. Studies in Avian Biology 34, Cooper Ornithological Society.

Kelly, J. P., K. Etienne, C. Strong, M. McCaustland, and M. L. Parkes. 2007. Status, trends, and implications for the conservation of heron and egret nesting colonies in the San Francisco Bay area. Waterbirds 30: 455-478.

Kelly, J. P., K. Etienne, and J. E. Roth. 2005a. Factors influencing the nest predatory behaviors of common ravens in heronries. Condor 107: 402-415.

Kelly, J. P., K. Etienne, and J. E. Roth. 2005a. Factors influencing the nest predatory behaviors of common ravens in heronries. Condor 107: 402-415.

Kelly, J. P., H. M. Pratt, and P. L. Greene. 1993. The distribution, reproductive success, and habitat characteristics of heron and egret breeding colonies in the San Francisco Bay area. Colonial Waterbirds 16:18–27.

Kelly, J. P., D. Stralberg, K. Etienne, and M. McCaustland. 2008. Landscape influences on the quality of heron and egret colony sites. Wetlands 28: 257-275.

Kushlan, J. A. 2000. Heron feeding habitat conservation. Pp. 219-235, in J. A. Kushlan and H. Hafner (eds.), Heron Conservation. Academic Press, San Diego, CA.

Kushlan, J. A. and J. A. Hancock. 2005. The Herons. Oxford University Press, New York, NY, USA.

Liu, L., P. Abbaspour, M. Herzog, N. Nur, and N. Warnock. 2007. San Francisco Bay Tidal Marsh Project Annual Report 2006: Distribution, abundance, and reproductive success of tidal marsh birds. PRBO Conservation Science, Petaluma, CA.

Liu, L., J. Wood, N. Nur, D. Stralberg, and M. Herzog. 2009. California Clapper Rail (*Rallus longirostris obsoletus*) population monitoring: 2005-2008. Report to California Department of Fish and Game from PRBO Conservation Science. PRBO Conservation Science, Petaluma, CA.

Marshall, J. T., and K. G. Dedrick. 1994. Endemic Song Sparrows and yellowthroats of San Francisco Bay. Studies in Avian Biology No. 15:316-327.

Martin, T.E. and G.R. Geupel. 1993. Nest-monitoring plots: Methods for locating nests and monitoring success. Journal of Field Ornithology 64:507-519.

Mayfield, H.F. 1975. Suggestions for calculating nest success. Wilson Bulletin. 87:456-466.

Nordby, J. C., A. N. Cohen, and S. R. Beissinger. 2009. Effects of a habitat-altering invader on nesting sparrows: An ecological trap? Biological Invasions 11:565-575.

Nur, N. and W. J. Sydeman. 1999. Demographic processes and population dynamic models of seabirds: Implications for conservation and restoration. Current Ornithology 15:149-188.

Nur, N., S. Zack, J. Evans, and T. Gardali. 1997. Tidal marsh birds of the San Francisco Bay region: status, distribution, and conservation of five Category 2 taxa. Report of the Point Reyes Bird Observatory, 4990 Shoreline Hwy., Stinson Beach, CA 94970 to USGS-Biological Resources Division. Now: PRBO Conservation Science, Petaluma, CA.

Nur, N., M. Herzog, and A. Pawley. 2007. Avian demographic parameters provide metrics of tidal marsh restoration success at site-specific and program-wide scales. Paper presented at the 2007 State of the Estuary Conference, Oakland, CA.

Nur, N., S. L. Jones and G. R. Geupel. 1999. Statistical Guide to Data Analysis of Avian Monitoring Programs. Biological Technical Publication, US Fish & Wildlife Service, BTP-R6001-1999.

Parnell, J. F., D. G. Ainley, H. Blokpoel, B. Cain, T. W. Custer, J. L. Dusi, S. Kress, J. A. Kushlan, W. E. Southern, L. E. Stenzel, and B. C. Thompson. 1988. Colonial waterbird management in North America. Colonial Waterbirds 11:129–69.

Pratt, H. M. 1970. Breeding biology of Great Blue Herons and Common Egrets in central California. Condor 72: 407-416.

Pyle, P., N. Nur, and D. F. DeSante. 1994. Trends in nocturnal migrant landbird populations at southeast Farallon Island, California, 1968-1992. pp. 58-74 **in** *A Century of Avifaunal Change in Western North America*. N. Johnson & J. Jehl, Ed. Studies in Avian Biology **15**.

Steere, J. T. and N. Schaefer. 2001. Restoring the Estuary: Implementation Strategy of the San Francisco Bay Joint Venture. SFBJV, Oakland, CA.

Shuford, W. D, and T. Gardali, Eds. 2008. California Bird Species of Special Concern. Studies of Western Birds No. 1. Western Field Ornithologists, Camarillo, CA, and California Dept. Fish & Game, Sacramento.

Spautz, H., N. Nur, D. Stralberg, and Y. Chan. 2006. Multiple-scale habitat relationships of tidal-marsh breeding birds in the San Francisco Bay estuary. Studies in Avian Biology 32:247-269.

Spautz, H., and N. Nur. 2008a. San Pablo Song Sparrow, *Melospiza melodia samuelis*. pp. 412-418 in *California Bird Species of Special Concern.* W. D. Shuford and T. Gardali, Eds. Studies of Western Birds 1. Camarillo and Sacramento, California.

Spautz, H., and N. Nur. 2008b. Suisun Song Sparrow, *Melospiza melodia maxillaris*. pp. 405-411 in *California Bird Species of Special Concern.* W. D. Shuford and T. Gardali, Eds. Studies of Western Birds 1. Camarillo and Sacramento, California.

Stralberg, D., D. L. Applegate, S. J. Phillips, M. P. Herzog, N. Nur, and N. Warnock. 2009. Optimizing wetland restoration and management for avian communities using a mixed integer programming approach. Biological Conservation 142:94-109.

Stralberg, D., M. P. Herzog, N. Nur, K. A. Tuxen, and M. Kelly. 2010. Predicting avian abundance within and across tidal marshes using fine-scale vegetation and geomorphic metrics. Wetlands 30:475-487.

Takekawa, J.Y., I. Woo, H. Spautz, N. Nur, J. L. Grenier, K. Malamud-Roam, J. C. Nordby, A. N. Cohen, F. Malamud-Roam, and S. E. Wainwright-De La Cruz. 2006. Environmental threats to tidal-marsh vertebrates of the San Francisco Bay estuary. Studies in Avian Biology 32:176-197.

Takekawa, J. Y. 2002. Data collection protocol: Waterfowl. Pp. 42-48 **in** *San Francisco Estuary Wetlands Regional Monitoring Program Plan: Data Collection Protocol: Wetland Bird Monitoring*. http://www.wrmp.org/protocols.html. Accessed January, 2011.

Thomas, L., S. T. Buckland,E. A. Rexstad,J.L. Laake,S. Strindberg,S. L. Hedley, J. R.B. Bishop, T. A. Marques, K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5-14.

U.S. Fish and Wildlife Service. 1998. The North American Waterfowl Management Plan. Expanding the Vision. 1998 Update. Accessed January, 2011. http://www.fws.gov/birdhabitat/NAWMP/files/NAWMP1998.pdf

Wittenberger, J. F. and G. L. Hunt, Jr. 1985. The adaptive significance of coloniality in birds. Avian Biology 8:1–78.

State of San Francisco Bay 2011
Appendix H

ECOLOGICAL PROCESSES – Flood Events Indicator
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background and Rationale

The San Francisco Estuary receives more than 90% of its freshwater inflow from the California's two largest rivers, the Sacramento River flowing from the north and the San Joaquin River from the south (Kimmerer 2002). Following winter rainstorms and during the height of the spring snowmelt in this vast watershed, the estuary's tributary rivers may flood, spilling over their banks to create ecologically important floodplain habitat and sending high flows of fresh water into the estuary. These seasonal high flows transport organisms, sediment, and nutrients to the Bay, increase mixing of Bay waters and create productive brackish, or "low-salinity" water habitat in the Bay's upstream Suisun and San Pablo regions, conditions favorable for many native fish and invertebrate species (Kimmerer 2002). High flows, as well as rapid increases in flows, are also important triggers for reproduction and movement for many estuarine fishes and for anadromous species like salmon that migrate between the ocean and rivers through the estuary. Just as high flows into the Bay create large areas of low salinity habitat (see Estuarine Open Water Habitat indicator), they also improve habitat conditions in riverine migration corridors for both adult fish moving upstream as well as young fish moving downstream.

In the Bay's Sacramento-San Joaquin watershed, several factors have had and are having substantial impacts on the frequency, magnitude and durations of high flow events into the estuary. First, flows in most of the Bay's largest tributary rivers have been greatly altered by dams (see Freshwater Inflow Index). Many of these dams were built for the purpose of reducing flood events and to store the mountain runoff for later use and export to other regions in the state. However, these upstream water management operations have deprived the estuary and its tributary rivers of an important physical and ecological process, regular seasonal flooding, that we now know is an essential component of the health of the estuary, its watershed and the plants and animals that depend on these habitats. Further, by physically blocking the flow of sediment, these dams are also starving riverine and estuarine wetlands and marshes of the materials they need to sustain (and restore) themselves. Second, the effects of climate change on flows in the watershed are already detectable and are predicted to increase. With warmer temperatures, increasing proportions of the precipitation in the watershed comes as rain, which runs off immediately, rather than snow, which melts later in the season. In the rivers and the Bay, the result is more frequent but shorter duration high flow events earlier in the year driven by rain runoff rather than the long duration spring snowmelt flood flows of the past. Third, large amounts of water are extracted from the rivers and the Delta upstream of the Bay. Collectively, these diversions can remove large percentages of the total flow, even during of relatively high flow (see Freshwater Inflow Index). This reduces the amounts of fresh water that flow into the estuary and can decrease inflows to levels below important threshold for habitat creation and sediment transport.

The Flood Events indicator uses three measurements to assess the frequency, magnitude and duration of flood events and high inflows to the estuary.

II. Data Source

The Flood Events indicator was calculated for each year using daily freshwater inflow data (referred to as "Delta outflow") from the California Department of Water Resources (CDWR) DAYFLOW model. DAYFLOW is a computer model developed in 1978 as an accounting tool for calculating historical Delta outflow, X2 and other internal Delta flows.[1] DAYFLOW output is used extensively in studies by State and federal agencies, universities, and consultants. DAYFLOW output is available for the period 1930-2010.

III. Methods and Calculations

The Flood Events indicator uses three measurements to assess the frequency, magnitude and duration of the occurrence of high inflow, or flood events, in the San Francisco Estuary each year.

For each year, frequency was measured as:

> # of years in the past decade (i.e., ending with the measurement year) with freshwater inflows >50,000 cubic feet per second (cfs)[2] for more than 90 days during the year.

For each year, magnitude was measured as:

> average inflow (cfs) during the 90 days of highest inflow in the year.

For each year, duration was measured as:

> # days during the 90 days of highest inflow that inflow>50,000 cfs.

For each year, the Flood Events indicator was calculated by combining the results of the three measurements into a single number by calculating the average of the measurement "scores" described in the Indicator Evaluation and Reference Conditions section below.

VI. Indicator Evaluation and Reference Conditions

The San Francisco Estuary Partnership's Comprehensive Conservation and Management Plan's (CCMP) goals for "increase[ing] freshwater availability to the estuary", "restor[ing] healthy estuarine habitat" and "promot[ing] restoration and enhancement of stream and wetland functions to enhance resiliency and reduce pollution in the Estuary" are non-quantitative. However, examination of historical flow and flood data records provide useful information for establishing ecologically relevant threshold levels and reference conditions for flood frequency, magnitude and duration.

Selection of the inflow threshold for a flood events, defined as the 5-day running average of freshwater inflow>50,000 cfs, was based on three rationales: 1) examination of DAYFLOW data suggested that flows above this threshold corresponded to winter rainfall events as well as some periods during the more prolonged spring snowmelt; 2) examination of DAYFLOW data also suggested that this inflow level corresponded to substantial inflows to the Delta from the Yolo Bypass, the main flood management overflow channel for the estuary's largest tributary, the

[1] More information about DAYFLOW is available at http://www.water.ca.gov/dayflow

[2] Freshwater inflow levels were measured as the 5-day running average of "Delta outflow."

Sacramento River; and 3) flows of this magnitude shift the location of low salinity habitat[3] downstream into Suisun and upper San Pablo Bays (depending on antecedent conditions), providing favorable conditions for many estuarine invertebrate and fish species. Examination of "pre-dam" flow data (1930-1943, before major storage and flood control dams were constructed on the estuary's main tributary rivers in the Sacramento-San Joaquin watershed), indicated that flood flow conditions occurred in 57% of years with median durations of 95 days per year of flows>50,000 cfs. Therefore, the frequency reference condition was set at five years out of 10 years (50%) and the duration reference condition at 90 days per year. Measured values that were above these reference conditions were interpreted to correspond to "good" conditions. An additional "lower" reference condition was established to denote "poor" conditions. Measured values that were between the two reference conditions were interpreted to correspond to "fair' conditions. Table 1 shows the reference conditions and associated interpretations for the indicator metrics.

Results of indicator and its component measurements were analyzed using analysis of variance and simple linear regression to identify differences among different time periods and trends with time.

V. Results

Results of the three component measurements of the Flood Events indicator are shown in Figure 1.

The frequency of occurrence of flood events has declined (Figure 1, top panel).
Frequency of occurrence of high inflow flood events in the San Francisco Estuary has declined significantly (regression, $p<0.001$). The first major decline occurred during the 1940s and 1950s, coincident with completion of large storage and flood control dams on the estuary's largest rivers, with frequency falling from an average of 5.8 years out of 10 years with floods in the 1940s (1939-1949) to an average of 1.7 flood years per decade in the 1950s and 1960s. Frequency declined again in the 1970s, 1980s and early 1990s, dropping to an average of just 1.3 flood years per decade (1970-1994). Frequency increased slightly during the late 1990s, concurrent with an unusually wet sequence of years, but then declined again in the 2000s. For the past three decades, flood frequency conditions have been consistently "poor." In the decade ending in 2010, the estuary experienced only one year (2006) with a flood event.

Flood magnitude has not changed (Figure 1, middle panel).
Flood magnitude, as measured by average inflows during the 90 days with highest inflows per year, is highly variable and, over the 81-year data record, it has not changed significantly (regression, $p>0.5$). High inflows during the "pre-dam" period (1930-1943) were, on average, 80,361 cfs compared to 64,697 cfs during the last two decades and not significantly different (Mann-Whitney Rank Sum test, $p>0.24$). High inflows during the most recent decade are somewhat lower, 48,003 cfs on average, bit not significantly different (t-test, $p>0.1$)

The duration of flood events has not changed (Figure 1, bottom panel).

[3] The location of low salinity habitat in the San Francisco Estuary is often expressed in terms of X2, the distance in km from the Golden Gate to the 2 ppt isohaline.

The number of days per year with inflows above the flood threshold is also highly variable and, over the 81-year data record has not changed significantly (regression, $p>0.15$). However, compared to the "pre-dam" period (1930-1943), which had an average of 82 days per year of inflows above the flood threshold, floods during the most recent decade (2001-2010) are significantly shorter, at just 27 days per year (t-test, $p<0.05$). In 2010, a year with median hydrological conditions, there were only 9 days with inflows >50,000 cfs.

Results of the Flood Events indicator are shown in Figure 2.

High inflow flood event conditions have declined.
Results of the indicator reveal a steady and significant decline in high inflow flood event conditions (regression, $p<0.001$), from a roughly equal mix of "good," "fair" and "poor" conditions prior to the 1960s to mostly "fair" and "poor" conditions by the 1980s. Conditions improved during the late 1990s, during a sequence of unusually wet years but declined again in the 2000s. Since 2001, conditions have been "poor" in all years except 2006, the 6th wettest year in the 81-year data record. Declining flood event conditions were driven almost entirely by the significant drop on flood frequency, which has fallen more than 75%.

Based on the Flood Events indicator, CCMP goals to restore healthy estuarine habitat and function have not been met.
The indicator shows that, for the past five decades, flood event conditions, an important physical and ecological process in the estuary, have been mostly "fair" or "poor." Since the early 1990s, when the CCMP was implemented, flood conditions have never been "good" and have been "poor" in 67% of years.

VI. Peer Review

The Flood Events indicator builds upon the methods and indicators developed by The Bay Institute for the 2003 and 2005 Ecological Scorecard San Francisco Bay Index and for the San Francisco Estuary Partnership Indicators Consortium. The Bay Institute's Ecological Scorecard was developed with input and review by an expert panel that included Bruce Herbold (US EPA), James Karr (University of Washington, Seattle), Matt Kondolf (University of California, Berkeley), Pater Moyle (University of California, Davis), Fred Nichols (US Geological Survey, ret.), and Phillip Williams (Phillip B. Williams and Assoc.). The versions of the Flood Events indicator presented in this document was reviewed and revised according to the comments of Bruce Herbold and Peter Vorster (The Bay Institute).

VII. References

Kimmerer, W. J. 2002. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25:1275-1290.

Table 1. Quantitative reference conditions and associated interpretations for results of the three Flood Events indicator metrics. The primary reference condition, which corresponds to “good” conditions, is in bold.

Flood Events Indicator			
Metric	**“Good”** **Score=3**	“Fair” Score=2	“Poor” Score=1
Frequency	**>5 years out of 10 years**	>2 years out of 10 years	≤2 years out of 10 years
Magnitude	**Inflow>100,000 cfs**	Inflow>50,000 cfs	Inflow≤50,000 cfs
Duration	**>90 days**	>45 days	≤45 days

Figure 2. Changes in the frequency, magnitude and duration of occurrence of flood events in the San Francisco Estuary, from 1930-2010. Black lines and symbols show the annual Index values, solid red line shows the 10-year running average for the Index. Horizontal dashed lines shows the reference conditions and associated interpretations.

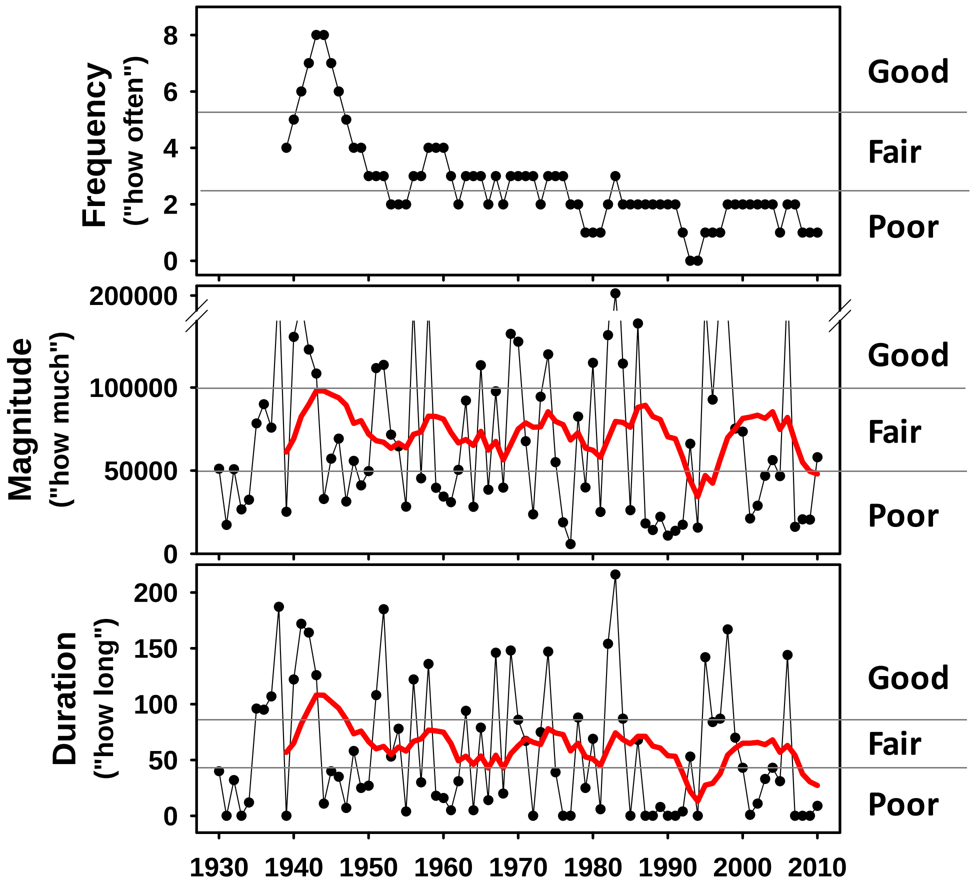
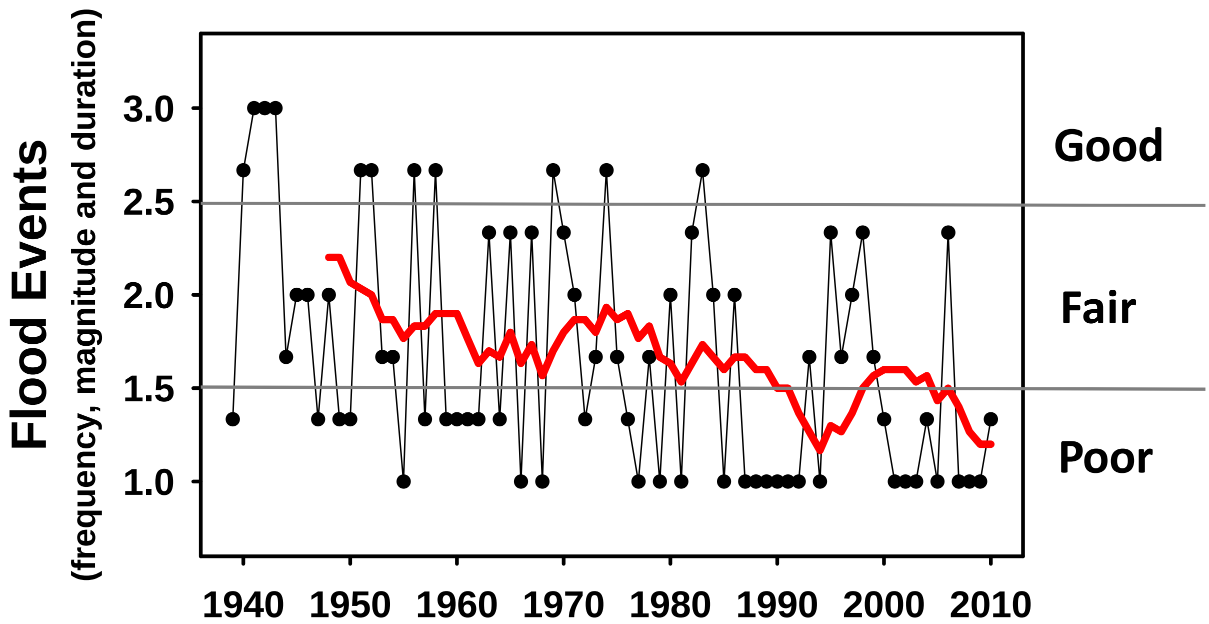


Figure 3. Changes in the Flood Events indicator from 1939-2010. Black lines and symbols show the annual indicator values, solid red line shows the 10-year running average for the indicator. Horizontal dashed lines shows the reference conditions and associated interpretations.



State of San Francisco Bay 2011
Appendix I

ECOLOGICAL PROCESSES – Food Web
Technical Appendix

Prepared by John Kelly and Nadav Nur, PRBO Conservation Science

Heron and Egret Brood Size

Background and Rationale:
Audubon Canyon Ranch has monitored brood size, prior to fledging, in Great Blue Heron and Great Egret nests across all known nesting colonies (40-50 sites) in the northern San Francisco Estuary, annually, since 1991. The number of young produced in successful heron and egret nests depends on the number of young hatched in the nest and the extent of subsequent brood reduction (i.e., mortality of nestlings during the brood-rearing period). Both parameters (young hatched per nest and survival of those young), reflect the amount of suitable foraging habitat and/or supply or availability of prey, in surrounding wetlands, especially that which is needed to provision nestlings with food (Frederick 2002, Kushlan and Hancock 2005). The Heron and Egret Brood Size Indicator is sensitive to changes in the extent and quality of foraging habitat, and is likely to be influenced by changes in land-use, hydrology (especially water circulation and depth), geomorphology, environmental contamination, vegetation characteristics, and the availability of suitable prey (Kushlan 2000). The two target species reflect differences in feeding habitat preference: Great Egrets preferentially forage in small ponds in emergent wetlands and areas with shallow, fluctuating water depths for foraging. In contrast, Great Blue Herons forage along the edges of larger bodies of water and creeks and are less sensitive to water depth (Custer and Galli 2002, Gawlik 2002). Previous work in the northern San Francisco Estuary demonstrated that prefledging brood size in herons and egrets is influenced by the extent of wetland habitat types as far as 10 km from nest sites (Kelly et al. 2008). Thus, this indicator reflects wetland condition over large spatial scales. The conspicuousness of heron and egret nesting colonies and the visibility of nests and broods—especially when nestlings are too young to leave the nests but old enough to have survived the period when most brood size reduction occurs—facilitates the use of brood size as an effective index of breeding productivity.

Data Source:
The Heron and Egret Brood Size Indicator was calculated using data from ongoing regional heron and egret studies by Audubon Canyon Ranch (Kelly et al. 1993, 2007). The data, which reflect brood size in successful nests at all known colony sites, provide an effective index of regional and subregional heron and egret productivity.

Methods and Calculations:
The Heron and Egret Brood Size Indicator includes metrics calculated for Great Egrets and Great Blue Herons. It is based on the number of young in completely visible nests when Great Blue Heron nestlings are known to be 5-8 weeks old and Great Egrets are known to be 5-7 weeks old (Pratt 1970, Pratt and Winkler 1985). The indicator measures changes or differences in brood size prior to fledging among nests that successfully fledge one or more young. Brood size counts are sampled in approximate proportion to colony size and averaged annually (1991-2008) among nests within and across the three major subregions of northern San Francisco Bay (Central San Francisco Bay, San Pablo Bay, and Suisun Bay). Brood size estimates are based on observations at most of the 40-50 colony sites within foraging range (i.e., 10 km) of the historic tidal wetland boundary (ca.1770–1820; San Francisco Estuary Institute 1999; Figure H1). The Brood Size Indicator is calculated by first determining for each species and each region the proportional change between the year in question and the benchmark value (five-year reference period) for that species. Then the geometric mean across species was calculated and, finally, this was

converted to percent change. The species-specific benchmark value was derived from the 1991-1995 data (Great Blue Heron: 2.01±0.088 young; Great Egret: 2.26±0.107 young, weighted equally across years).

Goals, Targets and Reference Conditions:

CCMP goals to "restore" and "enhance" the ecological productivity and habitat values of wetlands are non-quantitative. However, the use of time series back to 1991 allows the specification of appropriate quantitative reference conditions. Differences or trends in nest density can be quantified and used for assessment.

<u>Maintenance of current resource levels</u>

- Target: current 3-year mean (2006-2008) ≥ 5-year reference mean (1991-1995).

<u>Enhancement of resources with wetland restoration</u>

- Target: current 3-year mean (2006-2008) ≥ highest 5-year *subregional* reference mean (1991-1995)

Results:

Results of the Heron and Egret Brood Size Indicator are shown in Figure H4.

Current brood sizes (2006-2008) declined from reference levels (1991-1995).

Brood sizes in the northern San Francisco Estuary were significantly lower in 2006-2008, relative to 1991-1995 reference levels (t_{745} = -9.9, P < 0.001; Figure H4), with 8.4% and 17.1% fewer young produced in successful Great Blue Heron and Great Egret nests, respectively. Therefore, the proposed target associated with overall resource enhancement was also not achieved: regional productivity per nest was significantly less (t_{570} = 5.1, P < 0.001) than the highest subregional 1991-1995 level (Suisun Bay, 4.6% above regional reference value).

Changes in brood size differ among subregions.

During the 1991-1995 reference period, brood sizes were significantly smaller in San Pablo Bay than in other subregions (multiple comparisons, P < 0.001). In recent years (2006-2008), the Brood Size Indicator revealed significantly smaller broods in Suisun Bay than in other subregions (P < 0.02), suggesting a shift in relative per capita productivity among subregions (Figure H4). In addition, brood sizes in Suisun Bay in 2006-2008 were significantly smaller than the regional 1991-1995 average (t_{353} = -8.3, P < 0.001), with nests producing 14% fewer Great Blue Heron young and 19% fewer Great Egret young. The productivity of nests in San Pablo Bay in the recent years was also significantly lower than in the reference period (t_{273} = -3.7, P < 0.001), with average declines of 5.2% in Great Blue Herons and 10.5% in Great Egrets. In the Central Bay, the productivity of Great Egret nests declined by 13.8% (t_{56} = 3.8, P < 0.001) relative to reference levels, but the productivity in Great Blue Heron nests in recent years was similar to brood size during the reference period (P > 0.05).

Based on brood size estimates for Great Blue Heron and Great Egret, CCMP goals of restoring or enhancing wetland productivity and associated wetland habitat values have not been met in the region or within any subregion.

Recent productivity in successful nests of both species declined by 8-17% relative to the 1991-1995 reference period, with declines generally observed across the subregions. Subregional differences in productivity suggest opportunities for habitat restoration or enhancement, especially in Suisun Bay.

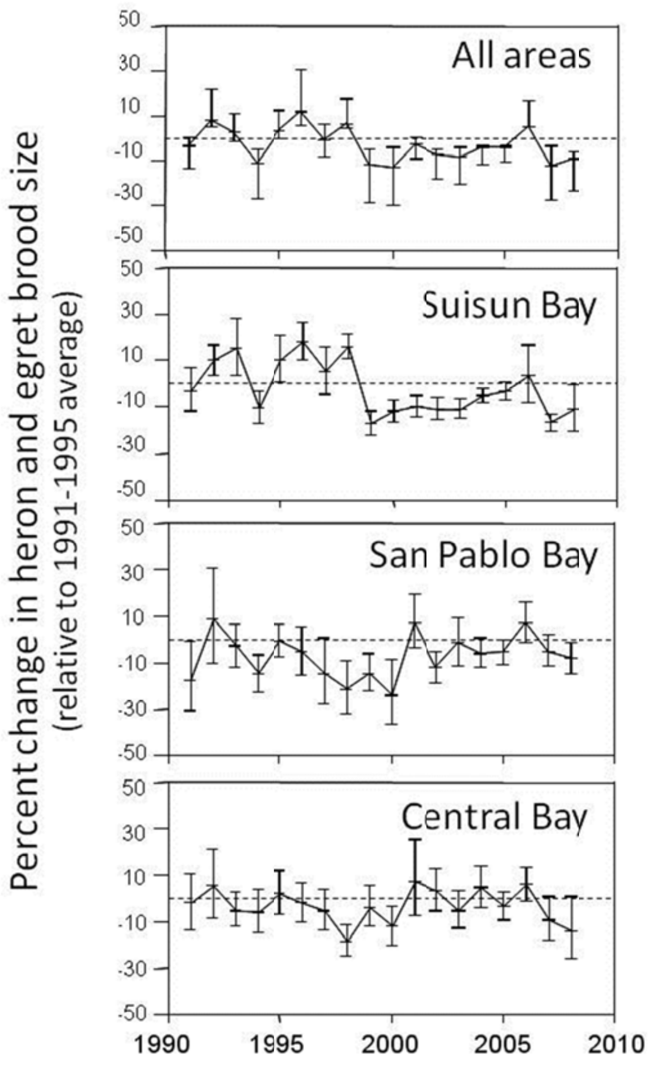


Figure H4. Annual percent change in heron and egret brood size, 1991-2008, relative to the average brood size (dashed line) for the reference period, 1991-1995, in the northern San Francisco Estuary, by bay region and for all 3 regions combined.

Peer Review: The above indicator was evaluated using methods and analysis described in Kelly *et al*. (2007).

References

Custer, C. M., and J. Galli. 2002. Feeding habitat selection by Great Blue Herons and Great Egrets nesting in east central Minnesota. Waterbirds 25:115–124.

Frederick, P. C. 2002. Wading birds in the marine environment. Pp. 617–55. In E. A. Schreiber and J. Burger (eds.) Biology of Marine Birds. CRC Press, Boca Raton, FL, USA.

Gawlik, D. E. 2002. The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72:329–346.

Kelly, J. P., H. M. Pratt, and P. L. Greene. 1993. The distribution, reproductive success, and habitat characteristics of heron and egret breeding colonies in the San Francisco Bay area. Colonial Waterbirds 16:18–27.

Kelly, J. P., K. Etienne, C. Strong, M. McCaustland, and M. L. Parkes. 2007. Status, trends, and implications for the conservation of heron and egret nesting colonies in the San Francisco Bay area. Waterbirds 30: 455-478.

Kelly, J. P., D. Stralberg, K. Etienne, and M. McCaustland. 2008. Landscape influences on the quality of heron and egret colony sites. Wetlands 28: 257-275.

Kushlan, J. A. 2000. Heron feeding habitat conservation. Pp. 219-235, in J. A. Kushlan and H. Hafner (eds.), Heron Conservation. Academic Press, San Diego, CA.

Kushlan, J. A. and J. A. Hancock. 2005. The Herons. Oxford University Press, New York, NY, USA.

Pratt, H. M. 1970. Breeding biology of Great Blue Herons and Common Egrets in central California. Condor 72: 407-416.

Pratt, H.M. and D.W. Winkler. 1985. Clutch size, timing of laying, and reproductive success in a colony of Great Blue Herons and Great Egrets. Auk 102: 49 – 63.

State of San Francisco Bay 2011
Appendix J

STEWARDSHIP – Urban Water Use
Technical Appendix

Peter Vorster, The Bay Institute

Background and rationale

Urban or municipal water use by residential, commercial, industrial and institutional (schools, government) customers accounts for about 90 percent of the 1.1 million acre-feet total annual water use in the Bay Area.[1] About 85% of the urban use is supplied from watersheds outside of the Bay-draining watersheds, predominantly from the Delta watershed with smaller amounts from the watersheds of the Russian River and Tomales Bay. The local bay-draining watersheds provide both surface water and groundwater to urban users in the Santa Clara Valley, the Alameda Creek watershed and the North Bay. Groundwater supplies about 15% of the urban uses although some of the groundwater is derived from the imported supply that is used to recharge the local groundwater basins. Groundwater is also the primary supply for water used by agriculture in the Bay Area.

Good stewardship of our water in the Bay Area means using it more efficiently by reducing the amount of water needed for any goal while still accomplishing that goal. More efficient water use has many actual and potential benefits for the Bay Area including:

- Reduces dependence on ecologically harmful water diversions from rivers and streams;
- Reduces the financial and energy costs of water and wastewater treatment, pumping, transporting and storing water supplies
- Relieves competition for limited supplies and the need to develop new supplies;
- Reduces pollutant loads from irrigating lawns, gardens and crops;
- Reduces the vulnerability of our supplies to disruption by earthquakes, droughts, floods, rising sea level, and regulatory requirements to protect endangered species.
- Qualifies water agencies for State grant funding if water use efficiency targets are met

In short, more efficient water use can reduce the human "footprint" on the natural water balance.

This indicator assesses municipal water use in the Bay Area by determining if the region is using more or less water over time and by determining whether we are getting more or less efficient by calculating the per person (per capita) use of water. Both total municipal water use and the use by just the residential customers are evaluated. Residential use, which consists of what is used in single family and multi-family residences for indoor uses (waste elimination, washing clothes and dishes, bathing, drinking) and outdoor uses (irrigation and cleaning), is the factor most directly controlled by individuals and families, whose decisions to use water more efficiently in and around the home can collectively create large-scale benefits. Institutional (primarily schools and governments) and commercial users can have both an indoor and outdoor component, depending on the nature of the business while industrial users are primarily using water indoors for a manufacturing process including energy generation. The water use that is measured by this indicator is the direct consumption of the water used inside and outside of the homes, businesses, and factories in the Bay Area but it does not measure our total water footprint, which is the volume of water that is required to produce all the goods and services that we consume and is many times greater than our direct consumption.[2]

---

[1] The terms "urban" and "municipal" water use are used interchangeably and refers to the use by communities that are supplied by public water districts and private water companies in contrast to the rural areas that are primarily self-supplied with groundwater.

[2] The average yearly water footprint of an American is about 655,000 gallons per year or about 18 times greater than the 36500 gallons per year or the roughly 100 gallons per day the average Bay Area resident consumes through the water supply system. Water footprints of all nations for the period 1997 - 2001 have been first reported Chapagain, A.K. and Hoekstra, A.Y. "Water footprints of nations". *Value of Water Research Report Series No. 16* (UNESCO-IHE)

Residential per capita use can be used to compare water use within and across watershed boundaries or among water agencies.[3] Per capita use derived from the total municipal use measures, along with the residential use, different proportions of commercial, industrial, and institutional uses by the different municipalities and thus make the comparisons across boundaries and what we as individuals use less accurate. The total municipal per-capita use for the Bay Area is a reasonable indicator of how the region as a whole is managing its water supplies over time and is also the metric that is used to assess compliance with the recently passed State legislation that establishes urban water use targets.

Data sources

This indicator requires measurements of total municipal and residential water use and population[4]. Data on water use and population for the period 1986-2009 was compiled in order to evaluate how the Bay Area urban use is affected over time by climate, plumbing codes, conservation measures and economic conditions. 1986 is just prior to the 1987-92 drought, the longest drought experienced by Bay Area municipalities. Major plumbing code changes were also instituted in the early 1990's. More recently the Bay Area experienced a 3-year dry period and economic downturn that also affected water use. Data for 2010 had not been reported for many of the Bay Area agencies when this was compiled in the spring of 2011.

All of the Bay Area municipal water agencies measure the water use of their customers in order to bill them based upon the volume of use but the reporting of that data is not always consistent and available since 1986. Municipal water use by retail water agencies is separated into different sectors or types of use, often distinguished by the size and type of water meter. Residential water use is accounted for separately from commercial, industrial, institutional and dedicated landscaping use. Residential customers are usually separated into single family and multi-family accounts and must be combined to derive the total residential use. Except for the residential water use compiled for the Santa Clara Valley Water District, a wholesaler of water in the South Bay, the residential water use is derived from the retail agency deliveries of water to their residential customers.[5]

Total municipal and residential water use and population data for the 1986-2009 period were compiled for Contra Costa Water District (CCWD), East Bay Municipal Utilities District (EBMUD), Alameda County Water District (ACWD), San Francisco Public Utilities District (SFPUC), Zone 7 Water Agency (Zone 7), Santa Clara Valley Water District (SCVWD), Bay Area Water Supply and Conservation Agencies (BAWSCA- an association of the water agencies that wholesale water from the SFPUC), Marin Municipal Water District (MMWD), and the City of Napa (Napa), which together serve about 93% of the 6.6 million people that reside in the municipalities in the local Bay-draining watersheds. Table 1 lists the agencies, the type of service provided (wholesale or retail or both), the geographic region served, population, and the sources of water.

Data was obtained either directly from the water agencies (either from data sent to me or obtained from their Urban Water Management Plans), or from a compilation done for the Bay Area Water Agencies Coalition (BAWAC- a coalition of the major Bay Area water agencies) or from the Department of

---

[3] This assumes that the agencies are defining the single family and multi-family residential customer class similarly, which is generally true although some agencies separate mobile home parks and dedicated landscaping meters at multi-family complexes.

[4] The volume of water use shown for this indicator is in acre-feet per year. An acre-foot is equal to 325,851 gallons. The data is reported by the water agencies in gallons or cubic feet or occasionally acre-feet.

[5] The data obtained from the Santa Clara Valley Water District assumes the residential water use in their retail districts is 52% of the total water use in every year.

Water Resources Public Water System Survey (DWR PWSS).[6] Municipalities and areas not included because data back to 1986 was not available include Novato, Petaluma, Sonoma Valley, Napa Valley not including City of Napa, Vallejo, American Canyon, Benicia, Fairfield, and Suisun City; these areas have about 450,000 people.

In addition to the DWR PWSS, the California Urban Water Conservation Council (CUWCC) also collects water use data from their member agencies, including most of the Bay Area retailers and wholesalers but their database was not functioning at the time of this compilation in the spring of 2011. The water use data the agencies report to the DWR PWSS and the CUWCC is not always consistent with the same data contained in agency reports including Urban Water Management Plans, thus where possible data was obtained by direct inquiry to the water agencies.

Methods and calculations

The average daily water use per person – gallons per capita per day (gpcd) – is calculated by converting the reported monthly, bi-monthly or annual residential water use data into gallons, dividing by the appropriate number of days to get a daily use and then dividing that result by the population using that water to get the gpcd. It is assumed for purposes of this calculation that only the population reported to reside within the service area of the district consumes the residential water and that visitors to the area are consuming water from non-residential accounts (i.e. commercial or institutional accounts).[7]

Goals, targets, and reference conditions

As noted above, in order to evaluate how the Bay Area urban use is affected over time by climate, plumbing codes, conservation measures and economic conditions, water use was assessed beginning in 1986. 1986 is just prior to the 1987-92 drought, the longest drought experienced by Bay Area municipalities and prior to major plumbing code changes instituted in the early 1990's and is used as a reference condition in this assessment from which to measure changes in total water use, population, and per-capita use.

The Water Conservation Act of 2009, Senate Billx7-7 (2009 Act) established a goal of reducing urban per-capita water use by 20% by 2020 with an interim goal of a 10% per-capita reduction by 2015. This first legislatively-proscribed urban water use target in California provides that targets can be calculated by one of four methods. Method 3 target is ninety-five percent of the applicable hydrologic region target derived from the State's April 30, 2009, draft 20x2020 Water Conservation Plan.[8] The calculated target for the San Francisco Bay region in 2020 is 124 gpcd (95% of 131 gpcd), which can be used to assess progress for the region, although this indicator is not meant to be used to determine 2009 Act compliance. A water agency can choose the method to establish its target, which is described in Methodologies for Calculating Baseline and Compliance Urban Per Capita Water Use, Feb 2011, available on the DWR web site http://www.water.ca.gov/wateruseefficiency/sb7/ established for tracking the implementation of the legislation.

[6] Because the Santa Clara Valley Water District did not compute residential water use data for 2008 and 2009, additional data for those years had to be compiled for water agencies in Santa Clara Valley that do not receive SFPUC water and thus whose data was unavailable from BAWSCA. These agencies include the San Jose Water Company, Great Oaks Water Company, California Water Service Agency- Los Altos, and San Jose Municipal Water Company- Evergreen.

[7] It is possible that some of the visitors using the water in the municipalities are using residential water (e.g. bed and breakfasts) but that there is no way of determining that for this project. If visitors are using residential water in significant quantities then the gpcd will be somewhat higher.

[8] The 20 by 2020 Water Conservation Plan follows from the 2008 governor's executive order requiring state agencies to develop a plan to reduce statewide per capita urban water use by 20 percent by the year 2020.

The Bay Institute's 2003 Ecological Scorecard (TBI 2003) concluded that residential use – particularly indoor use -- lends itself to establishing per-capita benchmarks and for comparative evaluation within the region since per-capita indoor use should be roughly the same throughout the region while outdoor use varies primarily due to lot size and climate. Differences in total urban use are a function of the mix of residential and non-residential uses and the different types of commercial and industrial uses and thus is harder to establish comparative targets for efficient use. Based upon a review of several end-use studies, TBI (2003) uses an indoor use target of 40 gpcd in its scoring of residential water use.

Results

Table 2 and Figures 1 and 2 document the fluctuation and eventual overall decline in total water use and per-capita use in the San Francisco Bay region in the 1986-2009 period. Total urban water use in the Bay Area is 20 percent less today than it was 25 years ago, a remarkable achievement given that the population has increased by 20 percent. This is primarily a result of greater efficiency of use, combined more recently with a dampening of water demand due to the economic downturn. The increased efficiency has been achieved through mandates for more efficient water-using appliances, and by Bay Area residents and businesses reducing their use in response to requests for conservation during the recent 2007 to 2009 dry period. Residential use did not decline as much - only 10% in the 1986-2009 period- reflecting the fact that residential growth in the region has been greater in the hotter inland areas with greater per-capita use. Commercial and industrial water use has also declined proportionally more due to the shrinkage of manufacturing and industry as well as economic incentives to decrease potable water use, including the greater use of recycled water.

Although data for the entire Bay Area is only available through 2009, data from selected suppliers for 2010 and 2011 indicates that usage is continuing its downward trend as cooler and wetter springtime weather in those years reduced demand. The outdoor water use, which can represent up to half of the total annual use, is sensitive to the variations in precipitation and temperature, particularly in the spring and fall.

The total per capita use of 132 gpcd in 2009 shows that the Bay region has already made significant progress toward meeting the 2020 regional urban water use target of 124 gpcd with many of the individual agencies well below the regional target. Although a rebounding economy and years with less precipitation are factors that will likely increase urban water use at some point in the future, if recent per-capita usage can be maintained or improved, the mandate for a 20 percent reduction should be easily achieved by 2020.

The change in water use and per-capita use for the individual agencies is shown in Table 3.
This table shows the considerable geographic variation in the water use and the trends over time around the region. The variation in water use is largely explained by the climatic differences between the cooler Bay-side versus the warmer inland areas and residential lot size differences between the smaller lots in the older cities and larger lots in the newer suburbs; SFPUC and Zone 7 Alameda County represent the two extremes with a greater than two-fold difference in the total and per-capita water use.   Variations in water use is also reflective of the relative proportion of the different types of uses- residential versus non-residential uses and variations within the commercial and industrial sectors- in the region (e.g. Santa Clara and Contra Costa Counties have more water-using industry than Marin or Napa Counties).   The water use trends over time also reflect the relative growth patterns in the region in the past quarter century.   Residential growth has been proportionally much greater in the warmer inland areas of Eastern Alameda and Contra Costa Counties than in the inner Bay Area and is

reflected in the increase of residential water use in the water districts serving those areas. The per-capita total and residential use, however, has decreased in all areas.

The decrease in the regional urban water use has reduced the demand on imported supplies, but it has not translated into greater inflows to the Bay from the Delta (see flow indicator). Competition for the limited supplies from the Delta and it tributary watersheds means that other urban and agricultural users will divert any additional water from the reduced demand. If demand stays at these reduced levels due to continued conservation, reduced economic activity or wetter conditions, the water agencies will continue to experience declining revenues and thus water rates will have to be increased to balance revenues with costs. The challenge for the agencies is how to structure rates so that users are not penalized for using less. The potential for greater efficiency of use in the region- upwards of 15% or more, particularly in outdoor water use, is still high and therefore it is possible to accommodate population increases over the next decade without significant increases in total demand.

Table 1: Water Agencies in the San Francisco Bay Region

Agency	**Type**	**County / region served**	**Population in 2009**	**Primary sources of water**
Alameda County Water District (*ACWD*)	Retail	South Alameda	332,000	SWP, SFPUC, and ground water
Bay Area Water Supply and Conservation Agencies (*BAWSCA*)[9]	Association	San Mateo, north Santa Clara, south Alameda	1,719,028 (869,164)[10]	SFPUC, SWP, CVP, local surface and ground water
Contra Costa Water District (*CCWD*) (includes treated and wholesale service areas)	Retail and Wholesale	North, central, and east Contra Costa	457,544	CVP, and direct diversion from the Delta
East Bay Municipal Utility District (*EBMUD*)	Retail	North Alameda, north and central Contra Costa	1,400,000	Mokelumne River and local surface water
Marin Municipal Water District (*MMWD*)	Retail	South and central Marin	190,600	Lagunitas Creek, and Russian River surface water
San Francisco Public Utilities District (*SFPUC*)	Retail and Wholesale	San Francisco	848,601	Tuolumne River and local runoff in Alameda and San Mateo County
Santa Clara Valley Water District (*SCVWD*)	Wholesale	Santa Clara	1,822,000[11]	SFPUC, SWP, CVP, local surface and ground water
Zone 7 of the Alameda County Flood Control and Water Conservation District (*Zone 7*)	Wholesale	East Alameda	216,000	SWP, local surface and ground water
City of Napa	Retail	Napa	85,814	SWP, local surface water

[9] BAWSCA does not deliver water but is an association from the 29 cities, water districts and other agencies that purchase all or a portion of their water from the City and County of San Francisco (SFPUC) Hetch Hetchy water system.
[10] BAWSCA includes ACWD and agencies that are part of SCVWD. The bracketed number represents the population *excluding* those entities.
[11] 2010 population

Table 2- Total and Residential Water Use for the San Francisco Bay Region

Category	Year	Population served		Total Use (AF)		Residential Use (AF)	
			Population Served	Gallons per capita day (GPCD)	Total Water Use (AF)	Gallons per capita day (GPCD)	Residential Water Use (AF)
	1986		4,899,583	200	1,095,075	107	588,575
Drought Period	1987		4,960,101	201	1,115,781	106	588,479
	1988		5,026,287	187	1,054,355	97	545,169
	1989		5,100,478	166	947,070	90	514,009
	1990		5,165,134	170	981,503	89	514,094
	1991		5,195,112	148	859,548	77	449,694
	1992		5,246,028	149	876,048	82	482,040
	1993		5,313,206	153	908,995	86	513,663
	1994		5,354,939	160	957,448	89	531,421
	1995		5,382,104	160	961,710	90	541,861
	1996		5,436,714	167	1,016,822	94	572,703
	1997		5,507,039	173	1,068,384	97	600,421
	1998		5,581,163	162	1,011,697	90	562,702
	1999		5,650,259	168	1,065,797	94	594,227
	2000		5,730,602	171	1,096,438	95	610,171
	2001		5,799,308	169	1,101,009	96	625,916
	2002		5,854,638	167	1,097,317	95	623,551
	2003		5,890,232	162	1,068,050	93	615,995
	2004		5,961,281	163	1,091,353	93	617,870
	2005		5,995,794	155	1,043,993	88	589,487
	2006		6,047,734	155	1,047,702	88	593,850
Dry Period	2007		6,116,940	158	1,081,633	89	607,079
	2008		6,114,426	145	993,462	86	587,141
	2009		6,146,286	132	909,842	78	534,279
Percent Change (%)			**20%**	**51%**	**-20%**	**38%**	**-10%**
Annual Change (%)			**0.8%**	**-2.1%**	**-0.8%**	**-1.6%**	**-0.4%**

Table 3: Total and Residential Water Use for Individual Agencies in the San Francisco Bay Area

Agency	Population change since 1986	Water Use: Total (AF[12])	Water Use: Residential (AF)	Water Use: Resid. as % of total[13]	Change in total water use 1986-2009: Overall total % change	Change in total water use 1986-2009: Residential total % change	*Per capita* water use: Total (GPCD)	*Per capita* water use: Residen-tial (GPCD)	Change in *per capita* water use 1986-2009: Total GPCD % change	Change in *per capita* water use 1986-2009: Residential GPCD % change
Alameda County Water District (*ACWD*)	**+28**%	47,000	29,100	62%	**+4**%	**-4**%	126	78	**-34**%	**-44**%
Bay Area Water Supply and Conservation Agencies (*BAWSCA*)[14]	**+13**%	127,821	81,793	64%	**+2**%	**0**%	133	85	**-13**%	**-15**%
Contra Costa Water District (*CCWD*)	**+ 32**%	105,600	53,900	51%	**-26**%	**+11**%	206	105	**-87**%	**-32**%
East Bay Municipal Utility District (*EBMUD*)	**+21**%	208,325	118,744	57%	**-16**%	**-14**%	127	76	**-47**%	**-44**%
Marin Municipal Water District (*MMWD*)	**+12**%	25,982	16,905	65%	**-24**%	**-16**%	122	79	**-42**%	**-32**%
San Francisco Public Utilities District (*SFPUC*)	**+13**%	81,128	48,184	59%	**-39**%	**-16**%	86	51	**-60**%	**-33**%
Santa Clara Valley Water District (*SCVWD*)	**+22**%	352,000	175,500	52%	**-7**%	**-9**%	187	86	**-37**%	**-39**%
Zone 7 Alameda County (*Zone 7)*	**+48**%	46,785	29,753	64%	**+43**%	**+40**%	193	123	**-11**%	**-17**%
City of Napa	**+23**%	14,865	9,118	61%	**+13**%	**+12**%	155	95	**-13**%	**-15**%

[12] Note on units: AF = acre-feet (325,831 US Gal., or 1233.48 m^3); GPCD = gallons per person per day
[13] Residential water use as % of total water use not including any recycled water
[14] BAWSCA values exclude ACWD and agencies that are part of SCVWD

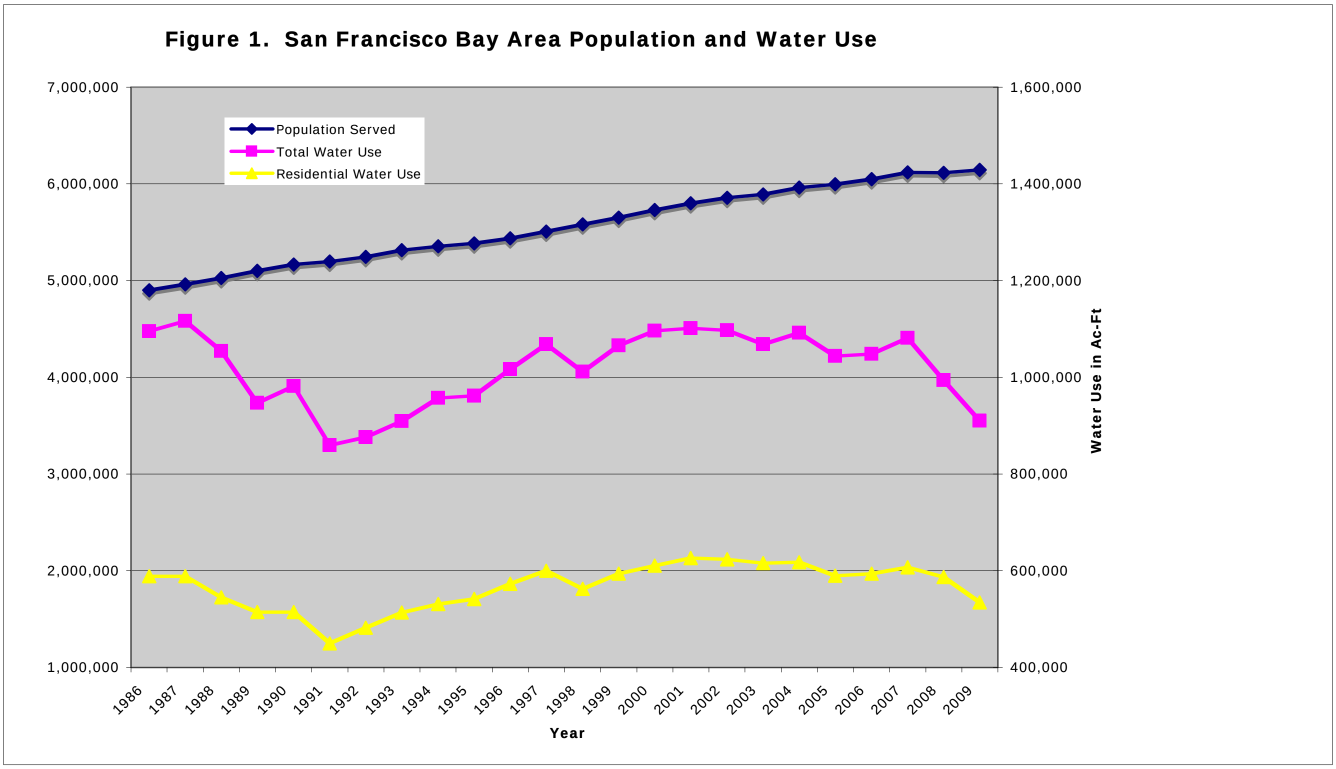


Figure 1. San Francisco Bay Area Population and Water Use
Population Served
Total Water Use
Residential Water Use
7,000,000
6,000,000
5,000,000
4,000,000
3,000,000
2,000,000
1,000,000
1,600,000
1,400,000
1,200,000
1,000,000
800,000
600,000
400,000
Water Use in Ac-Ft
1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009
Year

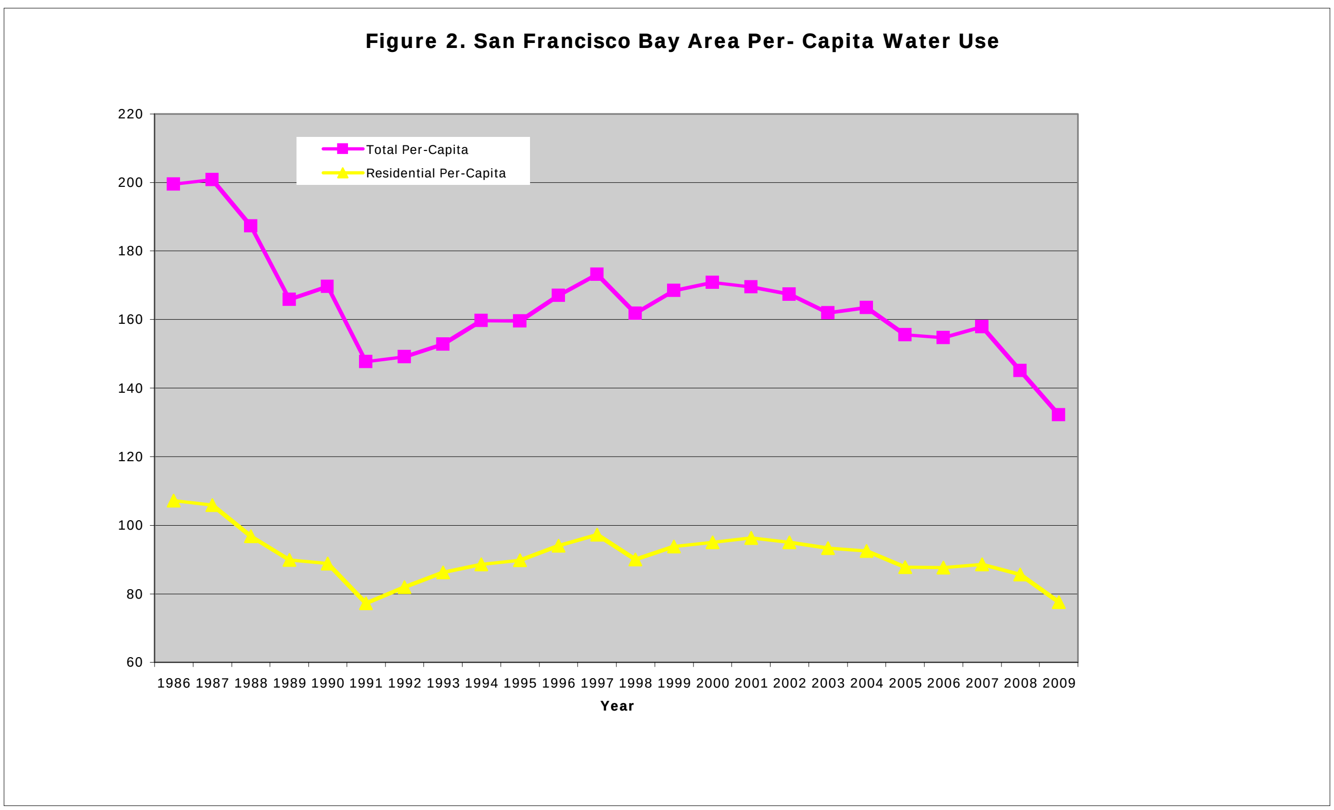


Figure 2. San Francisco Bay Area Per- Capita Water Use
Total Per-Capita
Residential Per-Capita
220
200
180
160
140
120
100
80
60
1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009
Year

State of San Francisco Bay 2011
Appendix K

STEWARDSHIP – Recycled Water Use Technical Appendix

Peter Vorster, The Bay Institute

TECHNICAL APPENDIX: RECYCLED WATER USE

Background and rationale

Most of the water that Bay Area communities consume is used once, treated, and discharged from 34 publicly owned treatment works (POTW) into the Bay and its tidal sloughs and streams (see Figure 1).[1] The Bay Area is fortunate to receive high quality surface water directly from relatively pristine Sierran watersheds and from the local watersheds draining the Coast Ranges. Supplies from the Delta are usually of good quality although they may have been "recycled" several times by upstream users before being pumped from the Delta. There has been a small amount of intentional recycling or reuse of the water from wastewater treatment plants for over 50 years, but the amount and uses of recycled water have grown substantially in the past several decades.[2]

In the Bay Area, recycled water from POTW's is used to irrigate landscapes, golf courses, and crops; for process water, including power plant and refinery cooling water and washdown water at commercial and industrial facilities; and to augment freshwater flow to wetlands. Proposed new uses of recycled water in the region include toilet flushing in commercial buildings, heating and cooling, and for groundwater recharge.

Water recycling demonstrates good stewardship because it uses the limited local and imported water supplies more efficiently, with the potential of reducing the need for new water diversions from the Bay's watershed. Compared to existing supplies, recycled water is much less sensitive to climate-induced supply variation and often consumes less energy than pumping water from the Delta or pumping groundwater (BACWA, 2007).[3] Water recycling supports the region's sustainability by providing a local and available source of water and because the wastewater is primarily discharged into or near the Bay, and not part of a downstream supply, it is a "new" source of water for the region and the State. Water recycling also reduces the amount of treated wastewater that is discharged into the Bay. In sum water recycling can meet multiple resource management and protection objectives.[4]

---

[1] Nearly all of the urban Bay Area is "sewered" and connected to publicly owned treatment works (POTW). The rural fringes of the Bay still rely on individual septic tanks or small facilities that discharge into groundwater. Some industrial users such as refineries, chemical companies, and shoreline businesses such as C&H Sugar discharge their wastewater directly into the Bay. Wastewater can be discharged into North Bay streams during the winter wet season when runoff is higher.

[2] Water recycling can refer to recycling of water from any source such as wastewater, irrigation water, gray water, or storm water but this analysis is specific to wastewater recycling from treatment plants. There is a very small but growing effort by residents and businesses to recycle greywater on-site to meet irrigation and plumbing needs.

[3] Energy consumption for recycled water depends on the distance and elevational difference of the end user and treatment plant.

[4] Recycling is not without its critics who note its high capital costs, water quality risks, and growth-inducing aspects.

Recycled water is quantified as either the recycled water produced at the POTW's, or the water supply that it replaces or creates. Recycled water that replaces water that otherwise would be delivered by a municipal supplier is considered a "potable offset." Recycled water can also be used in a way that does not offset potable water, such as for creating and enhancing freshwater marsh habitat at Hayward Marsh, Peyton Slough, Palo Alto Marsh and several North Bay streams.

Vineyards and dairies can also use recycled water from a POTW instead of pumping groundwater or withdrawing surface water from a nearby stream. A POTW may also treat its wastewater to recyclable standards but not have a market for the water and will apply it to formerly non-irrigated land to grow grass or forage crops instead of discharging it into the Bay. In all of these cases, the recycled water is providing a local water resource, expanding our region's available water portfolio, and providing economic, environmental or social benefits. For POTW's that normally discharge effluent to the Bay, any reuse will reduce the amount of that discharge.

Data sources

There is no consistent, reliable, and regularly reported data for recycled water use. State agencies including the State Water Resources Control Board, the Department of Water Resources and the Regional Water Quality Control Board periodically compile recycled water use but the data is not always consistent with data obtained directly from or reported by the wastewater recyclers and the water agencies that distribute the water. The data inconsistencies are due in part to the differing definitions of what is classified as recycled water with some agencies quantifying only the portion that offsets potable uses and other agencies quantifying all wastewater that is used for any beneficial use including in-plant use and land irrigation used for wastewater disposal. There is also not a consistent delineation between the different categories (commercial, industrial, irrigation) of recycled water use.

Multiple sources were consulted to obtain the most up-to-date information (generally 2010) on the quantity of recycled water used for the different types of end use including landscape irrigation, commercial, industrial, and agricultural water use as well as for wetland and wildlife habitat use (termed environmental enhancement in some delineations). The most reliable source of data is to obtain it directly from the wastewater treatment plant operator (e.g. Napa Sanitation District) or from the distributor or consumer of the water, either a water agency (e.g. SFPUC) or the direct consumer (e.g. turf farm). The most recent (2010) urban water management plans (UWMPs) were also consulted. The UWMP's reviewed in April or May 2011 were typically in a draft form, but included the mandated set of information on production and use of recycled water (Water Code Section 10633), and on projections for the recycled water production/use until the year 2035.[5] Lastly the "96-011" reports to the San Francisco Bay Regional Water Quality Control Board (Regional Board) were consulted where data for 2010 was available to fill in gaps and to corroborate previously obtained data.[6] The data

[5] Some of the UWMPs in the region were still being drafted and were unavailable.
[6] Order 96-011 from the San Francisco Bay Regional Board is the master recycled water permit for Region 2 (SF Bay) and is the name given to the reports that wastewater recyclers submit to the Regional Board.

compilation shown in Table 1 was derived from 18 contacts with treatment plant operators or water agencies, eight UWMPs, and two data points were derived from 96-011 reports.

Recycled water use from 2001 was compiled for The Bay Institute's 2003 Ecological Scorecard (TBI 2003). That data was obtained from the treatment plant operators and water agencies and from the 2001 State Water Board Recycled Water Survey. The 2001 data was reanalyzed for this report to insure consistency with the current assessment and thus includes data from the North Bay counties (Marin, Sonoma, Napa, Solano) whereas the 2003 report only included data from the five counties covered by the Bay Area Regional Water Recycling Program (San Francisco, San Mateo, Santa Clara, Alameda, Contra Costa).

Methods and calculations

In order to quantify the contribution by recycled water to reducing regional water demand, the recycled water volume used to offset (i.e., reduce by the same amount) the consumption of potable water or groundwater is distinguished from the recycled water used in ways for which potable water would not or was not being used. In the latter category ("non-offset recycled water") recycled water is quantified that is (a) directly discharged to a (restored) wetland or wildlife habitat (e.g., a duck pond), (b) has been used to irrigate previously dry-farmed land or irrigate pasture for forage and grazing. In addition where data was available, wastewater was quantified that was (c) "land applied" (instead of being discharged to a waterbody) but not providing any agricultural value, (d) used "internally" for the wastewater plant operations that would not normally replace potable water.[7] The latter two quantities are not included in the total recycled water use shown in Table 1 since they are not consistently quantified.

Goals, targets, and reference conditions

There are no standardized benchmarks or targets for assessing progress on recycled water use, however the following are approaches that have been used or could be used:

1. Comparing recycled water used to the total amount of water flowing into or out of wastewater treatment plants, usually expressed as a percentage. Regional Board records indicate that the 2009 inflow to the POTW's is nearly 615,000 ac-ft.
2. Comparing recycled water used to the amount of water treated to recyclable (Title 22) standards at the treatment plants (i.e. the amount potentially available for use). Depending on the end-use, recycled water often needs additional treatment above and beyond what is required for normal receiving water discharge. The amount of recycled water used can be less than the amount treated to the recyclable standards because demand for the water is reduced use due to seasonal and unusual variations in climate or reduced demand from businesses because of economic factors. The amount treated to recycled standards is not routinely reported but should be available from treatment plant operators.

[7] Wastewater treatment plants use treated wastewater for various in-plant functions and processes including cooling and filter flushing.

3. Comparing the recycled water used to published planning targets and projections. There are many plans and projections for recycled water use in the Bay Region, some of which include targets for recycled water use. In 1999 the Bay Area Regional Water Recycling Program projected that for the five county region (San Francisco, San Mateo, Santa Clara, Alameda, Contra Costa) Water recycling projects in the Bay Area could produce as much as 125,000 acre feet a year by 2010 and 240,000 acre feet a year by 2025 if funding were available and institutional constraints were reduced (BARWMP, 1999 and BACWA 2006). The North Bay Water Reuse program projects a recycled water potential of 36,500 ac-ft in Marin, Sonoma, and Napa Counties although the three alternatives analyzed for implementation project smaller amounts available for reuse (BACWA 2006, North Bay Water Reuse web-site project description http://nbwra.org/projects/3alternatives.html). The 2006 Bay Area Integrated Regional Water Management Plan identifies 27 projects that could produce up to 120 TAF/YR of recycled water by the year 2020 (BACWA 2006). The individual water agencies that prepare 2010 UWMP's are required to produce projections for the recycled water production and use through the year 2035. The plans and projections are based upon an assessment of future supply and demand for recycled water and, depending on the projection, a greater or lesser evaluation of the economic viability and funding availability. For this assessment the BARWRP projection for 2010 is the most viable to use, recognizing that many of the assumptions made in 1999 about demand and funding availability have not been realized.

Results

Results of this assessment are displayed in Table 1 and Figure 2. The data in Table 1 are listed by each "recycler" or producer of recycled water, i.e., a wastewater treatment plant, or a wastewater district –on a separate line. The recyclers are grouped by region (East Bay, South Bay, Peninsula, and North Bay) and by county (color code). The recycled water production in acre-feet (AF) is shown by-plant for 2010, and as total for the San Francisco Bay Area in 2001. Both 2010 and 2001 totals are further divided into use categories (shown in columns), with the categories grouped into those that offset water demand (summed in the "Potable offset" column), and those that do not. Also compiled but not shown on this table is the (2010) data source for each recycler, and a name and contact information of a person who provided the data or information specific to the plant.

Total recycled use in the Bay Area increased more than 50 percent from 2001 to 2010 to 46.1 thousand acre-feet (TAF), with the most significant increase in the use by refineries and power plants for process and cooling water. Nearly 36 TAF replaces potable use and stream and groundwater use, representing nearly 4 percent of the total urban and agricultural water demand in the Bay Area and more than doubling the potable offset in 2001. Most of the 36 TAF offsets potable supplies previously used for landscape irrigation and industrial uses, with around 3 TAF offsetting groundwater and surface water use by North Bay agriculture. The remaining recycled use does not offset potable uses but is used to sustain freshwater marshes around the Bay and to grow forage crops in the North Bay.

Recycled water use in 2010 was only about 30% of the 2010 target of 125 TAF established for the five counties in 1999 by the Bay Area Regional Water Recycling Program (BARWRP). This is primarily due to the cost of recycled water projects and funding limitations, reduced market demand, and customer/public acceptance. Currently, proposals for 27 projects with 120 TAF/YR of yield are in different phases of planning or funding procurement although this is still short of the 276 TAF of the potential market for recycled water that the BARWRP and North Bay Reuse Study identified for the year 2025. The 46.1 TAF of recycled water comprises about 7 percent of wastewater production from the POTW's. There is plenty of potential supply, although a portion of the wastewater stream may never be economically feasible to develop for recycling given the mismatch between wastewater discharge locations and recycled water market locations. This discrepancy results in a high conveyance cost for the recycled water product.

Significant expansion of recycled water use in the Bay region will depend on acquiring the necessary funding and overcoming the perceived and actual risks of using it for indirect potable use by recharging it in groundwater basins. These basins in the Santa Clara Valley, Alameda County, Sonoma Creek and Napa River Valleys supply groundwater for urban and agricultural uses. The use of recycled water in new development should also be maximized since it is generally less expensive to install the needed infrastructure compared to retrofitting it in an existing development.

Recycled water use is becoming an increasingly important part of the Bay Area's water portfolio and will help offset increased potable uses and hopefully replace enough existing potable uses to reduce our reliance on imported supplies and increase freshwater outflows to the Bay from the Delta. If the potential market for recycled water is fully realized, demand for imported water could be significantly reduced and the region's water supply would be far more reliable. To fully realize this potential, Bay Area residents and businesses will need to overcome their concerns about the perceived risks of recycled water and embrace it as one of the most viable means of achieving a more sustainable water future for the region.

References

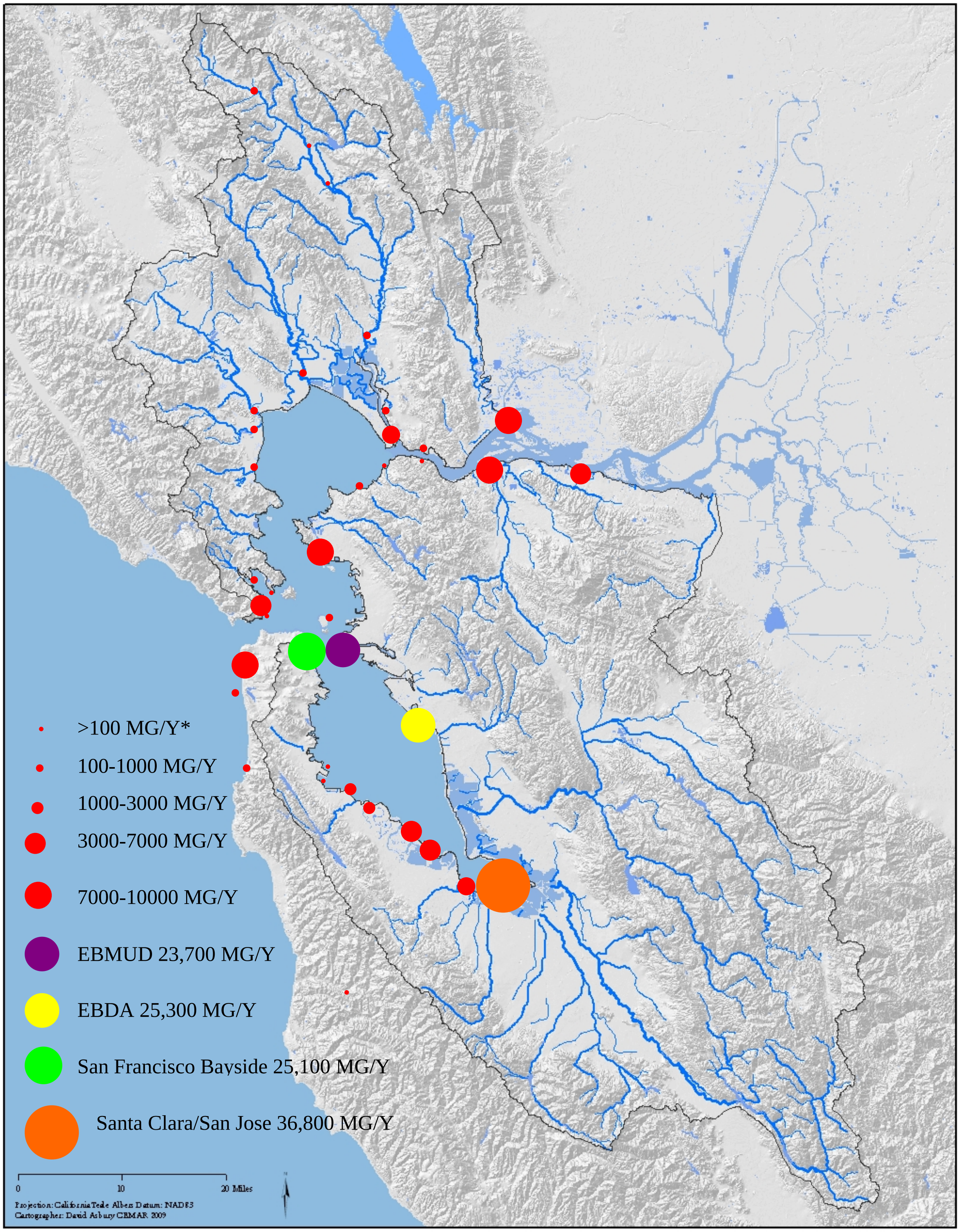
[BARWRP] Bay Area Regional Water Recycling Program. 1999. San Francisco Bay Area Regional Water Recycling Program Recycled Water Master Plan. Lafayette, CA.

[BAWAC] Bay Area Clean Water Agencies. 2006. Wastewater and Recycled Water. Functional Area Document. 125 pp. Available at http://bairwmp.org/docs/functional-area-documents/bay-area-clean-water-agencies-resolution-to-adopt

[BAWAC] Bay Area Clean Water Agencies. 2007. Importance of Recycled Water to the San Francisco Bay Area. Prepared for Bay Area Clean Water Agencies by M.Cubed, Oakland, CA. 34 pp. Available at http://bacwa.org/LinkClick.aspx?fileticket=3A7XYG3Bwyo%3d&tabid=105&mid=463

The Bay Institute (TBI) 2003 San Francisco Bay Index. 84 pp. Novato, CA. Available at www.bay.org

Figure 1. Locations of Region 2 POTW Discharges



* MG/Y = million gallons per year From Region 2 draft staff report on Water Recycling in the SF Bay Area

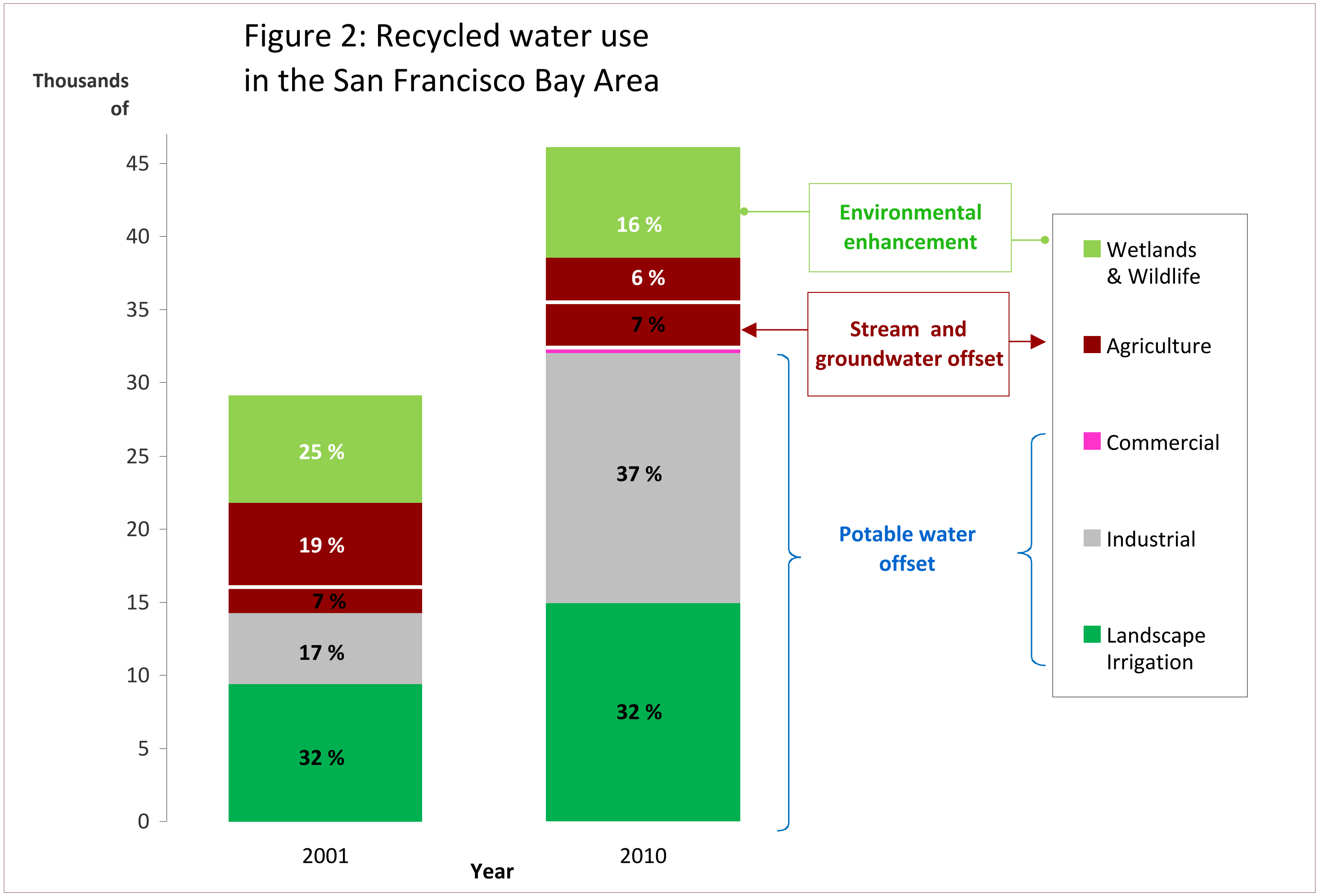


Figure 2: Recycled water use
in the San Francisco Bay Area
Thousands
of
45
40
35
30
25
20
15
10
5
0
25 %
19 %
7 %
17 %
32 %
16 %
6 %
7 %
37 %
32 %
2001
2010
Year
Environmental
enhancement
Stream and
groundwater offset
Potable water
offset
Wetlands
& Wildlife
Agriculture
Commercial
Industrial
Landscape
Irrigation

TABLE 1: RECYCLED WATER USE IN THE SAN FRANCISCO BAY REGION IN 2010 AND 2001

				All values in acre-feet for 2010 except where noted							
Region	County	Distributor / Customer	POTW Operator	Total	Potable Offset	Landscape Irrigation	Industrial	Commercial	Agriculture	Agriculture non-offset	Wetlands & Wildlife
East Bay	Alameda	EBMUD	EBMUD Main WWTP	560	560	549	0	11	0	0	0
East Bay	Alameda	EBMUD	San Leandro WPCP	448	448	448	0	0	0	0	0
East Bay	Alameda	Hayward	Oro Loma/Castro Valley	258	258	258	0	0	0	0	0
East Bay	Alameda	Livermore	Livermore, City of	800	800	773	24	1	3	0	0
East Bay	Alameda	Hayward Marsh	Union Sanitary District	3,362	0	0	0	0	0	0	3,362
East Bay	Contra Costa	CCWD	Central Contra Costa SD	614	614	614	0	0	0	0	0
East Bay	Contra Costa	CCWD	Delta Diablo Sanitation District	6,269	6,269	449	5,821	0	0	0	0
East Bay	Contra Costa	Peyton Slough	Mt. View Sanitation District	2,240	0	0	0	0	0	0	2,240
East Bay	Contra Costa	DSRSD	Dublin San Ramon Services District	1,729	1,729	1,675	54	0	0	0	0
East Bay	Contra Costa	EBMUD-R.A.R. E.	West County Wastewater District	4,124	4,124	202	3,922	0	0	0	0
East Bay	Contra Costa	EBMUD-Chevron	West County Wastewater District	4,482	4,482	0	4,482	0	0	0	0
East Bay	Contra Costa	EBMUD	from Dublin San Ramon	491	491	491	0	0	0	0	0
Peninsula	San Mateo	Redwood City	South Bayside System Authority	490	490	490	0	0	0	0	0
Peninsula	San Mateo	SFPUC	Daly City	547	547	547	0	0	0	0	0
South Bay	Santa Clara	San Jose etc.	San Jose/Santa Clara WPCP	7,767	7,767	5,030	2,277	460	0	0	0
South Bay	Santa Clara	Sunnyvale	Sunnyvale WPCP	776	776	762	8	6	0	0	0
South Bay	Santa Clara	Palo Alto	Palo Alto Regional WQCP	2,147	767	431	336	0	0	0	1,380
North Bay	Marin	MMWD	Las Gallinas SD	600	600	570	0	30	0	0	0
North Bay	Marin	North Marin WD	Novato SD	1,680	168	168	0	0	0	951	560
North Bay	Napa	American Canyon	American Canyon	68	68	27	0	0	41	0	0
North Bay	Napa	Napa and Ag	Napa Sanitation District	1,679	915	905	0	3	6	764	0
North Bay	Napa	Calistoga	Calistoga	211	153	153	0	0	0	58	0
North Bay	Napa	Yountville and Ag	Yountville	284	284	47	0	1	235	0	0
North Bay	Solano	Turf farm	Fairfield Suisun Sewer District	1,304	1,304	0	0	0	1,304	0	0
North Bay	Sonoma	Petaluma	Petaluma WR Facility	1,677	493	356	131	0	6	1,184	0
North Bay	Sonoma	Agriculture	Sonoma Valley County SD	1,500	1,500	0	0	0	1,500	0	0
East Bay				25,378	19,776	5,459	14,303	12	3	0	5,602
Peninsula				1,037	1,037	1,037	0	0	0	0	0
South Bay				10,690	9,310	6,223	2,621	466	0	0	1,380
North Bay				9,003	5,485	2,226	131	34	3,093	0	560
2010 total	9 counties			46,108	35,608	14,945	17,055	512	3,096	2,958	7,542
2010 minus N. Bay	5 counties			37,105	30,124	12,719	16,924	478	3	0	277 6,982
2001 total	9 counties			29,094	16,219	9,392	4,865	32	1,930	5,559	7,317

State of San Francisco Bay 2011
Appendix L

STEWARDSHIP - Volunteer Effort
Technical Appendix

Katherine Smetak, Center for Ecosystem Management and Restoration

Background and rationale: The success of local environmental conservation and restoration efforts relies in large part on public interest and involvement. Action by Bay Area residents to volunteer their time in local restoration activities is a direct expression of stewardship directed toward improving the health of the Bay. California's largest volunteer event is Coastal Cleanup Day, an annual event organized by the California Coastal Commission where volunteers collect debris from the state's marine environments, including the Bay Area's shoreline and waterways. The number of volunteers participating each year in the coastal cleanup event is presented in the State of the Bay report as an example indicator of volunteer effort.

Data sources: Data for this indicator was obtained from the California Coastal Commission, who maintain an electronic database of Coastal Cleanup Day results. The data can be viewed online at http://www.coastal.ca.gov/publiced/ccd/ccd11.html.

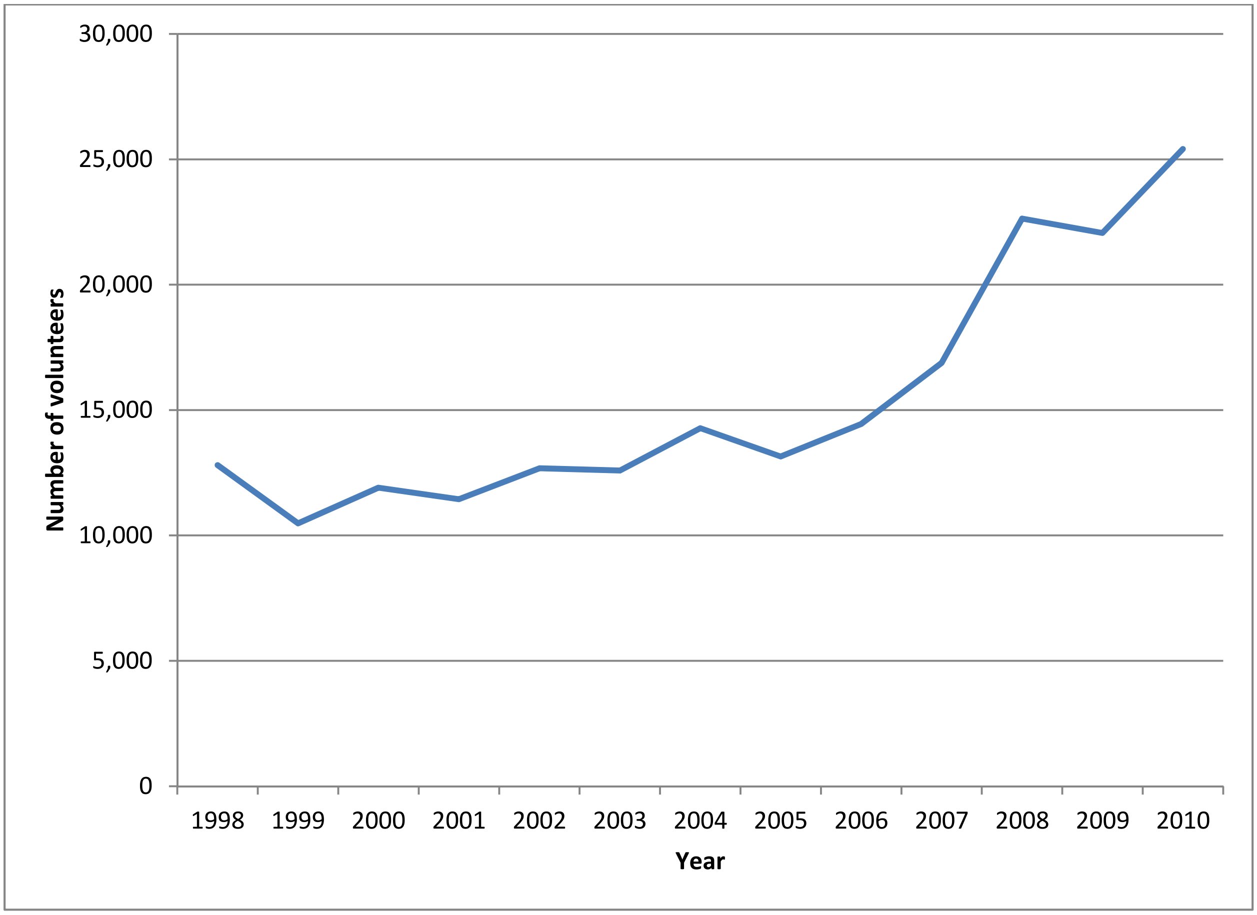
Methods and calculations: This indicator was measured as the total the number of volunteers participating in the Coastal Cleanup Day event in the nine-county region surrounding the San Francisco Bay (Marin, Sonoma, Napa, Solano, Contra Costa, Alameda, Santa Clara, San Mateo, and San Francisco counties) for each year between 1998 and 2010.

Goals, targets, and reference conditions: The number of volunteers in 1998 is used as the reference condition, with the goal of increasing volunteer effort each year.

Results: Results are shown in Figure 1. Since 1998, Coastal Cleanup Day participation in the San Francisco Bay area has nearly doubled.

Summary: Volunteer participation in stewardship activities, as represented by Coastal Cleanup Day, has increased in the last decade. The interest shown by Bay Area residents in volunteering their time to take part in stewardship activities is an important outcome of public outreach and education. Continued outreach and education efforts, combined with stewardship opportunities, will likely strengthen volunteer participation in the future, which will contribute to the ecological health of San Francisco Bay.

Figure 1. San Francisco Bay Area Coastal Cleanup Day Participation 1998 - 2010



State of San Francisco Bay 2011
Appendix M

STEWARDSHIP – Public Access
Technical Appendix

Katherine Smetak, Center for Ecosystem Management and Restoration

Background and Rationale: Access to the Bay is a prerequisite for public engagement leading to stewardship outcomes. The public access indicator assesses the extent to which access to the Bay is being provided by evaluating the increases in mileage over time of the San Francisco Bay Trail and the Bay Area Ridge Trail. The Bay Area Ridge Trail is included in the analysis with the rationale that people need to be engaged with the entire watershed and not just the Bay itself.

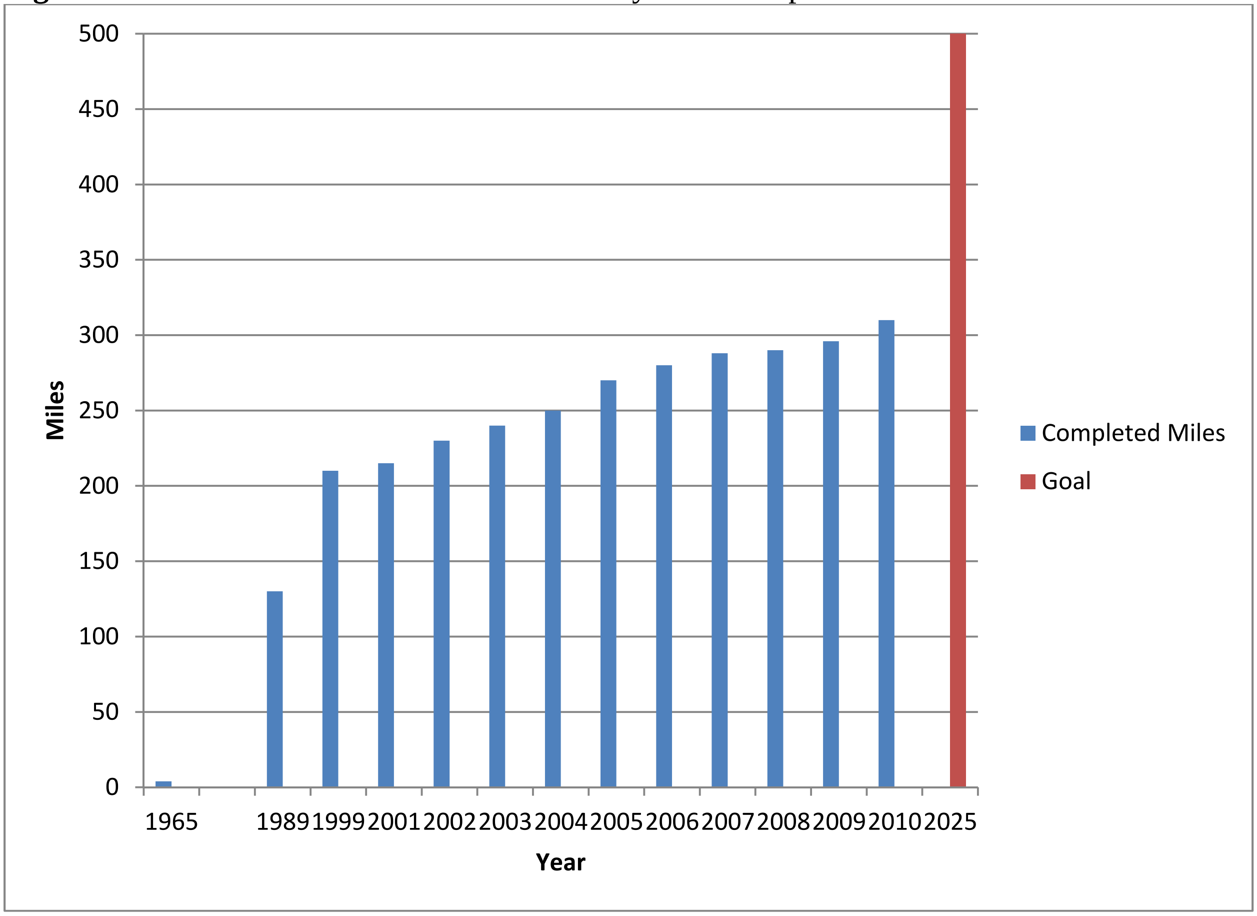
Data sources: Data for the San Francisco Bay Trail was obtained from the Association of Bay Area Governments. Data for the Bay Area Ridge Trail was obtained from the Bay Area Ridge Trail Council website (www.ridgetrail.org), which includes a history of trail completion and maps indicating the length of trail segments in miles.

Methods and Calculations: This indicator documents a trend over time from the initiation of the Bay Trail and Ridge Trail concepts through their present status. The indicator was calculated by evaluating miles of trail completed annually for the San Francisco Bay Trail and the Bay Area Ridge Trail. The indicator was assessed as the percentage of goals for trail completion that are being met.

Goals, Targets, and Reference Conditions: The Association of Bay Area Governments (ABAG) identified the goal in 1989 of establishing a 500-mile regional hiking and bicycling trail around the perimeter of San Francisco and San Pablo Bays. In 1989 time only four miles of trail existed. The Bay Area Ridge Trail Council identified the goal in 1987 of creating 550 miles of trail for recreational use along the ridgelines surrounding San Francisco Bay. In 2006, the Council identified the near-term goal of completing 400 miles of trail by 2010.

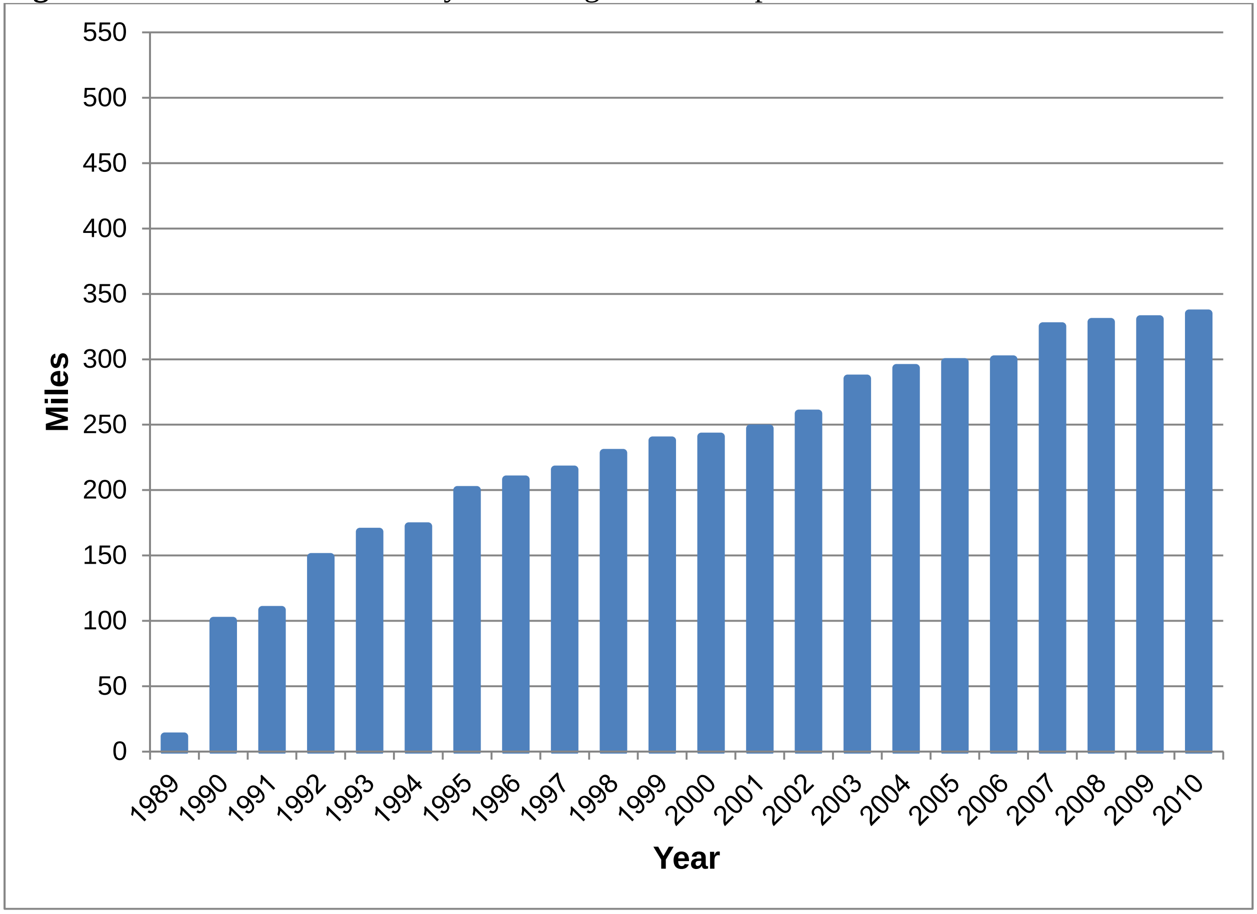
Results: Currently, 310 of 500 planned miles of the Bay Trail are complete, or 62 percent of the goal for the entire system (Figure 1). Since the dedication of the Ridge Trail’s first segment in 1989, 330 of 550 miles of trail have been completed, or 60 percent achievement of the goal for the entire system and 82 percent achievement of the near-term goal set for 2010 (Figure 2).

Figure 1. Cumulative Miles of San Francisco Bay Trail Completion 1965 – 2010*



*Data not available for 1966 – 1988

Figure 2. Cumulative Miles of Bay Area Ridge Trail Completion 1989 - 2010



State of San Francisco Bay 2011
Appendix N

STEWARDSHIP – Management Actions Technical Appendix

Katherine Smetak, Center for Ecosystem Management and Restoration

Background and rationale: One example of stewardship expressed through regulatory effort is the work done to alter disposal practices for material dredged from the Bay. An average of 6.84 million cubic yards of material was dredged from the Bay annually until recently, with the majority of the dredged material—roughly 80%—disposed of at three federally-designated in-Bay disposal sites. Concern regarding the impact of this activity on the Bay led to the creation of created the Long Term Management Strategy for the Placement of dredged material in the San Francisco Bay Region (LTMS) by the U.S. Environmental Protection Agency (USEPA), U.S. Army Corps of Engineers (USACE), San Francisco Bay Regional Water Quality Control Board (SFBRWQCB), San Francisco Bay Conservation and Development Commission (BCDC), State Water Resources Control Board (SWRCB), and other stakeholders, with the objective of maintaining economically viable waterways while minimizing the environmental impacts of dredging. The Dredged Material Management Office (DMMO) was established in 1996 to increase coordination between the member agencies of the LTMS and to consolidate handling of dredged material management issues to streamline the permitting process. In 1994, the USEPA established the San Francisco Deep Ocean Disposal Site as an alternative to in-Bay disposal, and the beneficial reuse of dredged material for wetlands construction, levee restoration, and landfill cover projects became another important alternative to in-Bay disposal.

This indicator is presented in the context of building towards a future assessment by SFEP that evaluates both the ecological condition of the Bay and our success at addressing known stressors through regulatory/management activity.

Data sources: This indicator was calculated using dredged material disposal volumes by location (in-Bay, upland/reuse, or deep ocean) reported in the *Dredging and Disposal Road Map* (LTMS 1999) and the *LTMS Management Plan* (LTMS 2001) for the years 1985 through 1999. These reports can be viewed online via the DMMO website (http://www.spn.usace.army.mil/conops/dmmo.htm). For the years 2000 through 2009, data for dredged material disposal volumes and disposal site locations were obtained from the US Army Corps of Engineers. Since 1999, the USACE has maintained a detailed dredging project tracking database, which includes information on project locations and volumes of dredged material.

Methods and Calculations: The success of management actions to reduce the negative impacts of dredging on Bay health is measured by examining the annual volume of in-Bay disposal of dredged material and the relative amount of disposal directed toward beneficial reuse. This indicator was calculated for the years 1985 through 2009.

Goals, Targets, and Reference Conditions: These indicators are evaluated using the goals of the 2001 LTMS Management Plan:

- In-Bay disposal is to be reduced to approximately 1.25 million cubic yards per year, to be implemented over a 12-year period (2000- 2012) with annual in-Bay disposal volume targets reduced by approximately 387,500 cubic yards every three 3 years.
- Each year, no more than 20 percent of dredged material is to be disposed of in-Bay, at least 40 percent is to be beneficially reused or disposed of at upland sites, and the remainder is to be disposed of at the deep ocean site.

Results: Management actions have led to a decrease in in-Bay disposal of dredged material since 1990 and an increase in beneficial reuse of dredged material since 2000 (Figures 1 and 2). The goal of reducing in-Bay disposal to 1.25 million cubic yards per year by 2012 is being met. Since 2000, the goal of disposing no more than 20 percent of dredged material in-Bay has been met in one year and was close to being met in 3 of the years evaluated, while the goal of disposing 40% of dredged material at upland or beneficial reuse sites has been met in five of the years evaluated.

References

LTMS. 1999. *Dredging and Disposal Road Map*. Prepared by SFBCDC, USEPA, USACE, SFBRWQCB, and CSWRCB. 18 pp.

LTMS. 2001. Long-Term Management Strategy for the Placement of Dredged Material in the San Francisco Bay Region *Management Plan 2001*.

Figure 1. In Bay Disposal of Dredged Material 1985 - 2009

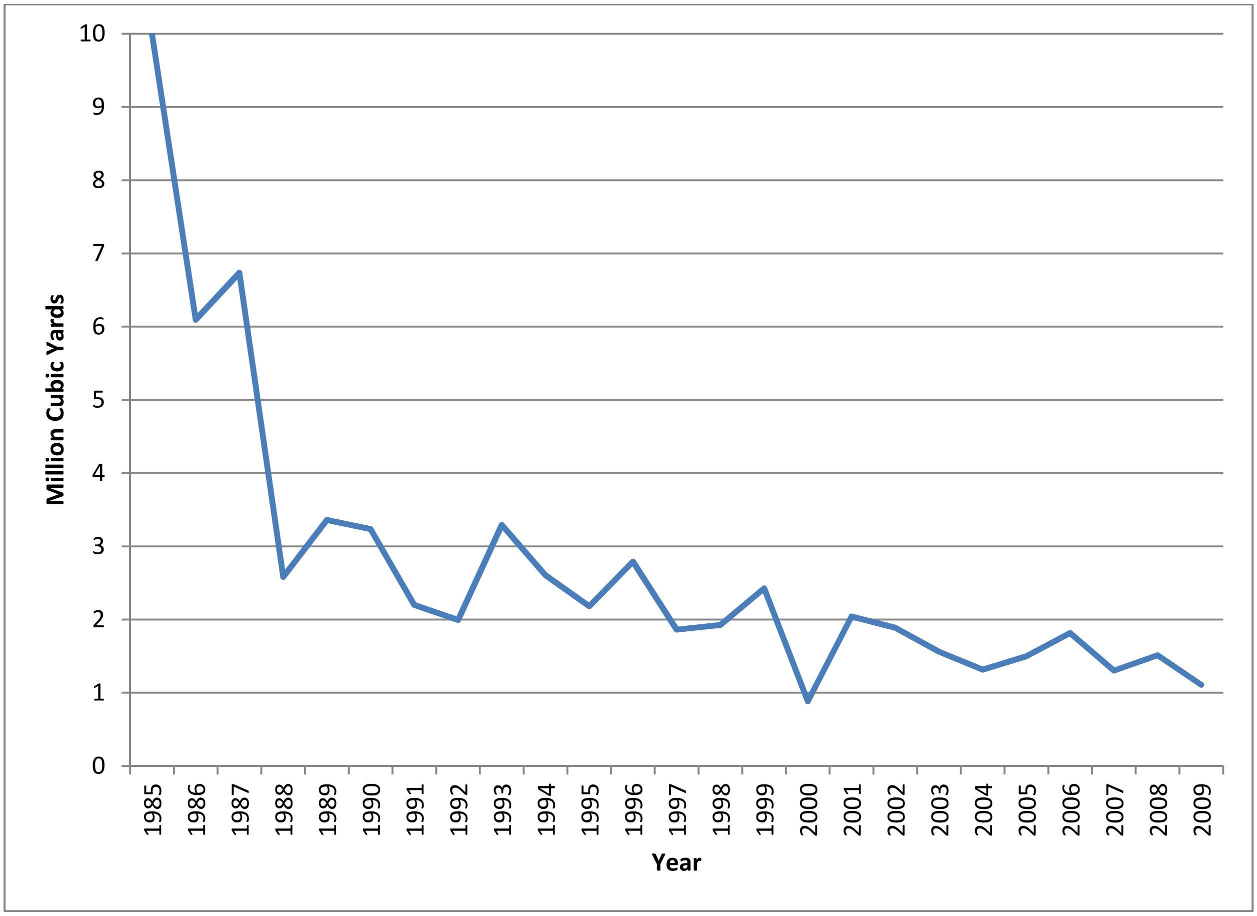
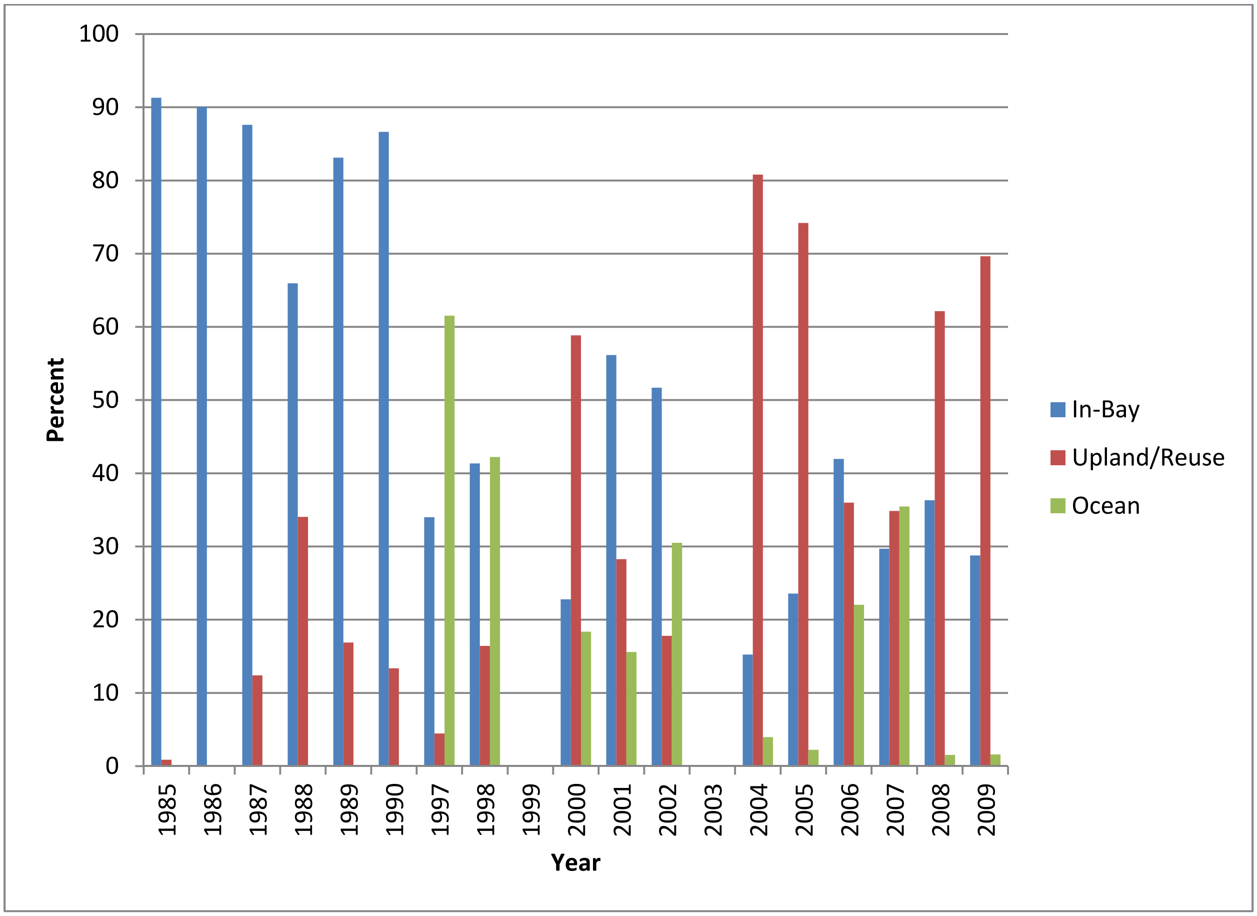


Figure 2. Distribution of Dredged Material by Site 1985 - 2009



State of San Francisco Bay 2011
Appendix O

Steelhead Trout Production as an Indicator of Watershed Health

Gordon Becker and Katherine Smetak, Center for Ecosystem Management and Restoration

Introduction
The Steelhead Trout Production indicator described in this document was not included in *The State of San Francisco Bay 2011* but could be utilized in future evaluations of the Bay by SFEP.

Background and Rationale
Urban and agricultural development over the last 150 years has diminished the quality and quantity of freshwater habitat in streams tributary to the San Francisco Estuary and led to a substantial decline in the region's salmonid abundance, including steelhead (*Oncorhynchus mykiss*) populations (Leidy *et al.* 2005). Steelhead in San Francisco Estuary tributaries are included in the Central California Coast Steelhead Distinct Population Segment (DPS) and are listed as threatened for purposes of the federal Endangered Species Act (Good *et al.* 2005). Efforts to protect and restore the estuary's tributary streams, including activities that benefit steelhead, such as modifying fish passage barriers, reducing sedimentation, and providing instream flows for habitat, are being undertaken throughout the region. Implementing monitoring programs to assess the effectiveness of these recovery efforts is essential to gauging the health of the estuary's watersheds.

A steelhead outmigrant monitoring program is proposed as an indicator of health in San Francisco Estuary watersheds. (This method commonly is called "smolt trapping" although all sizes and life history stages are sampled.) Monitoring has been identified by the National Marine Fisheries Service (NMFS) as critical to measure salmonid recovery, as smolt abundance is a key component in producing population size estimates and determining regional trends and population viability (NMFS 2010). Because steelhead complete one year or more of their life cycle in freshwater and are sensitive to changes in habitat conditions (e.g., flow, dissolved oxygen, temperature, turbidity, etc.), they are ideal indicators of stream health. Smolt trapping is an effective means of measuring the aggregate watershed condition for areas upstream of the trap and is a valuable tool for evaluating the success of restoration efforts and directing long-term management and restoration activities (Ketcham *et al*. 2005). In addition, although smolt trapping is focused on salmonids, it also provides additional information on the presence and relative numbers of other aquatic species.

Data regarding the outmigration of juvenile steelhead are virtually absent for watersheds tributary to the San Francisco Estuary, with the notable exceptions of information gathered recently by the Napa County Resource Conservation District and the Santa Clara Valley Water District. Becker *et al.* (2007) found that eight of the San Francisco Estuary's watersheds account for about 75 percent of steelhead habitat resources in the region. The drainages of Alameda, Coyote, San Francisquito, Corte Madera, Sonoma and Suisun creeks and the Guadalupe and Napa rivers comprise these "anchor" watersheds, in which monitoring juvenile steelhead outmigration is expected to produce robust data characterizing salmonid populations in the region. The collective data set could serve as an estuary-wide indicator of watershed health. Liermann and Roni (2008) suggest that monitoring programs implemented to determine whether restoration projects are increasing salmonid abundance must be established in multiple watersheds to produce results that can be generalized.

Data Sources

The steelhead trout outmigrant indicator will be calculated using data from the Napa County Resource Conservation District's (RCD) salmonid outmigrant monitoring program, initiated in 2009, and in the future will incorporate data from additional sampling sites in watersheds tributary to the San Francisco Estuary. The Napa County RCD conducted salmonid outmigrant monitoring at one sampling station in the lower mainstem Napa River in 2009 and 2010 using a rotary screw trap (RST) installed at a location allowing for the capture of migrating salmonids from all upstream tributaries, at approximately 1,500 feet upstream of the upper extent of tidal influence. The trap was operated in 2009 from March 17 through May 26 and in 2010 from February 17 through June 14 (NRCD 2009; NRCD 2010).

In 2012 (pending funding), outmigrant trapping will be expanded to include three additional locations in the Napa River watershed, one in Upper Penitencia Creek (Coyote Creek watershed), and two Sonoma Creek watershed sites Outmigrant trap locations in the Napa River watershed will be selected from candidate sites in Napa, Redwood, Milliken, Tulucay, Carneros, and Huichica creeks. Together with the sampling results from the mainstem Napa River monitoring location, new sampling results will account for steelhead smolt production in approximately 95 percent of all spawning and rearing habitat available to salmonids below major barriers in the watershed. If additional funding is secured, outmigrant traps are expected to be operated at locations in the Alameda, San Francisquito, and Corte Madera creek watersheds in the following year. Trapping will be conducted at all sampling stations during the 14 weeks between about February 15 and June 1. Results will be used to develop salmonid population estimates and track ecological responses to ongoing habitat restoration. A future goal is to expand the juvenile steelhead outmigrant monitoring program to include sampling stations in all eight anchor watersheds identified in Becker *et al.* (2007).

Methods and Calculations

The steelhead trout outmigrant indicator will be calculated using trap data from the mainstem Napa River sampling station operated by the Napa County RCD, and in the future will incorporate data from the additional sampling sites listed previously. Results will be normalized to reflect annual catch-per-unit-effort, or number of steelhead smolts captured per day of trapping in each sampling year, to correct for annual differences in trapping duration. The indicator will be calculated as the annual percent change in catch-per-unit-effort.

Goals, Targets and Reference Conditions

The steelhead smolt catch-per-unit-effort from sampling beginning in 2009 will be used as the reference condition against which change can be measured. Once trapping at additional sites has been established, results can be used to develop population estimates, to compare inter-annual variability among and within watersheds, and to correlate trapping results with environmental data (*e.g.*, streamflow) and restoration efforts.

Quantitative goals for recovery of steelhead populations exist for several watersheds tributary to the San Francisco Estuary, but may overestimate production potential. Further, due to the absence of historical data, reference conditions are not well understood. However, available information indicates a significant decline in the distribution and abundance of steelhead in the region (Leidy *et al.* 2005). Annual sampling will allow for the refinement of quantitative recovery goals in the watersheds where they are assigned, and development of goals for those watersheds without such assignments.

Results

Results from the Napa County RCD's salmonid outmigrant monitoring program for each year of sampling (2009 – 2010) at the mainstem Napa River site are shown in Figure 1. The catch-per-unit-effort was 1.78 steelhead smolts per day in 2009 and 2.48 steelhead smolts per day in 2010, representing a 39.3 percent increase in catch per unit effort of smolts from 2009 to 2010.

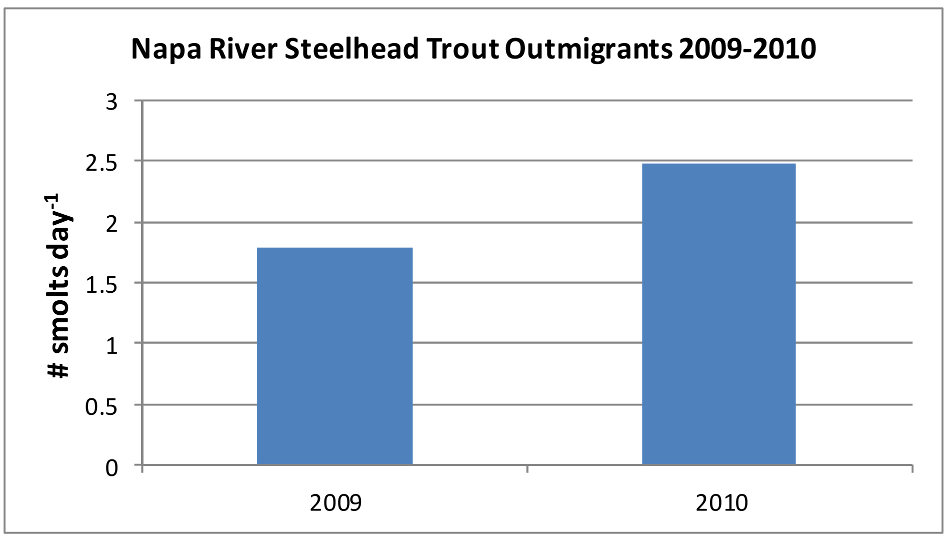


Figure 1. Results from salmonid outmigrant monitoring conducted on the mainstem Napa River in 2009 and 2010. The chart shows the number of steelhead smolts captured per day of rotary screw trap operation in each sampling year.

REFERENCES

Becker, G.S., I.J. Reining, D.A. Asbury. 2007. San Francisco Estuary Watersheds Evaluation: Identifying Promising Locations for Steelhead Restoration in Tributaries of the San Francisco Estuary. Center for Ecosystem Management and Restoration, Oakland, CA.

Bradford, M.J., G.C. Taylor J.A Allan. 1997. Empirical review of coho smolt abundance and the prediction of smolt production at the regional level. Tranactions of the American Fisheries Society. 126: 49-64.

Ketcham B.J., M.L Reichmuth, D.Fong, G.G. Brown. 2005. SOP 2 Spring outmigrant trapping program. Part of the Threatened and Endangered Aquatic Species and Stream Fish Assemblage Indicator – Stream Resources Monitoring Protocol. San Francisco Areas Network Inventory and Monitoring Program.

Leidy, R.A., G.S. Becker, B.N. Harvey. 2005. Historical distribution and current status of steelhead/rainbow trout (*Oncorhynchus mykiss*) in streams of the San Francisco Estuary, California. Center for Ecosystem Management and Restoration, Oakland, CA.

Liermann, M. and P. Roni. 2008. More Sites or More Years? Optimal Study Design for Monitoring Fish Response to Watershed Restoration. *North American Journal of Fisheries Management* **28**:3, 935-943

Moyle, P.B. 2002. Inland Fisheries of California (Revised and Expanded). University of California Press. pp 271-281.

[NMFS] National Marine Fisheries Service. 2010. Recovery Plan for the Evolutionary Significant Unit of Central California Coast Coho Salmon. Public Draft.

[NRCD] Napa County Resource Conservation District. 2009. Napa River Steelhead and Salmon Smolt Monitoring Program: Annual Report – Year 1.

[NRCD] Napa County Resource Conservation District. 2010. Napa River Steelhead and Salmon Smolt Monitoring Program: Annual Report – Year 2.