State of the Estuary Report 2015

Technical Appendix
WATER: Combined Water Quality
Safe for Swimming, Safe for Aquatic Life, Fish Safe to Eat

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I. GENERAL CONSIDERATIONS

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

The region is fortunate to have one of the best water quality monitoring programs in the world (the Regional Monitoring Program for Water Quality in San Francisco Bay) 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 Estuary water quality is based largely on information generated by the Bay Regional Monitoring Program. Other valuable sources of information are also available and were also considered.

Another major monitoring effort - the Regional Monitoring Program for the Delta - is beginning to collect samples in 2015. The Delta RMP will be a major source of information for future assessments of water quality in the Estuary. At present, however, there is a comparatively limited amount of readily available, systematic water quality data for the Delta. Also, the scope of the effort to conduct the present water quality assessment was limited due to a lack of funding. While this assessment represents an expansion relative to the 2011 State of the Bay Report with the inclusion of the Delta, only a few readily accessible Delta datasets could be incorporated.

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 Estuary. 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 and intensive study by scientists. 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 because they can be clearly assessed relative to the established guidelines.

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 thresholds 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 Estuary, 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.

II. EVALUATION SCHEME

This water quality element of the State of the Estuary Report addresses the three main beneficial uses of the Estuary 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 easilyunderstood by the public:

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

A fourth key question applies to the Delta: "Is Estuary water safe to drink?" Addressing this question in a quantitative manner was beyond the scope of this effort. A short summary of the issue is provided as a sidebar in the water quality chapter of the main report.

A. QUESTION 1: IS THE ESTUARY SAFE FOR AQUATIC LIFE?

"Aquatic life" as used here refers to all of the animal and plant species that live in or depend upon the Estuary, including algae, zooplankton, macroinvertebrates, fish, aquatic birds, and marine mammals. A varied group of indicators is most appropriate for addressing question 1, including a target from the Bay Mercury TMDL and Delta Methylmercury TMDL for methylmercury concentrations in small fish, qualitative narrative objectives that apply to the occurrence of toxicity in Estuary water, and numeric water quality objectives that are based on measurement of concentrations in water.

For each parameter, average values for each sampling year are compared to the targets. The degree of risk for pollutants in this category are based on assessments in published studies and other considerations discussed below for each pollutant.

Although water quality objectives to protect aquatic life exist for many pollutants in the Delta, a lack of systematic monitoring limited the scope of the assessment for that part of the Estuary.

B. QUESTION 2: ARE FISH FROM THE ESTUARY SAFE TO EAT?

This question refers to human consumption of fish from the Estuary. The appropriate indicators for this question are concentrations of pollutants of concern in the tissue of fish species that are popular for consumption by anglers. The Bay Regional Monitoring Program has conducted systematic and regular monitoring of Bay sport fish since 1994, providing a solid foundation for assessing this question (Davis et al. 2011). The California Surface Water Ambient Monitoring Program conducted fairly thorough monitoring of Delta sport fish in 2011 (Davis et al. 2013).

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 the Estuary. 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 also issued consumption guidelines for the Delta region in recent years (Gassel et al. 2007, 2008). 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 consumption advice. 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 consumption guidelines for the Bay and the published consumption guidelines for the Delta represent the definitive statements for the public on the safety of consuming Estuary fish. The intent of using ATLs in the State of the Estuary Report is to convey a message to the public that is consistent with and supports the consumption advice.

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 2008b) 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 Delta, 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.

C. QUESTION 3: IS THE ESTUARY 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.

FIB monitoring data for Delta beaches are not available through Heal the Bay.

Toxins produced by blooms of harmful algae such as *Microcystis* are another threat to the health of people enjoying contact recreation in the Estuary. Although studies measuring algal toxins in the Estuary have been conducted, and thresholds developed by the state are available for assessment (OEHHA 2012), routine and systematic monitoring of algal toxins in the Estuary is not being conducted. A synthesis of the studies that have been performed was beyond the scope of the present report.

III. IS THE ESTUARY SAFE FOR AQUATIC LIFE?

A. POLLUTANTS WITH APPROPRIATE THRESHOLDS

1. Methylmercury in Prey Fish

In addition to posing risks to humans who eat Estuary fish, methylmercury poses significant risks to Estuary wildlife. Extensive studies in Forster's Terns concluded that 48% of birds in the breeding season in this species in the Bay were at high risk of reproductive impairment due to methylmercury exposure (Eagles-Smith et al. 2009, Ackerman et al. 2014). 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 Ridgway's 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 Bay 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). The Delta Methylmercury TMDL also employs this same target based on Least Tern exposure and risk. 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 was a priority in the Bay RMP in recent 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 TMDLs, methylmercury in prey fish tissue is the key regulatory target for protection of aquatic life (the piscivorous California Least Tern). 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 Bay RMP. The extensive prey fish sampling that was conducted in recent years was summarized by Greenfield et al. (2013a,b). Systematic prey fish sampling has not recently been conducted in the Delta. Although extensive sampling was performed in 2000 and 2001 by U.C. Davis, these data were not used in this assessment because they were collected more than 10 years ago.

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. Sampling continued at four sites in 2011 to allow assessment of seasonal variation. The sampling focused on two species: Mississippi silverside and topsmelt. Samples were collected in all of the regional embayments.

Methods and Calculations The aquatic life methylmercury indicator (Figure 1) was calculated using available data from the Bay 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 intensive sampling year (2010), methylmercury concentrations in prey fish exceeded the 0.03 ppm target in approximately 95% of the samples collected. Similar results were obtained in 2008 and 2009, the other years 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 2010 was 0.050 ppm.

Evaluation of spatial and temporal trends focused on data from 2008-2010, which are based on larger sample sizes. Median concentrations in each region in 2010 ranged from a high of 0.099 in South Bay and Lower South Bay to a low of 0.033 ppm in Suisun Bay.

As discussed below in the Methylmercury in Sport Fish section, methylmercury concentrations in the Estuary food web have not changed perceptibly over the past 45 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 (Davis et al. 2012).

Methylmercury concentrations in prey fish in the Estuary are clearly elevated above the regulatory goal and represent a significant problem. There is no benchmark, however, that can be readily used to judge whether the state of the Estuary with regard to this indicator should be classified as "fair" or "poor", although "poor" would merit consideration.

2. Water Toxicity

Toxicity in water samples is a concern in the Delta. These toxicity tests suggest that pollutant concentrations in Delta waters are occasionally high enough to affect the abundance of aquatic invertebrates or fish. Pesticides (see sidebar in the main report) are often the cause of this toxicity.

Toxicity in Bay water samples was a concern in the 1990s, also driven by pesticide concentrations, but has not been observed within the past 10 years.

A narrative water quality objective in the Bay Basin Plan applies to water toxicity. The Basin Plan states: "All waters shall be maintained free of toxic substances in concentrations that are lethal to or that produce other detrimental responses in aquatic organisms. Detrimental responses include, but are not limited to, decreased growth rate and decreased reproductive success of resident or indicator species. There shall be no acute toxicity in ambient waters. Acute toxicity is defined as a median of less than 90 percent survival, or less than 70 percent survival, 10 percent of the time, of test organisms in a 96-hour static or continuous flow test. There shall be no chronic toxicity in ambient waters. Chronic toxicity is a detrimental biological effect on growth rate, reproduction, fertilization success, larval development, population abundance, community composition, or any other relevant measure of the health of an organism, population, or community."

The Basin Plan for the Sacramento River and San Joaquin River Basins has a similar narrative objective for toxicity: "All waters shall be maintained free of toxic substances in concentrations that produce detrimental physiological responses in human, plant, animal, or aquatic life. This objective applies regardless of whether the toxicity is caused by a single substance or the interactive effect of multiple substances. Compliance with this objective will be determined by analyses of indicator organisms, species diversity, population density, growth anomalies, and biotoxicity tests of appropriate duration or other methods as specified by the Regional Water Board."

The implicit quantitative goal associated with these objectives is a 0% incidence of toxicity in Estuary samples.

Data Source For the Bay, the water toxicity indicator is based on data from the RMP, available on the RMP website (www.sfei.org/rmp/data). The RMP measured water toxicity intermittently over the past 15 years, with sampling occurring in 2002, 2007, and 2011. In the most recent sampling, water toxicity was measured at 22 stations distributed 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. The test species was the mysid shrimp *Americamysis bahia*.

Water toxicity data for the Delta were retrieved from CEDEN. The compiled data consisted of a collection of datasets from various programs, including one-time studies (e.g., the Central Valley Water Board's Delta Island Monitoring Project and the Central Valley Water Board's SWAMP Delta Pyrethroid Study) and annual monitoring performed under the Central Valley Water Boards Irrigated Lands Regulatory Program. Test species have included invertebrates (*Ceriodaphnia dubia, Eurytemora affinis, Hyalella azteca*, and *Americamysis bahia*) and fathead minnows (*Pimephales promelas*). The number of samples for each year varied considerably, with a low of five in 2004 and a high of 118 in 2008.

Methods and Calculations The water toxicity indicator (Figure 2) is simply the percentage of the samples tested in each year that were determined to be toxic to at least one test species. 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 objectives pertaining to water toxicity is 0% incidence of toxicity in Estuary samples.

Results

No water toxicity has been observed in the Bay in recent sampling. The narrative objective for water toxicity has therefore been met consistently over the past 10 years.

In the Delta, the incidence of water toxicity has been greater than zero, but still infrequent, ranging from 0% of samples in 2004 (though only five samples were analyzed) to 12% in 2006 (based on 64 samples that year). The severity of the toxicity has also been low, with only 4% of samples having lower than 50% survival, and only 2% of samples with less than 10% survival.

Overall, the status of the Estuary with regard to water toxicity is fair: the goal is being met in the Bay but not quite being met in the Delta.

3. Copper in Water

Background and Rationale Copper pollution was a major concern in the Bay 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 concentrations. A remaining concern regarding possible impacts of copper on olfaction in salmonids was investigated by the National Oceanographic and Atmospheric Administration's Northwest Fisheries Science Center with funding from the RMP, and concluded that olfactory toxicity is of low concern in Bay waters (Baldwin 2015).

Copper toxicity is a greater concern in the Delta. Copper is one of the most widely applied herbicides in the Central Valley. Copper is toxic to fish at lower concentrations in fresh water than in saline water. Some concentrations measured in the Delta have exceeded levels at which effects could occur (3-5 ug/L) (Stephen Louie, personal communication).

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 Bay RMP. The data are available from the Bay RMP website (www.sfei.org/rmp/data).

Systematic data for copper are not available from the Delta, and a synthesis of the work that has been done was beyond the scope of this report.

Methods and Calculations The copper indicator was calculated for each year of Bay RMP monitoring from 1993 to 2012 (Figure 3). The time series plot shows the distribution of the data (dissolved concentrations in water) for each year sampled. The distribution is described with percentiles (5th, 25th, 50th, 75th, and 95th).

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 2012, except for four samples from South Bay in 2011. The South Bay is the only segment with concentrations approaching or exceeding the objectives. Concentrations in the South Bay over the last six years have been above the long-term average.

Overall, water quality with respect to copper in water is good, but warrants continued tracking, especially in the South Bay.

5. Other Priority Pollutants

In addition to the pollutants mentioned above, the Bay RMP monitors many other pollutants that are present at concentrations below water quality objectives and are considered to pose low risk to 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 Estuary.

The Bay 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" or "good" 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;
- others – cyanide.

B. POLLUTANTS WITHOUT APPROPRIATE THRESHOLDS

1. Invasive Species

Invasive species released from ship ballast water are considered a water pollutant under the Clean Water Act, and they are included on the 303(d) listings for the Bay and Delta 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 invasive species have been immense. Introductions of hundreds of invasive species have irreversibly altered the Estuary ecosystem in fundamental ways. Nonnative species introduced to the Estuary 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. These species are introduced through multiple vectors including: commercial shipping (including vessel fouling and ballast water), the aquaculture industry, live bait releases, intentional sport fishing introductions, release of aquarium pets and live seafood specimens, transfer via recreational watercraft, and association with marine debris. Vessel fouling and ballast water are responsible for the majority of the aquatic species invasions in California (Ruiz et al., 2011).

Invasive species introductions do not fit neatly into the assessment framework used for this report. 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. Commercial vessels are regulated for ballast water management and there are pending regulations for vessel fouling on commercial vessels. The anticipated switch to a ballast water discharge standard and the shift to ballast water treatment systems has been delayed due to the lack of available technologies, but the 98% compliance rate of California's current ballast water management program (which requires either retaining ballast water or conducting an open ocean exchange before discharging ballast water) is providing a significant risk reduction (Dobroski et al. 2015). Unfortunately, it will likely be several more years before technologies are available to meet a discharge standard, which would reduce risk even further. The pending vessel fouling regulations on commercial vessels (anticipated completion of the rulemaking is late 2015) will result in an additional reduction of risk. The other vectors could also be better managed by thoughtful regulation, or by a combination of regulations and public education and outreach.

Focusing on the significant goals mentioned above, progress over the next 5-10 years is likely in reducing invasive species introductions from ballast water and vessel fouling. 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 invasive species, and if invasions are allowed to continue additional large impacts are likely. This places invasive species in a "high concern" category.

2. Trash

Trash is a continuing problem in the Estuary 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 and to organisms that ingest plastic particles. Trash is a concern at 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, the Central Bay and South Bay shorelines were included on the 2010 303(d) List. 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 numeric 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 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 the first permit term, municipalities were 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 Estuary is a moderate concern in regard to impacts on aquatic life. Aggressive requirements in the municipal regional permit for stormwater in the Bay 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.

3. Nutrients

Nutrient concentrations in the Estuary are a major concern. Efforts are in progress to develop definitive numeric goals for the Estuary. This topic, which encompasses an array of indicators of nutrient impacts (dissolved oxygen depletion, harmful algae and algal toxins, chlorophyll abundance, and others), is summarized in a sidebar in the main report.

4. Other Suspected Threats

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

Selenium

Average selenium concentrations in the Bay food web in recent years are below thresholds for adverse effects in fish and wildlife, but a few samples have exceeded the thresholds. Concern for risks to aquatic life is the primary impetus for the North Bay Selenium TMDL that is in development by the San Francisco Bay Regional Water Board. Thresholds to protect aquatic life in the Bay are in development that will be more appropriate than existing water quality criteria. A TMDL for selenium in the San Joaquin River was completed by the Central Valley Regional Water Board and approved by US EPA in March 2002. The project area for this TMDL includes the area where the San Joaquin River enters the Delta.

Polycyclic Aromatic Hydrocarbons (PAHs)

Several locations are included on the 303(d) List due to PAH contamination. 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.

Perfluorooctanesulfonate (PFOS)

PFOS is also considered a potential risk to Estuary wildlife (SFEI 2013). 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.

Pesticides

Pesticides are of particular concern in urban creeks that flow to the Estuary and sometimes the water bodies into which they flow, such as the Delta, where recent studies have implicated pyrethroids as the cause of toxicity to invertebrate test organisms (Weston et al. 2014). A sidebar summarizing issues relating to pesticides is included in the main report. Data from routine, systematic monitoring of pesticides is not currently available for the Delta. However, the Delta Regional Monitoring Program began monitoring for pesticides in 2015. A detailed summary of the miscellaneous studies that have been done in the Delta was beyond the scope of this project.

5. Contaminants of Emerging Concern

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, pharmaceuticals, and chemicals in consumer products, and many of these make their way from our homes, businesses, and watersheds into the Estuary. Due to inadequate screening and regulation of these chemicals, some may cause toxicity in Estuary biota, either through direct exposure to contaminated water or sediment or through accumulation in the Estuary 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 wildlife and humans. The Bay RMP published a summary of the extensive information available on Bay CECs in the 2013 Pulse of the Bay (SFEI 2013). Several studies have also been conducted in the Delta. A review article on this topic was included in the 2011 Pulse of the Delta (Aquatic Science Center 2011).

The Bay RMP actively monitors contaminants of emerging concern that pose the greatest known threats to water quality. 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. 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 Estuary.

IV. ARE ESTUARY FISH SAFE TO EAT?

A. INTRODUCTION

For the Bay, no new sport fish data are available since the publication of the State of the Bay Report in 2011. Additional samples were collected in 2014, but the data were not available at the time this report was written. The text below regarding pollutants in Bay fish has therefore not changed from the 2011 version.

Sport fish pollutant data from 2011 are available from the Delta as a result of monitoring by the State Water Resources Control Board’s Surface Water Ambient Monitoring Program (SWAMP - Davis et al. 2013).

B. POLLUTANTS WITH APPROPRIATE THRESHOLDS

1. Methylmercury in Sport Fish

Background and Rationale

Methylmercury is one of the Estuary's most serious water quality concerns. Methylmercury is a primary driver of the fish consumption advisory for the Bay (Gassel et al. 2012), 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 was 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 accumulates in 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 (Davis et al. 2012), 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 Estuary on a grand scale is now beginning, raising concern that methylmercury concentrations could increase across major portions of the Estuary. 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 methylmercury TMDL for the Delta (Wood et al. 2010) established a water quality objective for methylmercury in muscle fillet of trophic level 4 (piscivorous) fish species.

The concentrations in sport fish were compared to OEHHA thresholds.

Data Source For the Bay, 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 five 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 Bay RMP collects sport fish from five popular fishing locations in the Bay (Figure 4). 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 5). In the most recent sampling in 2009, methylmercury was analyzed in striped bass, California halibut, and jacksmelt, but not white sturgeon or white croaker.

For the Delta, sport fish pollutant data are available as a result of monitoring by SWAMP in 2011 (Davis et al. 2013). These data are available via CEDEN (http://www.ceden.org/) and also from the My Water Quality Portal (http://www.mywaterquality.ca.gov/). This sampling included six locations and six species (largemouth bass, smallmouth bass, striped bass, Sacramento sucker, common carp, and white catfish - only two species from this list were collected at each location). The data presented in this report are for largemouth and smallmouth bass (or “black bass”), adjusted to a standard size of 350 mm.

Methods and Calculations For Bay fish, the sport fish methylmercury indicator (Figure 5) 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.

For the Delta, black bass were available at five of the six locations. The average of the five length-adjusted means was 0.43 ppm.

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

For the Bay, in data from the most recent sampling year currently available, 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, although concentrations in striped bass were right at the 0.44 ppm threshold. OEHHA advises that women between 18 and 45 years of age and children (1-17 years of age) do not eat several species of Estuary fish (including the popular striped bass), largely because of methylmercury contamination. The existence of a “no consumption” recommendation for popular species (rather than limited consumption) seems an appropriate trigger for classifying the state of the Estuary as poor with respect to methylmercury concentrations in sport fish.

Methylmercury concentrations in the Estuary food web have not changed perceptibly over the past 40 years. For the Bay, it is not anticipated that they will decline significantly in the next 30 years. For the Delta, declines are possible if methylmercury inputs can be reduced.

2. PCBs in Sport Fish

Background and Rationale

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.

The Delta is also on the 303(d) list because of PCB contamination, but a TMDL has not been developed.

Data Source The PCB indicator was calculated using data from the same RMP and SWAMP sport fish monitoring programs described for the methylmercury in sport fish indicator. The data are available from the RMP website (www.sfei.org/rmp/data), CEDEN (http://www.ceden.org/) and also from the My Water Quality Portal (http://www.mywaterquality.ca.gov/). Additional details on this sampling were provided in the methylmercury section. The two key Bay indicator species for PCBs have been sampled consistently over the years (Figure 6). For the Delta, the two best organics indicator species were Sacramento sucker (sampled at three locations) and common carp (sampled at two locations).

Methods and Calculations The sport fish PCBs indicator (Figure 6) 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.

For the Delta, Sacramento sucker were available at three of the six locations, and had an average concentration of 15 ppb. Common carp were available from two locations, with an average concentration of 5 ppb.

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 for the Bay, both of the PCB indicator species had average concentrations between 21 ppb and 120 ppb (Figure 6). 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. The existence of a “no consumption” recommendation for this popular group of species (rather than limited consumption) was considered an appropriate trigger for classifying the state of the Estuary as “poor” with respect to PCB concentrations in sport fish.

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.

For the Delta, though the data are limited, both of the indicator species had concentrations below 21 ppb, which put them in the “good” category.

3. Dioxins in Sport Fish

Background and Rationale

Recent sport fish monitoring indicates that dioxins are a concern in the Bay. Dioxins have not recently been measured in Delta sport fish.

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. (2004) 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. 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 7). 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 average concentration for each year sampled. 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 95th percentile rate (32 g/day, or 8 ounces per week) for all Bay fish consumers (Connor et al. 2004).

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 7). Median dioxin TEQ concentrations in white croaker have been over ten times higher than the target. Without ATLs for dioxins from OEHHA, however, there is an insufficient basis for determining that dioxins should be categorized as a high concern (i.e., having concentrations above a “no consumption” ATL). Therefore dioxins were placed in the “fair” 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.

4. Other Pollutants With Appropriate Thresholds

Several other pollutants have been measured in sport fish from the Bay and Delta and found to be present at concentrations of low concern. Legacy pesticides (DDT, dieldrin, and chlordane) and selenium have been measured in both the Bay and the Delta. PBDEs have been measured in Bay fish. More information on these pollutants in Bay fish was provided in Davis et al. (2011). Davis et al. (2013) presents and discusses the data for these pollutants in Delta fish.

C. 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 Estuary. As understanding advances, some of these contaminants emerge as posing risks to the health of humans and wildlife.

The Bay 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 Estuary, accumulating in the food web, and leading to human exposure and risk through consumption of sport fish. Past experience has shown that the Estuary 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 Bay 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 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.

V. IS THE ESTUARY SAFE FOR SWIMMING?

A. *BACKGROUND AND RATIONALE*

Recreation, including water sports, provides numerous physical, social, and psychological benefits to participants and spectators. Every year countless Bay-Delta region residents and visitors are drawn to Estuary 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 Estuary 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. Reliable and effective wastewater treatment occurs consistent with State and Federal standards throughout the Bay-Delta region, but wastewater treatment plant overflows occasionally occur in wet weather. Stormwater runoff is another pathway for input of pathogens into the Estuary, especially in wet weather.

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 the actual pathogens.

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 2014). 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. Routine bacteria monitoring does not occur at beaches in the Delta.

Toxins produced by blooms of harmful algae such as *Microcystis* are another threat to the health of people enjoying contact recreation in the Estuary. Although studies measuring algal toxins in the Estuary have been conducted, and thresholds developed by the state are available for assessment (OEHHA 2012), routine and systematic monitoring of algal toxins in the Estuary is not being conducted. A synthesis of the studies that have been performed was beyond the scope of the present report.

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 28 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. Special studies on bacterial contamination have been conducted by the Central Valley Water Board. Synthesis of this information was beyond the scope of this project.

Methods and Calculations Heal the Bay (2014) 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 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 1.

Results

Overall, the monitoring data and resulting grades (Table 2) indicate that conditions are excellent at most Estuary beaches most of the time. Conditions have been poor at 7% of beaches in summer, and 27% of beaches in wet weather at times during recent years..

Data for the summer beach season in 2013 are available for 28 beaches. In 2013, 22 of the 28 monitored beaches received an A or A+ grade, reflecting minimal exceedance of standards. Four of these beaches received an A+: Crown Beach Bath House, Crown Beach Windsurf Corner, Jackrabbit Beach and Candlestick Point, and Horseshoe Cove SW at Baker Beach. Most Bay beaches, therefore, are quite safe for swimming in the summer.

Six of the 28 beaches monitored in the summer in 2013 had grades of B or lower, indicating varying degrees of exceedance of bacteria standards. Aquatic Park and Lakeshore Park in San Mateo County received an F. These low grades indicate an increased risk of illness or infection.

Overall, the average grade for the 28 beaches monitored from April-October was an A-.

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, six of 22 beaches with data (27%) had grades of D or F. Many of the beaches (14 of 22, 64%), however, still had grades of A or A+. The overall average grade for these beaches in wet weather was a B (Table 2).

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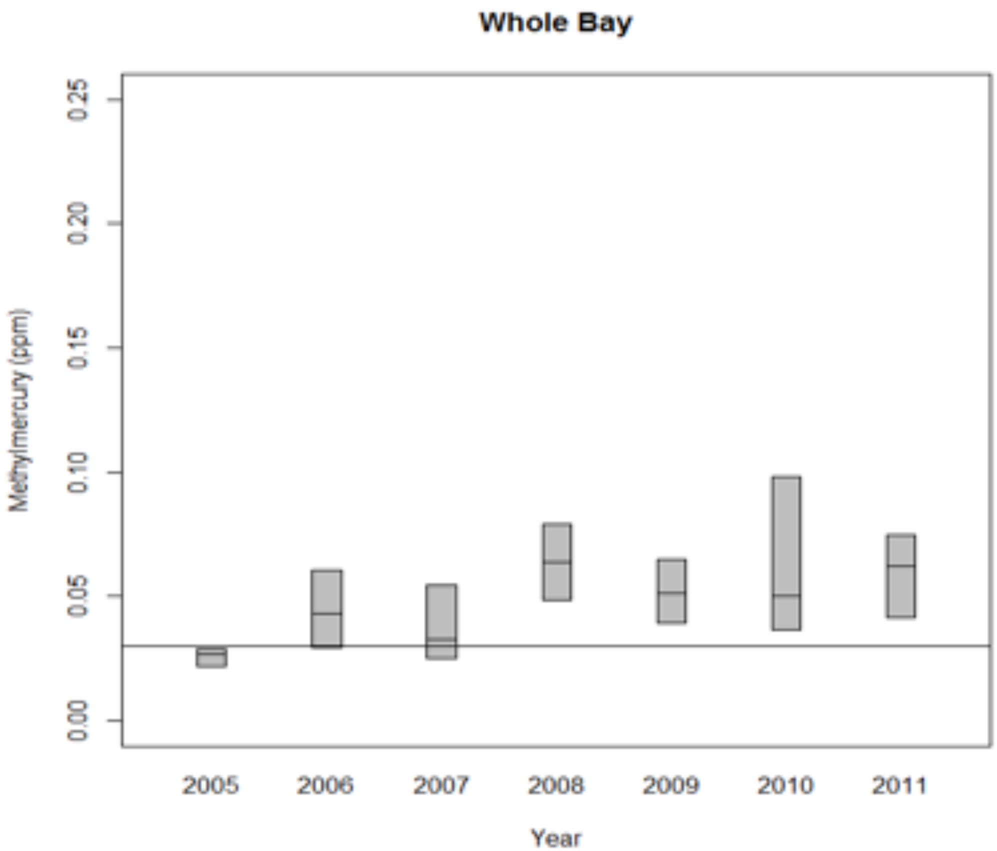
Figure 1. Methylmercury concentrations in small fish. Plots indicate the 25th, 50th, and 75th percentiles. Data for Mississippi silversides and topsmelt in the 3-5 cm size range sampled by the RMP. Reference line is the 0.030 ppm target from the Bay Mercury TMDL.



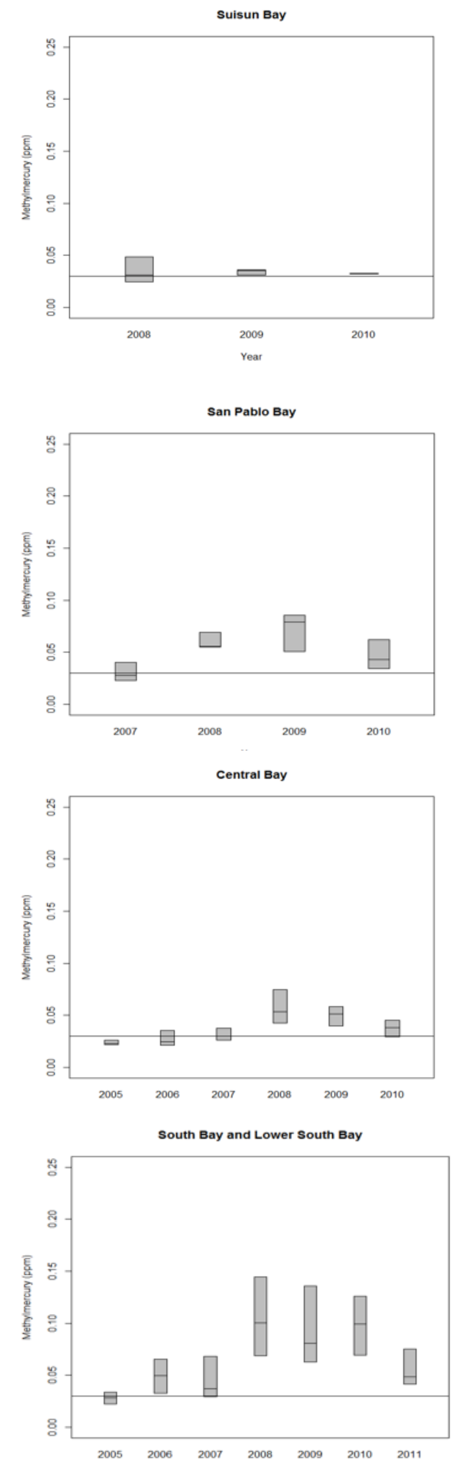
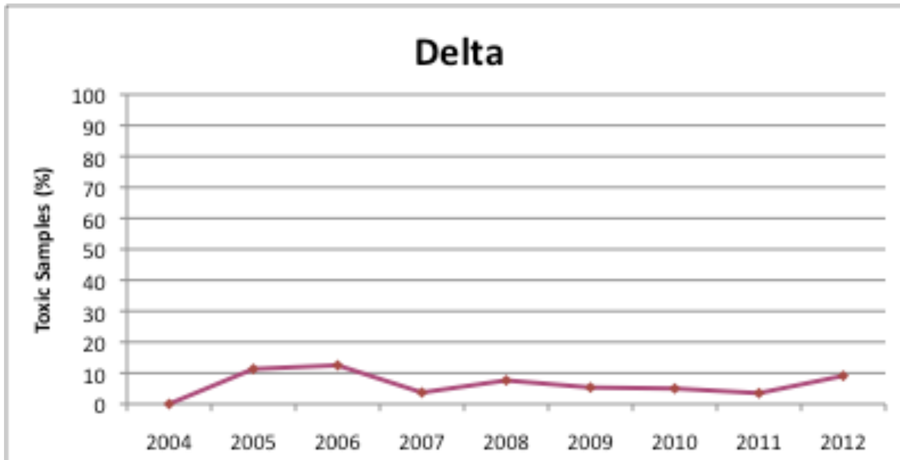


Figure 2. Percent of Estuary water samples exhibiting toxicity in laboratory assays. The RMP measured water toxicity in 2002, 2007, and 2011. In 2011, water toxicity was measured at 22 stations distributed throughout the Bay. Most of the samples are collected at randomly selected locations, with a few fixed historic stations included to continue long-term time series. The test species was the mysid shrimp *Americamysis bahia*. Water toxicity data for the Delta consisted of a collection of datasets from various programs, including one-time studies (e.g., the Central Valley Water Board's Delta Island Monitoring Project and the Central Valley Water Board's SWAMP Delta Pyrethroid Study) and annual monitoring performed under the Central Valley Water Boards Irrigated Lands Regulatory Program. Test species have included invertebrates (*Ceriodaphnia dubia, Eurytemora affinis, Hyalella azteca*, and *Americamysis bahia*) and fathead minnows (*Pimephales promelas*). The number of samples for each year varied considerably, with a low of five in 2004 and a high of 118 in 2008.



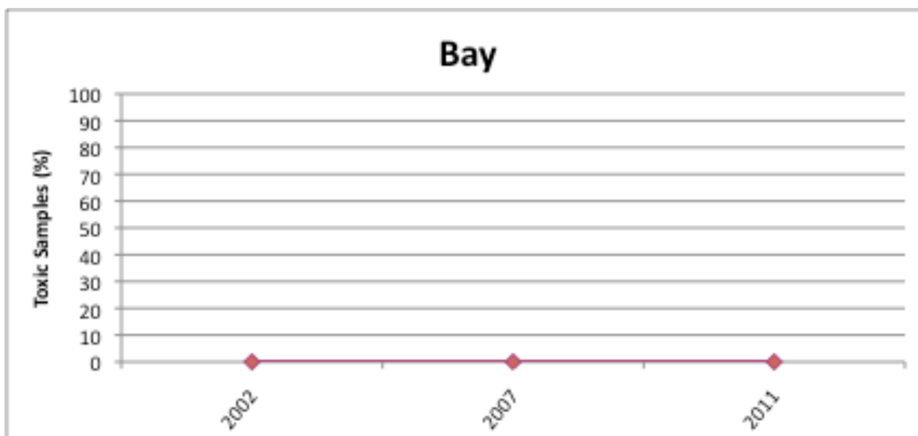


Figure 3. Dissolved copper concentrations in Bay water. Boxes indicate the 25th and 75th percentiles, whiskers the 5th and 95th percentiles. The water quality objective is a maximum of 6.9 ug/L in South Bay, and 6.0 ug/L in the other embayments.

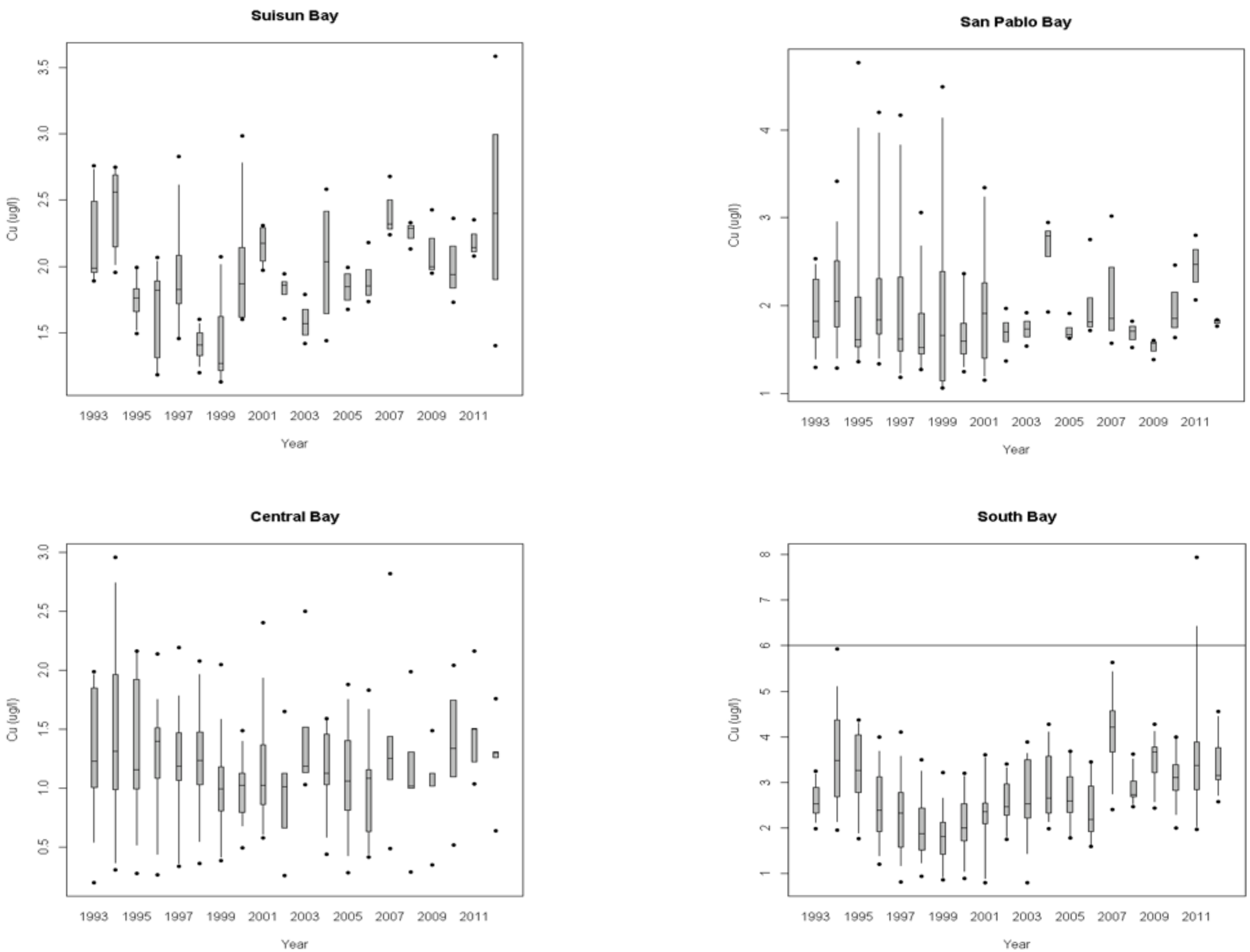


Figure 4. Locations of the five sampling stations in RMP sport fish monitoring and the six Delta stations sampled by SWAMP.

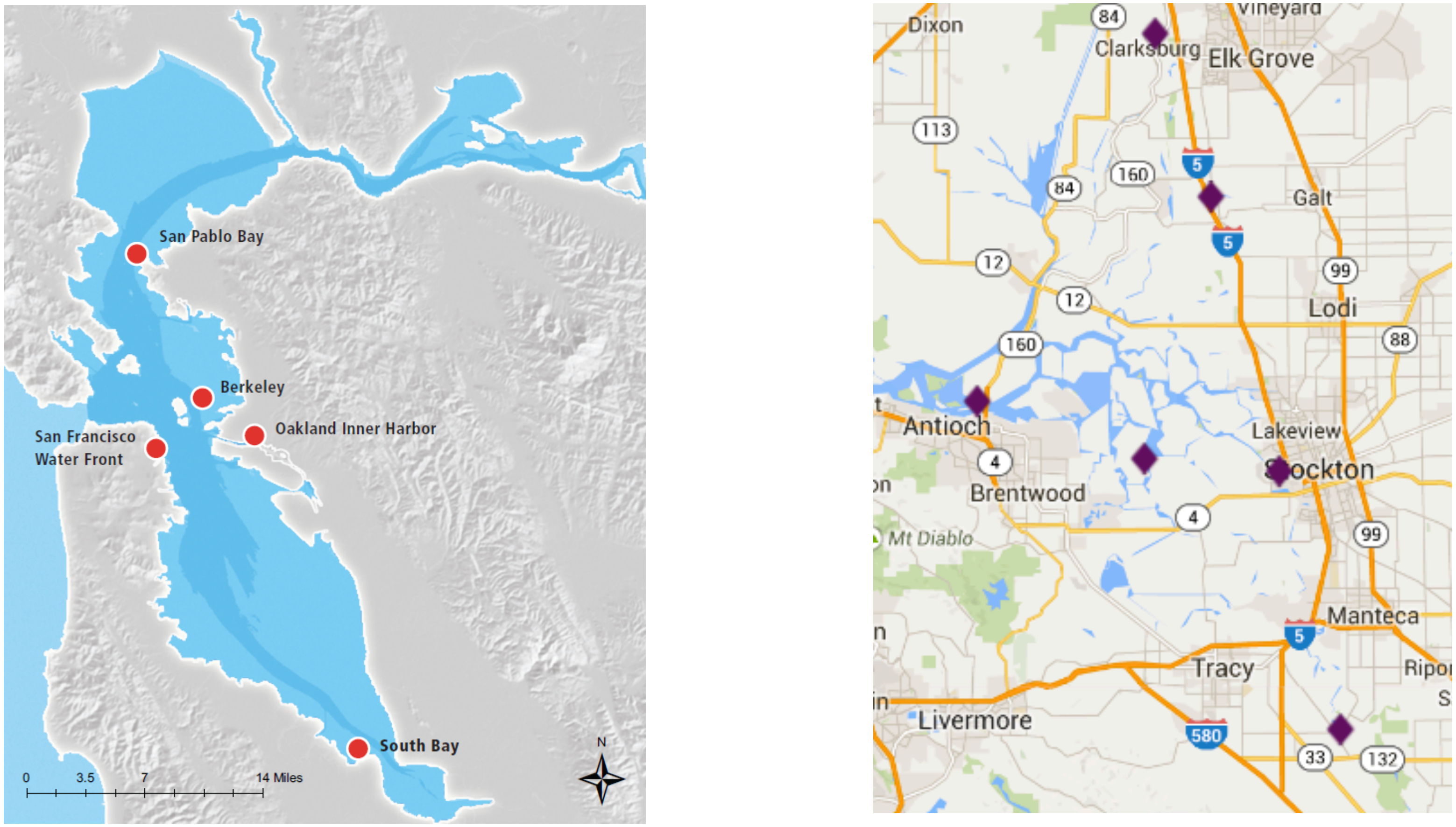


Figure 5. Average methylmercury concentrations in sport fish indicator species. Averages for striped bass based on concentrations for individual fish normalized to 60 cm. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for mercury is 0.44 ppm, and the two serving advisory tissue level is 0.07 ppm. Average concentrations for each species in the most recent sampling were between these two thresholds.

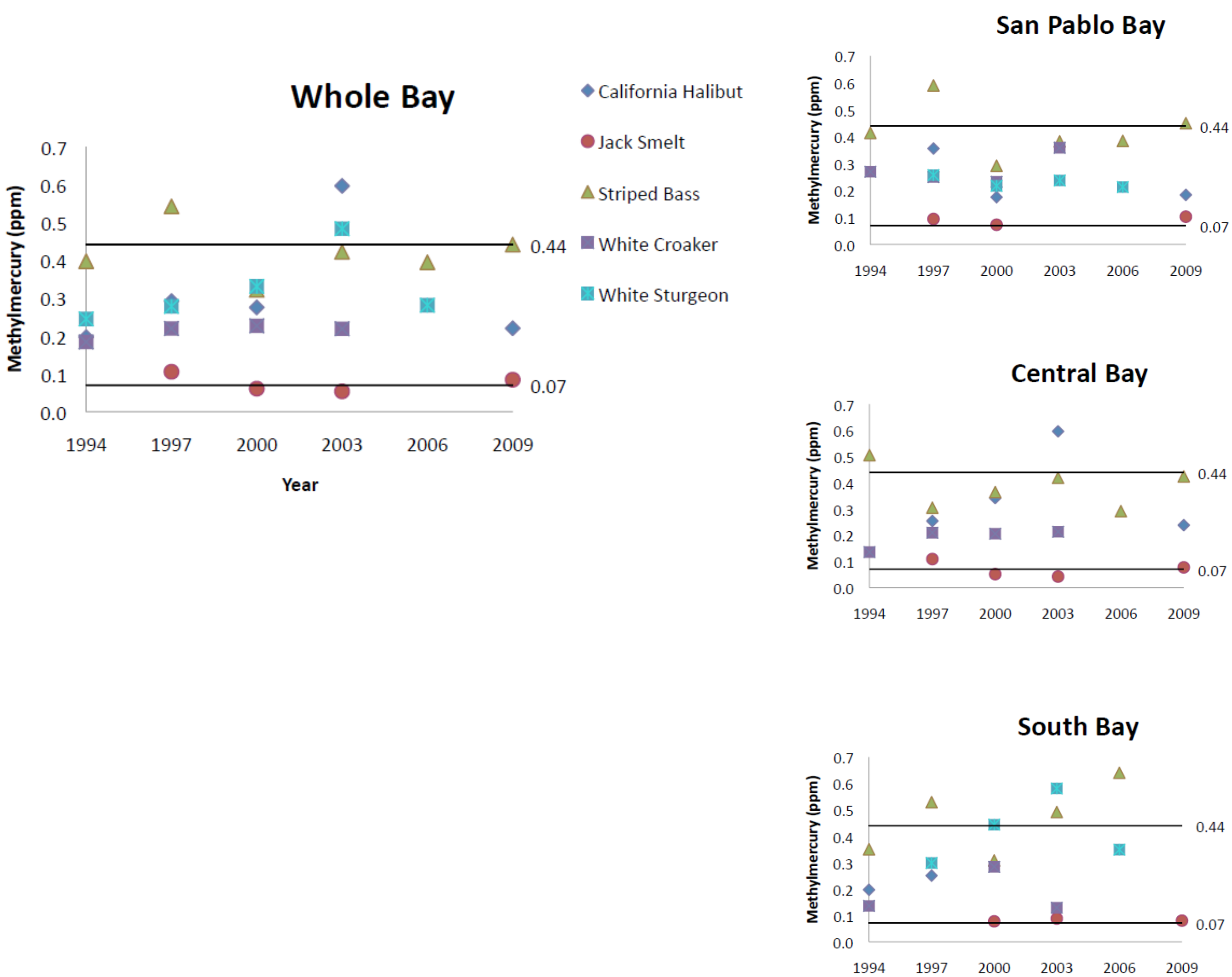


Figure 6. Average PCB concentrations in sport fish indicator species. Sport fish are not routinely sampled in Suisun Bay. The no consumption advisory tissue level for PCBs is 120 ppb, and the two serving advisory tissue level is 21 ppb. Average concentrations for both species in the most recent sampling 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.

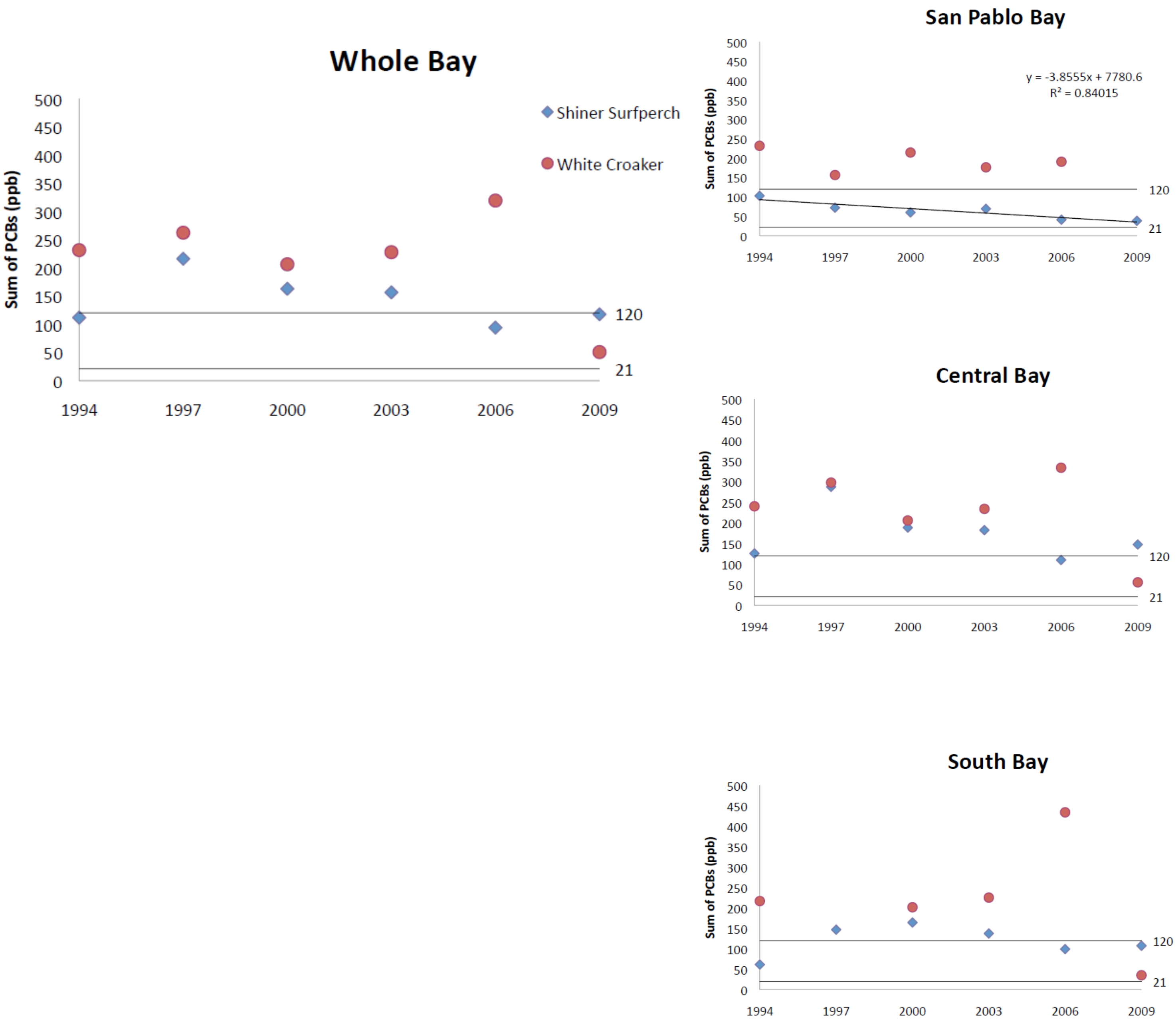


Figure 7. Average dioxin TEQ concentrations in shiner surfperch and white croaker, the key sport fish indicator species for organic pollutants. Sport fish are not routinely sampled in Suisun Bay. OEHHA has not established ATLs for dioxin TEQs. The San Francisco Bay Water Quality Control Board has developed 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.

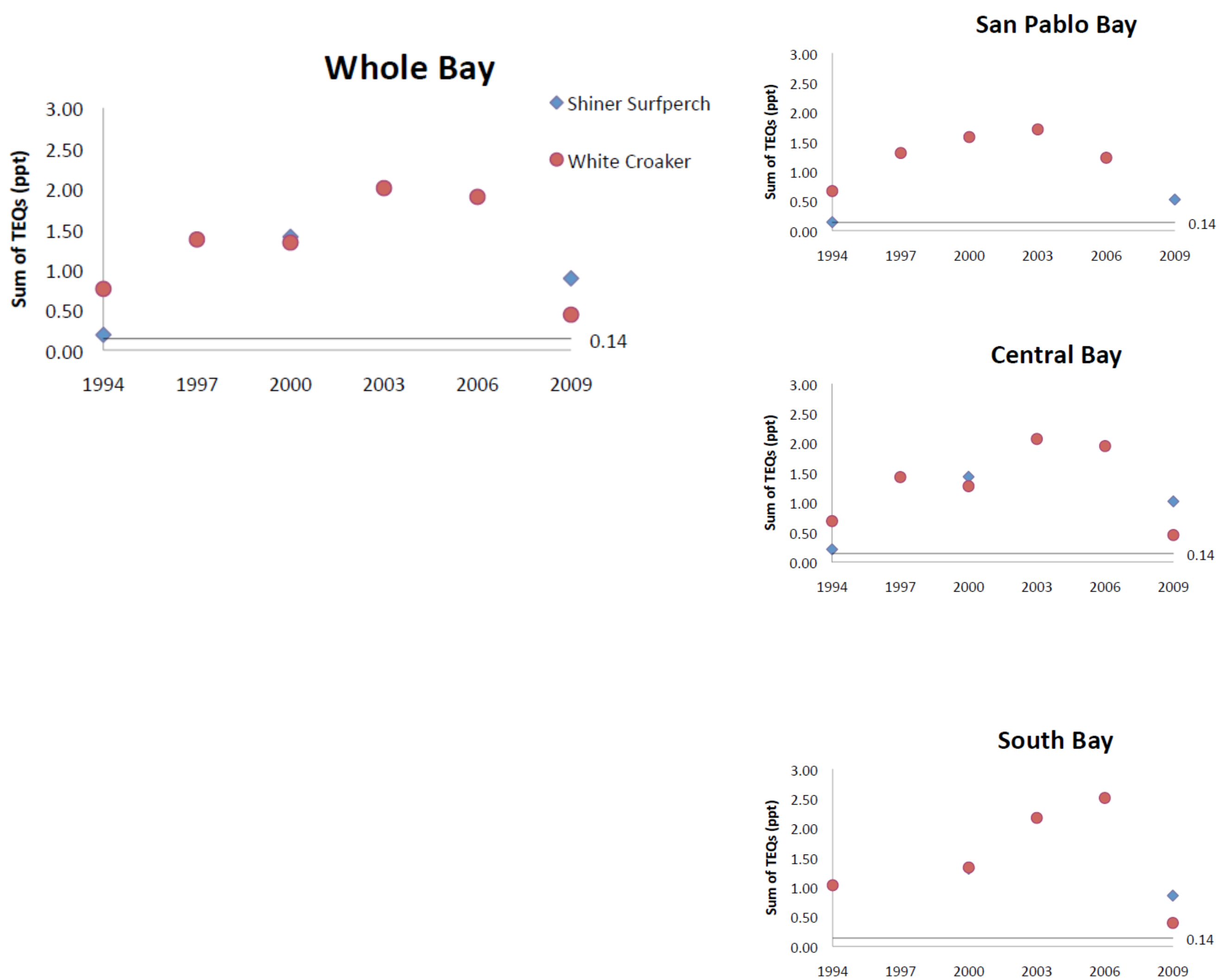


Table 1. California standards for fecal indicator bacteria.

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 2. Heal the Bay grades for San Francisco Bay Area beaches. From Heal the Bay (2014) and previous reports.

	2013			2012			2011			2010			2009		
San Mateo County	**Summer Dry (Apr-Oct)**	**Winter Dry (Nov-Mar)**	**Wet Weather (Year-Round)**	**Dry**	**Wet**	**ab411 (Apr-Oct)**	**Dry**	**Wet**	**ab411 (Apr-Oct)**	**Dry**	**Wet**	**ab411 (Apr-Oct)**	**Dry**	**Wet**	**ab411 (Apr-Oct)**
Oyster Point	A	A	D	A	D	A	A	B	D	A	D	A	A	F	B
Coyote Point	A	B	F	A	D	A	C	F	A	A+	C	A+	A	B	A+
Aquatic Park	F	F	F	F	F	F	F	F	F	D	F	D	F	F	F
Lakeshore Park	F	F	F	F	F	D	F	F	F	D	F	D	D	F	D
Kiteboard Beach	C	A	D	B	D	A	F								
Alameda Couty															
Alameda Point North	A		A+		C	A			A	A	C	A	A+	A	A+
Alameda Point South	A		A+		A+	A			A	A	A	A+	A	A	A
Crown Beach Crab Cove	A	F	A+												
Crown Beach Bath House	A+		A+		A+	A	A+	B	A	A+	A	A+	B	A	B
Crown Beach Windsurf Corner	A+		A+		A+	A		A	A	A+	B	A+	A	B	A
Crown Beach Sunset Road	A		A+		A	A	A	A	A	A+	B	A+	A	B	A
Crown Beach Shoreline Drive	A		A+		D	A	A	A	A	A	B	A+	A	C	A+
Crown Beach Bird Sanctuary	A	D	A		F	A	C	C	B	A	C	A	B	D	B
Contra Costa County															
Keller Beach North	A		A+		D	A	A	C	B	F	A	F	D	B	D
Keller Beach Mid-Beach									B	F	A	F	D	B	D
Keller Beach South	A		A		C	A	A	B	B	D	B	D	D	C	D
San Francisco County															
Crissy Field Beach West	A	A	A	A+	A+	A	A	A+	A	A	B	A+			
Crissy Field mid-Beach															
Crissy field Beach East	A	D	A	C	B	A	A	A	A	B	C	A+	A	B	A
Aquatic Park Beach 211 Station	A	C	A	F	A	A	B	A+	B	B	B	A	A	A	A
Aquatic Park Beach Hyde Street Pier	A	A	A+	A	A	A	A+	A	A	A	A	A+	A	A	A+
Candlestick Point Jackrabbit Beach	A+	A	C	A	A	A	A	D	B	A	B	A	A	C	A
Candlestick Point Windsurfer Circle	C	F	F	F	F	C	D	F	B	F	F	D	A	F	A
Candlestick Point Sunnydale Cove	B	A	C	D	C	A	B	F	A	C	F	D	C	F	B
Marin County															
Baker Beach Horseshoe Cove NE	A					A			B			A			A+
Baker Beach Horseshoe Cove NW	A					A			A			A			A
Baker Beach Horseshoe Cove SW	A+					A			A			A			A
Schoonmaker Beach	A					A			B			A+			A
Paradise Cove															
China Camp	A					A			A			A			A
McNears Beach	B														

State of the Estuary Report 2015

Summary

WATER QUANTITY – Freshwater Inflow Indicators and Index
Summary

Prepared by Christina Swanson
Natural Resources Defense Council
September 2015

State of the San Francisco Estuary 2015

WATER QUANTITY – Freshwater Inflow Indicators and Index Summary

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What are the indicators?

The Freshwater Inflow Index uses ten indicators to measure and evaluate the amounts, timing, and variability of freshwater inflow from the Sacramento-San Joaquin watershed to the Delta and the Bay. These indicators are designed specifically to look at various aspects of freshwater inflow conditions in the estuary, not the aquatic habitat conditions or ecological processes that result from or are affected by inflow. The ten indicators are also aggregated into a Freshwater Inflow Index, which combines the results of all the indicators into a single metric.

Five indicators measure aspects of the amounts of freshwater flow into the Delta and the Bay:

- Annual Delta Inflow;
- Spring Delta Inflow;
- San Joaquin River Inflow;
- Annual Bay Inflow; and
- Spring Bay Inflow.

One indicator measures the amount of water diverted directly from the Delta:

- Delta Diversions.

Four indicators measure the variability of freshwater flows into the Bay:

- Inter-annual Variation in Inflow;
- Seasonal Variation in Inflow;
- Peak Flow; and
- Dry Year Frequency.

In order to account for the watershed’s large year-to-year variations in hydrology, all of the indicators are measures of the alterations in freshwater inflow conditions, rather than measures of absolute amounts of inflow. Most of the indicators are calculated as comparisons of actual freshwater flow conditions to the freshwater flow conditions that would have occurred if there were no dams or water diversions, referred to as “unimpaired” conditions. By incorporating unimpaired inflow as a component of the indicator calculation, the indicators are “normalized” to account for natural year-to-year variations in precipitation and runoff.

Table 1.

Attribute	Indicators	Benchmarks
Water quantity (freshwater inflow to the estuary)	Alteration in the amounts, timing, patterns and variability of freshwater inflow to the Delta and the Bay.	Benchmarks (or reference conditions) are based on scientific literature on environmental flow requirements for riverine and estuarine ecosystems, including “presumptive standards” proposed by Richter et al. (2011) for river flows to maintain ecological integrity, the California’s State Water Resources Control Board 2010 Flow Criteria report that identified flows needed to protect public trust resources, historical inflow conditions, and regulatory standards for inflows, Delta diversion levels, and water quality.

Why is freshwater inflow important?

Estuaries are defined by the amounts, timing and patterns of freshwater inflow. In the San Francisco Bay estuary, freshwater inflows control the quality and quantity of estuarine habitat drive key ecological processes and significantly affect the abundance and survival of estuarine biota, from tiny planktonic plants and animals to shrimp and fish. The mixing of inflowing fresh water and saltwater from the ocean creates low salinity, or “brackish” water habitat for estuary-dependent species. Seasonal and inter-annual changes in inflow amounts trigger biological responses like 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.

Freshwater inflows to the San Francisco Bay estuary from its largest watershed, the Sacramento-San Joaquin watershed, are affected by a number of factors, including:

- Precipitation and runoff – flow amounts can vary from year to year by as much as an order of magnitude between wet and dry years;
- Dams – which capture and store runoff from the mountains for release into rivers at different times of the year and in different years);
- In-river diversions – which remove water from rivers for local agricultural or urban use or export to other regions in California, reducing the amount of water that flows to the estuary;
- Return flows and discharges – which add (or return) water to river flows, although water quality may be reduced by contaminants from agricultural runoff or wastewater;
- In-Delta diversions – which remove water from the upper reach of the estuary for local agriculture and urban use and for export to other regions in California, reducing the amount of water that flows from the Delta into the Bay;
- Climate change – warmer temperatures and shifts in precipitation from snow to rain have altered the amounts, timing and duration of seasonal flows in the estuary’s tributary rivers.

What are the benchmarks? How were they selected?

The benchmarks for the ten indicators were based on: 1) scientific literature on environmental flow requirements for riverine and estuarine ecosystems, including "presumptive standards" proposed by Richter et al (2011) for river flows to maintain ecological integrity (i.e., 80% of unimpaired flow as needed to maintain ecological integrity); 2) the California's State Water Resources Control Board 2010 Flow Criteria report that identified flows needed to protect public trust resources (i.e., 75% of unimpaired flow during winter and spring); 3) historical inflow conditions (i.e., before completion of major dams); and 4) SWRCB regulatory standards for inflows and Delta diversion levels.

What are the status and trends of the indicators and Index?

Freshwater inflows to the Delta and Bay have been highly altered, resulting in degradation of ecological condition and function in the estuary. The magnitude of alteration has increased for 9 of the 10 indicators during the 85-year record (and since development of dams and water diversion facilities and operations) and, for 5 of 10 indicators, even further during the last decade. Current freshwater inflow conditions are "very poor" for 6 of 10 indicators, "fair" for 3 indicators and "good" for only one indicator. As measured by the Freshwater Inflow Index, which combines the results of the 10 indicators into a single metric, freshwater inflow conditions for the San Francisco Bay Estuary are "poor."

Table 2.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend (long term; 1930-2014)**	**Trend since 1990**	**Current condition** (average for last 10 years)
Annual Delta Inflow	Partially met; goals achieved in 52% of years	Stable	Stable	Fair Inflow reduced by 26%
Spring Delta Inflow	Not met; goals achieved in 12% of years	Decline	Deteriorating	Poor Inflow reduced by 47%
San Joaquin River Inflow	Not met; goals achieved in 0% of years	Decline	Stable	Very poor Inflow reduced by 58%
Annual Bay Inflow	Not met: goals achieved in 12% of years	Decline	Deteriorating	Very poor Inflow reduced by 50%
Spring Bay Inflow	Not met; goals achieved in 12% of years	Decline	Deteriorating	Very poor Inflow reduced by 56%
Delta Diversions	Not met; goals achieved in 8% of years	Decline	Deteriorating	Poor 36% of inflow diverted
Inter-annual Variation in Inflow	Partially met; goals achieved in 40% of years	Decline	Mixed (variable)	Good Reduced by 10%
Seasonal Variation in Inflow	Not met; goals achieved in 28% of years	Decline	Deteriorating	Poor Reduced by 50%
Peak Flow	Partially met; goals achieved in 44% of years	Decline	Stable	Fair Reduced by 45 days/year

Dry Year Frequency	Partially met: goals met in 52% of years	Decline	Deteriorating	Poor Flow reductions triple dry year frequency
Freshwater Inflow Index	Not met; goals met in 12% of years	Decline	Mixed (variable)	Poor Only 1 of 10 indicators show “good” conditions

What does it mean? Why do we care?

Freshwater inflow to an estuary is a key physical and ecological driver, affecting the quality and quantity of habitat, primary and secondary productivity, and growth and survival of resident and migratory fish and wildlife. In recent years, freshwater inflows to the San Francisco Estuary have been cut by half on an annual basis and by 60% during the ecologically important spring season, and inter-annual and seasonal variability in inflows have been reduced. These man-made alterations in inflows have created chronic drought conditions in the estuary that, particularly in the estuary’s upstream region, impair ecological function, degrade habitat and productivity, and are a key contributor to increasingly serious fish population declines.

State of the Estuary Report 2015

Technical Appendix

WATER QUANTITY – Freshwater Inflow Indicators and Index
Technical Appendix

Prepared by Christina Swanson
Natural Resources Defense Council
September 2015

State of the San Francisco Estuary 2015

WATER QUANTITY – Freshwater Inflow Indicators and Index
Technical Appendix

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September 2015

I. Background

The San Francisco Bay Estuary, which extends upstream from the Golden Gate south to the South Bay and east through San Pablo Bay, Suisun Bay and the Delta to the limit of tidal influence in the Sacramento, Mokelumne and San Joaquin rivers, is the interface between California's largest rivers and the Pacific Ocean. It 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.

Estuaries are defined by the amounts, timing and patterns of freshwater inflow. In the San Francisco Bay estuary, freshwater inflows control the quality and quantity of estuarine habitat drive key ecological processes and significantly affect the abundance and survival of estuarine biota, from tiny planktonic plants and animals to shrimp and fish (Jassby et al. 1995; Kimmerer 2002, 2004; Kimmerer et al. 2008; Feyrer et al. 2008, 2010; Moyle and Bennett, 2008; Moyle et al., 2010; SWRCB 2010; and see Open Water Habitat and Flood Events indicators). The mixing of inflowing fresh water and saltwater from the ocean creates low salinity, or "brackish" water habitat for estuary-dependent species. Seasonal and inter-annual changes in inflow amounts trigger biological responses like 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 Estuary comes from the Sacramento and San Joaquin river basins, which provide >90% of total inflow in most years and have large impacts on salinity regimes in the estuary (Kimmerer 2002, 2004). 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.

Freshwater inflows to the Delta and the Bay from the Sacramento-San Joaquin watershed are affected by a number of factors, including:

- Precipitation and runoff – flow amounts can vary from year to year by as much as an order of magnitude between wet and dry years;
- Dams – which capture and store runoff from the mountains for release into rivers at different times of the year and in different years, and can change variability of seasonal

and inter-annual flows (nine of the ten largest Sacramento-San Joaquin watershed tributaries to the estuary are dammed and managed for flood control and water supply);

- In-river diversions – which remove water from rivers for local agricultural or urban use or export to other regions in California, reducing the amount of water that flows to the estuary;
- Return flows and discharges – which add (or return) water to river flows (return flow and discharge amounts are usually smaller than the amounts of water diverted);
- In-Delta diversions – which remove water from the upper reach of the estuary for local agriculture and urban use and for export to other regions in California, reducing the amount of water that flows from the Delta into the Bay;
- Climate change – warmer temperatures and shifts in precipitation from snow to rain have altered the amounts, timing and duration of seasonal flows in the estuary's tributary rivers.

The State of the Estuary Report uses ten indicators to measure and evaluate the amounts, timing and patterns of freshwater inflow from the Sacramento-San Joaquin watershed to the Delta and the Bay. These indicators are designed specifically to look at various aspects of freshwater inflow conditions in the estuary, not the aquatic habitat conditions or ecological processes that result from or are affected by inflow. The ten indicators are also aggregated into a Freshwater Inflow Index, which combines the results of all the indicators into a single metric.

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In order to account for the watershed's large year-to-year variations in hydrology, all of the indicators are measures of the alterations in freshwater inflow conditions, rather than measures of absolute amounts of inflow. Except for the Delta Diversions indicator, all of the indicators are calculated as comparisons of actual freshwater flow conditions to the freshwater flow conditions that would have occurred if there were no dams or water diversions, referred to as "unimpaired" conditions. By incorporating unimpaired inflow as a component of the indicator calculation, the indicators are "normalized" to account for natural year-to-year variations in precipitation and runoff. The Delta Diversions indicator compares Delta inflows to Delta outflows.

II. Data Sources and Definitions

A. Data Sources

Because most of the fresh water that flows into the San Francisco Bay Estuary comes from the Sacramento, Mokelumne and San Joaquin river basins (collectively the Sacramento-San Joaquin watershed), which provide >90% of total inflow in most years,[1] all of the Freshwater Inflow indicators were calculated using flow data from the Sacramento-San Joaquin watershed only.

The indicators were calculated for each year[2] using data from the California Department of Water Resources (CDWR) DAYFLOW model (for "actual flows), CDWR's Central Valley Streams Unimpaired Flows, and the California Data Exchange Center's (CDEC) Full Natural Flows (FNF) datasets (for "unimpaired flows"). DAYFLOW is a computer model developed in 1978 as an accounting tool for calculating daily historical Delta inflow, outflow and other internal Delta flows.[3] DAYFLOW output is used extensively in studies by State and federal agencies, universities, and consultants. DAYFLOW output is available for the period 1930-2014.[4] Annual and monthly unimpaired flow data for total Delta inflow, Delta outflow and San Joaquin River inflow are from the CDWR California Central Valley Unimpaired Flow dataset (1921-2003).[5] For 2004-2014, annual and seasonal unimpaired flows were calculated by regressions developed from the Central Valley unimpaired flow data (using the 1930-2003 period) and the corresponding unimpaired runoff estimates from the CDEC Full Natural Flows dataset[6] for the ten largest rivers in the watershed (for Delta inflows and outflows) and the four major San Joaquin Basin rivers for San Joaquin River inflows.[7] Figure 1 shows regressions of CDWR's unimpaired flows on Full Natural Flows for annual and spring (Feb-June) Delta inflow, annual and spring Delta outflow, and San Joaquin River inflow.

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[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] More information about DAYFLOW is available at www.water.ca.gov/dayflow.
[4] For actual flows, various indicators used DAYFLOW parameters for QTOT (for total Delta inflow), QOUT (net Delta outflow), and QSJR (San Joaquin River inflow).
[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, Tuolumne, Merced and San Joaquin Rivers. For the San Joaquin basin, the four rivers are the Stanislaus, Tuolumne, Merced and San Joaquin Rivers.

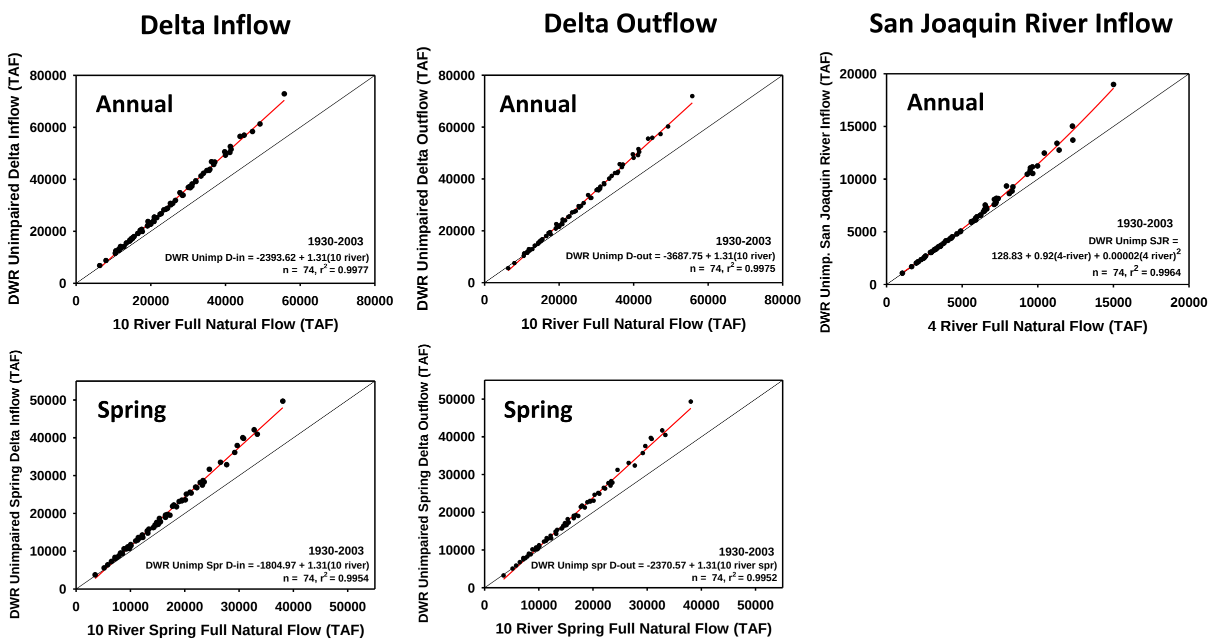

Figure 1. Regressions of CDWR unimpaired flows on Full Natural Flows for annual and spring (Feb-June) Delta inflow, annual and spring Delta outflow, and annual San Joaquin River inflow, 1930-2003.
Data sources: California Department of Water Resources, Central Valley Streams Unimpaired Flows, and California Data Exchange Center, Full Natural Flows.

B. Tidal Effects on Flows in the Delta

Flows in Delta channels and the Bay are influenced by tidal action as well as freshwater inflows from upstream and in-Delta diversions. The estuary experiences two tides every day, two high tides and two low tides, and magnitude of the high and low tides varies over a 28-day spring-neap cycle. Under conditions of low to moderate inflows, tidal flows in Delta channels can be an order of magnitude greater than the freshwater inflow and the direction of flow in the channels typically reverses twice daily with the tides. However, all daily flow data used to calculate the indicators (i.e., Dayflow data) have been filtered to remove tidal effects.

C. Definitions

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 and Delta, would have flowed into the Delta or Bay (see Figure 2). Unimpaired inflow is not the same as “natural” or “historical” inflow that would have occurred in the watershed prior to human development and land use changes; it is instead an estimate of what flows over the *existing landscape* would have been if there were no dams or diversions.

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, particularly in very wet years and during periods of high flows.

Post-water Development Inflow: Most of the major dams and water diversion facilities (such as the state and federal Delta pumping facilities) were completed and operational by 1970. Water export rates at the Delta pumping facilities increased rapidly during the 1970s, reaching "full operation" with export rates leveling off by 1980.

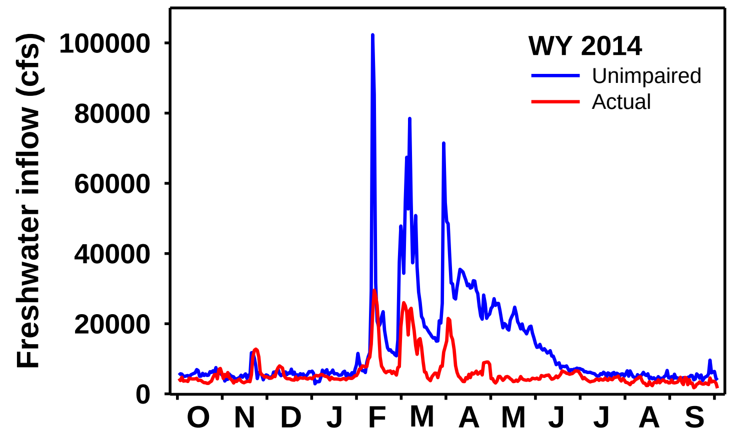

Figure 2. Unimpaired and actual freshwater inflows to the San Francisco Estuary in Water Year 2014.
Data sources: California Department of Water Resources, Dayflow, and California Data Exchange Center (CDEC), Full Natural Flows.

Delta Inflow vs. Bay Inflow: Delta inflow is the amount of water that flows into the Delta from the Sacramento-San Joaquin watershed. Bay Inflow (or Delta outflow) is the amount of water that flows from the Delta into the Suisun Bay region of San Francisco Bay. Bay inflow amounts are less than Delta inflow amounts because in-Delta diversions by local water users and the state and federal water export facilities remove of portion of Delta inflow before it reaches the Bay.

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, annual unimpaired inflows were classified for each year as one of five water year types: very wet, wet, median, dry and very dry. Year types were established based on frequency of occurrence during the period of 1930-2009, with each year

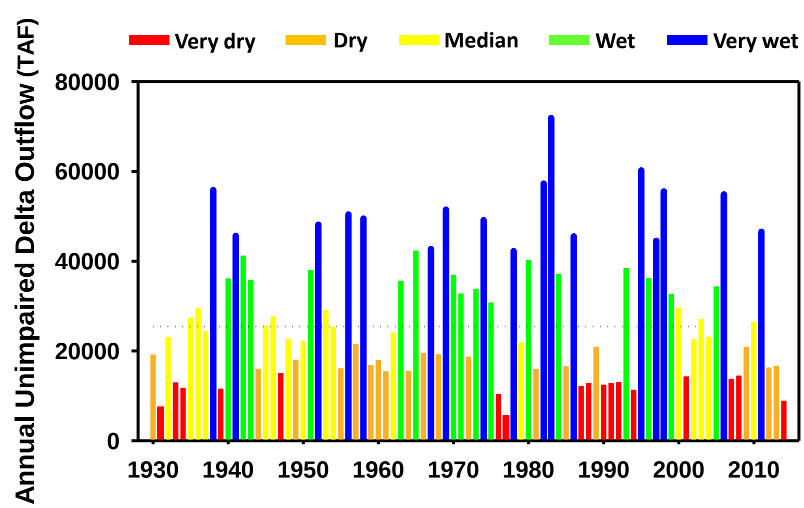

Figure 3. Annual unimpaired Delta outflow (TAF) for 1930-2014. Bars are colored to show frequency-based water year type (see text). Dotted line shows median unimpaired Delta outflow for the 1930-2003 period.
Data source: California Department of Water Resources, Central Valley Streams Unimpaired Flows.

type comprising 20% of all years. Figure 3 shows annual unimpaired Delta outflows to the Bay with year type classification shown by the different colors of the bars.

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" (SFEP 2007). These goals are non-quantitative; therefore we used information from the scientific literature, current regulatory standards and objectives, and historical and/or unimpaired conditions to identify and define levels of freshwater flows that promote restoration and enhance ecological function and resiliency.

There is a growing body of scientific literature on environmental flow requirements for riverine and estuarine ecosystems, including Arthington et al. (2006), Poff et al. (2010) and Richter et al. (2011). In particular, Richter et al. (2011) proposed conservative and precautionary "presumptive standards" for river flows to maintain ecological integrity, identifying 80% of unimpaired flow as needed to maintain ecological integrity and 90% of unimpaired to protect rivers with at-risk species.[8] In addition, 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 flow from the Sacramento-San Joaquin watershed should flow out of the Delta and into the Bay during the winter and spring seasons and that winter and spring lower San Joaquin River flows should be 60% of unimpaired San Joaquin River flow (SWRCB 2010).[9] The SWRCB has also established regulatory standards for minimum flow and maximum diversion levels for the Delta and Bay (SWRCB 2006). Information on historical conditions, prior to major water development in the watershed, was derived from DAYFLOW data from the pre-dam period.

For each indicator, a primary reference condition, the quantitative value against which the measured value of the indicator was compared, was established. For most of the indicators, this reference condition was developed based on recommendations of either Richter et al. (2011) or SWRCB (2010). The SWRCB 2006 regulatory standards (SWRCB 2006), pre-dam flow conditions and various metrics from unimpaired flow data (e.g., variability) were also used to inform development of reference conditions for some indicators. Measured indicator values that were higher than the primary reference condition were interpreted to mean that aspect of freshwater inflow condition, as measured by the indicator, met the CCMP goals and corresponded to "good" ecological conditions in that year. For the most recent 25 year period (since 1990, when the CCMP was being developed and established), CCMP goals were considered to be "fully met" is indicators met or exceeded he primary reference conditions in at

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[8] The standards proposed by Richter et al. (2011) were for daily flows.

[9] 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. On an annual basis, the majority of runoff in the watershed and unimpaired flows occur in the winter and spring.

least 67% of years; "partially met" if the indicators met or exceeded this level in 33-66% of years; and "not met" if indicators met or exceeded this level in less than 33% of years.

In addition to the primary reference condition, information on the range and trends of indicator results, results from the scientific literature and other watersheds, and known relationships between freshwater inflow conditions and physical and ecological conditions in estuaries was used to develop several intermediate reference conditions. The intermediate reference conditions were used to create a five-point scale that categorized and assigned a quantitative "score" to the indicator's measured value, ranging from zero (0), which was considered to correspond to "very poor" conditions with highly altered flow conditions, to four (4), which was considered to correspond to "excellent" conditions with minimally altered flow conditions. The primary reference condition was assigned a point value of three (3), corresponding to flow conditions that had been altered but which were sufficient to maintain ecological integrity and thus meet the CCMP goals. The size of the increments between the different 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. For each year, these scores of the ten indicators were averaged to calculate the Freshwater Inflow Index. Specific information on the primary and intermediate reference conditions for each indicator is provided in the following sections describing each of the indicators.

The results for each indicator and the Index are shown graphically, with all graphs showing the results for each year and each decade (e.g., 1950-1959). All graphs show the measured indicator (or Index) values and the indicator score using a consistent orientation on the Y axis, with values corresponding to good conditions shown above values corresponding to poorer conditions on the Y axis regardless of the unit of measure or numeric scale. To evaluate trends and differences over time and between other variables (e.g., water year types), indicator and Index results were analyzed using t-tests, analysis of variance and simple linear regression.

IV. Freshwater Inflow Indicators

A. Annual Delta Inflow

1. Rationale

The Delta receives freshwater inflow from more than a dozen rivers and streams, including the Sacramento, Mokelumne, Cosumnes, Calaveras and San Joaquin Rivers, as well as a number of smaller tributaries from the west side of the Sacramento Valley (including Putah and Cache Creeks). Collectively, these rivers drain more than 40% of the California landscape, from the Cascade Mountains in the north to the southern Sierra Nevada. From year to year, the amounts of flow from these rivers into the Delta can vary more than ten-fold, reflecting California's temperate climate and unpredictable cycle of droughts and floods. By the mid-1900s, nearly all of these rivers were dammed for water storage, flood control and/or hydropower, altering the amounts and timing of freshwater flows into the Delta. Runoff from rainstorms and the melting mountain snowpack that formerly flowed into the Delta in the winter, spring and early summer is now captured behind massive dams, and diverted from rivers and reservoirs for local and distant use. Flow from some rivers, such as the upper San Joaquin and the Calaveras, no longer even reaches the Delta in many years. In contrast, in some years (and in some seasons), water captured and stored in reservoirs in previous years is released and flows in to the Delta in excess of what would have flowed into the Delta under unimpaired conditions.

2. Methods and Calculations

The Annual Delta Inflow indicator measures the total amount of fresh water that flowed into the Delta each year from all of its tributary rivers, compared to the amount that would have flowed into the Delta from these rivers under "unimpaired" flow conditions, without the effects of dams or water diversions, for that year. Capture and storage of watershed runoff for release in subsequent years and diversion of water from the Delta's tributary rivers reduces annual Delta inflow; release of water captured and stored in watershed reservoirs in previous years and imports of water from the Trinity River watershed increase annual Delta inflow.

The indicator was calculated for each year (1930-2014) as the percentage of annual unimpaired Delta inflow that flowed into the Delta using the following equation:

Annual Delta Inflow indicator (% of unimpaired)
= (actual annual Delta inflow/unimp. annual Delta inflow) x 100

3. Reference Conditions

The primary reference condition for the Annual Delta Inflow indicator was established as 80%, the level identified by Richter et al. (2011) as needed to maintain the ecological integrity of most rivers. Annual inflows that were greater than 80% of unimpaired inflows were considered to reflect "good" conditions and meet the CCMP goals; annual inflows that were less than 50% of unimpaired inflows were considered to correspond to "very poor" conditions. The other reference condition levels were established based on Richter et al. (2011; 90% of unimpaired to protect rivers with at-risk species for "excellent" and minimally altered flows) and use of equal increments between the primary and lowest reference condition levels. Table 1 below shows the quantitative reference conditions that were used to evaluate the results of the Delta Inflow indicator.

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

Annual Delta Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>90% of unimpaired	"Excellent," minimal alteration	4
>80% of unimpaired	***"Good," meets CCMP goals***	***3***
>65% of unimpaired	"Fair"	2
>50% of unimpaired	"Poor"	1
≤50% of unimpaired	"Very Poor," extreme alteration	0

4. Results

Results of the Annual Delta Inflow indicator are show in Figure 4.

The total amount of fresh water flowing into the Delta each year has been reduced in almost all years.
On an annual basis, the percentage of the freshwater runoff from Sacramento-San Joaquin watershed that flows into the Delta has been reduced, averaging 78% of unimpaired Delta inflow for the period of 1930-2014. The greatest reduction in annual Delta inflow occurred in 2009, the third year of the recent three-year drought, when only 52% of unimpaired inflow reached the Delta. In 1976, a very dry year, annual Delta inflow was greater than it would have been under unimpaired conditions, 111% of unimpaired inflow, reflecting large releases of water stored in earlier years from Sacramento basin reservoirs. For the most recent 10-year period (2005-2014), an average of 74% of unimpaired inflow actually flowed into the Delta, similar to the amount for 2014, 75%; this level of freshwater inflow to the Delta corresponds to "fair" condition.

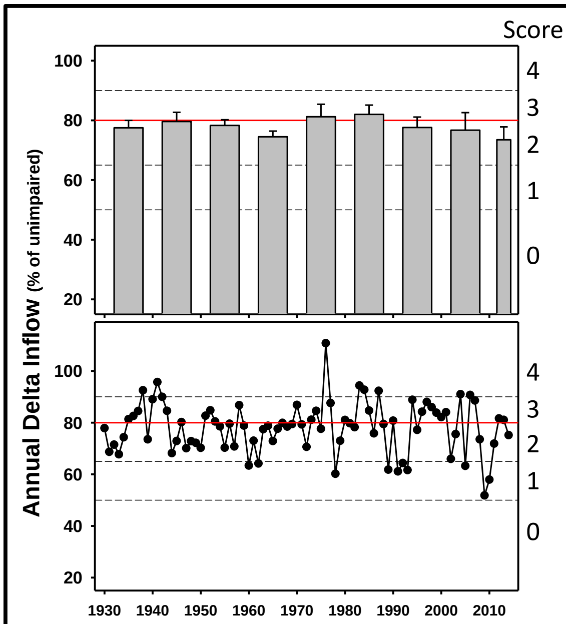

Figure 4. Results for the Annual Delta Inflow indicator, expressed as the percentage of unimpaired flow that actually flowed into the Delta for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The proportional reductions in annual Delta inflow to the estuary differ by water year type.
In general, the annual Delta inflow is higher in very wet years than in drier years. The greatest alterations to Delta inflow occur in dry years, when an average of 26% of unimpaired flow is diverted before reaching the Delta, significantly more than the 17% of unimpaired Delta inflow diverted in very wet years (ANOVA, $p<0.05$).

Annual freshwater flow into the Delta, as a percentage of unimpaired flow, has not changed over time.
The percentage of unimpaired flow that actually flowed into the Delta has not significantly changed over the past eight decades (regression, $p=0.7$). Since 1980, an average of 5.1 (±4.1 SD) million acre feet of water was diverted from the Sacramento-San Joaquin watershed before it reached the Delta.

Based on annual Delta inflows, CCMP goals to increase fresh water availability to the estuary have been partially met.
Since 1990, annual freshwater inflows to the Delta were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in 52% of years (13 of 25 years). Current

freshwater inflows to the Delta are generally comparable to the 80% of unimpaired level recommended by Richter et al. (2011) to maintain ecological integrity. However, annual Delta inflows in some recent years have been substantially below this level and lower than the lowest levels measured in previous decades. In addition, this indicator does not reflect within-year, or seasonal, alterations, which can be substantial.

B. Spring Delta Inflow

1. Rationale

Historically, two thirds of total annual freshwater inflow to the Delta occurred during the spring, as snow in the northern and central California mountain ranges melted and filled the Delta's tributary rivers. Prolonged high flows during this period are still the dominant feature of Estuary's hydrograph, the annual picture of the timing and amounts of flow (see Figure 2). However, since the early 1900s, growing numbers of large storage and flood control dams on most of the Delta's tributary rivers captured much of the snowmelt runoff for use later in the year, reducing Delta inflows during the spring (and increasing inflows during the summer and fall). Additionally, regulatory protections for flow, water quality and fisheries standards (SWRCB 2006) that reduce the percentage of Delta inflow that can be diverted by the state and federal export facilities have influenced management of seasonal reservoir releases.

2. Methods and Calculations

The Spring Delta Inflow indicator measures the total amount of fresh water that flowed into the Delta from all of its tributary rivers during the spring (February-June) of each year, compared to the amount that would have flowed into the Delta from these rivers under unimpaired flow conditions during that period, without the effects of dams or water diversions. Capture and storage of springtime watershed runoff for release later in the year or in subsequent years and diversion of water from the Delta's tributary rivers reduces spring Delta inflow; springtime release of water captured and stored in watershed reservoirs earlier in the year or in previous years and imports of water from the Trinity River watershed increase annual Delta inflow.

The indicator was calculated for each year (1930-2014) as the percentage of spring unimpaired Delta inflow that flowed into the Delta using the following equation:

Spring Delta Inflow (% of unimpaired)
= (actual Feb-June Delta inflow/unimpaired Feb-June Delta inflow) x 100

3. Reference Conditions

The primary reference condition for the Spring Delta Inflow indicator was established as 80%, the level identified by Richter et al. (2011) as needed to maintain the ecological integrity of most rivers. Spring inflows that were greater than 80% of unimpaired inflows were considered to reflect "good" conditions and meet the CCMP goals; annual inflows that were less than 50% of unimpaired inflows were considered to correspond to "very poor" conditions. The other reference condition levels were established based on Richter et al. (2011; 90% of unimpaired to protect rivers with at-risk species for "excellent" and minimally altered flows) and use of equal increments between the primary and lowest reference condition levels. Table 2 below shows the quantitative reference conditions that were used to evaluate the results of the Spring Delta Inflow indicator.

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

Spring Delta Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>90% of unimpaired	"Excellent," minimal alteration	4
>80% of unimpaired	***"Good," meets CCMP goals***	***3***
>65% of unimpaired	"Fair"	2
>50% of unimpaired	"Poor"	1
<u><</u>50% of unimpaired	"Very Poor," extreme alteration	0

4. Results

Results of the Spring Delta Inflow indicator are show in Figure 5.

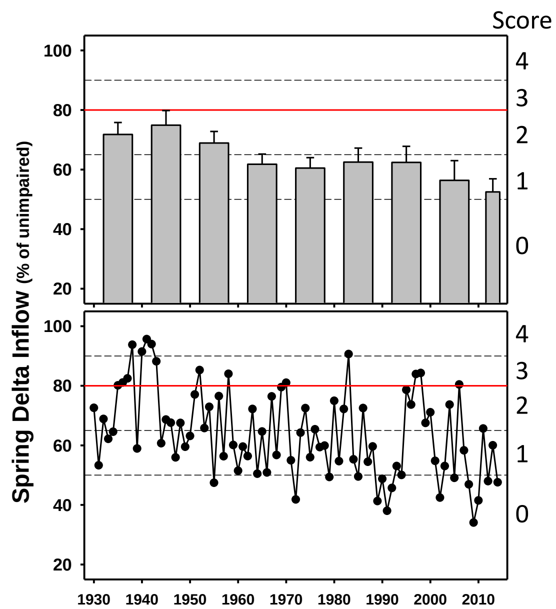

Figure 5. Results for the Spring Delta Inflow indicator, expressed as the percentage of spring (Feb-June) unimpaired flow that actually flowed into the Delta for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The amount of fresh water flowing into the Delta during the spring has been reduced.

The percentage of the springtime runoff from Sacramento-San Joaquin watershed that flows into the Delta has been significantly reduced. The greatest alteration in spring Delta inflow occurred in 2009, the third year of the recent three-year drought, when only 34% of unimpaired spring inflow reached the Delta. For the most recent 10-year period (2005-2014), on average only 53% of springtime unimpaired Delta inflow actually flowed into the Delta during the spring. During this period, spring Delta inflows were "good," greater than 80% of unimpaired, in only one year and "very poor," less than 50% of unimpaired in six years. In 2014, only 48% of unimpaired spring inflow reached the Delta, corresponding to "very poor" conditions.

The proportional reductions in spring inflow to the Delta differ by water year type.

The greatest alterations to freshwater inflows occur in dry years when springtime inflows are reduced by nearly half, 47%, on average compared to the average 20% reduction in very wet years (for the 1930-2014 period). Since 1970, the percentages of springtime unimpaired flow that reached the Delta during the spring averaged 52% in very dry years, 47% in dry years, 55% in median years, 63% in wet years and 76% in very wet years.

Spring flow into the Delta, as a percentage of unimpaired flow, has declined over time.
The percentage of unimpaired flow that actually flowed into the Delta during the spring has declined significantly over the past several decades (regression, $p<0.001$). Significant declines have occurred in all water years types except very wet years (regression, all tests, $p<0.05$; very wet year regression, $p=0.054$). Before construction of most of the major dams on the Delta's watershed (1930-1943, the pre-dam period), an average of 78% of springtime unimpaired flow actually reached the Delta. By the 1980s, the percentage had decreased significantly to just 63% (1980-1989 average; t-test, $p<0.05$). The average for the most recent 10-year period (2005-2014), 53%, is lower than spring Delta inflows during the 1980s but, because of large year-to-year variations, not significantly different (t-test, $p=0.15$).

Based on spring inflows, CCMP goals to increase fresh water availability to the estuary have not been met.
Since 1990, springtime freshwater inflows to the Delta were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in just 12% of years (3 of 25 years). Current spring inflows to the Delta are well below the 80% level recommended by Richter et al. (2011) as well as 75% level for Delta outflows identified by the SWRCB as necessary to protect public trust resources and estuarine health. Recent spring inflows are also frequently lower than those measured in the 1990s, when the CCMP was developed and established.

C. San Joaquin River Inflow

1. Rationale

The Delta's vast watershed extends more than 500 miles north to south, from the headwaters of the Sacramento River to the southern end of the San Joaquin basin. Historically, the southern portion of the watershed, San Joaquin River basin, provided just under a quarter (21%) of the total freshwater inflow to the Delta on average.[10] However, since the early 1900s, flows on most San Joaquin basin rivers have been stored behind increasingly large dams and diverted to supply water for San Joaquin Valley agriculture. Even before Friant Dam on the upper San Joaquin River near Fresno began operation in 1949, local water diversions dried up long stretches of the basin's mainstem river in some years. Since the 1950s, additional water has been imported into the San Joaquin Valley from the Delta and, in some areas, agricultural drainage water discharged into the river has added to flow levels, although the quality of drainage water can be very poor and even toxic.

2. Methods and Calculations

The San Joaquin River Inflow indicator measures the amount of water that flowed into the Delta from the San Joaquin River compared to the amount of water that would have flowed into the

[10] In some years, hydrological conditions (i.e., whether it's a wet or dry year) can differ between the basins. The San Joaquin River's contribution was higher in years when it was wetter in the southern basin than in the north and lower when the San Joaquin was drier than the Sacramento basin.

Delta from this river under unimpaired conditions, without the effects of dams, water diversions or water imports.[11] Capture, storage and diversion of San Joaquin watershed runoff by dams and on-river diversions reduces San Joaquin River inflow to the Delta; discharge of return water derived from water imported to the San Joaquin basin from the Sacramento River basin via the Delta increases San Joaquin River inflows.

The indicator was calculated for each year (1930-2014) as the percentage of annual unimpaired freshwater inflow from the San Joaquin Basin using the following equation:

San Joaquin River Inflow (% of unimpaired)
= (actual San Joaquin River inflow/unimpaired San Joaquin River inflow) x 100

3. Reference Conditions

The primary reference condition for the San Joaquin River Inflow indicator was established as 80%, the conservative level identified by Richter et al. (2011) as needed to maintain the ecological integrity of most rivers. Annual inflows that were greater than 80% of unimpaired inflows were considered to reflect "good" conditions and meet the CCMP goals; annual inflows that were less than 50% of unimpaired inflows were considered to correspond to "very poor" conditions. The other reference condition levels were established based on Richter et al. (2011; 90% of unimpaired to protect rivers with at-risk species for "excellent" and minimally altered flows) and use of equal increments between the primary and lowest reference condition levels. This primary reference condition is higher than the flow level identified by the SWRCB for seasonal San Joaquin River inflows to the Delta, 60% of unimpaired, and for Delta outflow, 75% of unimpaired, as needed to protect public trust resources (SWRCB 2010). However, the rationale used by the SWRCB for the lower flow levels was based only on minimum requirements to protect migrating salmonids, rather than the broader based objective of protecting ecological integrity used by Richter et al. (2011). Therefore, and for consistency with the other inflow indicators, the work of Richter et al. (2011) was used as the basis for the primary reference condition for this indicator. Table 3 below shows the quantitative reference conditions that were used to evaluate the results of the San Joaquin River Inflow indicator.

Table 3. Quantitative reference conditions and associated interpretations for results of the San Joaquin Inflow indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

San Joaquin River Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>90% change in SJR inflow	"Excellent," minimal alteration	4
>80% change in SJR inflow	***"Good," meets CCMP goals***	***3***
>65% change in SJR inflow	"Fair"	2
>50% change in SJR inflow	"Poor"	1
≤50% change in SJR inflow	"Very Poor," extreme alteration	0

[11] San Joaquin River inflow is measured at Vernalis.

4. Results

Results of the San Joaquin River Inflow indicator are show in Figure 6.

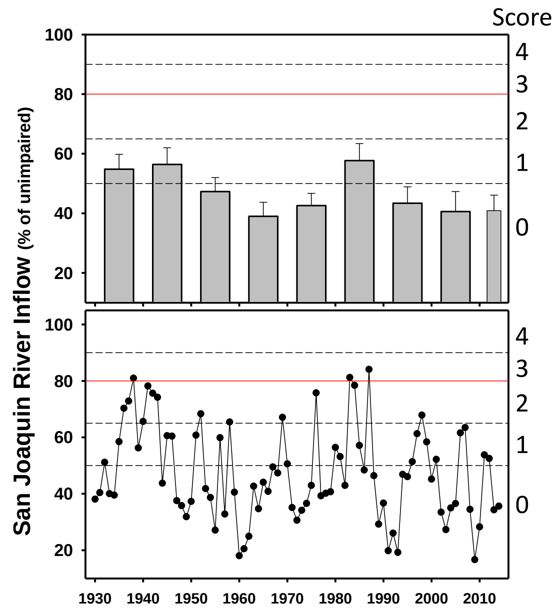

Figure 6. Results for the San Joaquin River Inflow indicator, expressed as the percent change in the San Joaquin River's contribution to total Delta inflow for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The amount of fresh water flowing into the Delta from the San Joaquin River has been reduced. The percentage of the annual runoff from San Joaquin River watershed that flows into the Delta has been substantially reduced, averaging just 47% of unimpaired inflow for the 1930-2014 period. The greatest reduction in San Joaquin River inflow occurred in 2009, the third year of the recent three-year drought, when only 17% of unimpaired inflow reached the Delta. Inflows were lower than 20% of unimpaired in several other years: 18% in 1960 (a dry year following a dry year), 19% in 1993 (a very wet year following a multi-year drought) and 20% in 1990 (a very dry year following several other very dry years). For the most recent 10-year period (2005-2014), on average only 42% of unimpaired San Joaquin River inflow actually flowed into the Delta. During this period San Joaquin River inflows were "very poor," less than 50% of unimpaired, in six of the ten years; in the other four years inflow were "poor," less than 65% of unimpaired. San Joaquin River inflows were at least 60% of unimpaired, the level identified by the SWRCB (2010) as necessary to protect public trust resources, in only two years during the last decade, and only nine years in the last 50 years (18% of years). In 2014, only 36% of unimpaired San Joaquin River flow reached the Delta, corresponding to "very poor" conditions.

The proportional reductions in San Joaquin River inflow to the Delta differ by water year type.
The greatest alterations to San Joaquin River inflows occur in dry years when annual inflows are reduced by nearly two thirds, averaging just 36% of unimpaired, significantly lower than inflows in very wet and wet years (ANOVA for the 1930-2014 period, $p<0.05$). Since 1930, the percentages of San Joaquin River inflow that reached the Delta averaged 46% in very dry years, 36% in dry years, 45% in median years, 52% in wet years and 59% in very wet years.

San Joaquin River flow into the Delta, as a percentage of unimpaired flow, has declined over time.
The percentage of unimpaired flow that actually flowed into the Delta from the San Joaquin River has declined significantly since the 1930s; inflows before most of the major dams were

completed (the pre-dam period, 1930-1943) were significantly higher, 60% of unimpaired, than those measured since 1970, which have averaged 46% (t-test, $p<0.01$).

The contribution of the San Joaquin River to total Delta inflow has been reduced.
Compared to unimpaired flow conditions, the fractional contribution of the San Joaquin River to total Delta inflow has been reduced by an average of 41% (1930-2014).[12] For the most recent ten-year period, 2005-2014, San Joaquin River's contributions to total Delta inflow were reduced by an average of 45%; in 2014 the San Joaquin River's contribution to total Delta inflow was less than half of what it would have been under unimpaired conditions.

San Joaquin River diversions constitute the majority of Sacramento-San Joaquin watershed runoff that is diverted before reaching the Delta.
Since 1980, an average of 3.3 (±1.9 SD) million acre feet of freshwater inflow was diverted from the San Joaquin River before it reached the Delta. This constitutes 65% of the reduction in Delta inflow from water diverted from the Sacramento-San Joaquin watershed prior to flowing in to the Delta and 30% of the total reduction in freshwater inflow to the Bay.

Based on San Joaquin River inflows to the Delta, CCMP goals to increase fresh water availability to the estuary have not been met.
Since 1990, freshwater inflows to the Delta from the San Joaquin River have not been "good," meeting or exceeding conditions considered to satisfy CCMP goals, in any year (0 of 25 years). Current San Joaquin River inflows to the Delta are much lower than the 80% level recommended by Richter et al. (2011) to maintain ecological integrity. They are also well below the 60% of unimpaired level identified by the SWRCB as necessary to protect public trust resources and estuarine health (SWRCB 2010). In 16 of the past 25 years (64% of years), San Joaquin River inflows were "very poor," cut by more than 50%.

D. Annual Bay Inflow

1. Rationale

Fresh water that flows out of the Delta, the upstream region of the estuary, provides >90% of the total freshwater inflow to the San Francisco Bay. As it enters the Bay, inflowing fresh water mixes with salt water from the Pacific Ocean and lower Bay, creating brackish water[13] habitat that is a key characteristic of estuaries, and the amounts, timing and seasonal and inter-annual variability of inflows function as physical and ecological drivers that stimulate productivity, reproduction and movement (Jassby et al. 1995; Kimmerer 2002; 2004 Feyrer et al. 2008; Moyle et al., 2010). In the Bay's Sacramento-San Joaquin watershed, annual runoff varies substantially

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[12] Change in the proportional contribution of the San Joaquin River to total Delta inflow as calculated as:

$$\text{SJR Inflow indicator} = \frac{\{[(\text{SJR-in as \%D-in})-(\text{unimp. SJR-in as \%unimp. D-in})]\}}{(\text{unimp. SJR-in as\%unimp. D-in})} \times 100$$

where SJR-in as %D-in is the percent contribution of total annual actual SJR inflow to total annual actual Delta inflow, and Unimp. SJR as %unimp. D-in is the percent contribution of total annual unimpaired SJR inflow to total annual unimpaired Delta inflow. The San Joaquin River's proportional contribution to Delta inflow is highly correlated to San Joaquin River inflow expressed as percent of unimpaired ($p<0.001$, Pearson product moment correlation coefficient=0.953).

[13] Brackish water is defined as water that has more salinity than fresh water, but not as much as seawater.

for year-to-year, but during the past century, freshwater inflows into the Delta and the Bay downstream have been greatly altered by upstream dams and water diversions. Nine of the ten largest rivers in the 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 rivers 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 Bay have affected the estuarine ecosystem and the plants and animals that depend on it.

2. Methods and Calculations

The Annual Bay Inflow[14] indicator measures the amount of fresh water from the Sacramento-San Joaquin watershed that flows into San Francisco Bay from the Delta each year compared to the amount that would have flowed into the Bay under unimpaired conditions. Capture and storage of watershed runoff for release in subsequent years and diversion of water from the estuary's tributary rivers and the Delta reduces annual Bay inflow; release of water captured and stored in watershed reservoirs in previous years and imports of water from the Trinity River watershed increase annual Bay inflow.

The indicator was calculated for each year (1930-2014) using data for total annual actual freshwater inflow and estimated total annual unimpaired inflow as:

Annual Bay Inflow (% of unimpaired)
= (actual annual Bay inflow/unimpaired annual Bay inflow) x 100

3. Reference Conditions

The primary reference condition for the Annual Bay Inflow indicator was established as 75%, a level based on the SWRCB's recommendation for freshwater inflows (or Delta outflows) needed to support public trust resources in the estuary. This level also corresponds to an average annual in-Delta flow depletion of 2.4 million acre-feet (approximately 10% of unimpaired Delta inflow) a level that is more than twice the amount of unimpaired in-Delta depletion.[15] Annual inflows that were greater than 75% of unimpaired inflows were considered to reflect "good" conditions and meet the CCMP goals; annual inflows that were less than 50% of unimpaired inflows were considered to correspond to "very poor" conditions. The other reference condition levels were based on equal increments between these two levels. Table 4 below shows the quantitative reference conditions that were used to evaluate the results of the Annual Bay Inflow indicator.

[14] Bay inflow is measured and frequently expressed as Delta outflow, or net Delta outflow.
[15] Unimpaired in-Delta depletion was calculated as (unimpaired Delta inflow – unimpaired Delta outflow).

Table 4. Quantitative reference conditions and associated interpretations for results of the Annual Bay Inflow indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Annual Bay Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>87.5% of unimpaired	"Excellent," minimal alteration	4
>75% of unimpaired	***"Good," meets CCMP goals***	***3***
>62.5% of unimpaired	"Fair"	2
>50% of unimpaired	"Poor"	1
<u>≤</u>50% of unimpaired	"Very Poor," extreme alteration	0

4. Results

Results of the Annual Bay Inflow indicator are show in Figure 7.

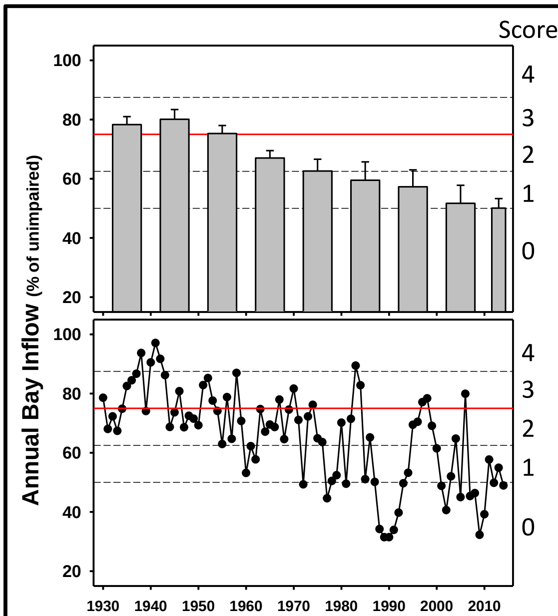

Figure 7. Results for the Annual Bay Inflow indicator, expressed as the percentage of unimpaired flow that actually flowed into the Bay from the Delta for 1930 to 2014(left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The amount of fresh water flowing into the San Francisco Bay from the Delta each year has been reduced.
On an annual basis, the percentage of the freshwater runoff from estuary's largest watershed that flows into the Bay has been substantially reduced. For the most recent 10-year period (2005-2014), on average only 50% of unimpaired inflow actually flowed into the Bay, with inflows less than 50% in seven of those years. In 2009, a dry year that followed two consecutive very dry years, annual Bay inflow was only 32% of unimpaired, the third lowest percentage of freshwater inflow in the 85-year data record. In 2014, a very dry year, only 49% of unimpaired inflow reached the Bay.

The proportional alteration in annual freshwater inflow to the Bay differs by water year type.
The greatest alterations to freshwater inflows (expressed as a percentage of estimated unimpaired inflow) occur in drier years. Since the 1970s, the percentages of unimpaired flow that reached the estuary averaged 45% in very dry and dry years, 52% in median years, 68% in wet years and 72% in very wet years.

Freshwater flow into the Bay, as a percentage of unimpaired flow, has declined over time.
The percentage of unimpaired flow that actually flows into the Bay has declined significantly over the past several decades (regression, $p<0.001$). Significant declines in the percentage of unimpaired inflow reaching the Bay have occurred in all water years types (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 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, 50%, is somewhat lower but, due to the large inter-annual variability associated with hydrology, not significantly different than flows during the 1980s. Since 1980, an average of 10.9 ($\pm$4.3 SD) million acre feet of freshwater inflow was diverted from either the Sacramento-San Joaquin watershed or Delta before it reached the Bay. Of this amount, reductions in Delta inflow constitute 48% percent of the reduction in Bay inflow and in-Delta diversions 53% percent.

Based on annual inflows, CCMP goals to increase fresh water availability to the estuary have not been met.
Since 1990, freshwater inflows to the Bay were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in just 12% of years (3 of 25 years). 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 1990s, the period during which the CCMP was developed and established. In 13 of the past 25 years (52% of years), Bay inflows were "very poor," cut by more than 50%.

E. Spring Bay Inflow

1. Rationale

Freshwater inflows to the Bay 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 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. Capture and storage of spring runoff for release later in the year or in subsequent years, and springtime diversion of water from the estuary's tributary rivers and the Delta reduces spring Bay inflows; springtime release of water captured and stored in watershed reservoirs in previous years and imports of water from the Trinity River watershed increase spring Bay inflow.

The indicator was calculated for each year (1930-2014) using data for February-June actual freshwater inflow and estimated total annual unimpaired inflow as:

Spring Inflow (% of unimpaired)
= (actual Feb-June inflow/unimpaired Feb-June inflow) x 100

3. Reference Conditions

The primary reference condition for the Spring Bay 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 and meet the CCMP goals; annual inflows that were less than 50% of unimpaired inflows were considered to correspond to "very poor" conditions. The other reference condition levels were based on equal increments between these two levels. Table 5 below shows the quantitative reference conditions that were used to evaluate the results of the Spring Inflow indicator.

Table 5. Quantitative reference conditions and associated interpretations for results of the Spring Bay Inflow indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Spring Bay Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>87.5% of unimpaired	"Excellent," minimal alteration	4
>75% of unimpaired	***"Good," meets CCMP goals***	***3***
>62.5% of unimpaired	"Fair"	2
>50% of unimpaired	"Poor"	1
≤50% of unimpaired	"Very Poor," extreme alteration	0

4. Results

Results of the Spring Bay Inflow indicator are show in Figure 8.

The amount of fresh water flowing in the Bay during the spring has been reduced.
The percentage of the springtime runoff from estuary's largest watershed that flows into the Bay has been significantly reduced. For the most recent 10-year period (2005-2014), on average only 44% of unimpaired inflow actually flowed into the estuary. In 2009, spring inflow only 27% of unimpaired, the seventh lowest percentage of freshwater inflow in the 85-year data record. In 11 of the past 20 years (55% of years), the percentage of unimpaired flow that flowed into the Bay during the spring was less than 50%. In 2014, only 36% of unimpaired inflow reached the estuary.

The proportional alteration in spring inflow to the estuary differs by water year type.
The greatest alterations to springtime freshwater inflows occur in drier years. Since the 1970s, the percentages of unimpaired flow that reached the estuary averaged 33% in very dry and dry years, 44% in median years, 67% in wet years and 72% in very wet years.

Spring flow into the Bay, 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 (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 springtime unimpaired flow actually reached the Bay. 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, 44%, is somewhat lower but, due to the large inter-annual variability associated with hydrology, not significantly different than flows during the 1980s.

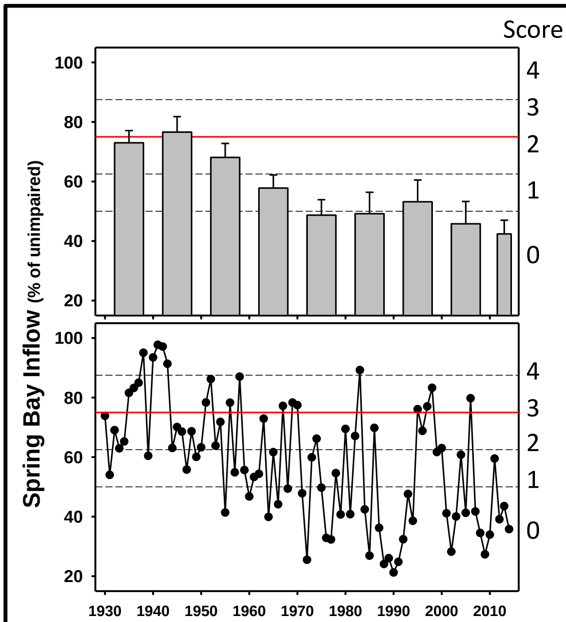

Figure 8. Results for the Spring Bay Inflow indicator, expressed as the percentage of spring (Feb-June) unimpaired flow that actually flowed into the Delta for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

Based on spring inflows, CCMP goals to increase fresh water availability to the estuary have not been met.
Since 1990, springtime freshwater inflows to the Bay were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in just 12% of years (3 of 25 years). Current spring inflows to the Bay are well below the 75% level identified by the SWRCB as necessary to protect public trust resources and estuarine health. In 64% of the past 25 years, spring inflows to the Bay have been cut by more than 50% and recent inflows are also somewhat lower than those measured in the 1990s.

F. Delta Diversions

1. Rationale

The Delta, now a complex network of interconnected river channels, sloughs, canals and islands, has been a site for water diversion for more than a century (CDWR 1995). The first Delta diverters were farmers irrigating the rich island soils and small local communities like Antioch. Today, there are more than 2,200 of these agricultural and local urban water diversions scattered throughout the Delta's 1152-square mile area. Beginning in the 1950s, the Delta also became the main "switching station" for much of California's managed water supply. Two giant pumping facilities located in the southern Delta – the Central Valley Project (CVP) operated by the U.S. Bureau of Reclamation and the State Water Project (SWP) operated by the California Department of Water Resources – divert and export large amounts of water into man-made canals for delivery to the San Francisco Bay area, San Joaquin Valley and Southern California.

Removal of water from Delta channels at a pipe or diversion canal can alter flow patterns and kill fish and other small animals trapped in the diverted water, particularly if the diversion rate is high relative to flow in the channel (Kimmerer 2008).

2. Methods and Calculations

The Delta Diversions indicator measures Delta diversions as the percentage of total Delta inflow that is diverted from the Delta for each year (1930-2014). Diversion of water from Delta channels reduces the amount of fresh water that flows into the Bay and can alter flow velocity and direction in Delta channels.

The indicator was calculated for each year (1930-2014) using data for actual annual Delta inflow and actual annual Delta outflow (or Bay inflow) as:

Delta Diversions indicator
= [(actual Delta inflow – actual Delta outflow)/actual Delta inflow]*100.

3. Reference Conditions

The primary reference condition for the Delta Diversions indicator was established as 13%. This level corresponds to the amount of in-Delta diversions that would result in Bay inflows that met or exceeded the primary reference condition for the Annual Bay Inflow indicator, 75% of unimpaired, when the primary reference condition for the Annual Delta Inflow indicator, 80% of unimpaired, was met or exceeded. This level is also more than double the average unimpaired in-Delta depletion rate (4%),[15] the average pre-dam in-Delta diversion rates (5% for the 1930-1943 period) and average pre-export pumping facilities period (6% for 1930-1958 period). In-Delta diversions that were less than 13% of actual annual Delta inflow were considered to reflect "good" conditions and meet the CCMP goals; annual diversions that were three times greater than this level, 39%, and more than six times greater than pre-export pumping facility in-Delta depletion rates and which would approach current regulatory standards limiting state and federal pumping facility exports to protect fish and wildlife (SWRCB 2006) in most years were considered to correspond to "very poor" conditions. The intermediate reference condition ("fair") was based on equal increments between these two levels and the upper ("excellent") reference condition was based on the average pre-export pumping facilities level. Table 6 below shows the quantitative reference conditions that were used to evaluate the results of the Delta Diversions indicator.

Table 6. Quantitative reference conditions and associated interpretations for results of the Delta Diversions indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Delta Diversions		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
<6% of Delta inflow	"Excellent," minimal alteration	4
<13% of Delta inflow	***"Good," meets CCMP goals***	***3***
<26% of Delta inflow	"Fair"	2
<39% of Delta inflow	"Poor"	1
≥39% of Delta inflow	"Very Poor," extreme alteration	0

4. Results

Results of the Delta Diversions indicator are show in Figure 9.

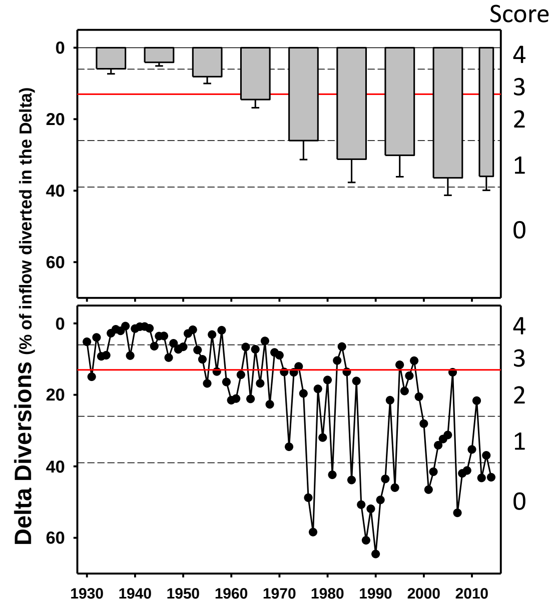

Figure 9. Results for the Delta Diversions indicator, expressed as the percentage of Delta inflow that is diverted in the Delta for 1930 to 2014(left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

A large percentage of the fresh water that flows into the Delta is diverted.
The amount of fresh water diverted from the Delta, expressed as percentage of annual Delta inflow, reached record highs during the past three decades. The highest proportional diversion rates occurred during droughts, exceeding 50% of inflow diverted in several years and a record 65% of inflow diverted in 1990. During the past ten years, Delta diversion rates have averaged 36% and, in 2014, 43% of total Delta inflow was diverted and did not flow into the Bay.

The percentage of Delta inflow that is diverted in the Delta differs with water year type.
Since 1970, when both the state and federal export facilities were operational, the percentage of Delta inflow diverted from the Delta differed significantly among all years types except very wet years compared to wet years (ANOVA, $p<0.05$ all comparisons except very wet v wet). The highest proportional diversions occur in very dry years, averaging 51%. Diversion rates are progressively lower with wetter years, averaging 42%, 34%, 18% and 14% for dry, median, wet and very wet years respectively.

The percentage of Delta inflow diverted from the Delta has increased over time.
The percentage of inflow diverted from the Delta has increased significantly during the past eight decades (regression, $p<0.001$) and since the 1970s, when both state and federal export facilities became operational (Mann Whitney, 1930-1969 v 1970-2014, $p<0.001$). Significant increases in Delta diversion rates occurred in all water year types (regression, all tests, $p<0.001$). Before construction of most of the major dams on the Delta's tributary rivers (1930-1943, the pre-dam period), an average of 5% of Delta inflow was diverted in the Delta. Not until the federal and then the state export facilities became operational in the 1950s and 1960s did Delta diversion rates begin to increase substantially.

Based on Delta diversion rates, CCMP goals to increase fresh water availability to the estuary have not been met.
Since 1990, Delta diversion rates were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in just 8% of years (2 of 25 years). Current Delta diversion rates, combined with upstream diversions that reduce Delta inflow, reduce freshwater inflows to the Bay to well

below the 75% of unimpaired level identified by the SWRCB as necessary to protect public trust resources and estuarine health. Since the 1990s, Delta diversion rates have increased, reducing freshwater availability to the estuary rather than increasing it; in 11 of the past 25 years (44% of years), total Delta diversions exceeded 39% of total Delta inflows.

G. 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), 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 (Moyle et al. 2010).

2. Methods and Calculations

The Inter-annual Variation in Inflow indicator measures the ratio, expressed as percentage, of the inter-annual variation in actual annual inflow to Bay (or Delta outflow) and that of unimpaired annual Bay 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.[16] Reductions in inflows from upstream and in-Delta diversions, particularly in median and wetter years, reduce the differences between annual inflow amounts in very wet years and dry years, making successive years more similar to each other in annual inflow amounts.

The indicator was calculated for each year (1939-2010) using actual annual Bay inflow (or Delta outflow) and unimpaired annual Bay inflow as:

Inter-annual Variation in Inflow (% of unimpaired)
= [(SD actual Bay inflow for year$_{(0\text{ to }-9)}$)/(SD unimpaired Bay inflow for year$_{(0\text{ to }-9)}$)] x 100.

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 Bay inflows and calculated unimpaired inflows that had been reduced by 25%, the level of inflow reduction used for the primary reference condition for the Annual Bay Inflow indicator, for the same period. Based on this calculation, the reference condition was set at 75%. Levels that were greater than this were considered to reflect “good” conditions and meet the CCMP goals; levels

[16] Inter-annual variation in inflow was not measured using the coefficient of variation (i.e., SD/mean) because for comparisons of actual to unimpaired inflows both the mean (of monthly inflow levels) and the variation around the mean (SD of monthly inflows) change.

that were less than 50%, more than double the reduction in inter-annual variability compared the primary reference condition, were considered to correspond to "very poor" conditions. The other reference condition levels were established based on equal increments of values based from these two levels. Table 7 below shows the quantitative reference conditions that were used to evaluate the results of the Inter-annual Variation in Inflow indicator.

Table 7. 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 italics.

Inter-annual Variation in Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
> 87.5%	"Excellent," minimal alteration	4
> 75%	***"Good," meets CCMP goals***	***3***
> 62.5%	"Fair"	2
> 50%	"Poor"	1
≤ 50%	"Very Poor," extreme alteration	0

4. Results

Results of the Inter-annual Variation in Inflow indicator are show in Figures 10 and 11.

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

The magnitude of inter-annual variability of unimpaired and actual freshwater inflows to the San Francisco Bay 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 similar annual flows (i.e., low inter-annual variation and low small standard deviation) (Figure 10). Beginning in the early 1980s, the unimpaired annual inflows became substantially more variable (1980-2004 average variability: 18,038 TAF) than annual unimpaired inflows during the earlier 40 years (1939-1979 average variability: 12,908 TAF). For the most recent decade, inter-annual variability levels have declined to level to levels comparable to the earlier period (2005-2014 average variability: 13,400 TAF). Inter-annual variation in actual annual flows showed a similar pattern (1939-1980 average: 12,082 TAF; 1980-2004 average: 15,579 TAF; and 2005-2014 average: 12,037 TAF).

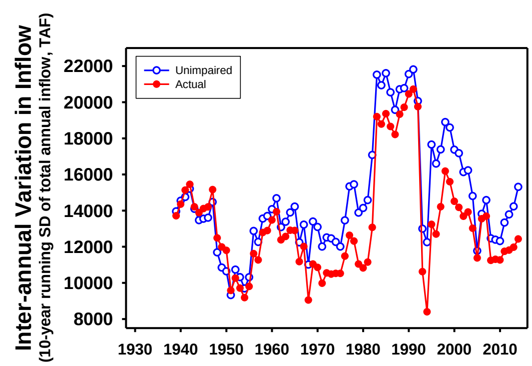

Figure 10. Inter-annual variation in actual and unimpaired freshwater inflows to the Bay for 1939-2014. Each point is the standard deviation for running 10-year periods ending in that year..

Inter-annual variability in inflows to the San Francisco Bay has been reduced. Inter-annual variability has decreased significantly during the past eight decades (regression, $p<0.01$). For the 1939-1967 period (the first 25 years of record), prior to completion of the most of the large dams in the watershed, the inter-annual variability of Bay inflows was essentially the same as for unimpaired inflows during the period, averaging 99% of unimpaired inter-annual variability. In contrast, the inter-annual variability of Bay inflows for the most recent 25 years, 1990-2014, is significantly lower than that of unimpaired inflows, averaging just 87% (t-test, $p<0.001$). The greatest reductions in inter-annual variation in Bay inflows occurred in the mid-1990s, following a prolonged drought when actual Bay inflows were reduced to record low levels (see Annual Bay Inflow indicator). In 2014, inter-annual variation in the most recent 10 years of Bay inflows was 81% of unimpaired inter-annual variation for that period.

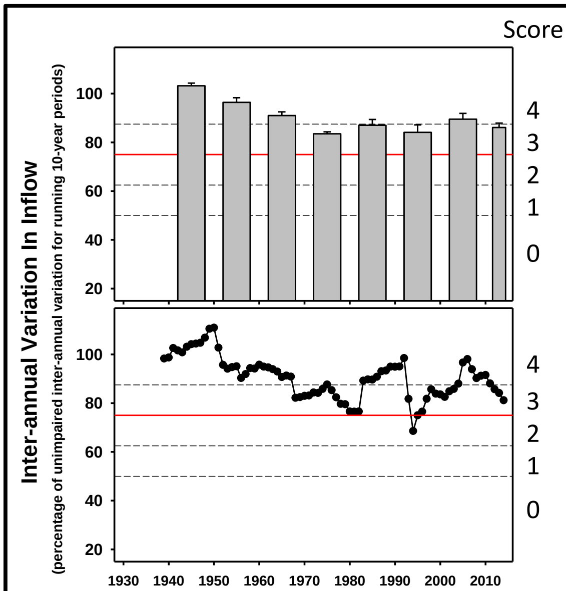

Figure 11. Results for the Inter-annual Variation in Inflow indicator, expressed as the percentage of unimpaired inter-annual variation of Bay inflows (calculated as the ratio of 10-year SD for actual inflow to the 10-year running SD for unimpaired inflows) for 1939 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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 fully met.

Since 1990, inter-annual variation in freshwater inflows to the Bay was "good," meeting or exceeding conditions considered to satisfy CCMP goals in all but two years, 1994 and 1995, 92% of years (23 of 25 years). However, this recent period also saw the greatest reductions in inter-annual variability measured during the past 85 years and, since the mid-2000s, inter-annual variation in Bay inflows has been declining.

H. Seasonal Variation in Inflow

1. Rationale

Freshwater inflow to the San Francisco Bay 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 2). These seasonal variations in inflow create different kinds of habitat, for example, seasonal high inflows create 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 ratio, expressed as a percentage, of the seasonal (or intra-annual) variation in actual monthly average inflow to the San Francisco Bay 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).[17] The standard deviation of monthly inflows is large in years with large seasonal changes in inflow, such as from a strong springtime snowmelt pulse, and low in years when springtime flows are low compared to summer and fall flows.

The indicator was calculated for each year (1930-2014) using average monthly unimpaired and actual Bay inflow (or Delta outflow) as:

Seasonal Variation in Inflow (% of unimpaired)
= [(SD of actual average monthly Bay inflow)/(SD in unimpaired monthly Bay inflow)] x 100.

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 Bay inflows and calculated unimpaired monthly inflows that had been reduced by 25%, the level of inflow reduction used for the primary reference condition for the Annual and Spring Bay Inflow indicators, for the same period. Based on this calculation, the reference condition was set at 75%. Levels that were greater than this were considered to reflect “good” conditions and meet the CCMP goals; levels that were less than 50%, more than double the reduction in seasonal variability compared the primary reference condition, were considered to correspond to “very poor” conditions. The other reference condition levels were established based on equal increments of values based from these two levels. Table 8 below shows the quantitative reference conditions that were used to evaluate the results of the Seasonal Variation in Inflow indicator.

[17] Seasonal inflow variation was not measured using the coefficient of variation (i.e., SD/mean) because for comparisons of actual to unimpaired inflows both the mean (of monthly inflow levels) and the variation around the mean (SD of monthly inflows) change.

Table 8. 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 italics.

Seasonal Variation in Inflow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
> 87.5%	"Excellent," minimal alteration	4
> 75%	***"Good," meets CCMP goals***	***3***
> 62.5%	"Fair"	2
> 50%	"Poor"	1
< 50%	"Very Poor," extreme alteration	0

4. Results

Results of the Seasonal Variation in Inflow indicator are show in Figures 12 and 13.

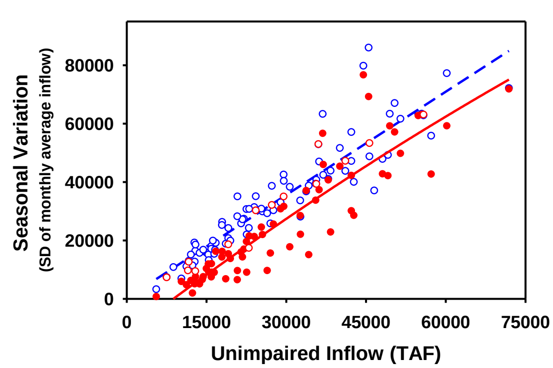

Figure 12. Seasonal variation in Bay inflows (SD of average monthly inflows, cfs, Y axis) is directly related to hydrology, as expressed by unimpaired inflow to the Bay (TAF, X axis). Seasonal variation in unimpaired inflows is shown in open blue circles. Seasonal variation in actual inflows is shown in open red circles (pre-dam period, (1930-1943) and solid red circles (1944-2014).

Seasonal variability in inflows to the San Francisco Estuary is directly related to hydrology.
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 very wet years and low in dry years (regression, both tests, $p<0.001$) (Figure 12).

Seasonal variability in inflows to the San Francisco Estuary has been reduced.
Seasonal variability of freshwater inflows to the Bay has declined significantly (regression, $p<0.001$) (Figure 13). The decline began in the mid-1940s, when the first of large storage dams in the estuary's watershed were completed, and since then each decade has seen progressive reductions in seasonal variation in Bay inflows. In the pre-dam period (1930-1943), actual seasonal variation in Bay inflows were 90% of seasonal variation of unimpaired inflows; by the 1980s the actual seasonal variation in inflows was significantly lower, averaging 66% of unimpaired seasonal variation (Mann Whitney Rank Sum test, $p<0.05$). Since then, seasonal variation has continued to decline, from an average of 62% in the 1990s to just 50% in the most recent 10 years (2005-2014). The greatest reduction in seasonal variation was in 1990, when actual seasonal variation was just 17% of unimpaired seasonal variation. In 2014, seasonal variation in Bay inflow was 28% of unimpaired seasonal inflow, the 5th lowest in the 85-year record.

Changes in seasonal variation in freshwater inflows to the Bay differ by water year type.
Seasonal variation in Bay inflows have significantly declined in all water year types except very wet years (regression, all tests except very wet, $p<0.01$). The greatest reductions in seasonal variation have occurred very dry and dry years, although in large reductions in seasonal variation have occurred in some recent wet years (e.g., seasonal variation was reduced by 61% in 2005, a

wet year). Since 1970, compared to unimpaired condition, seasonal variation in Bay inflows have averaged 39% in very dry years, 42% in dry years, 57% in median years, 77% in wet years and 86% in very wet years.

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.

Since 1990, seasonal variability of freshwater inflows to the Bay were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in just 32% of years (8 of 25 years). In 13 of the past 25 years (52% of years), seasonal variability of Bay inflows have been "very poor."

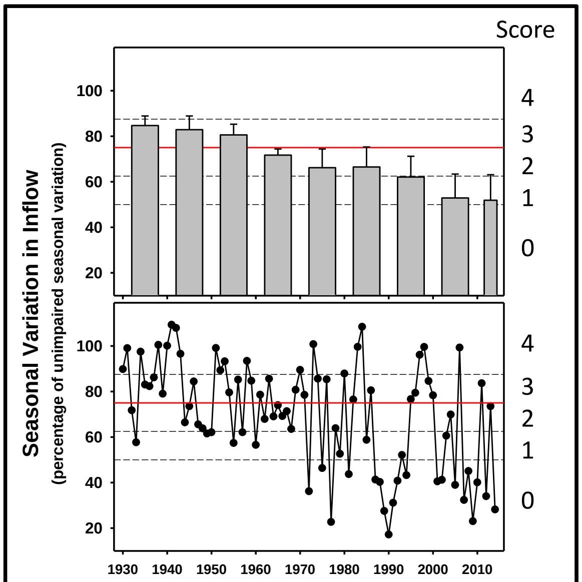

Figure 13. Results for the Seasonal Variation in Inflow indicator, expressed as the percentage of unimpaired seasonal variation of Bay inflows (calculated as the ratio of the SD for actual monthly inflows to the SD for unimpaired monthly inflows) for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

I. Peak Flow

1. Rationale

High, or "peak", freshwater inflows to the San Francisco Bay 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 Bay, 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 Bay 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[18] 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 the more prolonged spring snowmelt; therefore this indicator evaluated the estuary's responses to a key aspect of seasonal flow variation in its watershed.

[18] 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 indicator is calculated for each year (1930-2014) using the 5-day running average of actual Bay inflow (or Delta outflow) as:

> Peak flow (days)
> = (# days actual Bay inflow>50,000 cfs) – (# days predicted Bay inflow >50,000 cfs)

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 14). 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 14. 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.[19]

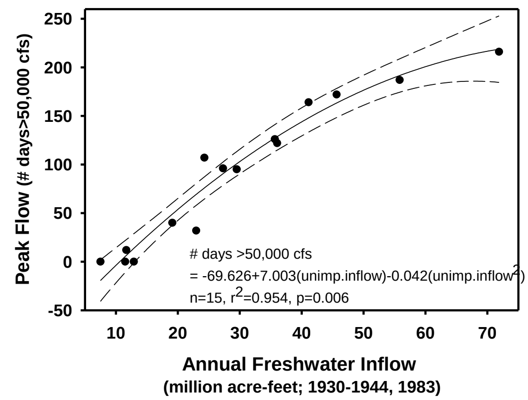

Figure 14. Actual (symbols) and predicted (regression with confidence limits) number of days with peak flow per year in relation to total annual Bay inflow for 1930-1944 and 1983. This relationship was used to establish the reference conditions for the Peak Inflows indicator.

3. Reference Conditions

Reference conditions were established based on the 95% confidence interval for the polynomial regression developed from pre-dam and 1983 data (see Figure 14 above). Over most of the range of annual freshwater inflows, the maximum value for the 95% confidence interval for predicted days of peak flows was 15 days; the primary 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 pre-dam inflows). 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 and meet the CCMP goals; reductions in days of peak flows that were more than double this level (or four times greater than the 95% confidence interval) were considered to correspond to "very poor" conditions. The other reference condition levels were established based on equal increments of values based from these two levels, with the upper reference conditions ("excellent") set at -15 days. Table 9 below shows the quantitative reference conditions that were used to evaluate the results of the Peak Flow indicator.

[19] This occurred in only four years: 1931, 1976, 1977 and 2014.

Table 9. 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 italics.

Peak Flow		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
> -15 days	"Excellent," minimal alteration	4
> -30 days	***"Good," meets CCMP goals***	***3***
> -45 days	"Fair"	2
> -60 days	"Poor"	1
≤ -60 days	"Very Poor," extreme alteration	0

4. Results

Results of the Peak Flow indicator are show in Figure 15.

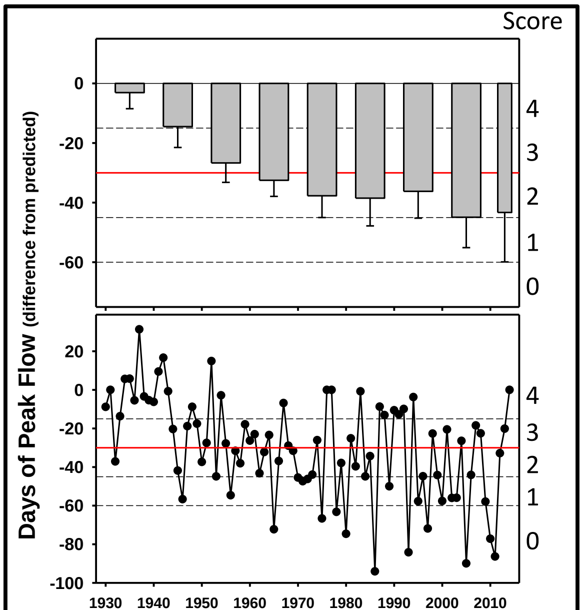

Figure 15. Results for the Peak Flows indicator, expressed as the number of days of peak flow different from predicted for 1930 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The frequency of peak flows into the San Francisco Bay varies with water year type. Actual peak flow frequency (as number of days per year) is highest in very wet years, when there are of 140 days of peak flow per year on average for the 85 year data record, lowest in very dry years (<2 days/year). Dry years have an average of 12 days/years, median years an average of 48 days/year and wet years an average of 85 days.

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 85-year period of record (regression, p<0.001). The decline began after 1943, immediately following completion of many of the large dams on the estuary's largest tributaries. Peak flow frequency has significantly declined in all water year types except very dry years (regression, p<0.05 all tests, regression for very dry years, p=0.16). On average, there are 36 fewer days of peak flows per year since the mid-1940s than during the 1930-1943 period. In the most recent ten year period (2005-2014), peak flow frequency was reduced by an average of 45 days per year. In 2014, a critical dry year in which no peak flows were predicted based on total annual Bay inflow, there were no days in which the 5-day average Bay inflow exceeded 50,000 cfs and the difference between actual and predicted peak flow frequency was zero.

Decreases in peak flow frequency differ with water year type.
Since 1943, the largest decreases in peak flow frequency have occurred in wet years, which have 55 fewer days of peak than predicted, a 43% decrease. In very wet years there are an average of 41 fewer days of peak flow in very wet years (24% decrease), 42 fewer days in median years (53% decrease), and 31 fewer days in dry years (75% decrease). Peak flows have been eliminated in most very dry years, cut by 95% to less than two day per year, compared to the predicted average of 11 days per year predicted.

Based on recent peak flow frequency, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have been partially met.
Since 1990, peak flow conditions in the Bay were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in 44% of years (11 of 25 years). However, peak flows were completely eliminated in 7 of 25 years (i.e., 0 days of peak flow in 28% of years) in which they would have occurred based on predictions from estimates of unimpaired conditions from pre-dam inflows.

J. Dry Year Frequency

1. Rationale

California's Mediterranean climate is characterized 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). 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, whether the result of hydrological drought or "man-made" drought from water diversion, 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 very 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 very dry years experienced by the estuary based on actual annual freshwater Bay inflow amounts. very dry (VD) years were defined as the driest 20% of years in the 80-year unimpaired Delta outflows dataset (1930-2009), with total annual unimpaired inflows to the estuary of less than 15,000 thousand acre-feet (TAF) (see Table 10).

Table 10. Frequency-based classification of water years based on estimated unimpaired annual San Francisco Bay inflow (Delta outflow) from 1930-2009.

Water Year Type	**Unimpaired inflow to the San Francisco Bay** (total annual, TAF)	**Years** (1930-2009)
Very dry (driest 20% of years)	≤15,000 TAF	1931, 1933, 1934, 1939, 1947, 1976, 1977, 1987, 1988, 1990, 1991, 1992, 1994, 2001, 2007, 2008
Dry	>15,000-21,500 TAF	1930, 1944, 1949, 1955, 1957, 1959, 1960, 1961, 1964, 1966, 1968, 1972, 1981, 1985, 1989, 2009
Median	>21,500-29,500 TAF	1932, 1935, 1936, 1937, 1945, 1946, 1948, 1950, 1953, 1954, 1962, 1979, 2000, 2002, 2003, 2004
Wet	>29,500-42,000 TAF	1940, 1942, 1943, 1951, 1963, 1965, 1970, 1971, 1973, 1975, 1980, 1984, 1993, 1996, 1999, 2005
Very Wet (wettest 20% of years)	>42,000 TAF	1938, 1941, 1952, 1956, 1958, 1967, 1969, 1974, 1978, 1982, 1983, 1986, 1995, 1997, 1998, 2006

For the indicator, actual annual freshwater inflows to the Bay for each year were categorized using this water year type classification scale; for example, a year with actual annual Bay inflow of less than 15,000 TAF was categorized as "very dry" even if the unimpaired inflow for that year was higher and placed that year in a different water year category based on its unimpaired inflow. For each year, the number of very dry years (i.e., inflow<15,000 TAF) 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 was calculated for each year (1939-2014) as the difference between the number of very dry (VD) years that occurred under unimpaired conditions and the number that occurred in actual conditions as:

Dry Year Frequency
= (# VD years, actual Bay inflow <15,000 TAF for year$_{(0 \text{ to } -9)}$) – (# VD years, unimpaired Bay inflow <15,000 TAF for year$_{(0 \text{ to } -9)}$)

3. Reference Conditions

The reference condition for the Dry Year Frequency indicator was established by calculating the average difference between very dry year frequency in unimpaired Bay inflows and for unimpaired Bay inflows that had been reduced by 15-25% (depending on water year type).[20] The results of this analysis showed that reductions in unimpaired Bay inflows at the level specified increased the frequency of very dry years by 1.5 years. Therefore, the primary reference condition was set at 2 years. Differences in the numbers of very dry years between 10-year sequences of actual and unimpaired flows that were 2 years or less were considered to reflect "good" conditions and meet the CCMP goals; differences in the numbers of very dry years between 10-year sequences of actual and unimpaired flows that were more than double this level were considered to correspond to "very poor" conditions. The other reference condition levels were established based on equal increments of values based from these two levels. Table

[20] 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%.

11 below shows the quantitative reference conditions that were used to evaluate the results of the Dry Year Frequency indicator.

Table 11. 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 italics.

Dry Year Frequency		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
<u><</u>1 additional year of VD conditions	"Excellent," minimal alteration	4
***<u><</u>2** additional years of VD conditions*	***"Good," meets CCMP goals***	***3***
<u><</u>3 additional years of VD conditions	"Fair"	2
<u><</u>4 additional years of VD conditions	"Poor"	1
>5 additional years of VD conditions	"Very Poor," extreme alteration	0

4. Results

Results of the Dry Year Frequency indicator are show in Figures 16 and 17.

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

While the classification of very dry (VD) year inflows is based on the bottom quintile from the 80-year unimpaired dataset, the frequency of very dry hydrological conditions (i.e., hydrological conditions that result in VD unimpaired freshwater inflow to the estuary) has been more variable over that period (Figure 16, upper panel). The number of VD 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 VD years under unimpaired conditions, was higher (Figure 16, bottom panel, and Figure17). The largest increases in VD year frequency occurred in the 1960s, a period during which there were no VD years based on hydrological conditions in the estuary's

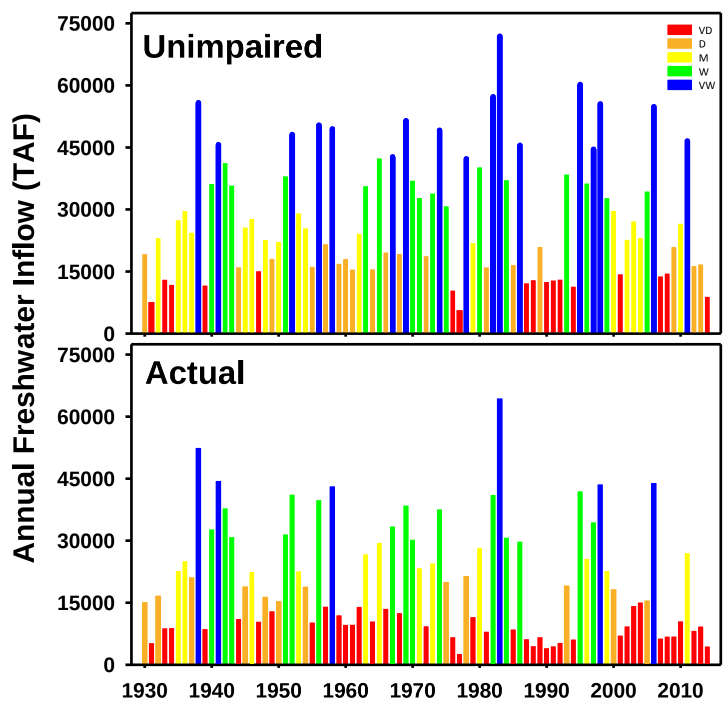

Figure 16. Freshwater inflow to the San Francisco Bay under unimpaired conditions (top panel) and actual conditions (bottom panel). Each histogram bar has been colored to the show the water year type, for example, red for very dry years with inflows<15,000 TAF and blue for very wet years with inflows >42,000 TSF. Water year type is based on frequency for unimpaired inflows and actual annual inflow amounts for actual inflows. See text for more details.

watershed, but during which the estuary received freshwater inflows comparable to VD conditions in an average of six out of 10 years. In the 1980s, an average of 1.8 years were very dry in the watershed but in the estuary an average of 4.4 years were very dry (i.e., there were an average of 2.6 more VD years out of 10 years than there were based on hydrological conditions in the estuary's watershed). Conditions during the most recent decade (2005-2014) were similar, with an average of 6.2 VD years out of 10 years for the estuary compared to just 2.2 VD years based on unimpaired conditions in the estuary's watershed. In 2014, the Bay had experienced critically low inflows in 70% of years in the past decade, a level of chronic, man-made drought conditions that had persisted since 2009.

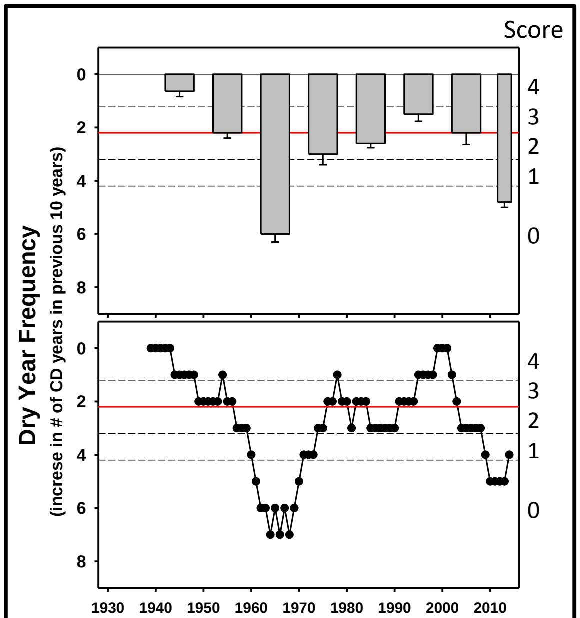

Figure 17. Results for the Dry Year Frequency indicator, expressed as the increase in the number of years of very dry actual inflow conditions (inflows<15,000 TAF) during the immediate past 10-year period, compared to the number of very dry years during that period under unimpaired conditions, for 1939 to 2014 (left Y axis) and indicator score (right Y axis). The top panel shows results as decadal averages+1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The frequency of freshwater inflow conditions in the San Francisco Estuary that are comparable to very 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 VD years has increased significantly (Wilcoxon Signed Rank test, $p<0.001$) (Figure 16). On average, the estuary experienced 2.8 more VD 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, man-made drought conditions, particularly during the 1960s and 2000s when conditions in the estuary's watershed were not chronically dry.

Based on recent very dry year frequencies in the estuary, CCMP goals to increase freshwater availability to the estuary and restore healthy estuarine habitat and function have been partially met.

Since 1990, dry year frequency conditions in the Bay were "good," meeting or exceeding conditions considered to satisfy CCMP goals, in 52% of years (13 of 25 years). However, all of these years occurred during the 1990s and early 2000s and reflected a sequence of several consecutive extremely dry years followed by several consecutive extremely very wet years. Since the early 2000s, when hydrological conditions were more moderate, the frequency of man-made drought conditions has increased. The CCMP goal has not been met in any of the past 11 years and, in the past decade, the Bay has experienced very dry inflow conditions in more than 60% of years.

V. Freshwater Inflow Index

The Freshwater Inflow Index combines the results of the ten indicators into a single number to measure the aggregate degree of alteration to the freshwater inflows to the San Francisco Bay Estuary.

A. Index Calculation

For each year, the Freshwater Inflow Index was calculated by averaging the quantitative scores of the ten indicators. Each indicator is weighted equally. For any single year, an index score that was between 2.5 and 3.5 was interpreted to represent "good" conditions in which, collectively (or an average), the different aspects of freshwater inflow conditions met the CCMP goals.

B. Results

Results of the Freshwater Inflow Index are shown in Figures 18, 19 and 20.

Freshwater inflows to the San Francisco Estuary are highly altered.

All of the ten indicators, which measured different aspects of freshwater inflow conditions, showed alteration in flows compared to estimated unimpaired conditions. Measured collectively using the Freshwater Inflow Index, the degree of flow alteration corresponds to "poor" conditions in most years since the 1970s.

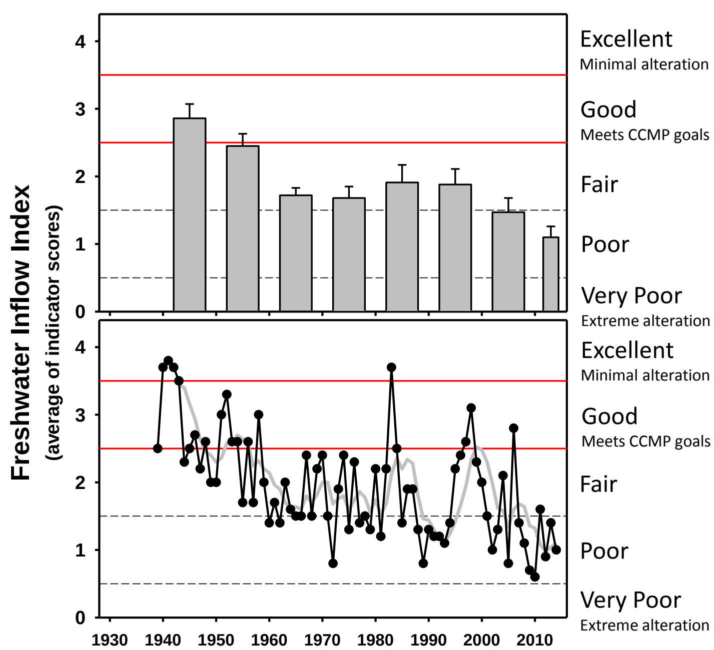

Figure 18. Results for the Freshwater Inflow Index for 1939 to 2014, expressed as the average score of the 10 component indicators. The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year, with the 5-year running average shown as the grey line. The horizontal red lines and dashed lines show the reference conditions and Index evaluation.

Freshwater inflow conditions in the estuary have declined over time.

Freshwater inflow conditions to the estuary have been increasingly altered over time; the Index has declined significantly (regression, $p<0.001$). The decrease in the Index is driven by declines in nine of the ten indicators of freshwater inflow conditions (i.e., all indicators except Annual Delta Inflow). 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 2.9 in the 1940s (1939-1949 average), to 2.4 in the 1950s, and 1.7

in the 1960s. The Index was relatively stable during the 1970s, averaging 1.7, somewhat higher and more variable during the 1980s and 1990s (1980-1989 average: 1.91; 1990-1999: 1.9) before declining again to an average of 1.5 in the 2000s and an average of 1.1 for the most recent five years. The Index has declined significantly in all water year types except very wet years (regression, p<0.01 for all year types except very wet; very wet years, p=0.09) (Figure 19). The lowest Index value, 0.6, occurred in 2010, a median year that immediately followed a dry year, 2009, which with an Index of 0.7 and was the second lowest in the 76 year record. With the exception of 2005, most of the other years with Index values below 1.0 were dry (1972, 1989, and 2012). Water Year 2005, a wet year following a median year, stands out however with an Index of 0.8, indicating that, in recent years, high levels of alteration to freshwater inflows can occur. The 2014 Index value, 1.0, was the same as in 2012 and the seventh lowest Index in the 76-year period for which it was measured.

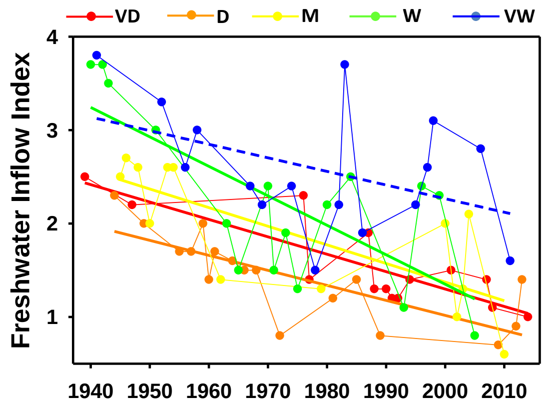

Figure 19. Results for the Freshwater Inflow Index for each water year type, from 1939 to 2014. Each color-coded plot shows the results for individual years (symbols) and the trend over time (connecting lines and regression line). Regressions (heavy solid lines) are significant for all years (p<0.01) except very wet years (heavy dashed line).

The Freshwater Inflow Index differs by water year type.

Since 1970, after most of the major dams in the estuaries watershed were completed and the Delta water export facilities became operational, the degree to which freshwater inflow conditions have been altered is significantly greater in dry, median and very dry years, compared to in very wet years and, for dry years, compared to wet years (ANOVA, all tests, p<0.05) (Figure 20).

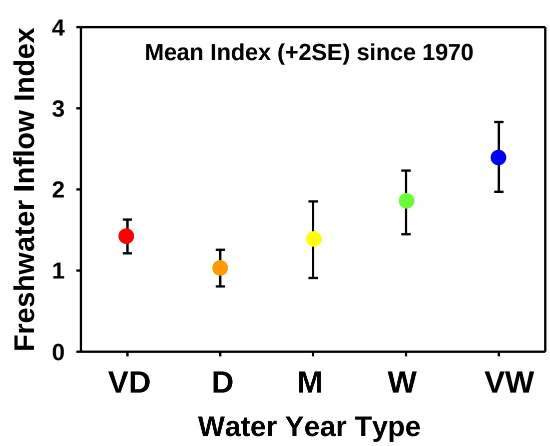

Figure 20. Post-water development (1970-2014) Freshwater Inflow Index for each water year type (mean±2 SE). Dry, Median and Very Dry years had significantly lower Index values than very wet years, and dry years have significantly lower index values that wet years (ANOVA, all tests p<0.05).

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 rarely “good” (12% of years since 1990), “fair” in some years (28% of years), and “poor” in most years (60% of years). Degraded inflow conditions reflect severe reductions in the amounts of freshwater inflow in most years, substantial reductions in seasonal variability of inflows, severe reductions in the frequency of peak flows and high frequencies of inflows comparable to very dry conditions, in effect, chronic man-made drought conditions resulting from water management operations in the estuary's watershed and upstream Delta region.

C. Summary and Conclusions

Collectively the ten indicators of the Freshwater Inflow Index provide a comprehensive assessment of the status and trends for freshwater inflow conditions to the San Francisco Bay and Sacramento-San Joaquin Delta from it largest watershed. Each of the indicators shows significant alterations to inflows to the 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 very dry hydrological conditions in the watershed. Table 12 summarizes the indicator results relative to the CCMP goals (as they are expressed by the reference conditions).

Table 12. Summary of results for the ten freshwater inflow indicators.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend since 1990**	**Current condition** (average for last 10 years)
Annual Delta Inflow	Partially met; goals achieved in 52% of years	Stable	Fair Inflow reduced by 26%
Spring Delta Inflow	Not met; goals achieved in 12% of years	Deteriorating	Poor Inflow reduced by 47%
San Joaquin River Inflow	Not met; goals achieved in 0% of years	Stable	Very poor Inflow reduced by 58%
Annual Bay Inflow	Not met: goals achieved in 12% of years	Deteriorating	Very poor Inflow reduced by 50%
Spring Bay Inflow	Not met; goals achieved in 12% of years	Deteriorating	Very poor Inflow reduced by 56%
Delta Diversions	Not met; goals achieved in 8% of years	Deteriorating	Poor 36% of inflow diverted
Inter-annual Variation in Inflow	Fully met; goals achieved in 92% of years	Mixed (variable)	Good Reduced by 10%
Seasonal Variation in Inflow	Not met; goals achieved in 32% of years	Deteriorating	Poor Reduced by 50%
Peak Flow	Partially met; goals achieved in 44% of years	Stable	Fair Reduced by 45 days/year
Dry Year Frequency	Partially met: goals met in 52% of years	Deteriorating	Poor Flow reductions triple dry year frequency
Freshwater Inflow Index	Not met; goals met in 12% of years	Mixed (variable)	Poor Only 1 of 10 indicators show “good” conditions

VI. References

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State of the Estuary Report 2015

Summary

HABITAT – Open Water Habitat Indicators

Summary

Prepared by Christina Swanson

Natural Resources Defense Council

September 2015

State of the San Francisco Estuary 2015

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What are the indicators?
The State of the Estuary Report uses two indicators to measure and evaluate the frequency, magnitude, and duration of the occurrence of good quality open water habitat conditions in estuary. The Delta Open Water Habitat indicator measures hydrodynamics and the occurrence of net downstream flow in the Delta. The Estuarine Open Water Habitat indicator measures the occurrence of low salinity conditions in the Bay's upstream embayment, Suisun Bay, during the ecologically important spring period.

Table 1.

Attribute	Indicators	Benchmarks
Habitat (Open Water Habitat)	Two indicators measure the frequency, magnitude and duration of: 1) net downstream flow in the western Delta (Delta Open Water Habitat indicator); and 2) low salinity habitat in Suisun Bay during the spring (Estuarine Open Water Habitat indicator).	Benchmarks (or reference conditions) are based on: 1) relationships between hydrodynamic conditions and entrainment of fishes at the state and federal water export facilities; 2) relationships between X2 and estuarine species survival and population abundance; 3) current regulatory standards for seasonal Delta outflow (i.e., State Water Resources Control Board, 2006 Water Quality Control Plan); and 4) unimpaired, pre-dam (before 1944) and pre-water export facility (before 1970) flow conditions.

Why is open water habitat important?
Most of the area of the San Francisco Bay Estuary is open water habitat. In the large, mostly shallow embayments – Suisun, San Pablo and South Bays – open water habitat conditions are largely defined by salinity, which varies seasonally and can range from near freshwater conditions in Suisun Bay to as salty as the adjacent Pacific Ocean in South Bay. In the Delta, with its narrow, relatively deep channels, large inputs of freshwater from the estuary's tributary rivers, and large water diversion facilities which extract large volumes of water, open water habitat conditions are more defined by hydrodynamics, or the movement patterns of water in its channels.

Both of these open water habitat features are affected by freshwater inflows from the estuary's Sacramento-San Joaquin watershed and by diversions of those flows upstream of the estuary and in the Delta. High river inflows push water through the Delta from the north, east and south to the west into Suisun Bay where it mixes with saltier water from the Bay and Pacific Ocean, creating the low salinity, brackish water habitat that is a defining feature of estuaries. Low river inflows and/or high rates of water extraction in the Delta can alter and even reverse flow patterns in Delta channels, changing open water channel habitat conditions and, by reducing freshwater inflows to Suisun Bay, reduce the quality and quantity of low salinity, estuarine open water habitat.

What are the benchmarks? How were they selected?

The benchmarks (or reference conditions) for the two indicators are based on: 1) relationships between Delta channel hydrodynamic conditions and entrainment of fishes at the state and federal water export facilities; 2) relationships between X2 (a measure of the location of low salinity habitat in the estuary) and estuarine species survival and population abundance; 3) current regulatory standards for seasonal Delta outflow (i.e., State Water Resources Control Board, 2006 Water Quality Control Plan); and 4) unimpaired, pre-dam (before 1944) and pre-water export facility (before 1970) flow conditions. The benchmark (or primary reference condition) for the frequency, magnitude and duration components of the two indicators was set to conditions that correspond with low entrainment mortality and moderately good species abundance and survival per the relationships identified above.

What are the status and trends of the indicators and Index?

The two open water habitat indicators show that the frequency, magnitude and duration of the occurrence of "good" quality open water habitat have declined significantly since the 1970s and are now poor in most years. Hydrodynamics conditions in the Delta have deteriorated consistently "good" prior to 1970 to "poor" or "very poor" in most (68%) years. Springtime low salinity habitat conditions are more variable but, since the 1990s, they have been "poor" or "very poor" in most years.

Table 2.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend (long term; 1930-2014)**	**Trend since 1990**	**Current condition** (average for last 10 years)
Delta Open Water Habitat	Not met; goals achieved in 4% of years	Decline	Deteriorating	Poor Frequency, magnitude and duration net downstream flow conditions too low to support native species in the Delta
Estuarine Open Water Habitat	Not met; goals achieved in 20% of years	Decline	Mixed	Poor Frequency, magnitude and duration of good quality low salinity habitat in the spring too low support to

				flow-dependent fish and invertebrates

What does it mean? Why do we care?

Open water habitat in the San Francisco Bay Estuary is used by many fish and invertebrates species, including all six of the Endangered Species Act listed fish species. The open water habitat conditions measured by the two indicators affect the species composition, survival and population abundance of many fish and invertebrates species in the Estuary. The declines in habitat condition measured by the two indicators, channel flow patterns in the Delta and seasonal low salinity habitat quality and quantity in the Bay's upstream embayment, Suisun Bay, are the result of human activities: water management in the Estuary's watershed and in the Delta. Regulatory standards for freshwater inflows to the Estuary and Delta water export rates affect both of these open water habitat characteristics but those standards have not provided habitat conditions that, according to this evaluation, meet the CCMP goals nor prevented the continuing decline in these habitat conditions. Restoration of the San Francisco Bay's estuarine ecosystem and recovery of its many threatened species will require improving open water habitat conditions.

State of the Estuary Report 2015

Technical Appendix

HABITAT – Open Water Habitat Indicators
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June 2015

State of San Francisco Estuary 2015

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I. Background and Rationale

The San Francisco Bay Estuary is large and geographically complex; the surface area of the entire waterbody is more than 1600 square miles (SFEI 1994). Therefore, not surprisingly, the physical and ecological characteristics of its open water habitats differ substantially among the estuary's different regions. In the Bay's large, mostly shallow embayments – Suisun, San Pablo and South Bays – open water habitat is largely defined by salinity, which can range from near freshwater conditions in Suisun Bay to as salty as the adjacent Pacific Ocean in South Bay (Kimmerer 2002, 2004). In contrast, the upstream region of the estuary, the Delta, is both highly channelized with open water habitat confined to narrow, relatively deep channels and, except during periods of extremely low freshwater inflows, predominately fresh water. In addition, the Delta receives large, localized inputs of freshwater inflow from the Sacramento, San Joaquin and eastside tributary rivers and is also the site of several water diversion facilities where large volumes of water are extracted from some Delta channels. Thus, in the Delta, hydrodynamics, or the movement patterns of water in its channels, is an important open water habitat characteristic.

Both of these open water habitat features are affected by freshwater inflows from the estuary's Sacramento-San Joaquin watershed and by diversions of those flows upstream of the estuary and in the Delta (see also Freshwater Inflow Index).[1] High river inflows push water through the Delta from the north, east and south to the west into Suisun Bay. There, the inflowing fresh water mixes with saltier water from the Bay and Pacific Ocean, creating the low salinity, brackish water habitat that is a defining feature of estuaries. Low river inflows and/or high rates of water extraction in the Delta can alter and even reverse flow patterns in Delta channels, changing open water channel habitat conditions and, by reducing freshwater inflows to Suisun Bay, reduce the quality and quantity of low salinity, estuarine open water habitat (Jassby et al. 1995; Kimmerer 2002, 2004; Feyrer et al. 2007; CCWD 20012).

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[1] Flows in Delta channels and the Bay are also influenced by tidal action. The estuary experiences two tides every day, two high tides and two low tides, and magnitude of the high and low tides varies over a 28-day spring-neap cycle. Under conditions of low to moderate inflows, tidal flows in Delta channels can be an order of magnitude greater than the freshwater inflow and the direction of flow in the channels typically reverses twice daily with the tides. However, all flow data used to calculate the indicators are daily averages and have been filtered to remove tidal effects.

In the Delta and Bay, the conditions of these different habitat characteristics – salinity and hydrodynamics – have been shown to be related to the abundance, survival and species composition of fish and invertebrates that live in and move through these habitats. For example, native fish species are more prevalent in Delta channels with higher flows (as well as high turbidity) than in channels with lower or altered flows, which favor non-native species (Feyrer and Healey 2003). Further, alteration or even reversal of natural flow movements in Delta channels induced by operations of the local water export facilities can lethally entrain fish and other small pelagic organisms, with entrainment rates directly related to the magnitude of "reverse" flows in some Delta channels (Grimaldo et al. 2009). Downstream in the estuary, the location of low salinity habitat in Suisun Bay rather than further upstream in the Delta during the spring corresponds to higher survival and population abundance of numerous estuarine fish and invertebrate species (Jassby et al. 1995; Kimmerer 2004; Feyrer et al. 2007).

The State of the Estuary Report uses two indicators to measure and evaluate the frequency (or "how often?"), magnitude ("how much?") and duration ("how long?") of the occurrence of good quality open water habitat conditions in estuary. The Delta Open Water Habitat indicator measures hydrodynamics and the occurrence of "reverse flow" conditions in the Delta. The Estuarine Open Water Habitat indicator measures the occurrence of low salinity conditions in the Bay's upstream embayment, Suisun Bay, during the ecologically important spring period. Figure 1 shows the locations for the measurements of these two indicators.

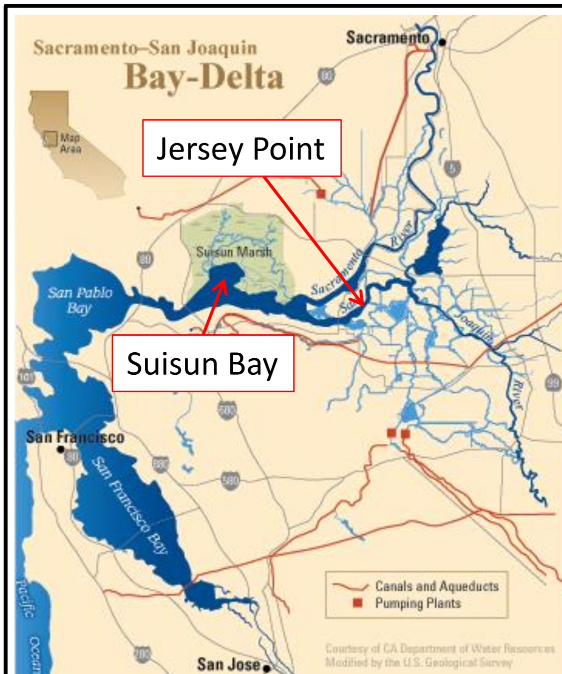

Figure 1. Open water habitats of the San Francisco Bay and Delta. The Delta Open Water Habitat indicator is measured in the western Delta at Jersey Point. The Estuarine Open Water Habitat indicator is measured in Suisun Bay, the upstream embayment of the San Francisco Bay.
Source: U.S. Geological Survey, available at: http://pubs.usgs.gov/fs/2010/3032/

II. Data Sources

The Delta Open Water Habitat and Estuarine Open Water Habitat indicators were calculated for each year using daily data from the California Department of Water Resources (CDWR) DAYFLOW model (using Qwest[2] for the Delta Open Water Habitat indicator and X2[3] for the Estuarine Open Water Habitat indicator). DAYFLOW is a computer model developed in 1978 as an accounting tool for calculating historical Delta outflow, X2 and other internal Delta flows.[4] DAYFLOW output is used extensively in studies by state and federal agencies, universities, and consultants. DAYFLOW

[2] Qwest is the estimated flow in the San Joaquin River at Jersey Point, located in the western Delta.
[3] X2 is a commonly used indicator of the location and quality of low salinity habitat in the San Francisco Bay Estuary. It represents the linear distance in kilometers upstream from the Golden Gate of the 2 ppt isohaline and it is calculated as a function of Delta outflow (or Bay inflow) from equations published in Jassby et al. (1995).
[4] More information about DAYFLOW is available at www.iep.ca.gov/dayflow.

output is available for the period 1930-2014. For the Delta Open Water Habitat indicator, additional information on interior Delta channel flows provided by the Contra Costa Water District was used to inform development of reference conditions and interpret indicator results (CCWD 2012).[5] For the Estuarine Open Water Habitat indicator, information on unimpaired Delta outflow (or Bay inflow) from CDWR's California Central Valley Unimpaired Flow dataset and calculated X2 conditions (Jassby et al. 1995) was used to inform development of reference conditions and interpret indicator results.[6]

III. 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, for a number of resident and migratory fish and invertebrate species, entrainment mortality, survival and population abundance are related to in-Delta hydrodynamic conditions and/or seasonal low salinity habitat conditions in the upper estuary, Suisun Bay. Therefore, the primary and intermediate reference conditions against which the measured values of the indicator component metrics were compared were based on relationships between hydrodynamic conditions and entrainment of fishes at the state and federal water export facilities (e.g., CCWD 2012), and relationships between X2 and estuarine species survival and population abundance (e.g., Kimmerer 2002, 2004), as well as examination of current regulatory standards, unimpaired flows, and pre-dam (before 1944) and pre-water export facility conditions (before 1970).

For each indicator and its frequency, magnitude and duration component metrics, a primary reference condition was established. Measured values that were higher than the primary reference condition were interpreted to mean that aspect of open water habitat conditions met the CCMP goals and corresponded to "good" ecological conditions. Specific information on the primary reference condition and additional intermediate reference conditions is provided below for each indicator.

Effects of Water Year Type on Flood Flows and the Indicators: 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.[7] Even in the current system, in which flows are highly altered by dams and water diversion, annual and seasonal flow volumes vary substantially between wet and dry years. However, for evaluation of these two

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[5] Data from Contra Costa Water District's Flow Index for Old and Middle River flows were kindly provided by Deanna Sereno, Contra Costa Water District.

[6] 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 2). Unimpaired inflow is not the same as "natural" or "historical" inflow that would have occurred in the watershed prior to human development and land use changes; it is instead an estimate of what flows over the existing landscape would have been if there were no dams or diversions. This 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.

[7] For these analyses, the water year type for each year was categorized based on the level of annual unimpaired Delta outflow and the frequency of occurrence of that level during the reference period of 1930-2009, with each year type comprising 20% of all years. Five water year types were used, each comprising 20% of all years: very wet (for the wettest 20% of years), wet, median, dry and very dry (for the driest 20% of years). For more information on this, see Freshwater Inflow Index and Figure 3.

indicators, water year type was not considered. Instead the indicators measure actual flow conditions for each year, and those measured levels are compared to reference conditions that do not vary with water year type. Therefore, measured values for frequency, magnitude and duration of Qwest and X2 and the evaluation results relative to habitat condition and the ecological services provided by those habitats (i.e., “good” v “poor”) are lower in dry years (and multi-year droughts) than in wetter years. (In contrast, the indicators of the Freshwater Inflow Index, which include measures of Delta inflow, outflow and in-Delta diversions, and spring inflow to the Bay, measure alteration in these flow conditions compared to unimpaired flow conditions and have therefore been at least partially normalized to account for differences in water year type.)

IV. Indicators

A. Delta Open Water Habitat

1. Rationale

The movement of water in Delta channels is influenced by the amounts of fresh water flowing in from upstream, the ebb and flood of the twice-daily tides, the amounts and locations of in-Delta water diversions, and the Delta’s geometry, including man-made channels and barriers. Before the massive transformation of the estuary’s Sacramento-San Joaquin watershed by humans, fresh water flowing into the Delta sloshed back and forth with the tides but ultimately moved downstream through Delta channels and west to the Bay (TBI 1998). Delta water diversions, particularly those located in areas of the Delta with low freshwater inflows, alter this flow pattern: when diversion rates are high, flows in some Delta channels may reverse, with water flowing “uphill” towards the point of diversion (CCWD 2012). Operations of the many barrier dams installed in Delta channels, most designed to deflect water towards diversion pumps, can further alter flow patterns and exacerbate reverse flows. Location of the large state and federal export pumps in the southern Delta, where freshwater inflows from the San Joaquin River were historically less than a quarter of the Delta total and have since been further reduced (see Freshwater Inflow Index, San Joaquin River Inflow indicator), concentrated the effects of their diversion operations in that portion of the Delta.

2. Methods and Calculations

The Delta Open Water Habitat indicator uses three component metrics to assess the frequency, magnitude and duration of occurrence of positive (or downstream) net flow conditions in the San Joaquin River in the western Delta at Jersey Point, referred to as Qwest, throughout the year. According to CDWR’s Dayflow model, Qwest is affected by Delta inflows from the San Joaquin, Cosumnes and Mokelumne Rivers, exports from the state and federal pumping facilities, cross Delta flows (e.g., through the Delta Cross Channel and Georgiana Slough), in-Delta depletions and diversions, and local precipitation. Net reverse flow, or negative flow, past Jersey Point indicates that higher salinity water from the Bay is being drawn into the interior Delta as a result of high depletions and exports compared to stream inflows, precipitation, and cross-Delta flows.

Frequency was measured as:

# of years in the past decade (i.e., ending with the measurement year) with Qwest>2500 cfs for at least 200 days during the year.

Magnitude was measured as:

average daily Qwest (cfs) during the year.

Duration was measured as:

# of days with Qwest>2500 cfs during the year.

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

3. Reference Conditions

The primary reference conditions for the component metrics of the Delta Open Water Habitat indicator were established as Qwest>2500 cfs for at least 200 days during the year in at least 6 out of 10 years. The Qwest level of 2500 cfs was based on comparison of Qwest and the CCWD's Flow Index for Old and Middle River flows, shown in Figure 2. This Qwest level roughly corresponds to reverse flows in Old and Middle Rivers (OMR), the two channels leading to the state and federal water export facilities, of approximately -2800 cfs. At OMR levels more negative than this, e.g., -5000 cfs, entrainment rates for fish and other small pelagic organisms in the central and south Delta increase markedly (CCWD 2012). The primary reference conditions for duration, 200 days, and frequency, ≥6 out of 10 years, specify that this Qwest level should occur for more than half of the year in more than half of all years. Qwest conditions that met or exceeded these levels were considered to reflect "good" conditions and meet the CCMP goals. Additional information about the relationship between Qwest and CCWD's OMR flow index, measured and modeled fish entrainment rates of the water export facilities (CCWD 2012), regulatory flow criteria for San Joaquin River inflows (SWRCB 2006), and pre-Delta export facilities Qwest flows were used to develop the other intermediate reference condition levels. Table 1 below shows the quantitative reference conditions that were used to evaluate the results of the component metrics for the Delta Open Water Habitat indicator.

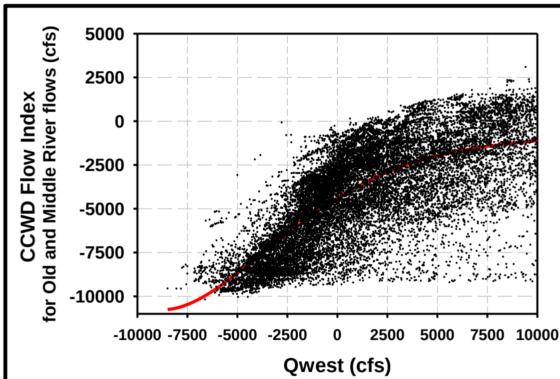

Figure 2. Relationship between daily Qwest (cfs) and daily CCWD Flow Index flows for Old and middle Rivers (cfs) (data from 1956-2014; graph axes are truncated to exclude extreme high and low flow levels)outside the range of the indicator reference conditions. The red line shows a best fit regression for visual reference.
Data sources: Contra Costa Water District for CCWD Flow Index, and CDWR Dayflow for Qwest.

Table 1. Quantitative reference conditions and associated interpretations for results for each of the three component metrics of the Delta Open Water Habitat indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Delta Open Water Habitat				
Quantitative Reference Conditions			Evaluation and Interpretation	Score
Frequency	Magnitude	Duration		
≥8 years out of 10	Qwest>5000 cfs	>275 days	"Excellent," similar to pre-water export conditions	4
6 or 7 years out of 10	***Qwest>2500 cfs***	***>200 days***	***"Good," meets CCMP goals***	***3***
4 or 5 years out of 10	Qwest>0 cfs	>125 days	"Fair," corresponds to >-5,000 cfs CCWD OMR flow	2
2 or 3 years out of 10	Qwest>-2500 cfs	>50 days	"Poor," predicted entrainment high	1
0 or 1 years out of 10	Qwest≤-2500 cfs	≤50 days	"Very Poor," chronic, severe reverse flows	0

4. Results

The frequency of positive net flow in the Delta is low (Figure 3, top panel).

Prior to the 1970s, open water habitat conditions in the Delta were characterized by consistent, net positive flows that met or exceeded the primary reference condition: San Joaquin River flows in the western Delta were greater than 2500 cfs for more than 200 days per year in 93% of years. Beginning in the 1970s, frequency of occurrence of these conditions declined significantly (regression, $p<0.001$), falling to just 21% of years during the last 25 years and just 12% of years during the last decade. Based on frequency of occurrence of net positive flows, open water habitat conditions in the Delta are poor.

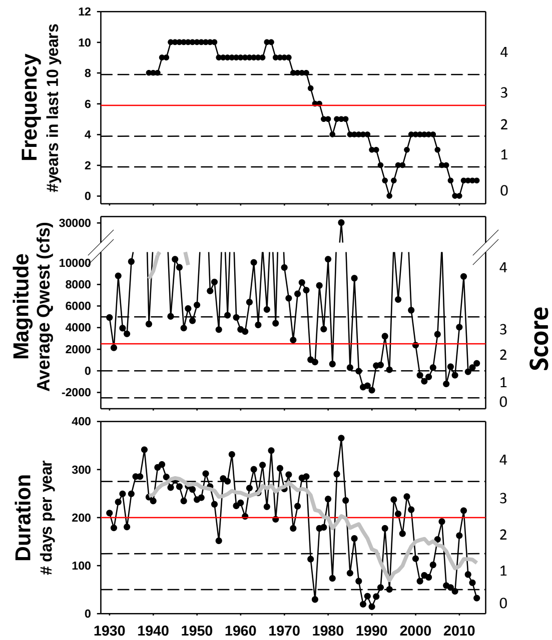

Figure 3. Results of the frequency (top panel), magnitude (middle panel) and duration (bottom panel) component metrics of the Delta Open Water Habitat indicator. Score is shown on the right Y axis. Each point shows the result for that year and, for the magnitude and duration metrics, the heavy solid grey line shows the 10-year running average. The horizontal red and dashed lines show the reference conditions for each metric and the numeric score is shown on the right Y axis.

The magnitude of net positive flows in the Delta is variable but it has declined over time (Figure 3, middle panel).

The magnitude of average annual flows in the western Delta is highly variable and largely a function of water year type (i.e., wet v dry). Prior to the 1970s, average annual Jersey Point flows were always positive, ranging from 3539 cfs in very dry years (i.e., the driest 20% of years) to 17,941 cfs in very wet years (the wettest 20% of years). Beginning in the 1970s and 1980s, western Delta flows declined significantly in all water years except very wet years (regression, all years except very wet, $p<0.001$; very wet years, $p=0.06$). Since 1980, average annual flows have been negative in very dry and dry years (the driest 40% of years), averaging -291 cfs in very dry years and -125 cfs in dry years. Average annual flows have declined 89% in median years, from 8817 cfs to just 1018 cfs, by 45% in wet years and by 23% in very wet years. Based on the magnitude of net positive flows in the western Delta, open water habitat conditions in the Delta have been good in only 40% of the last 25 years.

The duration of net positive flows in the Delta has declined over time and is low in most years (Figure 3, bottom panel).

The number of days or net positive flow >2500 cfs has declined significantly from an average of 256 days per year prior to 1970 to an average of 115 days per year during the last 25 years (Mann-Whitney, $p<0.001$). During the last 10 years, Qwest flows>2500 cfs have occurred for an average of 106 days per year and in 2014, a very dry year, for only 32 days. Duration of Qwest flows>2500 cfs declined significantly in all water year types (regression, $p<0.01$, all tests) and differed significantly among most water year types (ANOVA, $p<0.05$ for very dry v all other year types and dry years v very wet and wet year types): duration in very dry years declined from an average 217 days prior to 1970 to an average of just 45 days since 1990 compared the pre-1970 average for very wet years of 322 days and its decline to 232 days since 1990.

Results of the Delta Open Water Habitat indicator, which combines the results of the frequency, magnitude and duration metrics, are shown in Figure 4.

Delta open water habitat conditions, as measured by the hydrodynamic conditions, have declined from consistently good to predominantly poor.

Hydrodynamic conditions in the western Delta deteriorated sharply starting in the 1970s; by the mid-1980s, the frequency, magnitude and duration of Qwest flows>2500 cfs were "poor" in all but a few wet years. This period of decline coincides with the ramp-up to full capacity operations of the state and federal water export facilities in the Delta. Declining habitat conditions were driven by reductions in all three component measurements of the indicator. Frequency of occurrence of good open water habitat has been cut by 77%, from an average of 9 out of ten years prior to 1970, to just 2 years out of 10 in the last 25 years. The magnitude of net positive flows has declined 65%, from an average of 9202 cfs prior to 1970 to just 3241 cfs since 1990. The number of days with "good" hydrodynamic conditions has declined by 55%, from an average of 257 days per year to 116 days per year.

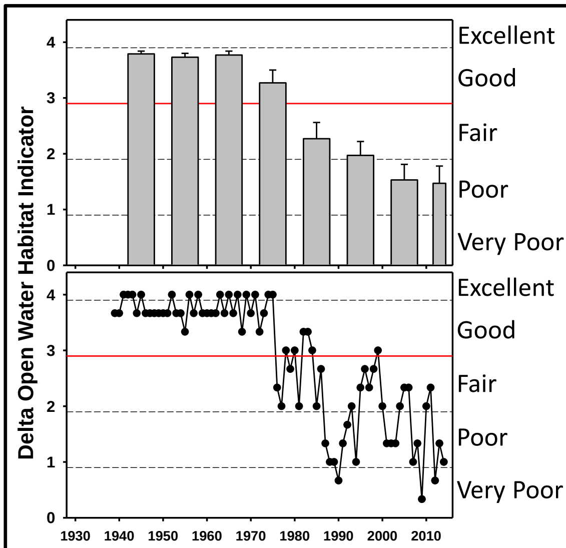

Figure 4. Results for the Delta Open Water Habitat indicator, which combines the results of the frequency, magnitude and duration component metrics (Figure 3) for 1939 to 2014. The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red and dashed lines show the reference conditions and the indicator evaluation categories are at right.

Based on the Delta Open Water Habitat indicator, CCMP goals to restore healthy estuarine habitat and function have not been met.

Since the early 1990s, when the CCMP was implemented, open water habitat conditions in Delta have been "poor" or "very poor" in 17 years (68% of years) and "good" in only one year (4% of years).

B. Estuarine Open Water Habitat

1. Rationale

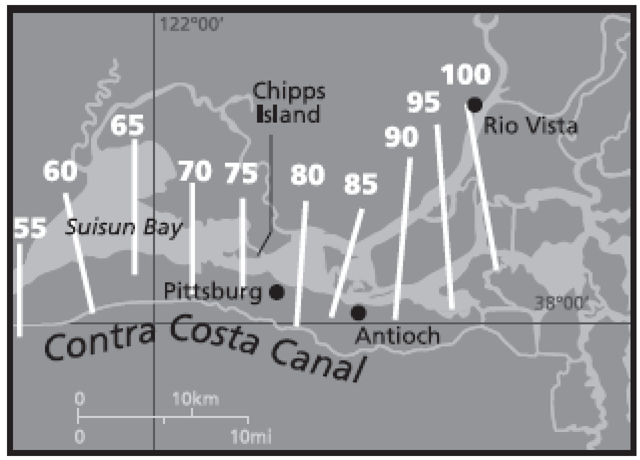

Figure 5. The location, quantity and quality of low salinity open water habitat is often measured in terms of "X2", the location in kilometers from the Golden Gate of the 2 parts per thousand isohaline. Based on survival and abundance of many estuary-dependent fish and invertebrate species, X2 locations at of downstream of 65 km provide good habitat conditions.
Figure from: The Bay Institute, 2003.

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 are largely determined by the amount of freshwater inflow. In the San Francisco Bay Estuary, 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; Reed et al. 2014).[8] 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 and closer to the Golden Gate (i.e., X2 is low), creating a large expanse of estuarine open water habitat (Figure 5). When springtime inflows are low, fresh and ocean waters mix farther upstream, X2 is higher and the quality and quantity of the estuary's low salinity habitat is reduced (Feyrer et al. 2007). 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 Estuary 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 Estuary'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 Freshwater Inflow Index).

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

[8] X2 can be measured directly as salinity but it is more frequently calculated using daily or monthly Delta outflow (or Bay inflow) data using equations first developed by Schubel et al. (1993). More recent analyses indicate that these equations may be underestimating and/or overestimating X2 under extreme flow conditions (Reed et al. 2014), however, the original X2 equation in CDWR's Dayflow data is still widely used.

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 Estuary, 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 Spring Bay Inflow indicator, 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.

2. 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.

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.

Magnitude was measured as:

> average daily X2 during the February-June period.

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 Reference Conditions section below.

3. Reference Conditions

The primary reference conditions for the component metrics of the Estuarine Open Water Habitat indicator were established as X2<65 km for at least 100 days during the February-June period in at least 6 out of 10 years. The X2 level of 65 km was based on review of the relationship between X2 and abundance and survival of selected estuary-dependent fish and invertebrate species that showed that open water habitat conditions with X2<65 km corresponded 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. Examination of unimpaired flow and X2 data yielded similar results: X2<65 km occurred in 83% of years for an average or 4.3 months during the spring in 7 to 8 years out of 10 years. Measured values that were above the primary reference condition were interpreted to correspond to "good" conditions. Other intermediate reference conditions were based on the pre-dam and unimpaired X2 data, abundance-X2 relationships, and current regulatory standards for seasonal Delta outflow (or Bay inflow; SWRCB 2006). Table 2 below shows the quantitative reference conditions that were used to evaluate the results of the component metrics for the Estuarine Open Water Habitat indicator.

Table 2. Quantitative reference conditions and associated interpretations for results for each of the three component metrics of the Estuarine Open Water Habitat indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Estuarine Open Water habitat				
Quantitative Reference Conditions			Evaluation and Interpretation	Score
Frequency	Magnitude	Duration		
≥8 years out of 10	X2<60 km	>130 days	"Excellent," similar to unimpaired conditions	4
6 or 7 years out of 10	***X2<65 km***	***>100 days***	***"Good," meets CCMP goals***	***3***
4 or 5 years out of 10	X2<70 km	>50 days	"Fair," similar to current regulatory standards	2
2 or 3 years out of 10	X2<75	>25 days	"Poor"	1
0 or 1 years out of 10	X2≥75 km	≤25 days	"Very Poor," spring inflows eliminated	0

4. Results

Results of the three component measurements of the Estuarine Open Water Habitat indicator are shown in Figure 6.

The frequency of occurrence of high quality estuarine open water habitat has declined (Figure 6, 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 2014, the estuary experienced only 3 years (2005, 2006 and 2011) in which estuarine open water habitat conditions were "good."

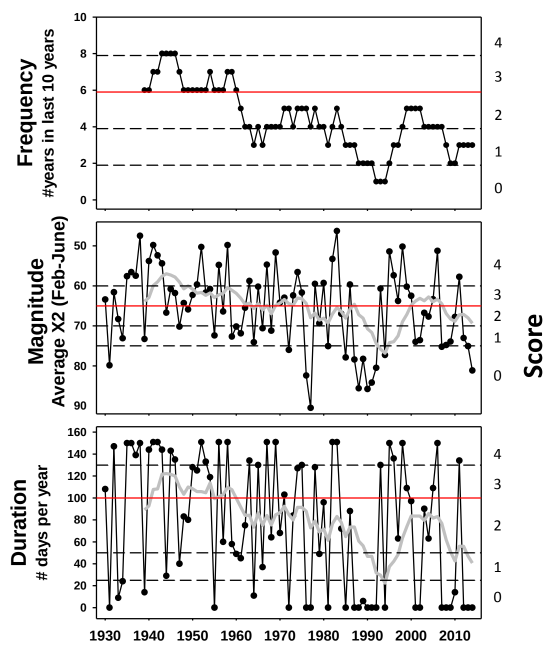

Figure 6. Results of the frequency (top panel), magnitude (middle panel) and duration (bottom panel) component metrics of the Estuarine Open Water Habitat indicator. Score is shown on the right Y axis. Each point shows the result for that year and, for the magnitude and duration metrics, the heavy solid grey line shows the 10-year running average. The horizontal red and dashed lines show the reference conditions for each metric and the numeric score is shown on the right Y axis.

The quality and quantity of estuarine open water habitat has declined (Figure 6, 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 last decade (2005-2014), X2 has averaged 69 km, significantly higher (i.e., poorer conditions) than during the 1940s and 1950s (t-test, $p<0.05$). In 2014, a very dry year, springtime X2 was 81 km, the sixth highest level in the 85-year data record. Average springtime X2 has significantly increased in all water year types except very dry years (regression $p=0.12$) and very wet years ($p=0.10$; regressions for dry, median and wet year type, $p<0.05$, all tests). The greatest upstream shifts in low salinity habitat have occurred in the dry and median years; since the pre-dam period, average spring X2 has increased by 5, 9, 10, 6 and 6 km for very dry, dry, median, wet and very wet years, respectively (averages are for 1990-2014).

The duration of occurrence of high quality estuarine open water habitat has declined (Figure 6, 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 41 days during the spring and, in six of those years, daily X2 was never downstream of 65 km. The number of days with X2<65 km declined significantly in all water year types except wet and very wet (the wettest 40% of years) (regression, $p<0.01$ for very dry, dry and median years; for wet and very wet years, $p=0.1$ and $p=0.17$, respectively). Spring days with X2<65 km have been eliminated in the driest 40% of years, falling from an pre-dam (1930-1943) average of 12 days in very dry years and 108 days in dry years to 0 days in each of these year types since 1990. In median years, the number of days with X2<65 km has been cut by two thirds, from a pre-dam average of 146 days per year to just 53 days per year since 1990.

Results of the Estuarine Open Water Habitat indicator, which combines the results of the frequency, magnitude and duration metrics, are shown in Figure 7.

Springtime estuarine open water habitat conditions have declined.

Results of the indicator reveal a steady and significant decline in the springtime estuarine open water habitat conditions (regression, $p<0.001$), from consistently "good" or "fair" 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. In the last 25 years, springtime open water habitat conditions have been "good" only during wettest 40% of years (wet and above normal year types) and consistently "poor" in nearly all of the rest of the years. 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 7 out of 10 years, or 70% of years, in the 1940s and 1950s to just 31% 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 last ten years. 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 41 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.

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 5 of 25 years (20% of years). In the remaining 80% of years, open water habitat conditions have been "fair" in 7 years (28% of years), and "poor" or "very poor" in 13 years (52% of years).

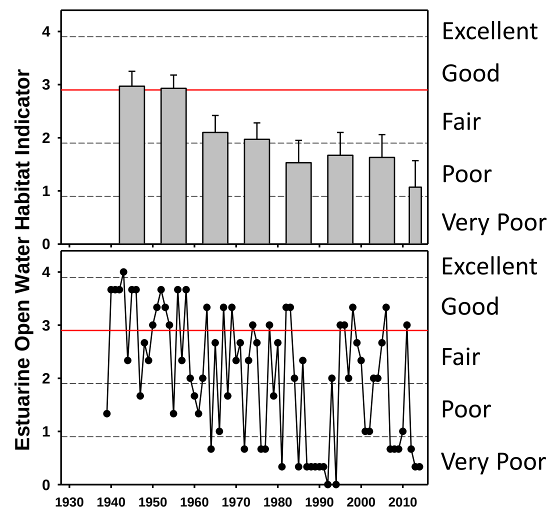

Figure 7. Results for the Estuarine Open Water Habitat indicator, which combines the results of the frequency, magnitude and duration component metrics (Figure 6) for 1939 to 2014. The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red and dashed lines show the reference conditions and the indicator evaluation categories are at right.

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State of the Estuary Report 2015

Summary

HABITAT – Eelgrass
Summary

Prepared by Caitlin Sweeney, San Francisco Estuary Partnership; Korie Schaeffer, NOAA Fisheries West Coast Region; Natalie Cosentino-Manning, NOAA Fisheries Restoration Center; Marilyn Latta, State Coastal Conservancy

SotER –Habitat Indicator

Subtidal - Eelgrass (*Zostera marina)*

1. Brief description of indicator and benchmark

Attribute	Indicator	Benchmarks
HABITAT		
Subtidal	• Eelgrass coverage (acres)	The benchmarks of 8,000 and 4,000 acres are based on the 2010 *San Francisco Bay Subtidal Habitat Goals Report* which established a goal of increasing native eelgrass populations in SF Bay within 8,000 acres of suitable subtidal/intertidal area of a 50-year time frame using a phased approach.

2. Indicator status and trend measurements

	STATUS	TREND	DETAILS
Eelgrass	Poor	Mostly Improving	Monitoring of eelgrass acreage since 2003 has shown a general expansion trend. However, current eelgrass acreage is significantly less than the estimated maximum potential coverage, based on a habitat suitability model. In addition, there has been a recent decline in eelgrass bed coverage that is a significant departure from the expansion trend.

3. Brief write-up of scientific interpretation

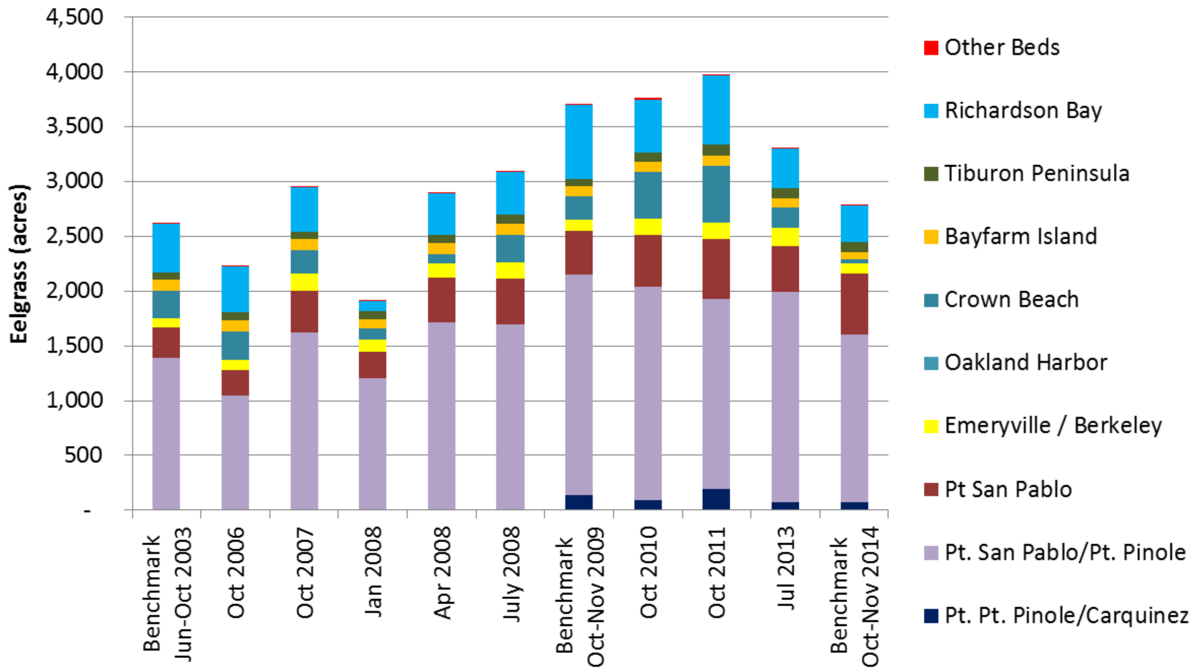
The indicator for health of the subtidal habitat of the San Francisco Bay is acreage of eelgrass beds. In San Francisco Bay, eelgrass is the most extensive type of submerged aquatic vegetation, or underwater flowering plants. Eelgrass performs a wide variety of functions in the Bay. Eelgrass beds provide shelter and food to small fishes of a variety of species, such a pipefish, kelpfish, staghorn sculpin, and multiple other species that are either bay resident, or which transit through the bay during portions of their life history. Eelgrass provides food for various species of birds both directly and indirectly. Eelgrass is also used as a preferred substrate for spawning by Pacific herring. Eelgrass beds mute wave energy, slow currents and trap sediment, reducing turbidity and shoreline erosion. Inventories of eelgrass bed coverage in the San Francisco Bay have been undertaken since 2003 under a comprehensive monitoring program, allowing the tracking of eelgrass trends over time.

The *San Francisco Bay Subtidal Habitat Goals Report* (Subtidal Goals Report) produced in 2010 contains restoration goals for native eelgrass in San Francisco Bay. The goals were based on a comparison of the current coverage of eelgrass (about 1% of the Bay) compared to the maximum potential coverage of eelgrass (about 9% of the Bay), determined by a habitat suitability model. The Subtidal Goals Report determined that the restricted extent of eelgrass beds may be limiting their support of valued ecosystem services and, furthermore, that restoration of eelgrass beds has been demonstrated and is feasible. The benchmark for eelgrass is based on the restoration goal in the Subtidal Habitat Goals report of increasing eelgrass populations in the Bay within 8,000 acres of suitable subtidal/intertidal area over a 50-year time frame using a phased approach under a program of adaptive management. The benchmarks under the phased approach are to increase eelgrass coverage by 25 acres within 5 years, 100 acres within 10 years, and up to 8,000 acres within 50 years.

The overall trend for eelgrass bed coverage since 2003 has been expansion. The 2009 baseline used for the Subtidal Goals report was 3,700 acres, and by 2011 acreage increased to just under 4,000 acres, thus meeting the initial 5 and 10 year goals of expansion of coverage by 25 and 100 acres. However, more recent monitoring data from 2013 and 2014 shows a significant decline of eelgrass bed coverage to 3,300 acres and approximately 2,790 acres, respectively. This is well below the 2009 baseline used in the Subtidal Goals Report. This recent decline is a significant departure from the expansion trend observed in the bay since 2003, leading to concerns about the possible end or even reversal of this trend. However, eelgrass beds are a dynamic habitat and can experience tremendous variability in coverage from year-to-year. In many instances, significant declines and increases, and even baywide distribution patterns may be attributed to specific environmental conditions or unique events. For instance, substantial declines in eelgrass, that were detected in October 2016, principally resulted from December 2005-January 2006 storms and subsequent flooding from the local watersheds and the Delta. This event depressed salinities throughout the north Bay for periods of over a month and loaded the Bay with considerable resuspendable sediment that exacerbated turbidity levels for an extended period of time. During the recent drought years, eelgrass has expended towards the Delta and even slightly into Suisun Bay, however some of the largest beds that have historically been more stable, have concurrently suffered some significant declines. In some cases, these declines are likely to be related to desiccation stress within intertidal areas, while in other areas, declines at the shallow margins of these extensive beds may be related to thermal stress from high residence time warm water. There may also be loss from disease, although no direct evidence of expansive bed damage has yet been noted in San Francisco Bay. Additional monitoring will better determine the current trend of eelgrass in the Bay and provide greater insights into both the natural and anthropogenic factors controlling the extent and distribution of eelgrass.

Eelgrass beds are subject to many threats over short and long time scales. In the Bay, eelgrass beds are strongly limited in maximum depth by allowable light penetration associated with turbidity of the water. In the Bay, turbidity of the water is related to both large-scale factors such as sediment supply from tributaries, as well as local effects such as increased turbidity from dredging and shipping activities. In addition, hardened shorelines reflect waves and increase their effects, which can break up eelgrass beds. The most recent decline in eelgrass bed coverage in the Bay raises concerns about the large-scale, long-term stability of eelgrass in the Bay, and the resulting potential loss of functions and services provided by eelgrass beds. In recent years, wasting disease has also become a significant factor affecting the area and distribution patterns of eelgrass within California bays and estuaries. The earliest evidence of wasting disease declines were noted in southern California in about 2006 with disease having now been noted in most California bays and estuaries, including San Francisco Bay. In Morro Bay, the system hardest hit by wasting disease, there has been a 97 percent decrease in eelgrass extent since 2007 likely a result of a combination of factors (e.g., disease, water quality, sedimentation) and efforts are now underway to foster recovery. The full ramifications of disease on eelgrass distribution and trajectory are not yet known, but are a factor of major concern with respect to achieving restoration goal.

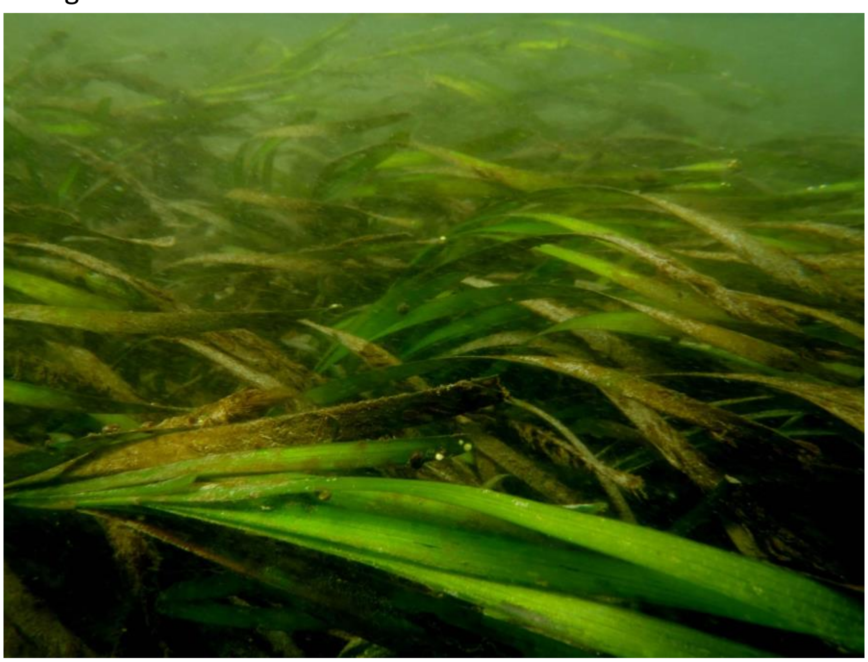
4. Related figures
5.
    a. Eelgrass extent and distribution from 2014 Regional Eelgrass Monitoring Report (Merkel & Associates 2015).



b. Table from Report with numbers - Eelgrass Occurrence by Survey/Monitoring Period and Bay Region

REGION	Benchmark Jun-Oct 2003 (ac.)	Oct 2006 (ac.)	Oct 2007 (ac.)	Jan 2008 (ac.)	Apr 2008 (ac.)	July 2008 (ac.)	Benchmark Oct-Nov 2009 (ac.)	Oct 2010 (ac.)	Oct 2011 (ac.)	Jul 2013 (ac.)	Benchmark Oct-Nov 2014 (ac.)
Pt. Pt. Pinole/Carquinez							136	93	190	68	77
Pt. San Pablo/Pt. Pinole	1,389	1,045	1,620	1,202	1,710	1,693	2,017	1,944	1,740	1,923	1530
Pt San Pablo	282	232	377	246	409	417	401	474	542	418	552
Emeryville / Berkeley	80	96	159	107	130	154	95	149	154	167	92
Oakland Harbor	0						-	1	2	2	0
Crown Beach	251	254	220	100	86	246	219	423	518	188	36
Bayfarm Island	102	110	96	90	104	107	88	93	93	81	70
Tiburon Peninsula	63	72	62	67	77	78	66	85	100	95	91
Richardson Bay	449	417	414	94	379	390	675	487	629	354	335
Other Beds	5	4	5	3	5	5	10	11	12	9	8
TOTAL	**2,623**	**2,231**	**2,955**	**1,910**	**2,901**	**3,089**	**3,707**	**3,760**	**3,982**	**3,306**	**2,790**

c. Photo of eelgrass



State of the Estuary Report 2015

Technical Appendix

HABITAT – Eelgrass

Prepared by Caitlin Sweeney, San Francisco Estuary Partnership; Korie Schaeffer, NOAA Fisheries West Coast Region; Natalie Cosentino-Manning, NOAA Fisheries Restoration Center; Marilyn Latta, State Coastal Conservancy

Eelgrass (*Zostera marina*) Indicator – Technical Appendix

Background and Rationale

<u>Discuss how the indicator relates to the ecological health of the estuary</u>

The indicator for health of the subtidal habitat of the San Francisco Bay is acreage of native eelgrass (*Zostera marina*) beds. In San Francisco Bay, eelgrass is the most extensive type of submerged aquatic vegetation, or underwater flowering plants. Eelgrass performs a wide variety of functions in the Bay. Eelgrass beds provide shelter and food to small fishes of a variety of species, such a pipefish, kelpfish, staghorn sculpin, and many other Bay resident species and fish that pass through the Bay during various periods in their life history. Eelgrass provides food for many species of birds both directly and indirectly. Eelgrass is also used as a preferred substrate for spawning by Pacific herring. Eelgrass beds also dampens wave energy and slow currents in a manner that results in trapping sediment, reducing turbidity, and protecting shoreline area from erosion. Inventories of eelgrass bed coverage in the San Francisco Bay have been undertaken since 2003 under a comprehensive monitoring program, allowing the tracking of eelgrass trends over time.

<u>Include historical information about the indicator and any current programs to evaluate it.</u>

The earliest known studies of eelgrass in the Bay were conducted in the 1920s. Though those studies were not intended to document the area of eelgrass distribution, they do indicate eelgrass beds in at least Marin County at the time, and there is anecdotal evidence that eelgrass may have been present elsewhere in the Bay (Boyer and Wyllie-Echeverria 2010).

The earliest Baywide survey for eelgrass in San Francisco Bay was conducted in 1987 using visual inspection and depth-sounding from small boats. (Wyllie-Echeverria and Rutten 1989). That survey reported 316 acres of eelgrass, located throughout the Bay.

In 2003, a Baywide Eelgrass Inventory and Resource Management Research Program was developed and jointly managed by the California Department of Transportation and NOAA's National Marine Fisheries Service (NMFS). The program has been the most comprehensive effort to inventory eelgrass in the San Francisco Bay over time. The program resulted in comprehensive baywide eelgrass inventories conducted by Merkel & Associates in 2003 and 2009 using sidescan and single beam sonar along with aerial surveys, and annual fixed position belt-transect surveys using sidescan sonar in years 2006, 2007, 2008, 2010, 2011 and 2013. The transects provide spatial and density information on a shoreline segment-by-segment basis that is scaled against the comprehensive mapping results of the "benchmark year" to evaluate changes in bed coverage, areal extent and regional distribution (Merkel 2013).

NMFS continues to support inventories of eelgrass in San Francisco Bay, including a 2014 survey, which was completed using interferometric sidescan sonar, allowing for the integration of bathymetric data collection, concurrent with eelgrass distribution mapping.

The overall trend for eelgrass bed coverage since 2003 has been expansion. By 2011, monitoring reported a Baywide acreage of just under 4,000 acres. However, the latest monitoring data from 2013 and 2014 shows decline of eelgrass bed coverage to 3,300 acres and 2,790 acres, respectively. These recent surveys show a significant departure from the expansion trend. However, eelgrass beds are a dynamic habitat and can experience tremendous variability in coverage from year-to-year and in response to large-scale climatic conditions. Additional monitoring will better determine the current trend of eelgrass in the Bay and facilitate understanding of the variability of eelgrass resources in the bay and the response to various stressors.

Explain why this indicator and this calculation approach were chosen.

This indicator was chosen because of the importance of eelgrass directly as a valuable ecological resource and as an indicator of health for the San Francisco Bay. Further, it was selected because of the long-term inventory and monitoring program which has a proven track record of being robust over time. Data on eelgrass bed coverage in the Bay have been collected using the same methodology and by the same entity since 2003, creating a long-term comparable dataset for eelgrass bed coverage.

Benchmark

Describe the benchmark and why it was chosen.

Discuss any limitations of the benchmark and how it might be improved in the future.

The benchmarks chosen for eelgrass in San Francisco Bay come from the 2010 San Francisco Bay Subtidal Habitat Goals Report (Subtidal Goals Report). The goals for eelgrass in the Subtidal Goals Report are based on a comparison of the 2009 coverage of eelgrass (3,700 acres or about 1% of the Bay), compared to the maximum potential coverage of eelgrass (23,440 acres or about 9% of the Bay). The maximum potential coverage of eelgrass was determined by a spatial-numeric habitat suitability model developed by Merkel & Associates (Merkel 2005). The model is based on bathymetry, current speed, exposure to wind waves, residence time, and the locations of extant eelgrass beds. Habitat characterized by the model as suitable for the establishment of eelgrass beds occurs at depths less than about 2 m in broad swaths along the shores of San Pablo, Central, and South Bays. About half of the maximum potential coverage of eelgrass was classified as moderately suitable to highly suitable. The Subtidal Goals Report developed restoration goals for eelgrass over a 50-year period based on the acreage of nearshore areas of moderate to high habitat suitability as predicted by the model. The benchmark of 8,000 acres within 50 years would increase eelgrass distribution within 50% of identified potential habitat. In addition, a phased adaptive management approach to eelgrass restoration was suggested in an effort to increase knowledge, and thus success, over time. The phased goals are to increase eelgrass coverage by 25 acres within 5 years, 100 acres within 10 years, and 8,000 acres within 50 years. The benchmark of 8,000 acres therefore represents the scoring break between "fair" and "good" for eelgrass health in the Bay. A second benchmark of 4,000 acres was chosen as the scoring break between "poor" and "fair" for eelgrass health. Of the years monitored, only one year (2011) comes close to meeting the "fair" benchmark, which is consistent with what the modeling shows for potential eelgrass habitat and consistent with how restoration efforts over time may be able increase eelgrass acreage in the Bay. It is important to note, however, that eelgrass beds are dynamic and acreages will vary from year to year. Trends for eelgrass are best evaluated over time using not only overall geographic extent, but also the stability of populations and the establishment of new beds.

The benchmark as developed by the Subtidal Goals Report may be refined in the future based on additional information on eelgrass restoration methods (including site selection). The benchmark may also be improved based on refinement of the habitat suitability model with additional data on Bay conditions and responses of eelgrass beds to the environment. Of high benefit to the suitability modelling would be enhancement of shallow bathymetric data within the Bay and potentially the integration of stochastic flood and sediment loading events.

Peer Review

<u>Describe how the indicator was vetted with other experts in the community as per the SOTER Peer Input Guidelines.</u>

The indicator and benchmark rely heavily on the information contained in the 2010 Subtidal Goals Project, a collaboration among the San Francisco Bay Conservation and Development Commission, the California Ocean Protection Council, the California State Coastal Conservancy, the National Oceanic and Atmospheric Administration, and the San Francisco Estuary Partnership. The Subtidal Goals Project underwent significant peer review. Contributors included multiple staff of the participating agencies, as well as additional experts from academia, non-profit organizations, and consulting firms who served on steering committees and provided input and review. The monitoring program for assessing the eelgrass distribution indicator was developed and tested over a three year period between 2006 and 2009. The monitoring program accuracy was verified by evaluating the transect-based estimates of eelgrass occurrence in the Bay against the measured distribution of eelgrass from the 2003 and 2009 benchmark comprehensive eelgrass surveys to determine the difference between calculated and measured eelgrass extent. The error checking process indicates that the monitoring program yields and overall error rate of approximately 1.5 percent for estimating baywide eelgrass extent. The monitoring program development was reviewed as draft and final documents by multiple agency reviewers at National Marine Fisheries Service and the California Department of Transportation during the program development.

In addition, the indicator as developed for the State of the Estuary Report was vetted with the following experts in the community: Marilyn Latta, State Coastal Conservancy; Korie Schaeffer, NOAA National Marine Fisheries Service, Natalie Consentino-Manning, NOAA NMFS Restoration Center; and Keith Merkel, Merkel & Associates. Ms. Latta, Ms. Schaeffer, and Ms. Consentino-Manning were three of the leads on the Subtidal Goals Project.

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State of the Estuary Report 2015

Sidebar Source Material

HABITAT – Oyster Beds

Prepared by Caitlin Sweeney, San Francisco Estuary Partnership; Korie Schaeffer, NOAA Fisheries West Coast Region; Natalie Cosentino-Manning, NOAA Fisheries Restoration Center; Marilyn Latta, State Coastal Conservancy

SotER –Sidebar

Intertidal and Subtidal – Native Oysters (*Ostrea lurida)*

1. <u>Indicator status and trend measurements</u>

	STATUS	TREND	DETAILS
Native Oysters	Poor	Mostly Improving	Monitoring density of oysters per square meter and acreage of native oyster beds and intertidal locations since 2000 has shown a general expansion trend. However, current native oyster bed acreage is significantly less than the estimated maximum potential coverage, based on a habitat suitability model. In addition, there has been a recent study showing significant native oyster populations persisting in the bay, what environmental variables and stressors are most related to their life cycle needs, and which sites have the highest potential for future protection and restoration in the face of climate change stressors such as warming air and water temperatures. Drought years appear to favor oysters, as extended low salinities can cause major mortality but not long-term population loss at sites especially in San Pablo Bay.

2. <u>Brief write-up of scientific interpretation</u>

The Olympia oyster (*Ostrea lurida*) has declined at many estuaries in its native range along the Pacific coast from Baja California to British Columbia. In the past decade, efforts have begun to conserve, enhance or restore Olympia oyster populations in California, Oregon and Washington. Olympia oysters range from Central Baja California, Mexico, to British Columbia, Canada (Polson 2009). Abundance varies enormously from scant, but persistent, populations consisting of a handful of individuals, to locations with nearly 100 percent cover of oysters on hard substrates at MLLW (Wassen et al 2014). In most locations, the size of the pre-European-contact population is unknown. However, there were sufficient populations in SF Bay prior to the Gold Rush to support a commercial fishery (Conte and Dupuy 1982). Based on a review of the former extent of commercial oyster grounds from the earliest available records (mid-1800s to early 1900s), Zu Ermgassen et al. (2012) estimated oyster grounds in Puget Sound, Humboldt Bay, SF Bay, Elkhorn Slough and Mission Bay to be at 1% of historic levels.

Shellfish beds provide several ecosystem functions and support several ecosystem services. The small native Olympia oysters do not commonly form tall, three-dimensional reefs, as do Virginia oysters, although they can add structure to hard substrates and may be able to colonize and overgrow soft substrates. In this sense they can be considered a "foundation species" or ecosystem engineer, altering their environment by increasing bottom roughness, reducing current speeds, and as a result, trapping sediments. Oysters also increase physical heterogeneity, which can increase diversity of other marine invertebrates and also result in higher fish diversity and abundances than in neighboring, less complex habitats. Increased abundance of native oysters can locally increase the number of other benthic invertebrates (Kimbro and Grosholz 2006 for Tomales Bay). With their associated invertebrates, oysters provide food for fish, birds, and crabs (State Coastal Conservancy 2010). In San Francisco Bay, native oysters and native mussels are the only native bivalves that form beds. Inventories of native oyster bed coverage and intertidal shoreline oyster habitat in the San Francisco Bay have been undertaken since 2000 by various entities, and since 2012 under a comprehensive monitoring program managed by the San Francisco Bay National Estuarine Research Reserve, allowing the tracking of native oyster trends over time.

The *San Francisco Bay Subtidal Habitat Goals Report* (Subtidal Goals Report) produced in 2010 contains restoration goals for native oysters in San Francisco Bay. The goals were based on a comparison of the current limited patchy shoreline coverage of native oysters (less than 1% of the Bay) compared to the maximum potential coverage of oysters (about 9% of the Bay), determined by identifying appropriate shoreline locations and extrapolating an area around those location out to a depth of 2 meters. The Subtidal Goals Report determined that the restricted extent of native oyster beds may be limiting their support of valued ecosystem services and, furthermore, that restoration of native oyster beds has been demonstrated and is feasible. The benchmark for native oyster beds is based on the restoration goal in the Subtidal Habitat Goals report of increasing oyster populations in the Bay within 8,000 acres of suitable subtidal/intertidal area over a 50-year time frame using a phased approach under a program of adaptive management. The benchmarks under the phased approach are to increase native oyster coverage by 100 acres within 5 years, 400 acres within 10 years, and up to 8,000 acres within 50 years.

Shells of native oysters occur in the vast shell middens at various sites around the bay along with those of mussels and clams, attesting to the pre-European settlement presence of the native oyster. However, the actual historical abundance of oysters is poorly known, in part because of confusion between native oysters and conspecific (?) Ostrea lurida brought from

Washington or Oregon and planted in the bay. Townsend (1893) referred to native oysters as very abundant and overgrowing the shells of eastern oysters, which had been introduced for aquaculture. Commercial harvest was important "since the days of the Spaniards" (Bonnot 1935), and native oyster reportedly made up about 15% of the total oyster harvest from San Francisco Bay in the late 1800s to early 1900s, producing up to 150 tons of meat per year during 1888-1904 (Barrett 1963).

The 2010 Subtidal Goals report did not include a historic baseline because none was available. The Report provided clear information regarding the restricted extent of oyster beds, which may be limiting their support of valued ecosystem services. Oyster restoration has been demonstrated and is feasible, although questions remain about the anticipated trajectory of restoration and associated response of ecosystem functions and services. The Goals Report concluded that restoration is warranted for oyster beds, but should be done within an experimental framework such as is being implemented via the Coastal Conservancy's San Francisco Bay Living Shorelines Project. Native oyster beds are a dynamic habitat and can experience tremendous variability in coverage from year-to-year. Additional monitoring will better determine the current trend of native oysters in the Bay.

In 2014, an interdisciplinary team led a project from the San Francisco Bay and Elkhorn Slough National Estuarine Research Reserves, in partnership with the State Coastal Conservancy, University of California-Davis, Smithsonian Environmental Research Center, and the California Department of Fish and Wildlife. The project evaluated oyster population densities at 12 sites in San Francisco Bay and also recorded information on oyster attributes, supportive environmental factors, and climate and other stressors summarized in the Site Evaluation Table from these results (Wassen et al. 2014). The sites include China Camp State Park, Loch Lomond Marina, Arambaru Island in Richardson Bay, Brickyard/Strawberry in Marin, Sausalito shoreline, Pt Orient in Richmond, Berkeley Marina, Oyster Pt, Coyote Pt; Pt Pinole Regional Shoreline, Eden Landing Ecological Reserve, and the San Rafael Shoreline. Oyster densities ranged from three to 961 oysters per meter squared at the 12 sites, and total population estimate of native oysters at MLLW at all of these sites combined is roughly 160,000 oysters. Of the sites evaluated, the top-scoring sites (based solely on oyster biology) for restoration in San Francisco Bay were Berkeley Marina (Shorebird Park area), Strawberry (Brickyard Cove), San Rafael Shoreline and Point Pinole Regional Shoreline. All of the high-ranked restoration sites also ranked high as conservation sites, but several additional sites ranked high for conservation: Richmond (Point Orient), Loch Lomond Marina, Sausalito (Dunphy Park) and Coyote Point Recreation Area.

Native oyster beds are subject to many threats over short and long time scales. The principal threats to native oysters seem to be high rates of sedimentation and high air temperatures that cause desiccation stress at low tide (?). Competition for space may be more important in the South Bay, where hard substrate is limited and in the subtidal zone, where fouling organisms such as non-native (?) sponges, tunicates, and hydroids are abundant. Intertidal substrate examined during surveys was on average ~40% bare, indicating that lack of attachment space may not limit abundance of intertidal oysters (State Coastal Conservancy 2010). Other potential limiting factors include contaminant effects, especially for intertidal beds that are vulnerable to oil spills, and predation by fish, birds (e.g., diving ducks), and possibly crabs. Small predatory snails (oyster drills) present a moderate to high source of mortality to young oysters in the South Bay and Richardson Bay, and a zero to low source of mortality. Diseases and parasites do not present a major threat, although this could change if population density increases and changes in water temperatures occur due to climate change. Heat stress in warm intertidal areas may reduce oyster survival in local areas. The full ramifications of these stressors on native oyster distribution and population trajectory are not yet known, but are factors of major concern with respect to achieving restoration goals.

a. Photo of native oyster bed (placeholder)

b. 

Native Oyster (Ostrea lurida) Sidebar – Technical Appendix

Note that this topic was very nearly an Indicator rather than a Sidebar in the 2015 SOTER, and so this additional research and information is now made available here.

Background and Rationale

Discuss how the indicator relates to the ecological health of the estuary

The Olympia oyster (*Ostrea lurida*) has declined at many estuaries in its native range along the Pacific coast from Baja California to British Columbia. In the past decade, efforts have begun to conserve, enhance or restore Olympia oyster populations in California, Oregon and Washington. Oysters increase physical heterogeneity, which can increase diversity of other marine invertebrates and also result in higher fish diversity and abundances than in neighboring, less complex habitats. Increased abundance of native oysters can locally increase the number of other benthic invertebrates (Kimbro and Grosholz 2006 for Tomales Bay). With their associated invertebrates, oysters provide food for fish, birds, and crabs (State Coastal Conservancy 2010). Oyster restoration has been demonstrated and is feasible, although questions remain about the anticipated trajectory of restoration and associated response of ecosystem functions and services. The San Francisco Bay Subtidal Habitat Goals Report (State Coastal Conservancy 2010).concluded that restoration is warranted for oyster beds, but should be done within an experimental framework such as is being implemented via the Coastal Conservancy's San Francisco Bay Living Shorelines Project. Native oyster beds are a dynamic habitat and can experience tremendous variability in coverage from year-to-year. Additional monitoring will better determine the current trend of native oysters in the Bay.

Include historical information about the indicator and any current programs to evaluate it.

Several surveys and studies of San Francisco Bay or portions of the Bay have been made over the last 10 years (Figure 2). These are briefly described below, and more information is available in Appendix 7-1 (Native Oyster Opportunities and Constraints Report) of the San Francisco Bay Subtidal Habitat Goals Report (State Coastal Conservancy 2010).

Oyster Distribution: UC Davis Study: From July 2006 to June 2007, Grosholz et al. (2007) surveyed most of the accessible rocky shoreline of San Francisco Bay for the presence of native oysters. A variety of information sources, including unpublished reports and anecdotal observations of oyster presence were used to guide survey site selection. In addition, various shoreline maps to determine substrate types and accessibility of potential sites were consulted. Sites were generally visited once, by 1-3 researchers, with appropriate substrate searched for at least half an hour. GPS points were recorded and qualitative notes on each site were taken. In addition, at a subset of sites, density was measured by counting oysters in 5-10 randomly placed 0.25 m^2 quadrats.

Oyster Distribution: Harris Thesis: In 2002-2003, Harris carried out a series of intertidal surveys for oysters. She reported oysters present from a number of sites in San Pablo Bay, including

Pinole Bayfront Park and China Camp in relatively high numbers, where no live oysters were found in 2006. The southern limit of the oyster population found by Harris was not different from that of the UC-Davis study. The sites for which she provided complete GPS data are included in Figure 2.

Population Density of Native Oysters: Polson Study In 2005-2006, Maria Polson surveyed for *O. lurida* in the intertidal zone along its entire known distribution. At each location, she carried out a 2 hour timed survey to locate the highest density patches. These densities were then measured by counting oysters in 10 .25-m2 quadrats. Polson had one site in San Francisco Bay, Point San Quentin. With a mean density of 36.7 ± 11.6 per quadrat, this site had some of the highest densities of any location along the West Coast. The next most dense sites were Mission Bay, CA (mean =22.8 ± 3.4) and Bahia de San Quintin, Baja, Mexico (mean = 20.7 ± 6.5). Links to Polson's work can be found in the 2006 West Coast Native Oyster Workshop Proceedings (http://www.nmfs.noaa.gov/habitat/restoration/publications/tech_glines.html).

Population Density of Native Oysters: UC Davis Study: In addition to the above-mentioned survey, a series of population measurements at 8-13 study sites was carried out by UC-Davis from 2006-2008. Sites were chosen to represent four broad geographical areas in San Francisco Bay: 1) North Bay; 2) Central Bay West; 3) Central Bay East; 4) South Bay. Sites were selected after the initial survey and represented areas with the most abundant oyster populations within each of region. In the North Bay, there were no sites with many live oysters. Here, and at one site in South Bay, sites with high numbers of recently dead oysters and/or oyster scars, which indicated that the site had at one time been good habitat, were selected. The oyster densities at the 13 sites in fall 2006 are represented graphically (Figure 4).

Oyster Recruitment: Save The Bay/ San Francisco State University Study: In 2001-2002, Save The Bay (Marilyn Latta, staff, volunteers) and San Francisco State University researchers (Professor Micheal McGowan, Aimee Good, Tripp McLandish, others) partnered with five community-based restoration and education organizations to survey intertidal oysters at five sites in San Francisco Bay. Simple shell strings made with Pacific oysters (*Crassostrea gigas*) were hung from docks and piers in 1-3 feet of water and monitored bi-monthly for presence/absence of native oyster settlement and other invertebrate species settlement. Water quality data was collected, including temperature, pH, dissolved oxygen, and salinity. The five sites included the north end of Richardson Bay in Tiburon (with Richardson Bay Audubon's Bayshore Studies Volunteers), the mouth of San Pablo Creek in Richmond (with The Watershed Project's San Pablo Area Watershed Awareness and Education and Restoration program); the mouth of Sausal Creek in Oakland (with the Friends of Sausal Creek), the mouth of Redwood Creek in Redwood City (with the Marine Science Institute), and at the Coyote Point Marina in San Mateo (with the Coyote Point Museum docent group). Native oysters were found at all sites except the mouth of San Pablo Creek. The project raised public awareness about native oyster presence in the bay, and generated media interest in the topic of native oyster restoration.

Oyster Recruitment: Save The Bay/ San Jose State University Study: In 2006-2007, recruitment data were generated by Save The Bay (Marilyn Latta, staff, volunteers) and San Jose State graduate student Sumudu Welaratna (Figure 9b). Four types of substrate were deployed, including 1) shell strings, 2) shell bags, 3) pvc plates attached to brick, and 4) collectors made from native oyster shell material mixed with Portland cement. Six sites were monitored bi-monthly, including the San Rafael Canal, Berkeley Marina, Oyster Point Marina, Palo Alto Baylands, the mouth of Permanente Creek, and the Ravenswood Pier. The south bay sites and the shell bag method had the highest recruitment densities.

Oyster Recruitment: UC Davis 2006-08 Study: From 2006-2008, UC Davis deployed standardized recruitment collectors at between 7 and 13 sites; in 2007 and 2008 members of the San Francisco Bay Native Oyster Working Group coordinated efforts with UC Davis to use similar methodology so that recruitment could be compared across more locations. These data are summarized below. Recruitment collectors (sets of 10 PVC tiles) were first deployed by UC Davis in July 2006 at the zero tide mark at 13 study sites. Seven oysters recruited to two sites, Berkeley (Shorebird Park) and Alameda (Encinal Boat Launch) (Figure 10). New recruits were seen in the late fall and winter at the UCD field sites after recruitment collectors had been removed. The lack of recruits was not surprising given the massive die-off of oysters earlier in the year. Recruitment was significantly higher in 2007, when recruitment collectors were placed at eight sites beginning in July. Recruitment collectors were checked every 2 months and left out continuously. Recruits first appeared in September in 2006 and August in 2007 and 2008.

Recruitment: San Francisco Bay Shared Oyster Protocol: In both 2007 and 2008 the San Francisco Bay Native Oyster Working Group (SFNOWG) conducted a recruitment study with a shared protocol. In 2007 SFNOWG compared the recruitment efficacy of shell, PVC plates and oyster "seameant." Shell consisted of 5 quart bags of shell, with an average shell surface area of 100 cm2. PVC plates were 100 cm2 sanded gray PVC. Oyster "seameant" was made by twice dipping 100 cm2 burlap in a cement mixture of 1 part Portland cement, 1 part ground native oyster shell (Jericho brand Pearl Powder), and 2 parts water. There was no significant difference in the number of recruits between the substrate types (Kruskal Wallis non-parametric test, $p=0.277$). There were differences between sites, but these might have been due to the fact that this study was conducted in a variety of ways including fixed versus floating substrate and varying depth of substrate.

Restoration Design and Oyster Recruitment: San Francisco Bay Living Shorelines Project: The State Coastal Conservancy, UC Davis, and others constructed a one acre pilot project in 2012. Monitoring includes tracking oyster recruitment, growth, survival, size, fecundity, and density. More than two million native oysters have recruited to the site, along with a wide diversity of invertebrates, fish and wildlife. Annual monitoring reports are available at www.sfbaylivingshorelines.org.

***Adult Oyster Density and Size: National Estuarine Research Reserve (NERR) Guidelines to Native Oyster Restoration and Conservation:* The San Francisco Bay NERR, UC Davis Bodega Marine Lab, and Smithsonian Environmental Research Center staff monitored oyster density and size distribution at each site on a quarterly timescale. They established a permanent 30 m transect within the densest oyster area and as close to 0 m MLLW as possible. At ten random points along this transect, they counted total number of oysters within a ¼ m^2 quadrat to determine density and measured up to 10 oysters to calculate size distribution. Size distribution data were used to calculate both the size-class diversity index and the mean upper quartile of oyster size. Density data were used in calculations for population estimates on suitable substrate over a 1 m by 300 m area at each site. This project provides the baseline data for the oyster indicator, and will conduct the ongoing annual monitoring that will be used to track future trends.

Explain why this indicator and this calculation approachsidebar topic were was chosen.

This indicator topic was chosen because of the importance of native oysters directly as a valuable ecological resource and as an indicator of health for the San Francisco Bay. Further, it was selected because of the monitoring program initiated by the San Francisco Bay National Estuarine Research Reserve that is already in place and has a proven track record of being robust over time. Data on native oyster bed and intertidal shoreline coverage in the Bay have been collected using the same methodology and by the same entity since 2012, creating the beginnings of a long-term comparable dataset for native oyster bed and intertidal shoreline habitat coverage.

Benchmark

Describe the benchmark and why it was chosen.
Discuss any limitations of the benchmark and how it might be improved in the future.

The benchmark chosen for native oysters in San Francisco Bay comes from the 2010 San Francisco Bay Subtidal Habitat Goals Report (Subtidal Goals Report). The goals for native oysters in the Subtidal Goals Report are based on the acreage of shoreline areas out to a depth of two meters where native oysters have been documented, and correlate with recent monitoring data regarding distribution. Native oysters would not be restored throughout these target areas, but at a subset of locations within these larger areas (Zabin et al Appendix 7-1 Subtidal Goals Report). The Subtidal Goals Report developed restoration goals for eelgrass over a 50-year period based on the acreage of nearshore areas of moderate to high habitat suitability. In addition, a phased adaptive management approach to native oyster restoration was suggested in an effort to increase knowledge, and thus success, over time. The phased goals are to increase oyster coverage by 100 acres within 5 years, 400 acres within 10 years, and 8,000 acres within 50 years.

The benchmark as developed by the Subtidal Goals Report may be refined in the future based on additional information on native oyster restoration methods (including site selection). The benchmark may also be improved based on refinement of the recommended restoration areas with additional data on Bay conditions and responses of native oyster beds to the environment. Of high benefit to the suitability modelling would be enhancement of shallow bathymetric data within the Bay and potentially the integration of stochastic flood and sediment loading events.

Peer Review

Describe how the indicator was vetted with other experts in the community as per the SOTER Peer Input Guidelines.

The indicator and benchmark rely heavily on the information contained in the 2010 Subtidal Goals Project, a collaboration among the San Francisco Bay Conservation and Development Commission, the California Ocean Protection Council, the California State Coastal Conservancy, the National Oceanic and Atmospheric Administration, and the San Francisco Estuary Partnership. The Subtidal Goals Project underwent significant peer review. Contributors included multiple staff of the participating agencies, as well as additional experts from resource agencies, academia, non-profit organizations, and consulting firms who served on steering committees and provided input and review. The monitoring program for assessing the native oyster distribution indicator was developed and tested over a three year period between 2012 and 2015. The monitoring program accuracy was verified by evaluating the transect-based estimates of oyster occurrence in the Bay. The monitoring program was developed by qualified staff of the San Francisco Bay and Elkhorn Slough National Estuarine Research Reserves and UC Davis Bodega Marine Laboratory, and development was reviewed as draft and final documents by multiple agency reviewers at National Marine Fisheries Service and the California Department of Fish and Wildlife n during the program development.

In addition, the indicator as developed for the State of the Estuary Report was vetted with the following experts in the community: Marilyn Latta, State Coastal Conservancy; Korie Schaeffer, NOAA National Marine Fisheries Service, Natalie Consentino-Manning, NOAA NMFS Restoration Center, Chela Zabin, UC Davis, and Matt Ferner, San Francisco Bay National Estuarine Research Reserve. Ms. Latta, Ms. Schaeffer, and Ms. Consentino-Manning were three of the leads on the Subtidal Goals Project.

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State of the Estuary Report 2015

Summary

HABITAT – Tidal Marsh

Prepared by Sam Safran
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State of the Estuary Report 2015
Tidal marsh habitat indicators- Content Deliverables, 6/23/2015
Sam Safran, San Francisco Estuary Institute

1. Brief description of indicator and benchmark

Attribute	Indicator	Benchmarks
Tidal marsh	Regional extent	The benchmark for tidal marsh regional extent in the Bay is 100,000 acres (a goal established by the 1999 *Baylands Ecosystem Habitat Goals Report* and approximately half the acreage circa 1800). A benchmark for tidal marsh regional extent in the Delta has not yet been determined. For context, we instead compare the current regional extent of Delta tidal marsh against three reference values: (1) half the tidal marsh acreage circa 1800 (~180,000 acres), (2) the current area of tidal marsh plus the area of diked land at intertidal elevations (~78,000 acres), and (3) the current area of tidal marsh plus the maximum acreage of tidal marsh restoration called for in the state's near-term habitat restoration initiative *California Eco Restore* (~17,000 acres).
Tidal marsh	Patch sizes	The benchmark is the historical (circa 1800) size distribution of tidal marsh patches, as measured by the percentage of tidal marsh area belonging to a patch >200 ha in size.

2. Indicator status and trend measurements

Indicator	Status	Trend	Details
Tidal marsh- Regional extent	Fair (Bay) to poor (Delta)	Improving	The historical decline of the Estuary's tidal marshes has ended and gradual restoration is underway, but there is still a long way to go. In the Bay, the extent of tidal marsh acreage is approximately halfway to the regional goal of 100,000 acres. In the Delta, where restoration efforts currently trail those underway in the Bay, the regional extent of tidal marsh is only a fraction of the historical acreage and clear regional goals are still needed (regional planning efforts are currently underway). There is now substantially less tidal marsh in the Delta than in the Bay (a reverse of the historical distribution).
Tidal marsh- Patch sizes	Good (Bay) to poor (Delta)	Unknown	In the Bay, the proportion of tidal marsh area belonging to patches large enough to support certain key ecological functions is very close to historical levels. In the Delta, however, this proportion has been reduced by more than two-thirds. More data are needed to determine recent trends.

3. Brief write-up of scientific interpretation

Tidal marsh- Regional extent

- *Provide 2-3 sentences to answer the question: What is this indicator?*

The regional extent of tidal marsh measures the combined area of all tidal marshes in the estuary and is derived from detailed maps of the estuary's wetlands. We report the regional extent of marsh in the Bay and the Delta separately.

- *Provide 2−3 sentences to answer the question: Why is it important?*

The regional extent of tidal marshes matters because many of the ecological and hydrological benefits they provide increase along with marsh extent. Put simply, as the total area of tidal marsh in the Estuary increases, so does the abundance and diversity of the plants and animals that utilize marshes, as well as the ecosystem services marshes provide for flood control, water quality, and recreation. Increasing the regional extent of marsh across the whole Estuary—from the South Bay to the North Delta—will ensure that marsh habitat exists along the full length of important ecological gradients (such as tidal influence, salinity, and vegetation), providing a range of options for marsh species. Tidal marshes in the Bay (which are salty or brackish) are not the same as tidal marshes in the Delta (which are fresh)—they have different physical characteristics, support different assemblages of plants and animals, and are subject to different stressors. Restoration in both regions is critical to provide the full suite of ecological functions provided by tidal marshes in the Estuary.

- *Provide 2−3 sentences to answer the questions: What is the benchmark? How was it selected?*

We utilize separate methods to evaluate the regional extent of tidal marsh in the Bay and in the Delta. For the Bay, we use a benchmark of 100,000 acres, a long-term tidal marsh acreage goal put forth by the 1999 *Baylands Ecosystem Habitat Goals Report*. This goal was the culmination of science-based public process that sought to evaluate the habitat needs of representative species and to identify changes needed to improve the Bay's ecological functioning and biodiversity. It is approximately half of the tidal marsh area that existed in the Bay at the beginning of the 19th century. A scoring break between Fair and Poor was arbitrarily (?) set at 50,000 acres, or half of the benchmark.

Since no similar quantitative goals exist for tidal marsh regional extent in the Delta, we instead provide three different reference values for context for a benchmark and goal that is yet to be set. (1) 180,000 acres or approximately half of the tidal marsh area that existed in the Delta at the beginning of the 19th century. This value is comparable to the one used to assess the regional extent of tidal marsh in the Bay. (2) 78,000 acres or the current area of tidal marsh plus the approximate area of diked lands in the Delta that are at intertidal elevations. This is the current area that would fall between high and low tide in the absence of levees and other water control structures and therefore exists at the right elevation for tidal marsh formation in the Delta. This acreage does not account for what percentage of the area will actually be available for restoration given other priority land uses. (3) 17,000 acres or the maximum area of tidal marsh that would exist in the Delta if the near-term habitat restoration goals laid out in the current version of *California Eco Restore* (the State's 5-year initiative for coordinating habitat restoration in the Delta) are met.

Provide 2−3 sentences to answer the question: What is the status and trend for this indicator?

The regional extent of tidal marsh in the Bay is characterized as "fair." In 2009 (the last year with standardized data), there were approximately 45,000 acres of tidal marsh in the Bay, which is 45% of the 100,000 acre goal. Since 2009, an additional 6,300 acres of land in the Bay have been opened to the tides. Much of this restored habitat is expected to transition into to tidal marsh in the future and, if counted in full, would bring the regional extent of tidal marsh to 51% of the 100,000 goal.

The regional extent of tidal marsh in the Delta is characterized as "poor." In 2002 (the last year with standardized data), there were approximately 8,000 acres of tidal marsh in the Delta. This area is only 4% of the 180,000 acre reference value (half the tidal marsh area circa 1800), 10% of the 78,000 acre reference value (the current area of tidal marsh plus the approximate area of diked lands in the Delta that are at intertidal elevations), and 47% of the 17,000 acre reference value (the maximum area of tidal marsh that would exist in the Delta if the near-term habitat restoration goals laid out in the current version of *California Eco Restore* are met). Although 260 acres of tidal wetlands have been restored since 2002, this relatively small area increases the percentages noted above by less than 2 percentage points.

- *Provide 4−6 sentences to answer the questions: What does it mean? Why do we care?*

In the Bay, the area of tidal marsh continues to increase towards the regional goal of 100,000 acres. A major milestone was passed in January 2015, when the levees of Cullinan Ranch were breached and the area of existing tidal marshes plus restored intertidal wetlands (much of which are expected to eventually develop into tidal marsh) moved past the goal's halfway mark of 50,000 acres. Looking forward, an additional approximately 24,000 acres of tidal marsh habitat in the Bay are currently planned as part of restoration, enhancement, and mitigation projects that have already been funded and/or permitted and therefore have a high probability of completion within the next 20-30 years.

Tidal marsh restoration efforts in the Delta trail those underway the Bay, as evidenced by the disparity in acres restored since the last standardized datasets indicating the extent of tidal marsh were produced. Part of this disparity can be explained by the extensive "subsidence" (sinking) of the Delta's peat islands—while these extensive areas once supported tidal marsh, many now sit 10-25 ft. below sea-level at an elevation that is much too low for tidal marsh vegetation establishment. (Subsidence is generally not as extreme in the Bay, although there are some diked areas in both North and South Bay where surface elevations would need to be increased to restore tidal marsh habitat). Because of the magnitude of subsidence in the Delta, lands that are at proper elevations for tidal marsh restoration are generally limited to the Delta periphery. Despite this, analyses of the landscape suggest that there are approximately 70,000 acres in the Delta of diked lands at the proper elevation for tidal marsh vegetation establishment. Looking forward, restoration and mitigation projects expected to break ground within the next two years would, if successful, add approximately 4,650 acres of tidal marsh to the current total. Clear regional habitat goals are still needed for the Delta in order to evaluate restoration progress. Planning efforts facilitated by the Delta Conservancy are currently underway.

Scientists are uncertain about how the Estuary's tidal marshes will fare in the future as sea-level rises ever more quickly. Although the Bay-Delta's tidal marshes have generally kept pace with sea-level rise over the las several thousand years, the rate of sea level rise and available sediment supply will have a major influence on whether they can continue to do so through the end of the century. Modeled scenarios of high sea-level rise rates and low sediment supply, which the latest evidence suggests is a likely trajectory, project that Bay tidal marshes will be unable to keep pace with rising tides and that their total regional extent will decrease; under scenarios of relatively low sea-level rise rates and high sediment supply, the total regional extent is projected to increase. Although similar projections have not been developed for the Delta, its tidal freshwater marshes (which have higher rates of organic matter production) are expected to be less sensitive to reduced sediment availability than the Bay's tidal salt marshes. Projections that assume marsh accretion can keep pace with estimated rates of sea-level rise in the Delta show an increase in the regional extent of tidal marsh over the next 50 years (assuming no major levee failures).

<u>Tidal marsh- Patch sizes</u>

- *Provide 2-3 sentences to answer the question: What is this indicator?*

Unlike the regional extent indicator, which assesses the total area of tidal marsh habitat, the tidal marsh patch sizes indicator assesses the size of individual patches of tidal marsh habitat in the Bay and Delta. Specifically, it measures the *distribution* of tidal marsh habitat into patches of different sizes by measuring the percentage of tidal marsh habitat belonging to patches larger than a particular size threshold. For the sake of this analysis we measure the proportion of total tidal marsh area belonging to patches >200 ha in size, a value that seems to be important for supporting the maximum possible densities of certain tidal marsh birds in the Estuary.

- *Provide 2−3 sentences to answer the question: Why is it important?*

The size of tidal marsh patches matters because when larger marshes are fragmented into smaller ones, their value as wildlife habitat tends to decrease. Larger marshes are more likely than smaller marshes to support a mosaic of marsh features (e.g., high marsh, low marsh, marsh pans), buffer native wildlife from nonnative predators, and have well developed tidal channel networks.

- *Provide 2−3 sentences to answer the questions: What is the benchmark? How was it selected?*

We developed a benchmark for both tidal marsh and tidal flat size by assuming that the historical distribution tidal marsh habitats is an appropriate measure for healthy tidal marsh habitats in the Estuary today. Considering this, the benchmark is the historical (circa 1800) size distribution of tidal marsh patches, as measured by the proportion of tidal marsh area belonging to a patch >200 ha in size. The benchmark is met if the current proportion is at least 80% of the historical proportion (measured separately for the Bay and Delta). A scoring break between Fair and Poor was arbitrarily (?) set at 40% of the historical proportion, or half of the benchmark.

- *Provide 2−3 sentences to answer the question: What is the status and trend for this indicator?*

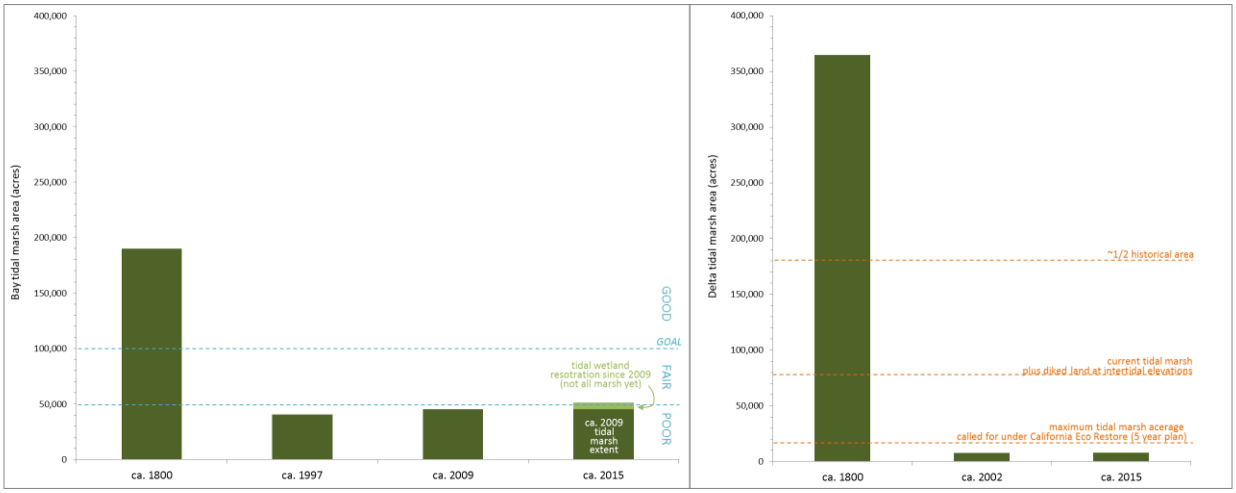
In general, the proportion of tidal marsh area belonging to patches smaller than 200 ha has increased, and the proportion belonging to patches greater than 200 ha has decreased, but this trend is much more pronounced in the Delta than in the Bay. In the Bay, the current proportion of total tidal marsh area belonging to patches greater than 200 ha in size is 88% of the historical proportion (considered "good"). In the Delta, the current proportion is only 30% of the historical proportion (considered "poor").

- *Provide 4−6 sentences to answer the questions: What does it mean? Why do we care?*

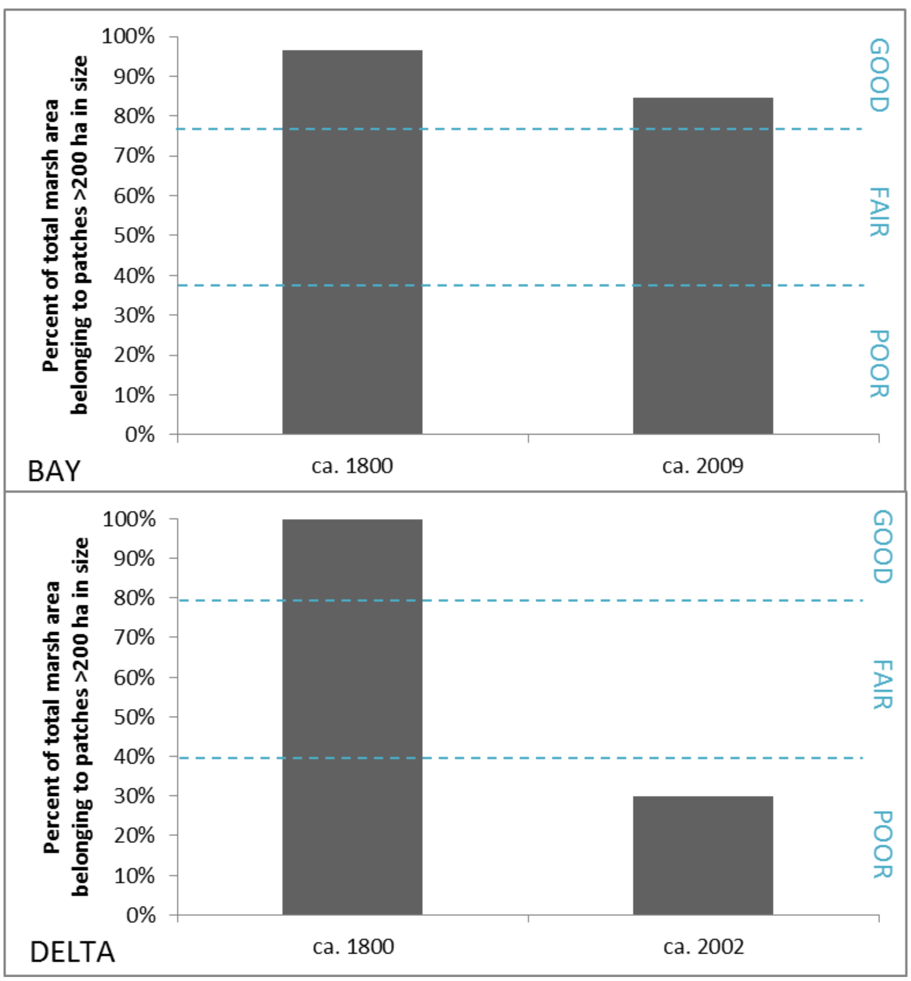
The decrease in the proportion of tidal marsh area belonging to patches greater than 200 ha is expected to have impacted resident tidal marsh birds like the endangered Ridgway's Rail, which only achieves its maximum population density in patches > 200 ha. Other species and ecological functions are likely impacted by the historical trend of fragmentation suggested by this indicator. Fragmented wetlands support smaller wildlife populations because of an increase in the relative proportion of "edge" habitat, with reduced population viability and a greater chance of local extinction within habitat fragments. The fact that the proportion of patches in the Bay larger than 200 ha is almost 90% of the historical proportion is reassuring, reflects the increasing size of individual tidal marsh restoration projects in the Bay over time, and highlights the need to restore and connect larger tidal marsh patches in the Delta.

4. Related figures

<u>Tidal marsh- Regional extent</u>



<u>Tidal marsh- Patch sizes</u>



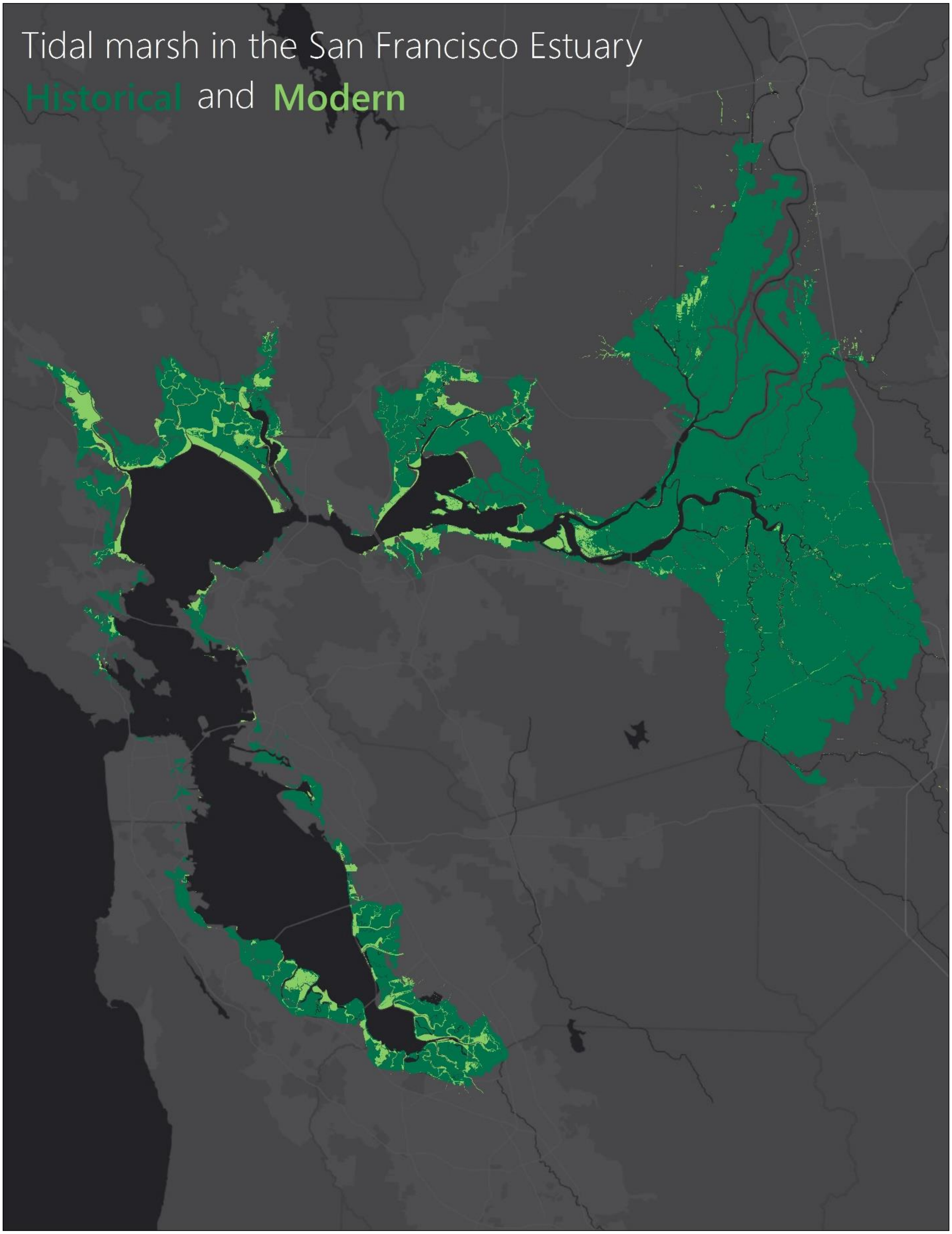
4. Related tables

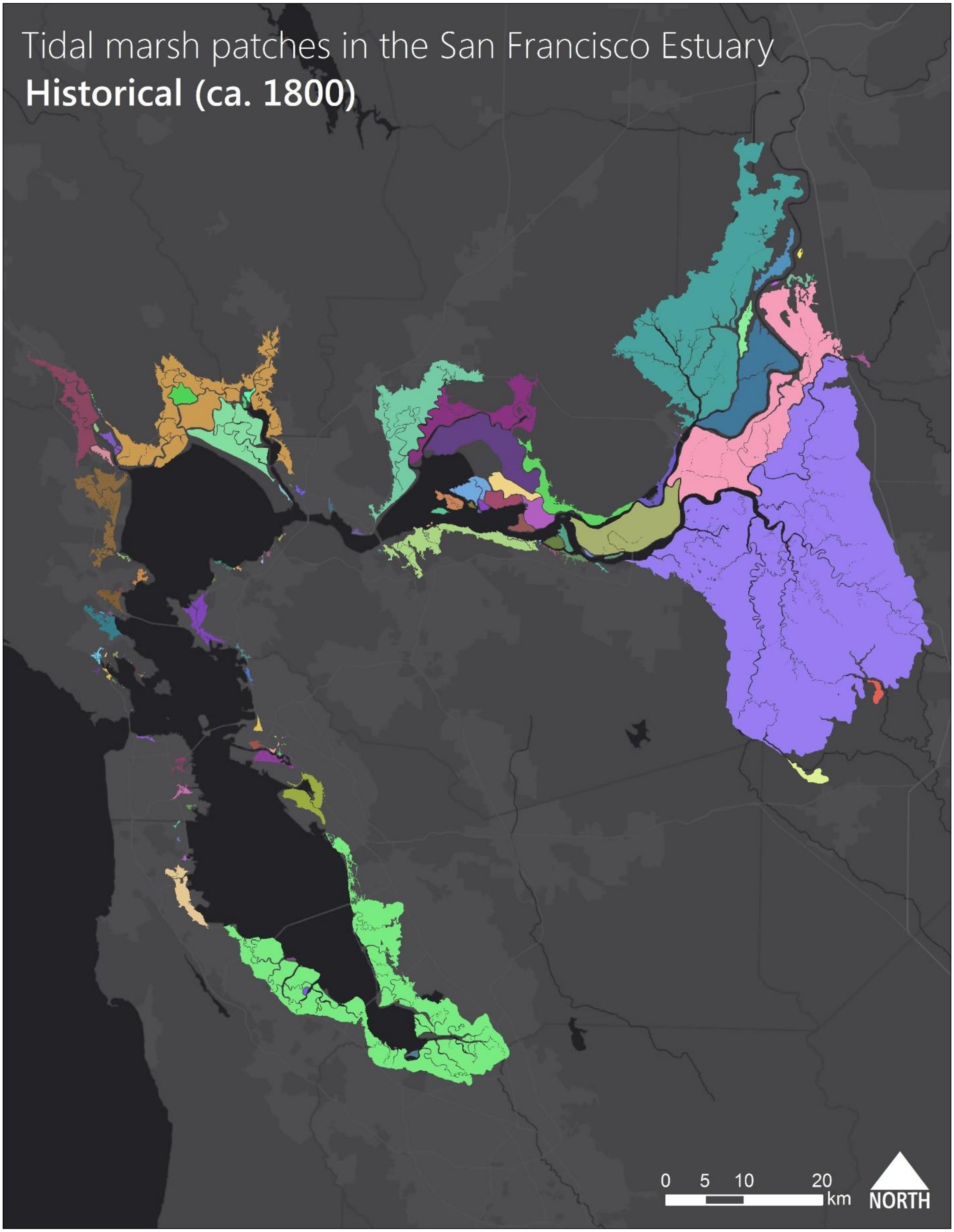
Table 1. Recent tidal wetland restoration. The areas listed below have been opened to tidal action since the datasets utilized in this study were developed (ca. 2009 for the Bay; ca. 2002 for the Delta). Although much of this restored tidal habitat is expected to transition into tidal marsh over time, these sites are not yet included in in the maps and charts summarizing the regional extent of tidal marsh.

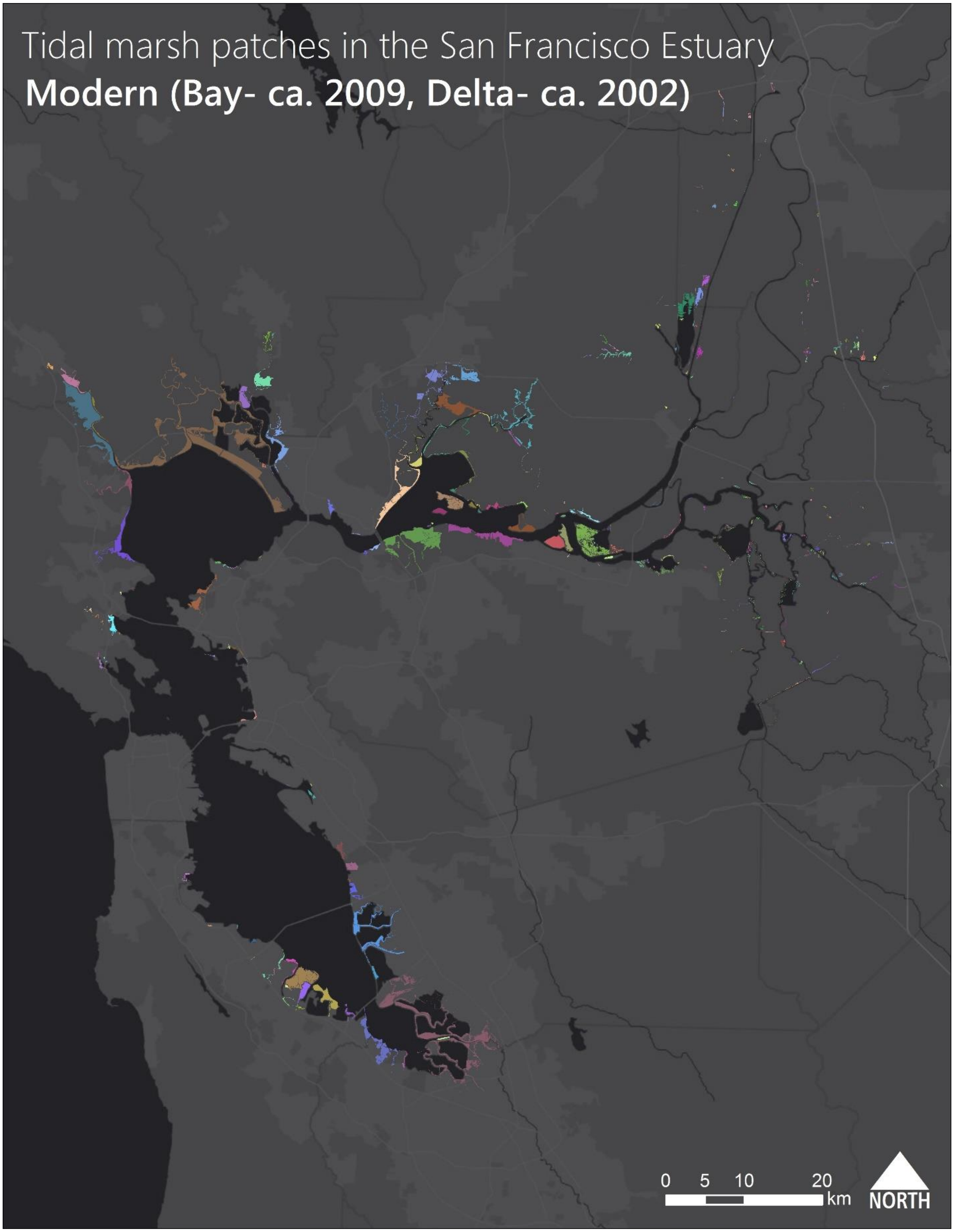
Site	Year opened to tidal action	Planned area of tidal wetland restoration (acres)
Bay (tidal wetland restoration since 2009)		
Napa Plant Site: Central Unit	2009	175
Alviso: Pond A6	2010	330
Napa Plant Site: South Unit	2010	1,080
Eden Landing: Ponds E8A/E9/E8X	2011	630
Alviso: Ponds A8/A7/A5	2012	1,400
Alviso: Pond A17	2012	130
Bair Island: Middle Bair	2012	646
Hamilton Marsh	2014	380
Bruener Marsh	2014	26
Cullinan Ranch	2015	1,549
Total (Bay)		6,346

Delta (tidal wetland restoration since 2002)		
Twitchell Island Setback Levee	2005	1
Sherman Island Setback Levee	2005	7
Liberty Island Conservation Bank and Preserve	2010	31
Cosumnes Floodplain Mitigation Bank	2011	73
Calhoun Cut	2014	147
Total (Delta)		259

5. Optional maps




Tidal marsh patches in the San Francisco Estuary
Historical (ca. 1800)
0 5 10 20
km
NORTH


Tidal marsh patches in the San Francisco Estuary
Modern (Bay- ca. 2009, Delta- ca. 2002)
0 5 10 20
km
NORTH

State of the Estuary Report 2015

Technical Appendix

HABITAT – Tidal Marsh

Prepared by Sam Safran
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Tidal marsh habitat indicators

Background and Rationale

Tidal marshes—including those found in the San Francisco Bay-Delta Estuary (the "Estuary")—provide a wide array of ecosystem services. They provide habitat and support food webs for wildlife, stabilize shorelines and protect them from storm damage, store floodwaters and maintain water quality, preserve biodiversity, store carbon, and offer profound opportunity for scientific study, education, recreation, and aesthetic appreciation (Costanza et al. 1997, Peterson et al. 2008, Palaima 2012, Zedler 2012).

Although tidal marshes have a wide array of functions, this study focuses on indicators that evaluate the Estuary's tidal marshes for their function as habitat for native wildlife. Specifically, the indicators selected here—the regional extent and patch sizes of tidal marsh—seek to help broadly assess the status of tidal marshes in the Estuary for their ability to support the life histories of native tidal marsh wildlife (defined as obligate or transitory plants or animals). It is worth mentioning, however, that although the focus here is on tidal marshes as habitat for native wildlife, the nature of the indicators (the regional extent of tidal marsh is perhaps the most fundamental measurement of tidal marsh habitat) means they likely integrate across the other services provided by the Estuary's tidal marshes. The focus on wildlife support is merited since much, if not most, of the interest and concern about tidal marshes relates to their function as habitat for native fishes, animals, and plants (e.g. BCDC 2008, SFBRWQCB 2010, SFEP 2011, USFWS 2013, SFEI-ASC 2014). Tidal marshes are especially valued for their contribution to the native biological diversity of the San Francisco Estuary. Many of the region's rare and endangered plants and animals rely on tidal wetlands for their survival, and legal mandates to protect these species provide the regulatory framework and funding behind a significant portion of tidal marsh restoration activities.

The San Francisco Bay and the Sacramento-San Joaquin Delta are often studied and managed as distinct entities. However, the Bay and Delta function as a unified and complex estuary, which crosses several ecologically significant physical gradients (e.g., in tidal influence, salinity, wave energy, suspended sediment). These physical gradients, in turn are manifested in gradients within the Estuary's tidal marsh ecosystems (e.g., in vegetation composition, physical structure, soils types, channel density). When planning for habitat restoration in the Estuary, these gradients are important to consider if we wish to support the full range of ecological functions provided by the estuary's tidal marshes. This analysis seeks to evaluate and inform restoration efforts by considering the Bay and the Delta's tidal marshes side by side in a single document. This said, we do report the status of the tidal marsh habitat indicators separately for the Bay and the Delta (a structure that is reflected throughout this State of the Estuary report). This distinction is driven by a few different considerations, including the following: freshwater and salt marshes are not equivalent (Odum 1988) and the state of the science surrounding each differs greatly within the Estuary; the Bay and Delta have different environmental histories and differences in current environmental stressors; the political realities, regulating authorities, regional goals, and history of restoration are different in the Bay and the Delta; and available data on tidal marsh extent are generally limited to one region or the other. Although the tidal marsh indicators are reported separately for each region, substantial effort was made to integrate the datasets before splitting them, ensuring a "seamless" divide in the analyses of each region.

The **tidal marsh regional extent indicator** measures the combined area of all tidal marshes in the estuary and is derived from detailed maps of the estuary's wetlands.

The importance of tidal marsh extent as an indicator is based on the notion that greatest threat to tidal marsh ecosystems and the species they support is habitat loss (USFWS 2013). Measuring the areal extent of an ecosystem is a simple way to assess its quantitative loss and a critical component of ecosystem conservation (which, in turn, is a complement to species-level conservation; Noss et al. 1995). The regional extent of tidal marsh matters because many of the ecological and hydrological benefits the habitat provides increase along with marsh extent. Put simply, as the total area of tidal marsh in the Estuary increases, so does the abundance and diversity of the plants and animals that utilize marshes. Increasing the regional extent of marsh across the whole Estuary—from the South Bay to the North Delta—will ensure that marsh habitat exists along the full length of important ecological gradients (such as tidal influence, salinity, and vegetation) and provide a range of options for the species that utilize tidal marshes.

The **tidal marsh patch sizes indicator** measures the percentage of tidal marsh habitat belonging to patches (useable areas of habitat separated from each other by non-useable areas of habitat; Fahrig and Merriam 1985) over a particular size threshold. For the analysis presented in the main body of this report, we utilize a threshold of 200 ha (494 acres), a value based on observed intertidal rail densities relative to patch size (described in greater detail below).

Studies of patch size are a basic quantitative proxy for qualitative changes to the structure and function of marsh habitat caused by fragmentation and are generally grounded in the equilibrium theory of island-biogeography and species-area relationships, which hold that all else being equal, smaller areas hold smaller populations, which are more vulnerable to extinction than larger populations (MacArthur and Wilson 1967; Soule 1987; Noss et al. 1995). Habitat fragmentation, which is technically separate from, but usually coincident with habitat loss, affects habitat connectivity, metapopulation dynamics, and the physical conditions within habitats (e.g. Saunders et al. 1991). When larger marshes are fragmented into smaller ones, their value as wildlife habitat tends to decrease. Speaking generally, 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). Larger marshes are more likely than smaller marshes to support a mosaic of marsh features (e.g., high marsh, low marsh, marsh pans), buffer native wildlife from nonnative predators, and have well developed tidal channel networks (all of these factors are, for example, positively associated with endangered Ridgway's Rail densities in San Francisco Bay; Liu et al. 2012).

Both tidal marsh indicators build off of previous work. The tidal marsh regional extent indicator relies heavily on the work done for the *Baylands Ecosystem Habitat Goals Project* ("Goals Project"; Goals Project 1999), which assessed changes in the regional extent of bayland habitats, including tidal marsh, between ca. 1800 and ca. 1997. The regional extent of tidal marsh in the Bay was updated for both *The State of San Francisco Bay 2011* (SFEP 2011) and the forthcoming *Baylands Ecosystem Habitat Goals Science Update* ("Goals Project Update"; report in press, scheduled for release in fall 2015). This indicator also builds on studies analyzing the regional extent of marsh in the Delta over time (Atwater et al. 1979, The Bay Institute 1998, Whipple et al. 2012, SFEI-ASC 2014).

Methods for delineating and evaluating historical (circa 1800) and existing (circa 1997) tidal marsh patches in the Estuary were first developed/reported by Collins and Grossinger (2004) in a report analyzing the landscape dynamics of South San Francisco Bay. Using the same methodology, historical and contemporary tidal marsh patches were delineated for the full Bay in *The State of San Francisco Bay 2011* report (SFEP 2011; an analysis led by Dr. Josh Collins). The methods were first applied to the Delta's marshes for the CDFW-funded "Delta Landscapes Project" and published in *A Delta Transformed* (SFEI-ASC 2014). An analysis of marsh patch sizes that considered Bay and Delta marshes together was first presented as a poster at the 2014 State of the Estuary Conference (Safran et al. 2014), but this effort did not distinguish between tidal and non-tidal marshes. This current report therefore represents the first known effort to evaluate the patch size distribution of tidal marshes across the full Estuary. An analysis comparing ca. 1800 and ca. 2009 marshes will also be included in the forthcoming Goals Project Update (report in press).

The analysis of tidal marsh presented in this report differs from that of its predecessor, *The State of San Francisco Bay 2011*, in three main ways. First, this report incorporates the tidal marshes of the Delta and therefore draws upon additional data sources to capture the expanded study extent. Second, although the guiding principles and general methodology used to determine tidal marsh regional extent and to delineate tidal marsh patches in this report are similar to those utilized in *The State of San Francisco Bay 2011* report, the technical implementation of the methodology differs. The nature of and reasons for these changes are detailed below. Finally, the final method/calculation used to evaluate/report tidal marsh patch sizes in the main body of report differs. In the 2011 report, the authors calculated changes in patch size-frequency. Although we present an updated calculation of tidal marsh patch size-frequency in this technical appendix, the metric presented in the main body of the report is instead the percent of total tidal marsh area belonging to patches >200 ha (494 acres) in size.

Benchmarks

Tidal marsh – regional extent

We utilize separate benchmarks to evaluate the regional extent of tidal marsh in the Bay and in the Delta. **For the Bay, we use a benchmark of 100,000 acres, a long-term tidal marsh acreage goal put forth by the 1999 *Baylands Ecosystem Habitat Goals Report***. This goal was the culmination of science-based public process that sought to evaluate the habitat needs of representative species and to identify changes needed to improve the Bay's ecological functioning and biodiversity. It is approximately half of the tidal marsh area that existed in the Bay at the beginning of the 19th century.

Since no similar quantitative goal exists for tidal marsh regional extent in the Delta, we instead three different provide reference values for context:

(1) *180,000 acres* or approximately half of the tidal marsh area that existed in the Delta at the beginning of the 19th century. In that it equals approximately one half of the historical habitat acreage, it is comparable to the benchmark used to assess the regional extent of tidal marsh in the Bay. The value was calculated by dividing the total area of tidal freshwater emergent wetland identified by Whipple et al. (2012) as occurring in the Delta ca. 1800 (364,810 acres) by two and then rounding to the nearest 10,000 acres.

(2) *78,000 acres* or the current area of tidal marsh plus the approximate area of diked lands in the Delta that are at intertidal elevations. This is the current area that would fall between high and low tide in the absence of levees and other water control structures and therefore exists at the right elevation for tidal marsh formation in the Delta. It was calculated by adding the area of diked lands at intertidal elevations in the Delta (70,000 acres) as reported by Siegel (2014) to the ca. 2002 area of tidal marsh reported in this analyses (7,638 acres, see below) and rounding to the nearest 1,000 acres. This value is meant to contextualize the upper bounds of tidal marsh regional extent based on existing elevations alone and does not take into consideration the acreage of land that will be available for tidal marsh restoration given other priority land uses in the region (such as agriculture). As with the other reference values, this value is not presented as a goal or benchmark.

3) 17,000 acres or the current area of tidal marsh plus the maximum amount of tidal marsh habitat that would be restored over the next five years under the State's current plan for habitat restoration in the Delta (*California Eco Restore*). *California Eco Restore* currently calls for 9,000 acres of tidal and sub-tidal habitat restoration over the next five years (California Natural Resources Agency 2015). The 17,000 acre reference value was determined by adding these 9,000 acres to the existing (ca. 2002) area of tidal marsh habitat in the Delta (7,638 acres; see below) and rounding to the nearest 1,000 acres. This calculation assumes that all 9,000 acres of proposed tidal and sub-tidal habitat restoration become tidal marsh. It therefore represents the *maximum* regional extent of tidal marsh habitat that would exist in the Delta after successful implementation of the current iteration of *California Eco Restore*.

Tidal marsh – patch sizes

The benchmark for the tidal marsh patch sizes indicator is the historical (circa 1800) size distribution of tidal marsh patches, as measured by the percentage of tidal marsh area belonging to patches >200 ha (494 acres) in size. Justification for using the historical patch size distribution of tidal marshes as a benchmark to assess current patch size distribution was provided by Dr. Josh Collins in *The State of San Francisco Bay 2011* (SFEP 2011, Appendix D). The flowing three paragraphs are an excerpted and slightly modified version of that justification:

Three basic assumptions underlie the decision to use the historical (ca. 1800) patch size distribution of tidal marshes as a benchmark to assess current and future patch size distributions. First, it is assumed that the current patch size 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 patch size distribution successfully 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, their core-area ratio has decreased (Safran et al. 2012; SFEI-ASC 2014), and the distance between patches has increased (Collins et al. 2005; Safran et al. 2012; SFEI-ASC 2014). All of these changes have, in theory, increased tidal marsh wildlife's risk of predation, exposure to external stressors, and required dispersal distances (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 overshadow the effect of fragmentation (Bender et al. 1998, Harrison and Bruna 1999)—this is one reason the tidal marsh patch size indicator must be considered alongside the tidal marsh regional extent indicator. 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 (SFEI-ASC 2014).

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 salinity regimes, temperatures, substrate colors, hydrology, vegetation, predators, and 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.

The specific method for calculating, visualizing, and comparing patch size distribution across time in this report differs from the methods utilized in its predecessor (SFEP 2011). The 2011 report presented the percentage of patches in each of six patch size categories and then measured whether or not the current percentage of patches in each class was within 25 percent of the historical percentage. To report a final benchmark, the report then measured what percentage of the classes passed this test. In this report, the patch size distribution is calculated as the percentage of total tidal marsh area belonging to patches >200 ha (494 acres) in size. Measuring patch size distribution in this way allows us to consolidate the measurement of each year into a single value (as opposed to a range), utilize a single benchmark (as opposed to a separate one for each size category), and conform to the form of other indicators in this report (with “up” on the bar chart corresponding to “good”). Our hope is that this method of calculation is a simpler measurement of patch size distribution and is easily accessible to the report’s general audience.

The 200 ha (494 acre) threshold is based on indications that this is an ecologically significant size threshold for intertidal rails (the wildlife group for which patch boundaries were defined, see below). Specifically, we draw on research intro the distribution and population trends of Ridgway Rail that suggests their population density increases with marsh area up to approximately 200 ha (494 acres), at which point rail densities plateau (Liu et al 2012, Wood et al. 2013; Figure 1). There are indications that densities of Black Rail might plateau at a lower marsh patch size (~100 ha) than observed for Ridgway’s Rail (Nadav Nur, personal communication). Other results also point to 100 ha as a meaningful tidal marsh patch size threshold for Black Rails—Spautz and Nur (2002) and Spautz et al. (2005) report a significant negative correlation between Black Rail presence and the distance to the nearest 100 ha (247 acre) marsh (significant relationships were not observed when testing Black Rail presence against the distance to marshes of 25 or 50 ha). Despite this information, we utilized the larger 200 ha (494 acre) threshold under the assumption that, when considering tidal marsh patch sizes in the San Francisco Estuary, Ridgway's Rail can serve as an umbrella species for Black Rail. The main premise of the umbrella species concept is that the requirements of demanding species encapsulate those of many co-occurring, less demanding species (Roberge and Per Angelstam 2006). Ozaki et al. (2006) relate the concept specifically to patch sizes when they define umbrella species as "those with large area requirements for which protection of the species offers protection to other species that share the same habitat."

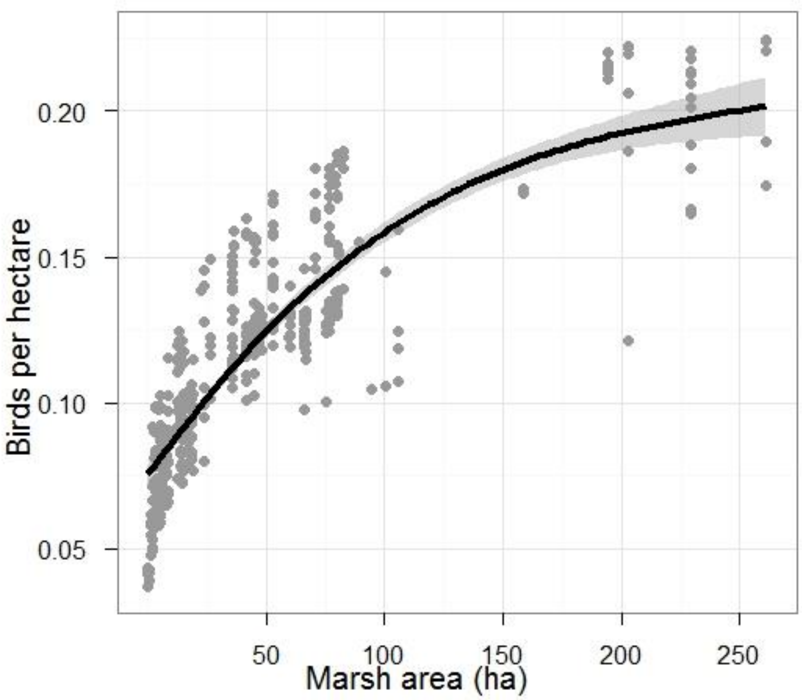


Figure 1 (*courtesy Julian Wood and Nadav Nur, Point Blue Conservation Science, adapted from Liu et al. 2012 and Wood et al. 2013).* The relationship between tidal marsh area and Ridgway Rail density. Rails appear to reach a maximum density at approximately 200 ha. This finding is used here to define a tidal marsh patch size threshold for the patch sizes indicator.

There are certain limitations to the benchmark, the first being its focus on tidal marsh as habitat for intertidal rails. Although, from the perspective of patch size, rails have relatively demanding habitat needs—a patch that is large enough for Ridgway's Rail, for example, should not be limiting (based on size alone) for small resident rodents—there are functions of marshes that are likely only realized at even larger sizes. One advantage of highlighting the full patch size-frequency distribution (as was done in the main body of the 2011 report) is that it involves no assumptions about the importance of any particular patch size and could therefore be used to assess a wider range of ecosystem services and ecological functions for which optimal size might differ (SFEP 2011, Appendix D). Additionally, although the benchmark is meant to measure the general distribution of patch sizes (aka, "a certain percentage of marsh area should belong to patches above a certain size"), it could give the impression that small tidal marsh patches are not valuable to wildlife. This is not the case and is not the intention of the benchmark. Small patches are likely important as "stepping stones" between larger patches, facilitating the movement and gene flow of marsh wildlife (e.g. Gilpin 1980, Simberloff et al. 1992, Fischer and Lindenmayer 2002, Murphy and Lovett-Doust 2004, Baum et al. 2004). Black rails, for example, have been observed in marsh patches as small as 2 ha (Hildie Spautz, personal communication). Finally, assigning a size threshold based on population density has some inherent limitations—density alone offers no indication of population resilience and demographic processes. The benchmark and size threshold used to analyze the distribution of tidal marsh patches should continue to be reevaluated as new information and techniques become available.

Data Sources

GIS data depicting the extent of tidal marshes in the Estuary were obtained from multiple regional wetland mapping efforts (Table 1).

Table 1. Geospatial datasets utilized in this study to determine the extent of tidal marshes in the Estuary.

Region/Year	Citation	Title	Source Institution	Year released	Years represented (range)	Year represented (primary)	Link (accessed 2/25/2015)
Bay							
ca. 1800	SFEI 1997a	EcoAtlas Baylands Maps ('Historical Baylands')	San Francsico Estuary Institute	1997	ca. 1800	ca. 1800	http://www.sfei.org/sites/default/files/EcoAtlas_SFEI.zip
ca. 1997	SFEI 1997b	EcoAtlas Baylands Maps ('Modern Baylands')	San Francsico Estuary Institute	1997	1985-1997	1997	http://www.sfei.org/sites/default/files/EcoAtlas_SFEI.zip
ca. 2009	SFEI 2011	Bay Area Aquatic Resource Inventory ('BAARI Baylands v01')	San Francsico Estuary Institute	2011	2005-2009	2009	ftp://dl.sfei.org/geofetch/BAARI.zip
Delta							
ca. 1800	Whipple et al. 2012	Sacramento-San Joaquin Delta Historical Ecology Investigation ('Historical Habitats Delta')	San Francisco Estuary Institute	2012	ca. 1800	ca. 1800	http://www.sfei.org/sites/default/files/Delta_Historical_Ecology_GISdata_SFEI_ASC_2012.zip
ca. 2002	CDWR 2013	Draft Bay-Delta Conservation Plan- Natural Communities	California Department of Water Resources	2013	2002-2010	2002	http://baydeltaconservationplan.com/Libraries/Dynamic_Document_Library/Public_Draft_BDCP_Chapter_2_-_Existing_Ecological_Conditions.sflb.ashx (Figure 2-14)

Boundary conditions defining the extent of the Bay and the Delta were enforced for each layer. Tidal marsh polygons were excluded from the Bay datasets if they were west of the Golden Gate or upstream of Broad Slough. Tidal marsh polygons were excluded from the Delta datasets if they were downstream of Broad Slough or outside of the Legal Delta boundary (although this latter condition did not ultimately exclude any areas mapped as tidal marsh in the Delta). Figure 2 provides a detailed view of the line dividing the Bay and the Delta at Broad Slough—it was derived from the eastern margin of the *Baylands Ecosystem Habitat Goals Project* study extent.

Figure 2. The dividing line (in yellow) between San Francisco Bay (the "Bay") and the Sacramento-San Joaquin Delta (the "Delta") utilized in this study.

The original source classifications we considered "tidal marsh" for this study are listed, by source, in Table 2. The crosswalk for the Bay sources was originally developed for the *Baylands Ecosystem Habitat Goals Update* (report in preparation, scheduled for release in spring 2015). The crosswalk for the Delta sources was originally developed for the Delta Landscapes Project (SFEI-ASC 2014).

Table 2. Original classifications considered "Tidal marsh" for this study, by source (see Table 1).

Bay ca. 1800 (SFEI 1997a; "CROSSWALK" field)
Tidal Marsh
Bay ca. 1997 (SFEI 1997b; "SHORT_DEFN" field)
Old High Tidal Marsh
Young High Tidal Marsh
Young High Tidal Marsh within Modern but not Historical extent
Young Low/Mid Tidal Marsh
Muted Tidal Marsh
Bay ca. 2009 (SFEI 2011; "CLICKLABEL" field)
Tidal Ditch
Tidal Marsh Flat
Tidal Panne
Tidal Vegetation
Delta ca. 1800 (SFEI 2012; "Habitat_Type" field)
tidal freshwater emergent wetland
Delta ca. 2002 (CDFW 2013; "SAIC_Type" field)
Tidal Freshwater Emergent Wetland
Tidal Brackish Emergent Wetland

The same tidal marsh datasets developed for the regional extent indicator were used for the tidal marsh patch sizes indicator.

Methods

Tidal marsh – regional extent

Determining the regional extent of tidal marsh

The total acreage of tidal marshes, as identified in the crosswalks reproduced in Table 2, was tabulated separately for each spatial dataset (Bay ca. 1800, Bay ca. 1997, Bay ca. 2009, Delta ca. 1800) using a Geographic Information System (GIS).

Determining the extent of recent tidal wetland restoration

To determine the acres of tidal wetlands that have been restored since the most recent standardized datasets were developed, we compiled a list of restoration sites that have been opened to tidal action since 2009 in the Bay and since 2002 in the Delta (the primary years of source imagery for the SFEI 2011 and CDFW 2013 datasets, respectively). Bay sites were initially identified using the EcoAtlas Project Tracker database (CWMW 2015) by querying projects within the administrative boundary of Regional

Board 2 with a planned habitat type of "Estuarine wetlands" and an event type entry of "Groundwork start" or "Groundwork end" since 2009. This resulting list was reviewed and edited by local scientists with knowledge of recent/ongoing restoration efforts (April Robinson and John Bourgeois, personal communication). Delta projects implemented since 2002 that seek to increase the acreage of tidal wetlands were initially identified by reviewing sources that summarize recent restoration efforts in the Delta (Cannon and Jennings 2014; CDWR 2012). The resulting list was also reviewed and edited by local scientists with knowledge of recent/ongoing restoration efforts (Kristal Davis-Fadtke, personal communication).

The planned area of tidal wetland restoration for each site was determined using publically available data (see Table 6). When available, we recorded the expected net gain in tidal wetland area (as opposed to total planned acreage of tidal wetlands). All sites were reviewed against the datasets used to determine the regional extent of tidal marsh to ensure the new sites were not already counted as tidal marsh.

For both the Bay and the Delta, the acreages of recent tidal wetland restoration were added to the acreage of tidal marsh determined for most recent standardized datasets to develop the regional extent totals for ca. 2015. This methodology assumes that the area of existing tidal marshes has not changed since 2009 in the Bay and since 2002 in the Delta, and that the only possible change in tidal marsh extent comes from intertidal wetland restoration. This assumption has obvious limitations. Future updates of this indicator will benefit from updated standardized regional maps of tidal wetland restoration. Finally, it is worth noting that, although the area of intertidal wetland restoration is included on the chart of tidal marsh regional extent, not all of this acreage is yet (or will ever become) tidal marsh. Although a significant portion of the tidal wetland restoration areas are expected to develop into tidal marsh over time (or already have), some percentage of the habitat will remain un-vegetated, either unintentionally or by design. Once available, the 2015 acreages reported here should be replaced by values derived from actual updated maps of the Estuary's tidal marshes.

<u>Determining the regional extent indicator status/score</u>

Throughout this report, a three-tiered "Good—Fair—Poor" system is used to assign a qualitative score to the status of each indicator. With few exceptions, the line between "Good" and "Fair" is set at each indicator's goal/benchmark and another means is used to establish the line between "Fair" and "Poor."

Rules and thresholds for determining the status of the regional extent of tidal marsh in the Bay are shown in Table 3. The line between "good" and "fair" was set at the regional goal established by the *Goals Project* (1999) and, without any ecologically sound justification for another value, the line between "fair" and "poor" was simply set at half this amount. Since no quantitative benchmarks were developed for determining the regional extent of tidal marsh in the Delta, we did not develop rules and thresholds for determining the status of the indicator in that region. For now, we assigned the Delta a score of "poor" based on the fact that the current regional extent is less than one half the lowest reference value utilized in this study (see Table 5), but the system for scoring this indicator should be reevaluated in the future once a benchmark or regional goal is determined.

Table 3. Rules employed for determining the status of regional extent of tidal marsh in the Bay. No rules were developed for assigning the status of the indicator in the Delta.

Status	Regional extent	Explanation
Good	>100,000 acres	The indicator receives a score of "good" when it exceeds the 100,000 acre regional goal established by the *Goals Project* (1999).
Fair	50,000-100,000 acres	The indicator receives a score of "fair" when it exceeds one-half of the regional goal.
Poor	<50,000 acres	The indicator receives a score of "poor" when it is less than one-half of the regional goal.

Tidal marsh – patch sizes

Defining individual marsh patches

Note: although the guiding principles and general methodology used to delineate tidal marsh patches in this report are similar to those developed by Collins and Grossinger (2004) and utilized in *The State of San Francisco Bay 2011* report, the technical implementation of the methodology differs. The precise patch boundaries identified and utilized by the two studies may therefore vary. See below for more details.

Patches were generated using the tidal marsh datasets spatial described above. Since tidal marsh patches can span the boundary between the Bay and the Delta (Figure 2), the Bay and Delta tidal marsh polygons were combined before defining patches. "Historical patches" were generated after combining the 'Bay ca. 1800' and the 'Delta ca. 1800' polygons. "Modern patches" were generated after combining the 'Bay ca. 2009' and the 'Delta ca. 2002' datasets. For the sake of this analysis, patches that ultimately spanned the boundary of the Bay and Delta ("transboundary patches"—of which there were two in the historical patches and one in the modern patches) were assigned to the Bay. For the charts in this report, the modern patches located in the Bay are said to be representative of conditions ca. 2009 and the modern patches in the Delta are said to be representative of conditions ca. 2002 even though the single "transboundary patch" assigned to the Bay was generated from polygons representative of both years. The vast majority of patches were generated from a single spatial dataset representative of a single point in time (ca. 2009 for the Bay and ca. 2002 for the Delta).

In the GIS, discrete tidal marsh polygons were aggregated into a single "patch" if they were located within 60 m of one another. Groups of polygons separated by less than this distance were identified and aggregated using ArcGIS's 'Aggregate Polygons' tool and then assigned unique patch identification values. The full work flow for this analysis was implemented/automated using a custom tool developed with ArcGIS's Model Builder software.

The 60 m threshold for grouping marsh polygons was derived from the rule set for defining resident intertidal rail patches developed by Collins and Grossinger (2004), which was based on the best available data on rail habitat affinities and dispersal distances. For additional information on the development of rules for defining tidal marsh patches and analyzing tidal marsh fragmentation, please refer to Collins and Grossinger (2004) and *The State of San Francisco Bay 2011*, Appendix D (SFEP 2011).

In the absence of more specific data, we made the assumption that the rules developed for defining intertidal rail patches in the salt marshes of South San Francisco Bay are also applicable to the freshwater marshes of the Delta. Unlike Collins and Grossinger (2004), our analysis also only considered roads and levees as dispersal barriers if the width of these features (as mapped in the habitat type layers) exceeded the 60 m distance threshold described above. Similarly, we also did not consider channels that receive perennial freshwater discharge to be barriers unless they exceeded the 60 m distance threshold. Finally, we did not employ the rule that “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” (Collins and Grossinger 2004). These modifications to the rule set increase repeatability of the patch size analysis, which is important for its use an indicator that will be re-measured at regular intervals in the future. The patch-generating process was developed into an automated model using ArcGIS’s model builder tool to maximize repeatability.

It is worth noting that this model of a binary landscape (marsh and non-marsh) greatly simplifies the complexities of how species interact with their surroundings. It assumes, for example, that all patches of tidal marsh are equally suitable for intertidal rails, that the routes of travel between patches are linear, and that the only barrier to wildlife movement is distance (D’Eon et al. 2002).

Final patch boundaries can be seen in the map of historical patches (Figure 3) and the map of modern patches (Figure 4).

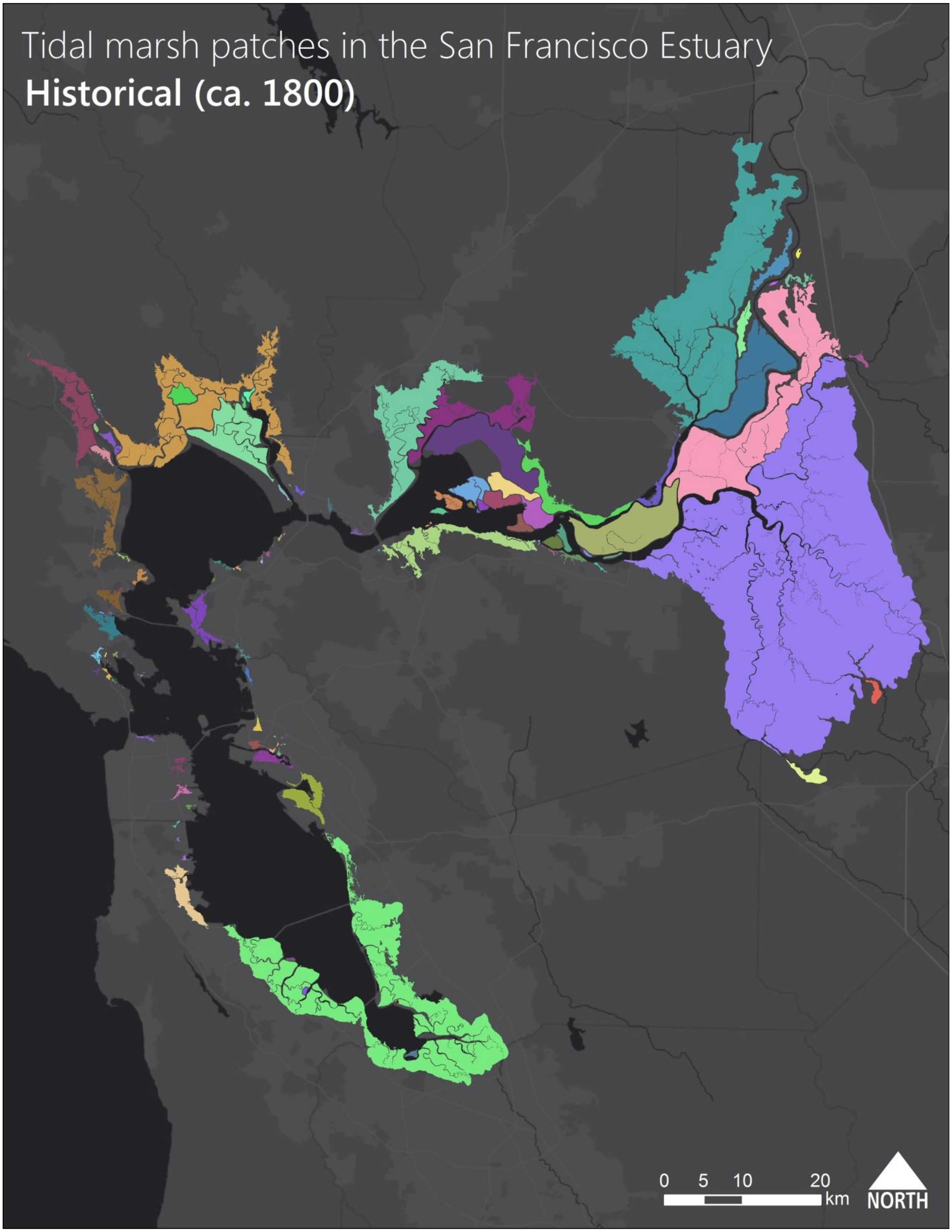


Figure 3. Tidal marsh patches in the historical San Francisco Estuary (ca. 1800). Each patch is given a different color. The rules for defining patches are described above. Compare with the map of modern tidal marsh patches in Figure 4.

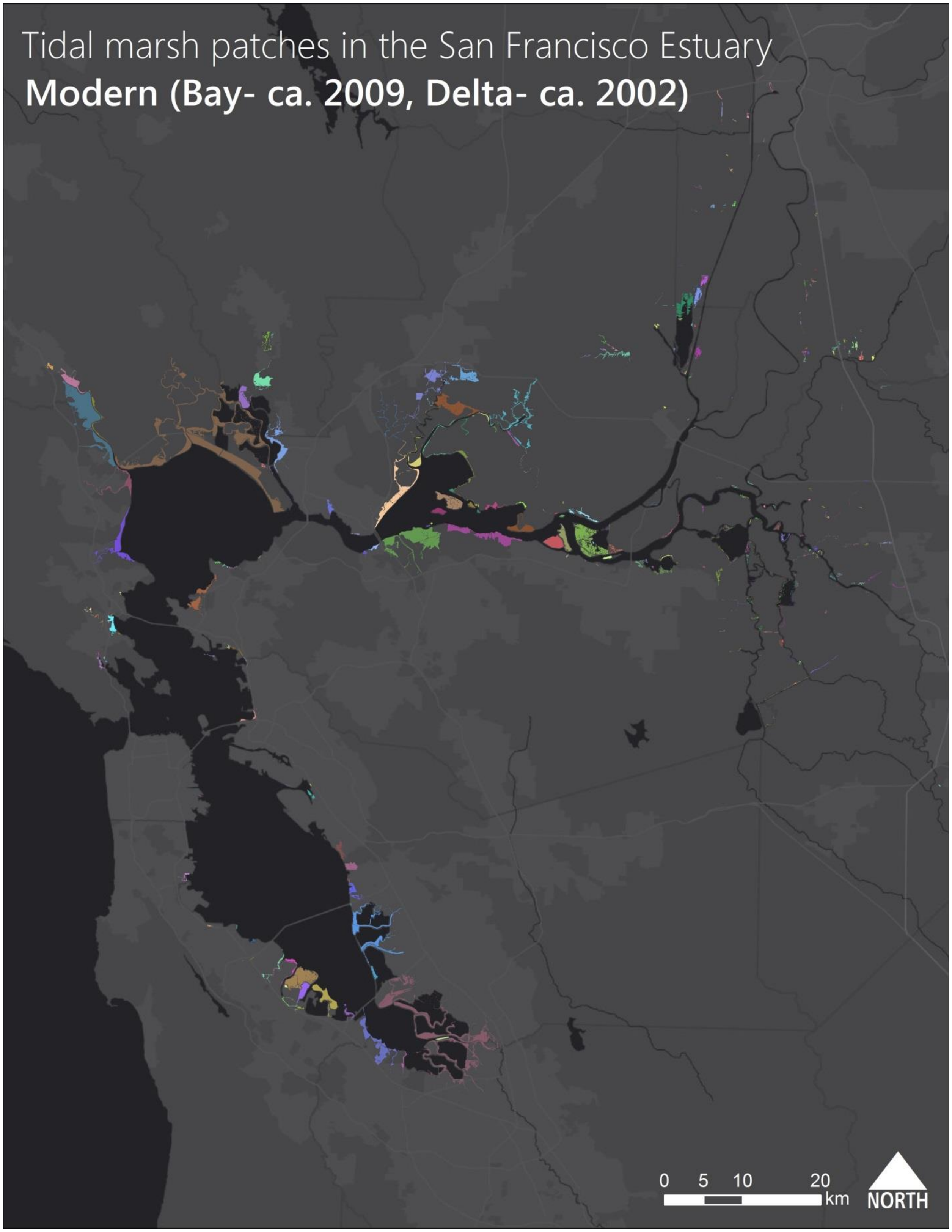


Figure 4. Tidal marsh patches in the modern San Francisco Estuary (ca. 2009 for the Bay and ca. 2002 for the Delta). Each patch is given a different color. The rules for defining patches are described above. Compare with the map of historical tidal marsh patches in Figure 3.

Measuring tidal marsh patch size distributions

After tidal marsh patch boundaries were defined, we calculated the size of each individual patch using ArcGIS. We assessed tidal marsh patch sizes using three methods: (1) calculating the percent of total marsh area belonging to a patch >200 ha (494 acres) in size, (2) calculating the patch size-frequency distribution, and (3) calculating the cumulative frequency distribution. For each method were only able to compare patch sizes at two points in time (ca. 1800 and ca. 2009 for the Bay, ca. 1800 and ca. 2002 for the Delta). The **percent of total marsh area belonging to patches >200 ha in size** was calculated for each time interval by summing the total area of tidal marsh belonging to patches greater than 200 ha (494 acres) and dividing by the total acreage of tidal marsh (the above section on benchmarks discusses how the 200 ha threshold was selected). For the **patch size-frequency distribution**, we calculated both the percent of total marsh patches and percent of total marsh area in each of the six size classes utilized in the 2011 report (SFEP 2011; refer to Appendix D for how patch size classes were selected). **Cumulative fraction functions** were generated using the patch size data for each region and time step and compared using two-sample Kolmogorov-Smirnov (K-S) tests (Kirkman 1996). All patch size distributions were non-normal. The resulting p-values were used to assess and compare the similarities and differences in patch size distributions.

Determining the patch sizes indicator status/score

Rules and thresholds for determining the status of the tidal marsh patch sizes indicator in both the Bay and the Delta are shown in Table 4. The line between “good” and “fair” was set at 80% of the historical proportion of marsh belonging to patches >200 ha (494 acres) in size. The line between “fair” and “poor” was simply set at half of this proportion (or at 40% of the historical proportion). From an ecological standpoint, these thresholds are, admittedly, somewhat arbitrary. They are guided by the notion that the size distribution of marsh is “good” if it is within some percentage of the historical distribution (either slightly below or slightly above). In *The State of San Francisco Bay 2011* (SFEP 2011), the proportion of tidal marsh patches in a size category was “good” if it was within 75 and 125% percent of the historical proportion. We follow this general guideline, but have changed the qualifying range to 80-120% of the historical proportion. This was done because we were seeking to create three scores (good-fair-poor) and 120% is evenly divisible by 3. Since, when using a 200 ha (494 acres) size threshold, the contemporary proportion cannot actually exceed 120% of the historical proportion, we did not define scores for current proportions that exceed the 120% of the historical proportion. This would only become necessary if the benchmark utilized a higher patch size threshold (and thereby decreased the historical proportion of marshes above the critical size).

Table 4. Rules employed for determining the status of the tidal marsh patch size indicator in the Bay and the Delta.

Status	Current proportion of total marsh belonging to patches > 200 ha	Explanation
Bay (historical proportion ca. 1800 = .964)		
Good	>0.771 (80-120% of historical proportion)	The indicator receives a score of "good" when the current proportion of tidal marsh belonging to patches >200 ha is 80-120% of the historical proportion. Since the upper bounds of this range exceeds 1, the indicator effectively receives a score of "good" when current proportions are greater than 0.771.
Fair	0.386-0.771 (40-80% of historical proportion)	The indicator receives a score of "fair" when the current proportion of tidal marsh belonging to patches >200 ha is 40-80% of the historical proportion (between 0.386 and 0.771).
Poor	<0.386 (0-40% of historical proportion)	The indicator receives a score of "poor" when the current proportion of tidal marsh belonging to patches >200 ha is <40% of the historical proportion (<0.386).
Delta (historical proportion ca. 1800 = .997)		
Good	>0.798 (80-120% of historical proportion)	The indicator receives a score of "good" when the current proportion of tidal marsh belonging to patches >200 ha is 80-120% of the historical proportion. Since the upper bounds of this range exceeds 1, the indicator effectively receives a score of "good" when current proportions are greater than 0.798.
Fair	0.399-0.798 (40-80% of historical proportion)	The indicator receives a score of "fair" when the current proportion of tidal marsh belonging to patches >200 ha is 40-80% of the historical proportion (between 0.399 and 0.798).
Poor	<0.399 (0-40% of historical proportion)	The indicator receives a score of "poor" when the current proportion of tidal marsh belonging to patches >200 ha is <40% of the historical proportion (<0.399).

Results

Tidal marsh- regional extent

The regional extent of tidal marsh for each region and time period is shown below both in Figure 5 and Table 5. Values for ca. 2015 were calculated for each region by adding the most recent regional extent of tidal marsh to the acreage of recent tidal wetland restoration (Table 6).

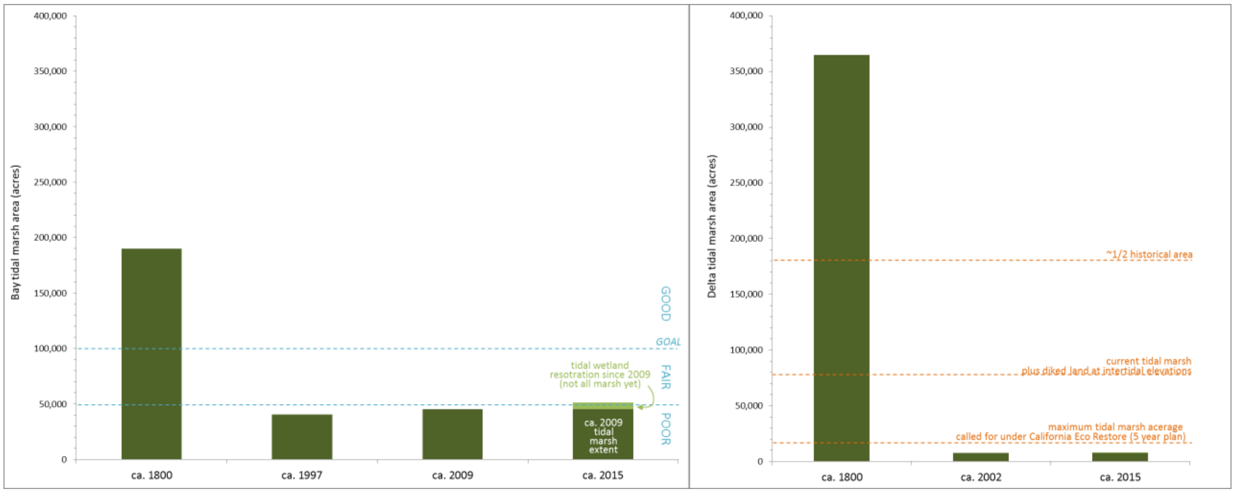


Figure 5. Tidal marsh regional extent in the Bay (left panel) and Delta (right panel) over time. Note that x-axes are not to scale. Circa 2015 regional extents are calculated by copying the previous time interval's regional extent and adding the extent of tidal wetland restoration that has occurred since (light green bar segments). Although much of this area is expected to transition into tidal marsh over time, some will remain unvegetated—it is shown to approximate progress since the last comprehensive spatial datasets of tidal marsh extent in the Bay and Delta were developed. Tidal wetland restoration since 2002 in the Delta is included, but is too small to be visible at this scale. Reference values on the Delta chart are colored orange to distinguish them from proper goals and benchmarks (colored blue).

Table 5. Regional extent of tidal marsh in the Bay and the Delta at multiple points in time. Data sources and the methods for defining regions are detailed above. In the main body of the report, values for ca. 2015 were calculated for each region by adding the most recent regional extent to the acreage of recent tidal wetland restoration (Table 6).

Year	Tidal marsh regional extent (acres)
Bay	
ca. 1800	190,113
ca. 1997	40,514
ca. 2009	45,052
Delta	
ca. 1800	364,545
ca. 2002	7,638

Historically, the area of (freshwater) tidal marsh in the Delta exceeded the area of (salt and brackish) tidal marsh in the Bay by a factor of nearly 2. Today, the reverse is true, and the area of tidal marsh in the Bay exceeds the area of tidal marsh in the Delta by a factor of nearly 6 (Figure 6).

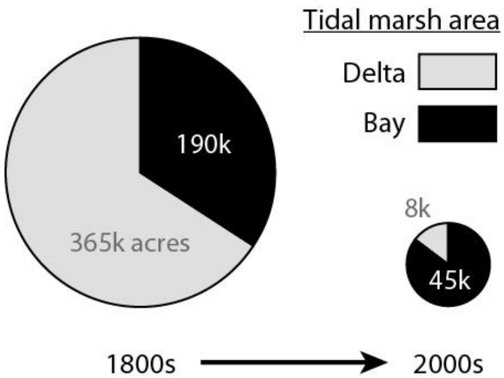


Figure 6. Historical (1800s) and modern (2000s) tidal marsh regional extent (in acres) by region. "2000s" data is ca. 2009 for the Bay and ca. 2002 for the Delta.

Based on the rules described in the methods section, the regional extent of tidal marsh in the Bay is characterized as "fair." Since it is below 50,000 acres, the ca. 2009 extent of tidal marsh alone would only qualify as "poor." The score of "fair" is based on the ca. 2015 regional extent value (51,398 acres), which combines the area of tidal marsh ca. 2009 with the area of tidal wetland restoration that has occurred since (Table 4), which together exceed the 50,000 acre threshold for "fair" (Table 3). This score is consistent with the ranking of "fair" previously reported for the indicator status in *The State of San Francisco Bay 2011* (SFEP 2011). The regional extent of tidal marsh in the Delta is characterized as "poor," since, as described in the methods section, the current regional extent is less than one half the lowest reference value utilized in this study. The system for scoring this indicator should be reevaluated in the future once a true benchmark or regional goal is determined.

Recent tidal wetland restoration

In the Bay, approximately 6,350 acres have been restored to tidal action since 2009 (Table 6). This figure does not include an additional approximately 24,000 acres of tidal marsh restoration that are currently permitted and/or funded, but have not yet broken ground (Goals Project Update, report in preparation; calculated by subtracting the acreage of restoration since 2009 determined for this study from the total permitted/funded acreage of post-2009 tidal marsh restoration identified by the Goals Project Update).

In the Delta, tidal wetland restoration since 2002 has totaled approximately 250 acres (Table 6). Since this list only includes restoration projects that have broken ground, it does not capture the nearly 5,000 acres of tidal marsh restoration planned for the Delta in the near future (

Table 7).

Table 6. Recent tidal wetland restoration. The areas listed below have been opened to tidal action since the datasets utilized in this study were developed (ca. 2009 for the Bay; ca. 2002 for the Delta). Although much of this restored tidal habitat is

expected to transition into tidal marsh over time, these sites are not yet included in in the maps and charts summarizing the regional extent of tidal marsh.

Site / Project	Year opened to tidal action	Planned area of tidal wetland restoration (acres)	Source
Bay (tidal wetland restoration since 2009)			
Napa Plant Site: Central Unit	2009	175	1
Alviso: Pond A6	2010	330	2
Napa Plant Site: South Unit	2010	1,080	1
Eden Landing: Ponds E8A/E9/E8X	2011	630	2
Alviso: Ponds A8/A7/A5	2012	1,400	2
Alviso: Pond A17	2012	130	3
Bair Island: Middle Bair	2012	646	4
Hamilton Marsh	2014	380	5
Bruener Marsh	2014	26	6
Cullinan Ranch	2015	1,549	7
Total (acres)		6,346	

Delta (tidal wetland restoration since 2002)			
Twitchell Island Setback Levee	2005	1	8
Sherman Island Setback Levee	2005	7	9
Liberty Island Conservation Bank and Preserve	2010	31	10
Cosumnes Floodplain Mitigation Bank	2011	73	11
Calhoun Cut	2014	147	12
Total (acres)		259	

Sources
1 CWMW 2015
2 SBSPRP 2015
3 USFWS 2011
4 measured from SFEI 2011
5 California State Coastal Conservancy 2008
6 NOAA 2014
7 USFWS n.d.
8 CDWR 2011 (“Twitchell Island Setback Levee Habitat Enhancement Project”)
9 CDWR 2011 (“Sherman Island Setback Levee Habitat Enhancement Project”)
10 ICF Jones & Stokes 2009
11 Personal communication, Jeff Mathews (Westervelt Ecological)
12 Personal communication, Kristal Davis-Fadtke (Delta Conservancy)

Table 7. Delta tidal marsh restoration projects planned for the near future. Together, these projects total approximately 4,650 acres. Projects and acreages come from Delta Conservancy scientists (Kristal Davis-Fadtke, personal communication).

Site / Project	EIR status	Planned area of tidal marsh restoration (acres)
Lower Yolo Ranch Tidal Restoration Project	Final EIR released July 2013	1,371
Prospect Island Tidal Habitat Restoration Project	Draft EIR expected 2015	1,528
North Delta Flood Control and Ecosystem Restoration Project	Final EIR released October 2010	1,200
Dutch Slough Tidal Marsh Restoration Project	Final EIR released March 2010; Final Supplemental EIR released September 2014	560
Total (acres)		4,650

Tidal marsh- patch sizes

Percent of total marsh area belonging to patches >200 ha in size

Historically, the proportions of tidal marsh in the Bay and the Delta belonging to patches >200 ha (494 acres) in size were both above 0.96 (Figure 7). This proportion has decreased over time (a greater percentage of total marsh area now belongs to patches <200 ha in size) in both the Bay and Delta, but the decrease is much more pronounced in the Delta. While nearly 100% of total marsh area in the Delta was once arranged in patches >200 ha (494 acres), this percentage has since dropped to less than 30%. Put another way, the patch size distribution in the Delta has skewed significantly towards patches that are too small to achieve maximum densities of intertidal rails (using the patch size threshold identified for Ridgway's Rail).

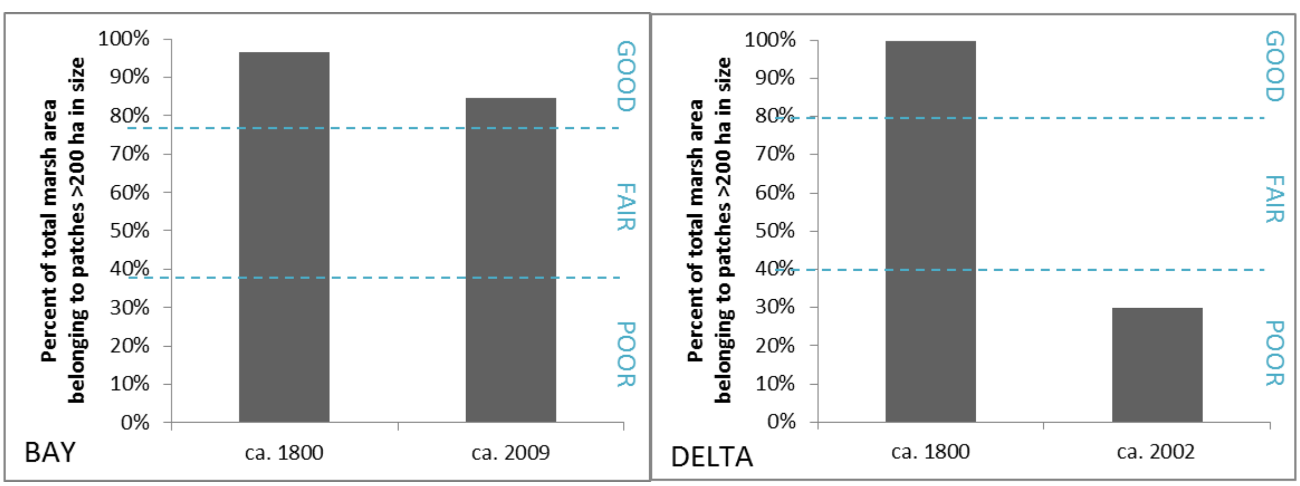


Figure 7. Percent of total marsh area belonging to patches >200 ha (494 acres) in the Bay (above) and Delta (below) over time.

It is worth mentioning that the overall results do not change dramatically if we use a smaller size threshold of 100 ha (the patch size at which Black Rails densities are known to plateau)—the proportion of total marsh area belonging to patches >100 ha has decreased in the Delta from 0.998 (ca. 1800) to 0.336 (ca. 2002) and in the Bay from 0.978 (ca. 1800) to 0.885 (ca. 2009). To document the effect of the patch size threshold on reported patch size distributions in the Bay and Delta over time, we include here the proportion of total marsh existing in patches of above 100 ha, 200 ha, 500 ha, 1,000 ha, and 10,000 ha (Table 8).

Table 8. Proportion of total tidal marsh area existing in patches above various minimum size thresholds.

Region	Historical (Bay-ca. 1800) (Delta- ca. 1800)	Modern (Bay- ca. 2009) (Delta- ca. 2002)
Proportion of total tidal marsh area in patches >100 ha		
Bay	0.98	0.88
Delta	1.00	0.34
Proportion of total tidal marsh area in patches >200 ha		
Bay	0.96	0.85
Delta	1.00	0.30
Proportion of total tidal marsh area in patches >500 ha		
Bay	0.93	0.62
Delta	0.99	0.21
Proportion of total tidal marsh area in patches >1,000 ha		
Bay	0.86	0.44
Delta	0.98	0.00
Proportion of total tidal marsh area in patches >10,000 ha		
Bay	0.42	0.00
Delta	0.90	0.00

Patch size-frequency distribution

Tidal marsh patch size-frequency plots generated for the historical and modern Bay and Delta were calculated with two different independent variables: the percentage of marsh patches (Figure 8) and the percentage of total marsh area (Figure 9) using the patch size classes identified in the 2011 report (SFEP 2011). The former measurement is considered a patch-centric approach ("what's the probability you'll land in a patch of a certain size if you're dropped in a *randomly selected patch*?"), while the latter measurement is effectively weighted by total area and considered "landscape centric" approach ("what's the probability you'll land in a patch of a certain size if you're dropped in a *randomly selected acre of marsh in the landscape?*") (McGarigal 2002). Measured either way, although the general shapes of the ca. 1800 and ca. 2009 tidal marsh patch size distributions in the Bay are similar, the current proportion of patches in the largest three size classes is still low. In the Delta, the difference between the historical and modern patch size distribution is more pronounced and heavily skewed towards the smallest size class. These trends are more pronounced when measured based on percent of total tidal marsh area (as opposed to percent of patches). Finally, it is worth noting that the patch size-frequency plots highlight just how small the 200 ha (494 acres) size threshold used in the tidal marsh patch size benchmark is relative to the historical range of patch sizes (more than 80% of the Bay's total tidal marsh extent and close to 100% of the Delta's was situated within patches larger than 5,000 acres).

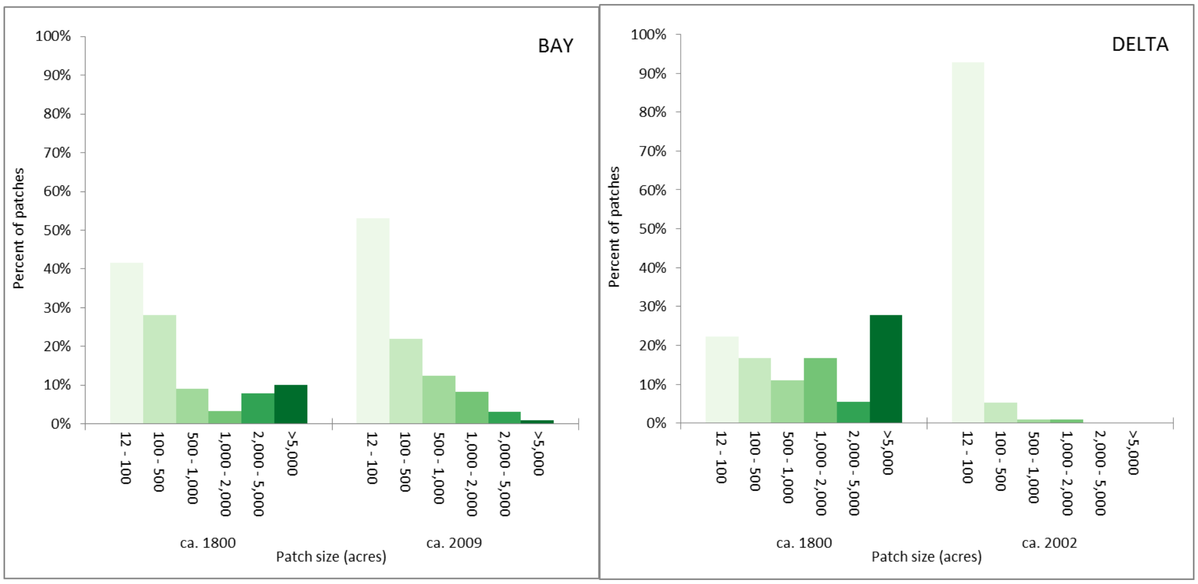


Figure 8. Patch size distributions of historical and modern tidal marsh patches in both the Bay (left panel) and Delta (right panel) as measured by the percent of tidal marsh patches in each of six patch size classes. A "patch-centric" measurement.

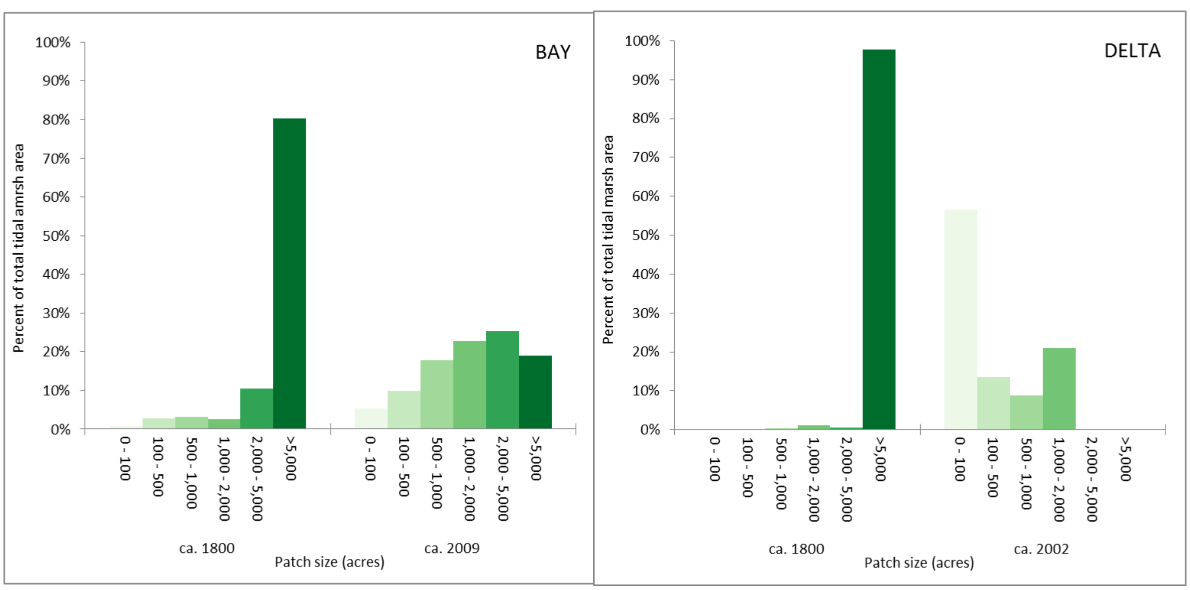


Figure 9. Patch size distributions of historical and modern tidal marsh patches in both the Bay (left panel) and Delta (right panel) as measured by the percent of total tidal marsh area in each of six patch size classes. A "landscape-centric" measurement.

Note that since the number of patches is highly sensitive to the minimum mapping unit [MMUs] of each dataset, we limited the smallest class in the charts of patch size-frequency distribution measured by the percent of patches (Figure 8) to 5 ha (equal to 12 acres and the largest minimum mapping unit employed by any of the source datasets). This effectively forced the modern datasets (with their slightly lower MMUs) to have the same MMU as the historical datasets. It is important to note that differences in minimum mapping unit have very little effect on the measurements calculated based on percent of total tidal marsh area (since the patches below 5 ha in are such a small percentage of the total tidal marsh area). We therefore utilized the full range of patch sizes when plotting the patch size-frequency distribution measured by the percent of total tidal marsh area (Figure 9).

Cumulative fraction functions

The cumulative fraction functions (Figure 10 - Figure 13) presented below to visualize and compare the full patch sizes across regions (Delta and Bay) and time (historical [ca. 1800] and modern [ca. 2020 or ca. 2009]). We generated four cumulative fraction plots (measuring the cumulative fraction of tidal marsh patches—not tidal marsh area—across the full range of patch sizes) comparing the historical Bay with the modern Bay (Figure 10), the historical Delta with the modern Delta (Figure 11), the historical Bay with the historical Delta (Figure 12), and the modern Bay with the modern Delta (Figure 13). Comparison of p-values suggests that the patch size distribution of the modern Bay is more similar to the patch size distribution of the historical Bay than the modern Delta is to the historical Delta (Figures 7 – 8). Additionally, comparison of p-values suggests that the distributions of tidal marsh patch sizes in the Bay and the Delta were more similar historically than they are today (Figures 9 -10).

As with the patch-size frequency measured with the percent of patches, we only considered patches above 5 ha for this analysis to force similar MMUs across all datasets (see the final paragraph of the previous section for further explanation).

Figure 10. Comparative cumulative fraction of tidal marsh patches (y-axis) across the full range of patch sizes (hectares; x-axis) in the ***Bay ca. 1800 (solid line) and the Bay ca. 2009 (dotted line)***. The maximum difference between the cumulative distributions, D, is 0.1606 with a corresponding P value of 0.168. The null hypothesis that the distributions are similar is

accepted ($p > 0.05$). In both conditions, patches are relatively evenly distributed across their full patch size range, but the maximum patch size in the historical Bay was an order of magnitude larger than in the modern Bay.

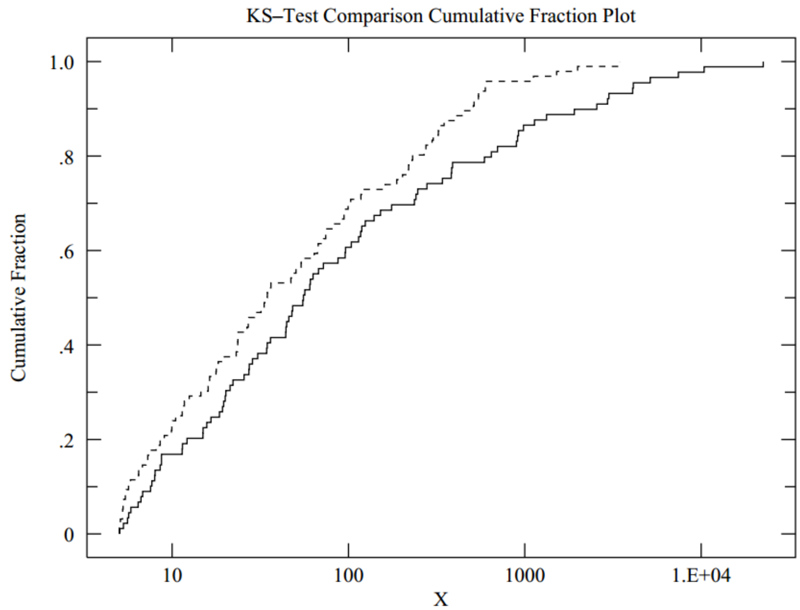


Figure 11. Comparative cumulative fraction of tidal marsh patches (y-axis) across the full range of patch sizes (hectares; x-axis) in the ***Delta ca. 1800 (solid line) and the Delta ca. 2002 (dotted line).*** The maximum difference between the cumulative distributions, D, is 0.8646 with a corresponding P value of 0.000. The null hypothesis that the distributions are similar is rejected ($p < 0.05$). Relative to historical conditions, the relative fraction of tidal marsh patches is skewed towards smaller patch sizes (more than 90% of patches less than 100 ha today versus ~30% historically). The maximum patch size in the historical Delta was two orders of magnitude larger than in the modern Delta.

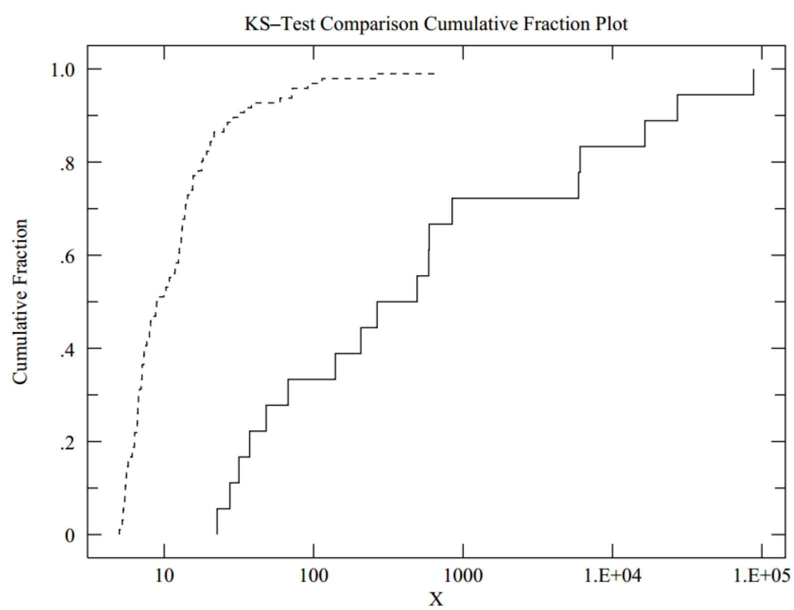


Figure 12. Comparative cumulative fraction of tidal marsh patches (y-axis) across the full range of patch sizes (hectares; x-axis) in the ***Bay ca. 1800 (solid line) and the Delta ca. 1800 (dotted line).*** The maximum difference between the cumulative distributions, D, is 0.3408 with a corresponding P value 0.047. Although the null hypothesis that the distributions are similar is rejected ($p < 0.05$), the p value is non-zero. Both distributions show similar distributions across their relative ranges in patch

sizes, but the relative fraction of historical Delta patches at any given size is lower than in the Bay (the historical Delta's patches skew larger). Maximum patch sizes in the historical Bay and the historical Delta were within 1 order of magnitude of each other.

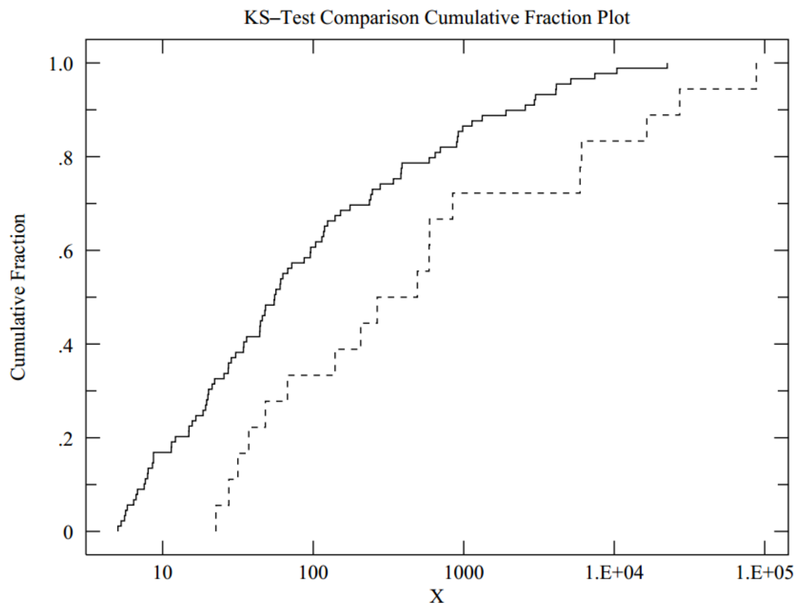
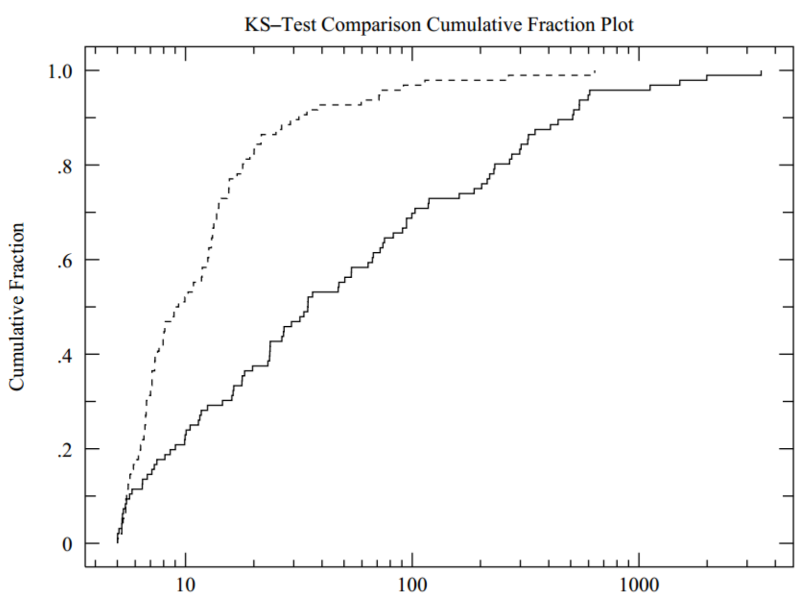


Figure 13. Comparative cumulative fraction of tidal marsh patches (y-axis) across the full range of patch sizes (hectares; x-axis) in the ***Bay ca. 2009 (solid line) and the Delta ca. 2002 (dotted line)***. The maximum difference between the cumulative distributions, D, is 0.4896 with a corresponding P value of 0.000. The null hypothesis that the distributions are similar is rejected ($p < 0.05$). Relative to the modern Bay, the relative fractions of tidal marsh patches in the modern Delta are skewed towards smaller patch sizes. The maximum patch size in the modern Bay is an order of magnitude larger than in the modern Delta.



Supplemental citations

For the sake of readability, text in the main body of the State of the Estuary report is presented without citations. To document the source for uncited material, key sentences from the Tidal Marsh section that are not otherwise reiterated above are copied here and supplemented with their supporting citations.

Page XX, paragraph XX:

> Part of this disparity can be explained by the extensive "subsidence" (sinking) of the Delta's peat islands—while these extensive areas once supported tidal marsh, many now sit 10-25 ft. below sea-level at an elevation that is much too low for tidal marsh vegetation establishment **(Ingebritsen et al. 2000)**.

Page XX, paragraph XX:

> Although the Bay-Delta's tidal marshes have generally kept pace with sea-level rise over the last several thousand years **(see Parker et al. 2011)**, the rate of future sea level rise and available sediment supply will have a major influence on whether they can continue to do so through the end of the century. Modeled scenarios of high sea-level rise rates and low sediment supply, which the latest evidence suggests is a likely trajectory, project that Bay tidal marshes will be unable to keep pace with rising tides and that their total regional extent will decrease; under scenarios of relatively low sea-level rise rates and high sediment supply, the total regional extent is projected to increase **(Stralberg et al. 2011)**. Although similar projections have not been developed for the Delta, its tidal freshwater marshes (which have higher rates of organic matter production) are expected to be less sensitive to reduced sediment availability than the Bay's tidal salt marshes **(Orr et al. 2003)**. Projections that assume marsh accretion can keep pace with estimated rates of sea-level rise in the Delta show an increase in the regional extent of tidal marsh over the next 50 years (assuming no major levee failures; **CDWR 2013, Appendix 3B)**.

Peer Review

This work has benefitted from review by staff at the Delta Science Program and Delta Conservancy, who provided comments on an earlier draft. Additionally, the methods for defining tidal marsh patches were reviewed as part of the development of the *Delta Transformed Report* by a technical review group of 19 scientists (SFEI 2014; referred to in the report as the "Landscape Interpretation Team").

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State of the Estuary Report 2015

Sidebar Source Material

HABITAT – Woody Riparian

Prepared by Sam Safran
San Francisco Estuary Institute

Sam Safran, San Francisco Estuary Institute

Woody riparian habitat in the Delta *(490 words)*

Riparian habitats are transitional areas between terrestrial and aquatic or wetland ecosystems. Here we look at a specific subset of riparian habitat in the Estuary—woody riparian habitat along rivers and streams in the Delta. Despite comprising only a small proportion of the Delta's total extent, these riparian habitats offer a wide range of ecological functions that support a wide range of species. Their complex array of trees, shrubs, and understory plants, for instance, provides a suite of food resources and sites for resident and migratory birds (like Swainson's Hawks and Least Bell's Vireos) to forage, nest, and roost. Riparian habitats can also serve as movement corridors for far-ranging mammals (like coyotes and now-extirpated grizzly bears), as well as smaller mammals (like ringtails). Riparian habitats support aquatic species (like Chinook salmon) by shading a complex shoreline and contributing organic matter to the aquatic environment. Delta riparian habitats support a number of species that are endemic to (only found in) the riparian forests of the Central Valley; these include the riparian brush rabbit, riparian woodrat, and valley elderberry longhorn beetle.

The regional extent of woody riparian habitat within the legal boundary of the Delta is currently 36% of its extent just prior to significant Euro-American landscape modification (ca. 1800). This number alone, however, does not tell the full story of how woody riparian habitats have been altered in the Delta. Historically, woody riparian habitat formed a continuous band along the banks of the Delta's major fluvial channels, which provided a predictable corridor for wildlife to move between terrestrial areas and wetlands. Remaining patches of this habitat are small, scattered, highly fragmented, and mostly found on artificial levees, an arrangement that breaks down the formerly predictable connections between habitat types. In addition, the overall width of existing woody riparian habitats has notably decreased: while more than 50% of the historical habitat (measured by length) was wide enough to be considered "suitable" for the endangered Western Yellow Billed Cuckoo (at 200 m), today only 5-8% of existing habitat currently meets this width threshold. Width is important because the number and level of ecological functions provided by riparian habitats generally increase as the habitat becomes broader.

These measurements of woody riparian habitat in the Delta have not been developed into quantitative indicators for three major reasons. (1) Benchmarks for both regional extent and width have not yet been established for the Delta. (2) The methods used to calculate the length of riparian habitat at various widths are labor intensive and not yet sensitive enough to meaningfully measure the small incremental changes expected in the near future. (3) The best single statistic to evaluate the width of riparian habitat over time has not been determined. Although we report the percent of habitat above 200 m based on information concerning the needs of cuckoos, a better statistic would more conclusively reflect the relationship between riparian habitat width and the diversity and abundance of native riparian wildlife.

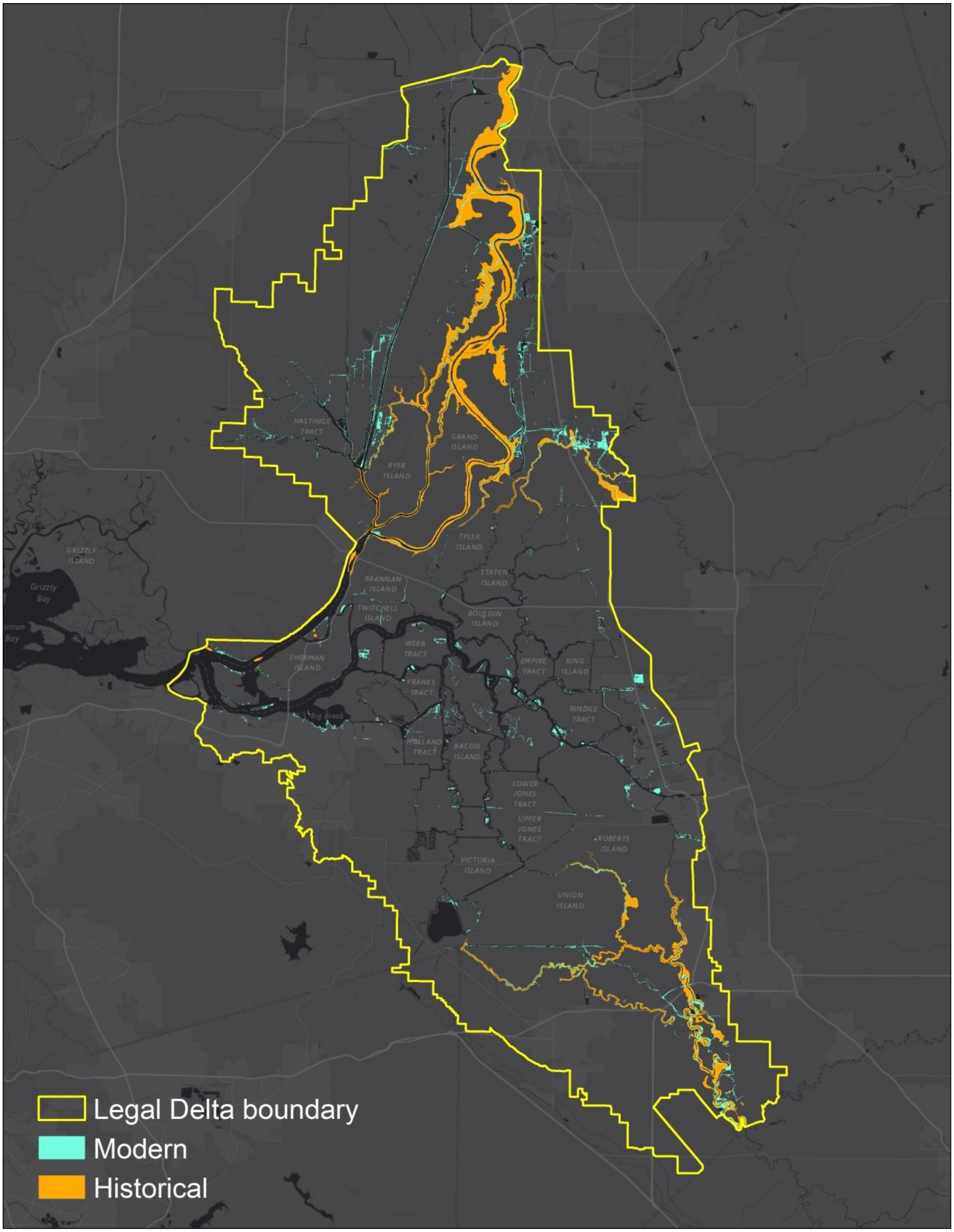

Figure 1. Woody riparian habitat mapped within the legal Delta ca. 1800 ("historical"; orange polygons) and ca.2002 ("modern"; teal polygons). Historically, riparian habitat formed continuous bands along the Delta's major rivers; its modern arrangement looks quite different.

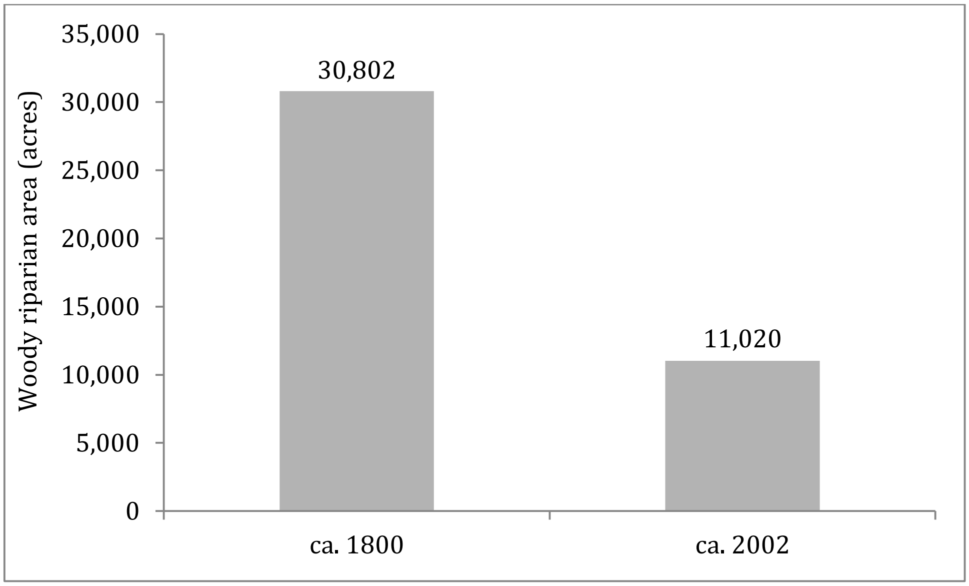

Figure 2. The regional extent of woody riparian habitat in the Delta circa 1800 and 2002. Net area has decreased from 30,802 acres to 11,020 acres (a decrease of 64%).

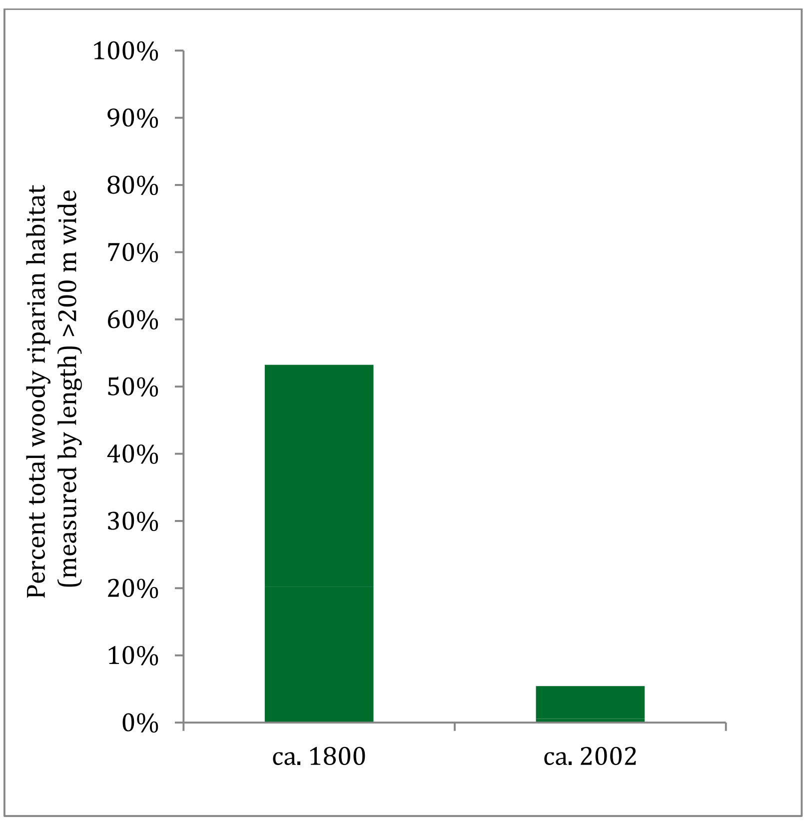

Figure 1. Changes between ca. 1800 and ca. 2002 in the proportion of woody riparian habitat >200 m wide. Habitats >200 m wide are considered "suitable" for Western Yellow-billed Cuckoos (Laymon and Halterman 1989). The proportion of woody riparian habitat in the Delta meeting this width threshold has decreased over time (from 53% to less than 8%).

Sam Safran, San Francisco Estuary Institute

Woody riparian habitat

Background and Rationale

Large bodies of work attempt to define and clarify the term "riparian," which can have a wide range of meanings and applications (see NRC 2002). For the purpose of this work, we consider "riparian habitats" to be the transitional areas between terrestrial and aquatic ecosystems (NRC 2002, RHJV 2004, CVJV 2006). This said, the analyses presented here focus on a narrow subset of riparian habitats, namely *woody riparian habitats associated with streams*. A convenient definition of riparian habitats for our purposes, then, comes from the Riparian Habitat Joint Venture, which says that riparian habitats are "those plant communities supporting woody vegetation found along rivers, creeks and streams" (RHJV.org, accessed 5/1/2015).

Although riparian habitats provide wide range of functions (Gregory et al. 1991, Naiman et al. 2005), the measurements and discussion presented here are tailored towards assessing one function in particular—the provision of habitat for native wildlife. Specifically, the measurements described below—the regional extent and width of woody riparian habitat—seek to help broadly assess the status of woody riparian habitat in the Delta for its ability to support the life histories of native riparian wildlife (including both resident and transient species). This focus is important because, in California alone, over 225 species of birds, mammals, reptiles, and amphibians depend on riparian habitats (CVJV 2006). In the Delta specifically, woody riparian habitats support a number of endemic species, including the riparian brush rabbit, riparian woodrat, and valley elderberry longhorn beetle (CDWR 2013). Among other ecological functions, woody riparian habitats in the Delta provide important vertical structure and food resources for numerous resident and migratory birds to forage, nest, and roost (e.g., Finch 1989) and also support aquatic species (like Chinook salmon) by creating a complex shoreline and contributing organic matter to the aquatic environment (e.g., Opperman 2002). Woody riparian habitats likely served as movement corridors for far-ranging terrestrial mammals in the Delta such as coyotes, mule deer, and grizzly bear (now extirpated), as well as for smaller mammals like gray fox, long-tailed weasels, and ringtails (Whipple et al. 2012, SFEI-ASC 2014; also see Brinson et al. 2002). The ecological functions provided by woody riparian habitat in the Delta are described in greater detail by SFEI-ASC (2014).

Motivation for assessing the health of woody riparian habitat is not only driven by its importance for native wildlife, but also by the decline of the ecological functions it provides associated with habitat loss. California riparian forests were identified by Noss et al. (1995) as an endangered ecosystem (defined as an ecosystem type that has experienced 85-98% decline), and in the Central Valley an estimated 89-99% of riparian habitat has been lost or severely degraded over the last 150 years (Smith 1977, Katibah 1984, Barbour et al. 1991, Noss et al. 1995, Vaghti and Greco 2007). The trends in the larger Central Valley are mirrored in the Delta, where decreases in riparian forest since the early 1800s are estimated at approximately 75% (Whipple et al. 2012, SFEI-ASC 2014). According to DeSante and George (1994), riparian habitat loss may be the most important cause of population declines among songbird species in western North America (as cited in CVJV 2006).

Given these trends, the sidebar in this the main document of this report presents an updated calculation of the **regional extent of woody riparian habitat** in the Delta at two points in time (ca. 1800 and ca. 2002). The regional extent of woody riparian habitat matters because, at the broadest level, we expect the diversity and abundance of riparian wildlife to increase with the area of riparian habitat (but not necessarily linearly). As noted by Noss et al. (1995), quantifying the areal extent of an ecosystem is a critical component of ecosystem conservation. This work builds off of other efforts to calculate the regional extent of woody riparian in the Delta during historical and modern time periods (Whipple et al. 2012, SFEI-ASC 2014), but it is unique for its exclusive use of publically available datasets and application to a standardized study extent (the legal extent of the Delta, which also defines SFEP's Study Area). To lay the groundwork for the measurement's future possible development into a scored indicator, the calculations were carried out with a fully documented and automated workflow. See the "Sources and Methods" section below for additional detail.

We also present calculations of how the **width of woody riparian habitats** has changed over time. The width of riparian areas is important because the number and level of ecological functions provided by riparian habitats generally increase as the habitat becomes broader (Collins et al. 2006). Wide riparian corridors are likely than narrow corridors to provide more and better habitat (RHJV 2004), partially because wide corridors are more likely to have areas of "core" habitat buffered from edge effects and to express a full range of latitudinal gradients (e.g. hydroperiod, moisture, light, disturbance frequency, and vegetation composition). The analysis presented here is derived from a previous analysis of woody riparian habitat width in the Delta carried out by SFEI-ASC (2014). Although the same methods are applied for this version, it utilizes the riparian habitat datasets developed for the new iteration of the regional extent measurements. See the "Sources and Methods" section below for additional detail.

Riparian areas were previously assessed in *The State of San Francisco Bay 2011* (SFEP 2011). In that report, width was analyzed to assess the health of two Bay Area watersheds (the Napa River Watershed in Napa County and the Coyote Creek Watershed in Santa Clara County). The analysis used the Riparian Buffer Decision Tool to define and then measure the width of the maximum extent of riparian areas along creeks. Since the analysis did not seek to measure width for the purpose of assessing any single riparian function, the riparian areas measured for the 2011 report were not necessarily vegetated. The approach utilized in this report, which specifically measures woody riparian habitat for its function as habitat for native wildlife, therefore has a different intent and methodology. The results of these two analyses should not be directly compared.

While the two other analyses of riparian width referenced above measured the total length of woody riparian habitat belonging to various width classes, this analysis of riparian width measures the *percent* of woody riparian habitat (by length) that exceeds 200 m in width. Two components of this calculation are worth discussing, with special attention paid to its possible future development into a scored indicator. First, the motivation for assessing the *percent* of habitat above a particular width is to report the measurement of riparian width distribution as a single value for each measured point in time (as opposed to presenting the full distribution of widths for each year) . This format (one value per year) is the preferred format for scored indicators. Second, the *200 m threshold* is derived from the work of Laymon and Halterman (1989) who (based on occupancy and nest predation rates) defined riparian habitat > 200 m wide as "suitable" for Western Yellow-Billed Cuckoo nesting. The focus on Western Yellow-billed Cuckoos is, in turn, motivated by the species' demanding habitat needs (relative to other riparian bird species). Recommendations put forth in the Riparian Bird Conservation Plan state that, "when considering a suite of species, managers should use the species with largest territory needs (e.g., Western Yellow-billed Cuckoo) to set the minimum patch size requirement" (RHJV 2004). Under this guidance, cuckoos are assumed to be an umbrella species (one whose needs encapsulates those of many co-occurring, less demanding species and whose protection should also offer protection to other species that share the same habitat; Ozaki et al. 2006, Roberge and Per Angelstam 2006).

As noted in the RBCP, however, "quantifying a specific target width of riparian habitat is extremely complex; the effect of riparian width varies by bird species and riparian type and is only one of many variables affecting species occurrence and reproductive success" (RHJV 2004). Notable concerns with the approach presented here (measuring the percent of riparian habitat that is wide enough to support cuckoos) include general concerns with using a two-dimensional measurement to assess the ecological functioning of riparian habitat and, additionally, using any one species to guide this approach. More specific concerns relate to the limited geographic applicability of cuckoos and the applicability of the 200 m threshold. Before any measurement of riparian width can be developed into an indicator to assess the health of riparian habitat in the Estuary, better agreement needs to be reached on an approach/statistic that more conclusively reflects the relationship between the shape of riparian habitat and the diversity/abundance of native riparian wildlife.

Benchmarks

Woody riparian habitat- regional extent

Since the regional extent of woody riparian habitat in the Delta is not yet being developed into an indicator, no benchmark is reported. The lack of an agreed upon benchmark is, in fact, one reason the measurement is not presented as a scored indicator. In the interest of developing this measurement into a full indicator in future iterations of this report, we discuss below some of the options considered/researched for use as a benchmark.

In 2006, the Central Valley Joint Venture established methods for setting conservation objectives for breeding riparian songbirds in the Central Valley (CVJV 2006). Acreage targets for riparian songbirds were determined for each of the eight basins in the Central Valley based on (1) existing and restorable riparian habitat, (2) population estimates and targets, (3) recommended values of nest success, (4) species distribution and richness, and (5) annual rates of riparian restoration. Three main factors prevented simple usage of these targets as benchmarks in this report. First, the SFEP Study Area is spread across (but does not completely cover) multiple basins for which the riparian habitat objectives were developed. The study extents of the two efforts should be rectified before using the CVJV objectives as a benchmark (i.e., can the CVJV methods/targets be applied specifically to the legal Delta/SFEP study extent?). Second, it remains unclear if “riparian habitat” as measured for the CVJV objectives can be directly compared to the “riparian habitat” measured for this work. The CVJV report predated the development of the natural community and vegetation maps utilized for this report, utilized a separate set of data sources, and may therefore be measuring riparian habitat differently. This should be evaluated and clarified before directly incorporating the CVJV habitat objectives as benchmarks. Third, the CVJV benchmarks focus specifically on the needs of breeding riparian songbirds. The regional extent benchmark should, ideally, also consider the needs of other riparian wildlife and plants (or explain why a focus on songbirds is sufficient).

A second option for a regional extent benchmark would be to set it at a certain percentage of the woody riparian habitat acreage that existed prior to significant Euro-American landscape modification. As reported below, 30,802 acres (ca. 1800) of woody riparian habitat were mapped within the extent legal Delta by Whipple et al. (2012). One half of this area would be ~15,000 acres, which, from the simple perspective of proportion of historical extent, would be comparable to the regional goal developed by the Goals Project for the area of tidal marsh in the Bay. This effort established 100,000 acre tidal marsh habitat goal that was approximately *one half* of the historical tidal marsh area. It is important to note, however, that the marsh goal was not simply selected as a percentage of historical acreage, but instead was the product of a science-based public process that sought to evaluate the habitat needs of representative species and to identify changes needed to improve the Bay’s ecological functioning and biodiversity. Future iterations of this report should consult the CVJV and RHJV to determine if an acreage goal developed with these standards is available for the SFEP Study Area.

Woody riparian habitat- width

Like the measurements of riparian habitat regional extent, the measurements of riparian habitat width are not yet being developed into scored indicators. As such, no benchmark for riparian width is reported here. In previous iterations of this report, the benchmark for assessing riparian width was the historical (circa 1800) width distribution of Delta woody riparian habitats. Justification for this method was provided in Appendix D of the *State of San Francisco Bay 2011* (SFEP 2011). If the statistic used in this iteration of the report to compare riparian width distributions over time (the percent of habitat > 200 m wide) is developed into a scored indicator in the future, one possible benchmark would therefore be the historical proportion of woody riparian habitat > 200 m in width. This value is reported below in the results section and referenced in the woody riparian habitat sidebar in the main body of this report.

Data sources and Methods

Woody riparian habitat- regional extent

The regional extents of riparian habitat in the Delta reported here were calculated from regional maps of Delta habitat types and vegetation. Data sources and detailed descriptions of how acreages were derived from these sources are provided below.

The source for GIS data indicating the **historical (ca. 1800) regional extent of woody riparian habitat** in the Delta was SFEI-ASC's *Sacramento-San Joaquin Delta Historical Ecology Investigation* (Whipple et al. 2012). This dataset classifies the historical Delta (ca. 1800 or prior to significant Euro-American landscape modification) into 17 habitat types, the majority of which are based on modern classification systems. The historical habitat types considered to be woody riparian habitat for this *State of the Estuary* report were 'valley foothill riparian' and 'willow riparian scrub/shrub' (Table 1; also see page 6 for additional details on historical habitat crosswalks).

The source for GIS data indicating the **contemporary (ca. 2002) regional extent of woody riparian habitat** in the Delta was the California Depart of Fish and Wildlife Vegetation Classification and Mapping Program's "Vegetation and land use classification and map of the Sacramento-San Joaquin River Delta" (Hickson and Keeler-Wolf 2007). This mapping effort utilized true color 1-foot resolution aerial photography from the spring of 2002 (and from the summer of 2005 in some marginal areas) to classify 129 fine-scale to mid-scale vegetation mapping units within the extent of the legal Delta. Although the dataset is derived from imagery that is now more than a decade old, it is still the most comprehensive (with respect to extent and resolution of vegetation mapping units) available for the Delta. We assigned the dataset a date of ca. 2002 based on the primary date of the source photography. The mapping units considered to be woody riparian habitat for this *State of the Estuary* report are listed in Table 1 (see pages 6-7 for additional details and discussion).

Table 1. Classes considered "woody riparian habitat" for this study, by source.

Delta ca. 1800 (Whipple et al. 2012; "Habitat type")
valley foothill riparian
willow riparian scrub/shrub
Delta ca. 2002 (Hickson & Keeler-Wolf 2007; "MAPUNITS")
Acer negundo - Salix gooddingii
Alnus rhombifolia / Cornus sericea
Alnus rhombifolia / Salix exigua (Rosa californica)
Arroyo Willow (Salix lasiolepis)
Baccharis pilularis / Annual Grasses & Herbs
Black Willow (Salix gooddingii)
Black Willow (Salix gooddingii) - Valley Oak (Quercus lobata) restoration
Blackberry (Rubus discolor)
Box Elder (Acer negundo)
California Wild Rose (Rosa californica)
Coast Live Oak (Quercus agrifolia)
Coyotebush (Baccharis pilularis)
Fremont Cottonwood (Populus fremontii)
Hinds walnut (Juglans hindsii)
Mexican Elderberry (Sambucus mexicana)
Narrow-leaf Willow (Salix exigua)

Oregon Ash (Fraxinus latifolia)
Quercus lobata - Acer negundo
Quercus lobata - Alnus rhombifolia (Salix lasiolepis - Populus fremontii - Quercus agrifolia)
Quercus lobata - Fraxinus latifolia
Quercus lobata / Rosa californica (Rubus discolor - Salix lasiolepis / Carex spp.)
Restoration Sites
Salix exigua - (Salix lasiolepis - Rubus discolor - Rosa californica)
Salix gooddingii - Populus fremontii - (Quercus lobata-Salix exigua-Rubus discolor)
Salix gooddingii - Quercus lobata / Wetland Herbs
Salix gooddingii / Rubus discolor
Salix gooddingii / Wetland Herbs
Salix lasiolepis - Mixed brambles (Rosa californica - Vitis californica - Rubus discolor)
Santa Barbara Sedge (Carex barbarae)
Temporarily or Seasonally Flooded - Deciduous Forests
Tree-of-Heaven (Ailanthus altissima)
Valley Oak (Quercus lobata)
Valley Oak (Quercus lobata) restoration
White Alder (Alnus rhombifolia)
White Alder (Alnus rhombifolia) - Arroyo willow (Salix lasiolepis) restoration

The two historical habitat types—"valley foothill riparian" and "willow riparian scrub/shrub"—considered woody riparian habitat in this report are those explicitly identified by Whipple et al. (2012) as riparian habitat types (this follows the methods used to evaluate historical riparian habitat in the Delta by SFEI-ASC [2014]). Importantly, we do not include historical areas classified as "willow thicket," which, although dominated by woody vegetation, were distinguished by Whipple et al. from the woody riparian habitat categories:

> This category ["willow thicket"] includes broad stands of willow (*Salix* spp.), and occasional larger trees (e.g., cottonwood, *Populus fremontii*) that are usually associated with distributary channel networks at the base of alluvial fans and the margins of freshwater emergent wetlands (see discussion of "willow grove" in Goals Project 1999). Often, willow thickets (historically referred to as "sinks," "sausal," or "swamps") grade into freshwater emergent wetland such that the boundary between the two is indistinct. These areas are differentiated from the willow riparian scrub or shrub class because they share hydroperiod characteristics akin to freshwater emergent wetland, withstanding frequent flooding, prolonged periods of inundation, and saturation at or near the surface. They are also not generally linearly oriented along channels, but are larger and more rounded or ovate in plan form and are associated with distributary systems. They therefore occupy lower-elevation floodplain positions relative to riparian forest habitat types. [Whipple et al. 2012:43].

The list of contemporary map units to consider woody riparian habitat was developed by Whipple et al. (2012), who created a crosswalk to relate the classifications utilized in the CDFW vegetation dataset (Hickson and Keeler-Wolf 2007) to the historical habitat types. The map units listed in Table 1 and considered woody riparian habitat in this analysis are those crosswalked by Whipple et al. (2012) to the historical “valley foothill riparian” and “willow riparian scrub/shrub” habitat types. Importantly, the original crosswalk was developed with the assistance of CDFW staff familiar with both mapping efforts (Todd Keeler-Wolf, personal communication and Daniel Burmester, personal communication, as cited in Whipple et al. 2012). As noted by Whipple et al. (2012), the willow-dominated communities were somewhat challenging to crosswalk. Ultimately, to facilitate comparison with the historical dataset, the team attempted to group the modern map units based on whether the willows were part of a backwater swamp community (willow thicket), the dominant species along channel banks (willow riparian forest, scrub, or shrub), or were part of a forest with oaks (valley foothill riparian forest).

Not all of the contemporary areas classified as woody riparian habitat (based on dominant vegetation alone) are hydrologically connected to an adjacent channel. To distinguish between functionally riparian vegetation and hydrologically disconnected riparian-type vegetation, we applied the methods developed for the Delta Landscapes Project (SFEI-ASC 2014) to roughly distinguish hydrologically connected and disconnected “riparian” habitat. An individual polygon was considered hydrologically connected if it shared an edge with a polygon classified as water (see Table 2). Riparian polygons that were connected to water through other riparian polygons (see Table 1) and/or areas of freshwater emergent wetland (see Table 2) were also considered hydrologically connected. Ultimately, only polygons considered hydrologically connected based on this exercise were counted towards the regional extent of woody riparian habitat (hydrologically disconnected polygons excluded from the regional extent calculations totaled 20% of the ca. 2002 areas identified in Table 1). This step was meant only to approximate hydrologic connectivity at a coarse level—it does not, for example, distinguish between standing water and creeks, nor does it consider topography or flood frequency. These limitations should be resolved before the regional extent of woody riparian habitat becomes a scored indicator.

Table 2. Polygons considered “water” and “freshwater emergent wetland” for the assessment of woody riparian habitat hydrologic connectivity. Riparian areas were only considered “hydrologically connected” if they (A) shared an edge with one of the polygons considered “water” below or (B) were connected to a polygon considered “water” through areas classified as “freshwater emergent wetland.” Only hydrologically connected riparian areas were counted towards the regional extent of woody riparian habitat and included in the assessment of riparian widths.

Map units (Hickson and Keeler-Wolf 2007) considered "water" for analysis of riparian hydrologic connectivity
Algae
Brazilian Waterweed (Egeria - Myriophyllum) Submerged
Floating Primrose (Ludwigia peploides)
Generic Floating Aquatics
Hydrocotyle ranunculoides
Ludwigia peploides
Milfoil - Waterweed (generic submerged aquatics)
Pondweed (Potamogeton sp.)
Shallow flooding with minimal vegetation at time of photography
Tidal mudflats
Water

Water Hyacinth (Eichhornia crassipes)
Map units (Hickson and Keeler-Wolf 2007) considered "freshwater emergent wetland" for analysis of riparian hydrologic connectivity
American Bulrush (Scirpus americanus)
Broad-leaf Cattail (Typha latifolia)
California Bulrush (Scirpus californicus)
Common Reed (Phragmites australis)
Cornus sericea - Salix lasiolepis / (Phragmites australis)
Hard-stem Bulrush (Scirpus acutus)
Mixed Scirpus / Floating Aquatics (Hydrocotyle - Eichhornia) Complex
Mixed Scirpus / Submerged Aquatics (Egeria-Cabomba-Myriophyllum spp.) complex
Mixed Scirpus Mapping Unit
Narrow-leaf Cattail (Typha angustifolia)
Polygonum amphibium
Salix lasiolepis - (Cornus sericea) / Scirpus spp.- (Phragmites australis - Typha spp.) complex unit
Scirpus acutus - (Typha latifolia) - Phragmites australis
Scirpus acutus - Typha angustifolia
Scirpus acutus -Typha latifolia
Scirpus acutus Pure
Scirpus californicus - Eichhornia crassipes
Scirpus californicus - Scirpus acutus
Scirpus spp. in managed wetlands
Smartweed Polygonum spp. - Mixed Forbs
Typha angustifolia - Distichlis spicata

The SFEP study extent (the legal Delta) was enforced by clipping both layers to the legal Delta boundary as digitized by Hickson and Keeler-Wolf (2007). The total acreage of woody riparian habitat was then tabulated separately for each spatial dataset (Delta ca. 1800, Delta ca. 2002) using a Geographic Information System (GIS).

Woody riparian habitat- width

Measurements of woody riparian width reported in this *State of the* Estuary report were adapted from the analysis of woody riparian width by SFEI-ASC (2014) in the report titled *A Delta Transformed: Ecological Functions, Spatial Metrics, and Landscape Change in the Sacramento-San Joaquin Delta* (SFEI-ASC 2014). For detailed methods, please refer to Appendix A of that document (pages 91-93). A very general description of the methods can be found below.

We measured historical (ca. 1800) and modern (ca. 2002) riparian habitat widths in GIS by generating transects at 100 m intervals perpendicular to channel centerlines. These transects were then intersected with adjacent riparian polygons to “trim” the transects at the outer boundaries of the riparian polygons. Transects were also trimmed at artificial levee centerlines to further limit the analysis of riparian width to hydrologically connected habitat. The length of each trimmed transect was then measured to determine the width of riparian habitat along the corresponding 100 m channel segment. Calculated widths correspond to the combined width of woody riparian habitat on both sides of the channel centerline (but exclude the width of the channel itself). To tailor the analysis to the study extent utilized in this report (which differs slightly from the study extent utilized in the SFEI-ASC [2014] report), we simply substituted in the riparian polygons developed for this report's calculation of woody riparian habitat regional extent (described above) before intersecting them with the pre-existing transects.

As detailed by SFEI-ASC (2014), the nature of the historical and modern datasets required two different (although generally similar) methods to determine riparian habitat width. It is important to note that, due to the complicated shape and distribution of woody riparian habitat in the modern Delta, the particular results of the contemporary riparian habitat width calculations are highly contingent on the channel and levee shapefiles utilized in the analysis and on extensive manual GIS work to implement a complicated ruleset. So, while the current methodology is suitable for assessing the relative magnitude of large scale changes that have occurred in the width of riparian habitats areas over the last ~160 years, it is not currently suitable for assessing the scale of changes likely to be occur between smaller intervals of time in the future. This is the primary reason why the riparian habitat analysis described here has not been developed into a scored indicator for this *State of the Estuary* report. To achieve this higher standard, a new, more sensitive, and preferably automated methodology that can measure the length of contemporary riparian habitat at different widths (within a smaller range of range of variability) should be developed.

Results

Woody riparian habitat- regional extent

The extent of woody riparian habitat in the Delta ca. 1800 and ca. 2002 are show below in Figure 2-Figure 1. The current (ca. 2002) regional extent of woody riparian habitat within the legal boundary of the Delta is 36% of its extent just prior to significant Euro-American landscape modification ca. 1800.

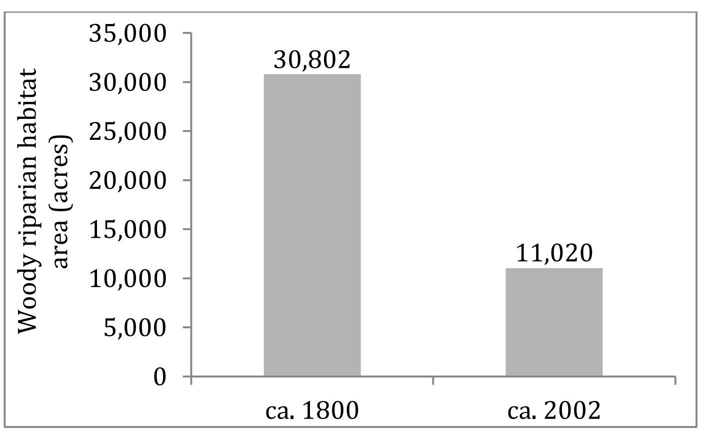

Figure 2. The regional extent of woody riparian habitat in the Delta circa 1800 and 2002. Net area has decreased from 30,802 acres to 11,020 acres (a decrease of 64%).

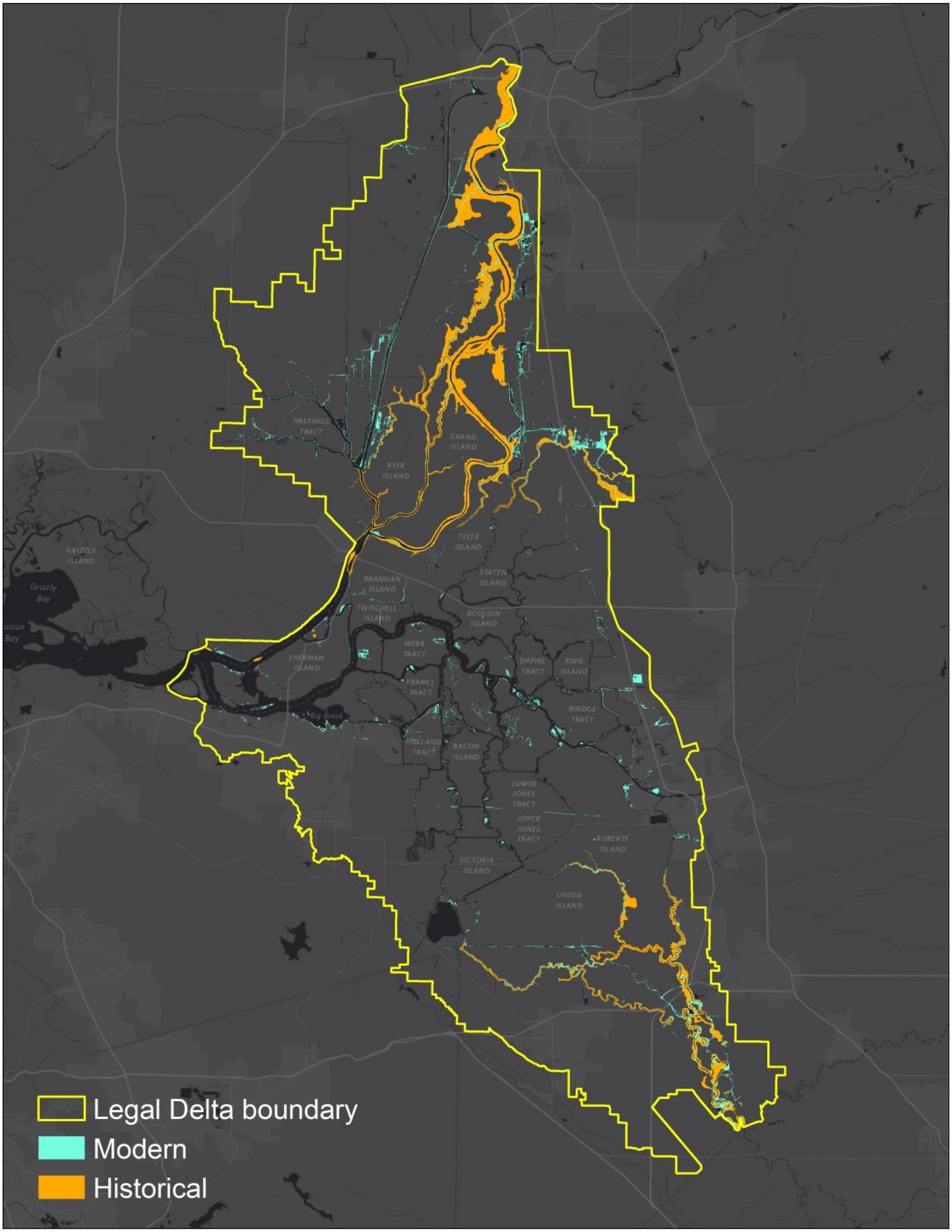

Figure 3. Woody riparian habitat mapped within the legal Delta ca. 1800 ("historical"; orange polygons) and ca.2002 ("modern"; teal polygons).

It is important to note that the source for ca. 1800 woody riparian habitat extent only covers 86% of the legal Delta. As a result, the ca. 1800 estimate likely underestimates the extent of riparian habitat within the full study extent. Of the 14% of the legal Delta unmapped by Whipple et al. 2012, 9% was mapped by the East Contra Costa Historical Ecology Study (Stanford et al. 2011). Although creeks within the legal extent of the Delta in East Contra Costa County are not thought to have supported broad riparian forests during the period preceding Euro-American development of the region, some woody riparian habitat is known to have existed along the lower reaches of Kirker Creek, Lower Marsh Creek, West Antioch Creek, and Willow Creek (Stanford et al. 2011). Together, within the legal Delta, these creeks supported approximately 13.7 km of riparian habitat with at least sparse (>10%) tree cover (as measured from 1939 aerial photographs), likely dominated by blue and valley oaks adapted to intermittent flow conditions. However, without polygonal data depicting the areal extent of this habitat, it could not be counted using the methods outlined above. Estimates for the total area of these riparian habitats are approximately 85 acres (a value calculated by assuming the 13.7 km of riparian habitat had an average width of 25 m, a value in turn estimated from 1939 aerials photographs of Marsh Creek). In the remaining 6% of the legal Delta without detailed historical ecology maps, only the unmapped region along the Stanislaus River near its confluence with the San Joaquin River likely contained significant tracts of woody riparian habitat. Based on coarse mapping done by the Bay Institute, we roughly estimate the unmapped riparian habitat in this region at less than 500 acres.

Estimates of unmapped riparian habitat were not included in the regional extent measurements shown above and are only included here to provide a rough sense for the scale of current data gaps. These gaps should be filled (or the estimates refined) before the regional extent of woody riparian habitat becomes a scored indicator. For a more direct comparison, we also report here the area of woody riparian habitat found only in the areas mutually mapped by the two source datasets: 30,802 acres ca. 1800 and 10,864 acres ca. 2002. Only 156 acres of contemporary woody riparian habitat within the legal Delta are found outside of the historical and modern dataset's mutually mapped area.

The measurement of contemporary (ca. 2002) woody riparian habitat regional extent reported here (11,020 acres) is approximately 5,000 acres less than measurements of riparian habitat reported in the proposed Bay-Delta Conservation Plan (CDWR 2013), which identifies 16,174 acres of the "valley foothill riparian" natural community within the legal Delta boundary. Differences in underlying data sources account for very little of the differences in reported regional extent of riparian habitat (99.6% of the BDCP valley foothill riparian area is derived from the same CDFW source as our analysis). Instead, approximately 55% of the difference is attributable to the exclusion of hydrologically disconnected areas of riparian habitat in this study, which totaled ~2,800 acres. The remaining difference in area is attributable to which vegetation map units (from Hickson and Keeler-Wolf 2007) were considered riparian habitat in each study. In addition to the map units considered woody riparian habitat here (Table 1), the BDCP "valley foothill riparian" natural community includes map units we instead considered "willow thicket" (see page 6), the most extensive being "*Cornus sericea – Salix lasiolepis / (Phragmites australis)"* (823 acres within the legal Delta) and "Salix lasiolepis - (Cornus sericea) / Scirpus spp.- (Phragmites australis - Typha spp.) complex unit" (488 acres). One other extensive map unit not considered woody riparian habitat in our analysis, but included in the BDCP valley foothill riparian class is “Intermittently or Temporarily Flooded Deciduous Shrublands” (537 acres within the legal Delta), which was considered by Whipple et al. (2012) as “Agriculture/ Non-native/ Ruderal” and not riparian habitat.

Updated vegetation and natural community datasets are needed to assess how the extent of woody riparian habitat has changed in the Delta since ca. 2002.

Woody riparian habitat- width

Of the 340 km of woody riparian habitat mapped along channels in the Delta ca. 1800, 181 km (53%) were found to be > 200 m wide (or “suitable” for Western Yellow-Billed Cuckoo nesting according to Laymon and Halterman [1989]). Of the 671 km of woody riparian habitat mapped along channels in the Delta ca. 2002, only 36 km were found to be > 200 m wide (5%). Changes over time in the proportion of woody riparian habitat >200 m wide are shown in Figure 1.

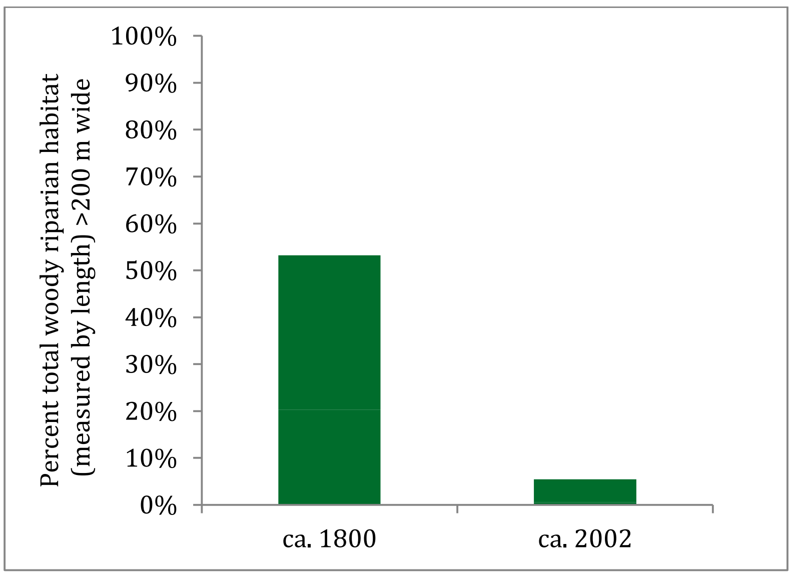

Figure 4. Changes between ca. 1800 and ca. 2002 in the proportion of woody riparian habitat >200 m wide. Habitats >200 m wide are considered “Suitable” for Western Yellow-billed Cuckoos (Laymon and Halterman 1989). The proportion of woody riparian habitat in the Delta meeting this width threshold has decreased over time.

It is important to note that these numbers are highly sensitive to the minimum width of riparian habitat mapped by each dataset. Although the contemporary dataset (Hickson and Keeler-Wolf 2007) mapped distinct linear vegetation polygons as narrow as 10 m, the narrowest riparian areas mapped by Whipple et al. (2012) were 15 m wide. If we force both datasets to have equivalent minimum mapping units by excluding areas mapped as less than 15 m wide, we find the contemporary percentage of woody riparian habitat length >200 m wide increases from 5% to 8% (as expected, the historical dataset is essentially unaffected) and the total length of contemporary riparian habitat decreases from 671 km to 477 km (in other words, by length, 29% of the mapped contemporary woody riparian habitat is <15 m). Because of this, in the main body of this report, we report a range of values for the contemporary proportion of woody riparian habitat >200 m wide (5-8%). Ways to incorporate uncertainty associated with differences in minimum mapping units should be determined before developing the width measurements into a scored indicator.

Two important findings are indicated by the measurements of the length of riparian habitat by width. First, although the total area of woody riparian habitat has decreased (see previous section on regional extent), its length has increased. As evident in Figure 1, many kilometers of woody riparian habitat line artificial levees today in locations where waterways historically met freshwater emergent wetland (most notable in the central Delta). Second, although the total length of woody riparian habitat in the Delta has increased, a much smaller proportion of the total length is now wide enough to support certain ecological functions (like cuckoo nesting). This finding is just one indication of how woody riparian habitats have changed in ways not reflected by analyses of regional extent alone. Other reports (Whipple et al. 2012, SFEI-ASC 2014), for example, have quantified an increase in both the absolute area and relative proportion of riparian scrub (with a corresponding net decrease in the absolute area and relative proportion of riparian forest). Changes in hydroperiod, disturbance regime, and species composition are other changes not directly measured by these results that are expected to have impacts on riparian habitat for native wildlife.

Peer Review

The methods used to calculate woody riparian habitat extent and width presented here were previously reviewed as part of the development of the *Delta Transformed Report* by a technical review group of 19 scientists (SFEI 2014; referred to in the report as the "Landscape Interpretation Team"). Although drafts of this document were provided to Delta Conservancy and Delta Science Program staff, the iteration of the work presented here was not independently reviewed for the *State of the Estuary* report. The work has benefitted from the *State of the Estuary* Advisory Committee, who provided feedback on the methods and initial results.

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Summary

WILDLIFE – Benthic Invertebrates

Prepared by Elizabeth Wells
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1. Brief description of indicator and benchmark

Table 1.1

Attribute	Indicator	Benchmarks
Benthic invertebrates	**1.** Diversity: number of native species **2.** Community composition: percent of all species that are native **3.** Community composition: percent of all individuals that are native	• Benchmark for native diversity is 1981-86. Good ≥ 1981-86 average, "Poor" ≤ historical average -1 standard deviation. • Benchmark for community composition (both by species and individuals) for "Good" ≥ 75% native, "Poor" ≤ 50% native.

2. Indicator status and trend measurements

Table 1.2

Indicator	Status	Trend	Details
1. Benthic invertebrate diversity: number of native species	Good	No change	All sites had "Good" native species diversity and were not significantly different from the historical period.
2. Benthic invertebrate community composition: native/nonnative species	Mixed	No change or deteriorating	The Delta site (D28A) was "Good," the confluence site (D4) was "Fair" and the Suisun Bay site (D7) was "Poor". D7 has significantly decreased in proportion of native species since the historical period.
3. Benthic invertebrate community composition: native/nonnative individuals	Fair or poor	No change or improving	The Delta site (D28A) was "Fair", with a significant increase since historical times. The confluence site (D4) was "Fair" and the Suisun Bay site (D7) was "Poor"; neither had a significant trend in the proportion of native individuals

3. Brief write-up of scientific interpretation

The benthic invertebrate indicators give a summary of the status and trends of the community composition and native species diversity of the benthic (i.e. bottom-dwelling) community of the upper part of the San Francisco Estuary. The data used to construct these indicators is EMP benthic monitoring data from the three longest-sampled sites (D28A in the Delta, D4 at the confluence, and D7 in Suisun Bay) from 1981-2013. The three sites were analyzed independently because of the large differences in benthic communities between regions (Peterson and Vayssieres 2010, Thompson 2013). The data analyzed for the indicators comes from benthic grab samples, which have been collected, identified to species, and counted in the same way for the whole period of the monitoring program.

Benthic invertebrate indicators are important because the benthic community is a key part of estuary foodweb dynamics and nutrient cycling, and because benthic species are a classic bioindicator of estuary health (Gibson *et al.* 2000). The filter and deposit feeders of the San

Francisco Estuary and Sacramento-San Joaquin Delta have a large effect on how phytoplankton either continues into the fish food supply, or is diverted into the benthic community, with potentially large community effects (Alpine and Cloern 1992; Jassby 2008; Kimmerer and Thompson 2014). Benthic invertebrates are more localized indicators of estuary health than plankton or fish, and are sufficiently sensitive and have quick enough life cycles that changes in benthic community patterns can indicate large recent changes in nutrient loading, toxic substances, or sedimentation patterns (Gibson *et al.* 2000).

We chose our three indicators because they are unambiguous indicators of environmental health. Loss of native diversity has been associated with ecosystems that are less productive, have less ecological function and provide fewer ecological services, and are less resilient in the face of stresses (Worm *et al.* 2006, Cardinale *et al.* 2012). Similarly, ecosystems that have higher proportions of non-native species and individuals are characterized by lower environmental health and services than more intact ecosystems, and an increase in non-native species may lead to lower native biodiversity (Pimentel *et al.* 2005, Butchart *et al.* 2010, but see Gurevich and Padilla 2004).

The benchmark for native diversity and community composition was based on the historical period of 1981-86, chosen because 1981 was the earliest year-round monitoring at all sites, and the 1986-87 invasion of the Asian overbite clam (*Potamocorbula amurensis*), along with several other non-native species at roughly the same time, marked a drastic community shift at D4 and D7. Current (2009-2013) native diversity that was equal to or higher than the historical average was counted as “Good”, and the upper boundary for “Poor” native diversity was set at one standard deviation below the historical average, with “Fair” all values between these two. For community composition, the upper boundary of the “Poor” status was set at 50% native for both species and individuals (following the example of the 2011 State of the Bay Fish indicators), and the lower boundary of “Good” was set at or above 75% native in order to give equally sized intervals to “Good” and “Fair”. Trends for all three indicators were determined by determining whether the current status differed significantly from the historical benchmarks.

The status and trends for the various benthic indicators are variable but give a generally worrying overall picture. While native diversity has remained good, and has remained steady compared with 1981-86 historical levels (Figure 1), a large proportion of the community’s species and individuals are now non-native species at some sites (Figure 2). This is especially true at site D7 in Suisun Bay, a major site of *Potamocorbula amurensis* invasion, and where over the last five years native species were 50% of the species diversity but native individuals were only 5% of the total count. The current community composition was considerably better at D4 in the confluence (74% of species and 74% of individuals were native) and at D28A in the Delta (88% of species and 67 % of individuals were native.

The patterns we see in the benthic invertebrate indicators are important because they are a clear indication that the estuary and Delta are not in a pristine state, and are extremely unlikely to return to anything like a pristine state. The San Francisco Estuary is one of the most invaded in the world (Cohen and Carlton 1998, Ruiz *et al.* 2011), and with the addition of many non-native species we can expect changes to ecological services and functions such as food web dynamics that support valued fish, nutrient cycling, and water filtration that removes sediment and contaminants. We do not know exactly how the current benthic community functions differently from the historical one: many of the non-native species were introduced long before regular monitoring. While it is heartening that there has been no large net loss of native diversity at the species level, management of species such as salmonids and smelt should take into account the potential changes in benthic-pelagic food web interaction compared with historical conditions, as assumptions of similar function in the current and historical benthic community may be deeply flawed (Sommer *et al.* 2007).

Figure 1. Indicator 1: Native diversity

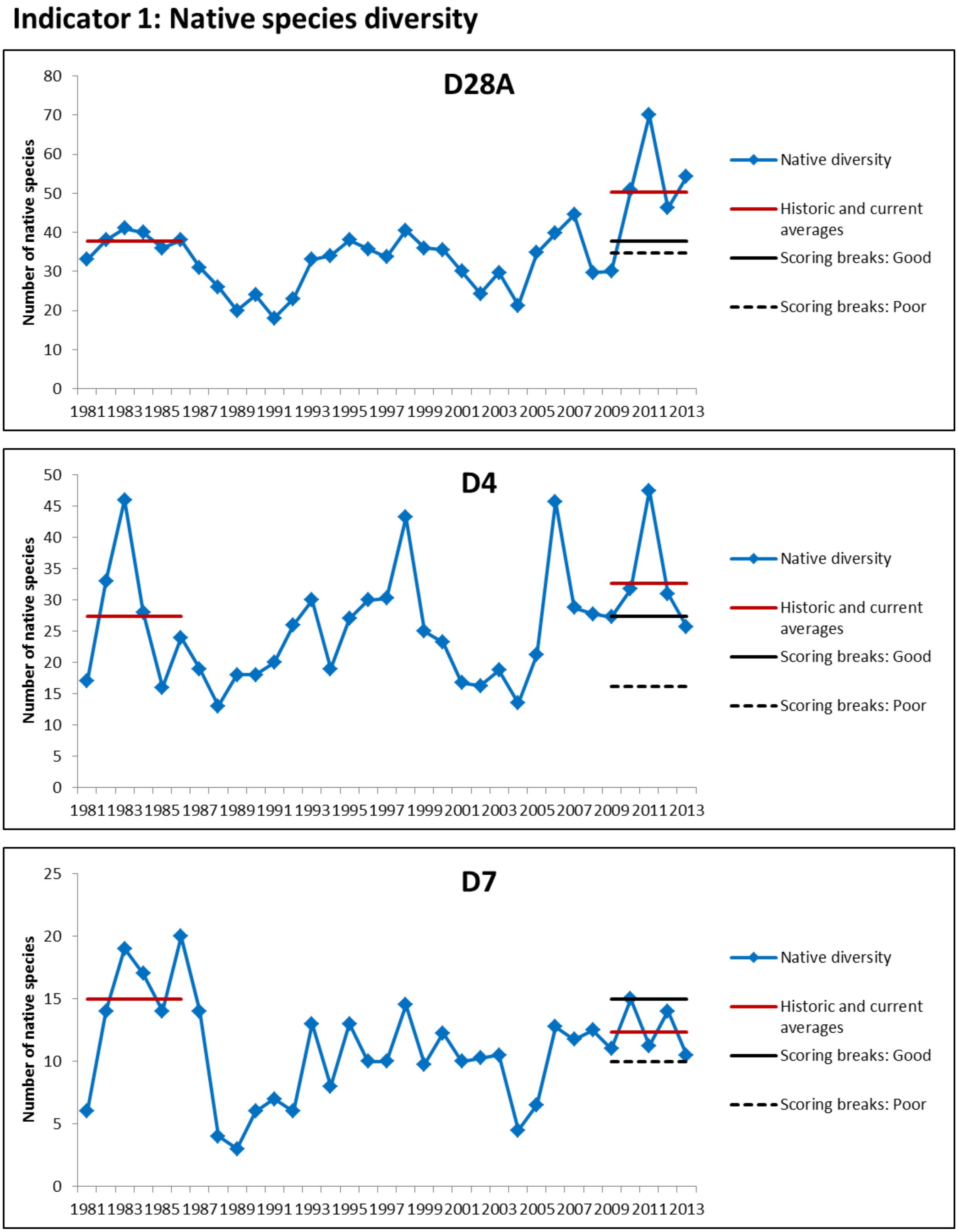
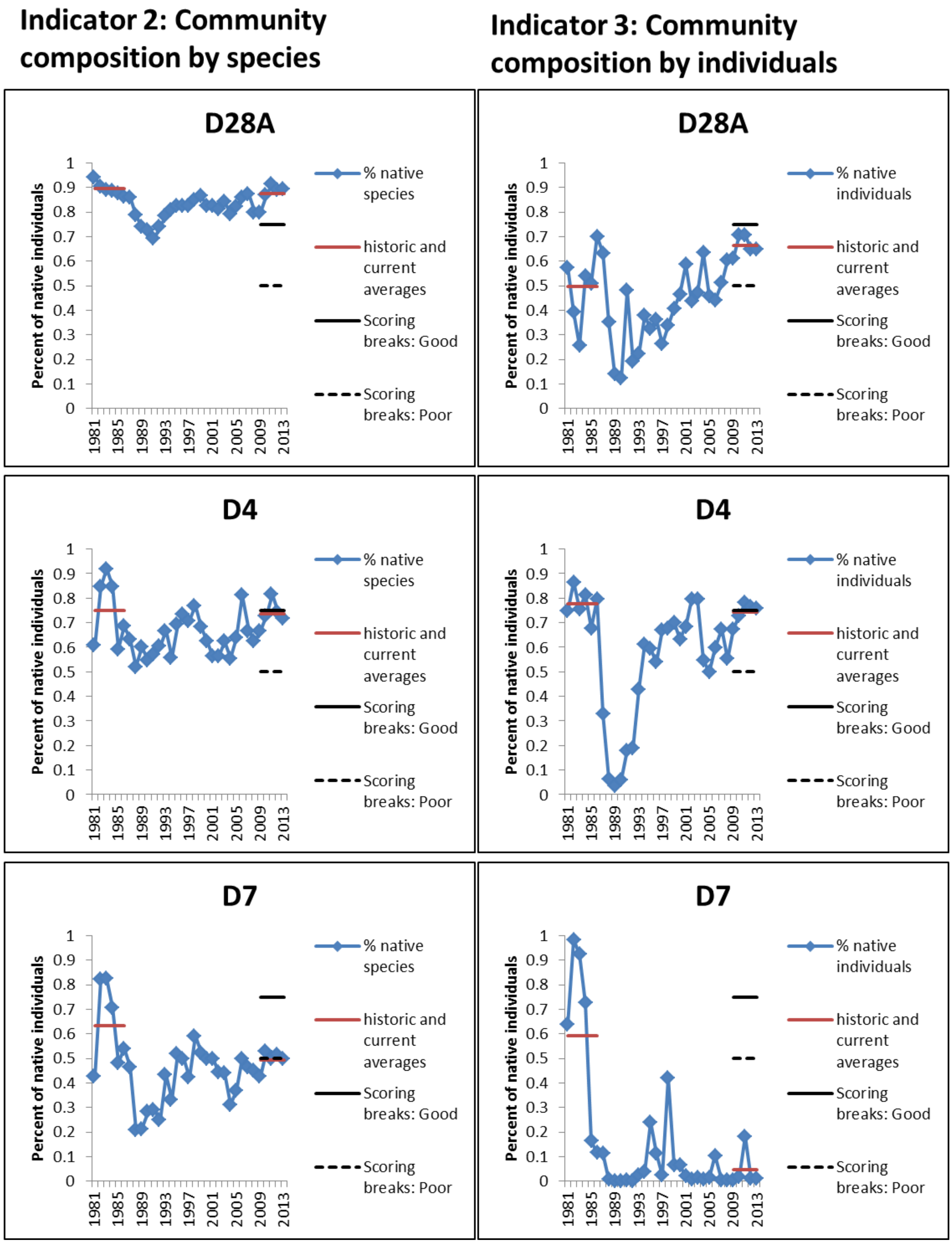


Figure 2. Indicators 2 and 3: Community composition by species and by individuals



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Technical Appendix

WILDLIFE – Benthic Invertebrates

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Benthic Invertebrates Technical Appendix

I. Background and Rationale

Benthic (bottom-dwelling) invertebrate indicators are an important part of assessing estuary health because the benthic community is a key part of estuary foodweb dynamics and nutrient cycling, and because benthic species are classic bioindicators (Gibson *et al*. 2000, Holt and Miller 2010). The filter and deposit feeders of the San Francisco Estuary and Sacramento-San Joaquin Delta have a large effect on how phytoplankton either continues into the fish food supply, or is diverted into the benthic community, with potentially large community effects (Alpine and Cloern 1992; Jassby 2008; Kimmerer and Thompson 2014). San Francisco Bay and the Delta comprise one of the most invaded estuaries in the world (Cohen and Carlton 1998, Ruiz *et al*. 2011) as well as having experienced major changes and degradation in the forms of altered water flow, channelization and hardening, pollution, agriculture, and development. Benthic invertebrates are more localized indicators of estuary health than plankton or fish, and are sufficiently sensitive and have short enough life cycles that changes in benthic community patterns can indicate large recent changes in nutrient loading, toxic substances, or sedimentation patterns (Gibson *et al*. 2000, Gomez Gesteira and Dauvin 2000).

The benthic invertebrate indicators give a summary of the status and trends of the native species diversity and community composition of the benthic community in the Sacramento – San Joaquin Delta and the upper part of the San Francisco Estuary. One indicator measures the native species diversity, or "how many species?" are found in the estuary. Two indicators assess the community composition, or "what kinds of species?", comparing the number of native vs. non-native species and individuals.

Because the San Francisco Estuary and Sacramento – San Joaquin Delta covers conditions from marine to completely fresh water, there are distinct groupings of invertebrate communities along the salinity gradient sites (Peterson and Vayssieres 2010, Thompson 2013). These completely distinct communities displayed different patterns and cannot be compared directly, so all indicators were analyzed separately for each of three long-term monitoring sites: D28A (on Old River in the south Delta), D4 (at the confluence of the Sacramento and San Joaquin Rivers), and D7 (in Suisun Bay).

III. Data Source

The data used to construct these indicators is EMP benthic monitoring data from 1981-2013, which was derived from analysis of benthic grab samples. A standard-sized PONAR grab sampler (152mm x 152mm, or 6 inches x 6 inches) was used to take 3 replicate grabs at each site (1981-1995), which was increased to 4 replicate grabs at each site in later years (1996-present). The samples were sieved over an 0.5mm sieve in the field, preserved in 10% formalin and transferred to 70% ethanol, and were then identified to species and enumerated by Hydrozoology. For further details about the sampling protocols, please see the California Department of Water Resources page on benthic sampling methods: http://www.water.ca.gov/bdma/meta/benthic.cfm

The stations used are the three longest continuously sampled sites in the EMP benthic monitoring program. While seven other sites are currently monitored, and several others have been monitored historically, including them in this analysis proved difficult statistically due to the varying periods of study and conclusions from the analysis could not be interpreted unambiguously. The sites used for this analysis are listed in Table 1 and are placed on a map in Figure 1.

III. Benchmarks

The benchmarks for all three indicators were based on a historical period of 1981-86. While monitoring began in 1975 at some sites, 1981 was the earliest year-round monitoring at all sites, and the 1986-87 invasion of the Asian overbite clam (*Potamocorbula amurensis*), along with several other non-native species at roughly the same time, marked a drastic community shift at D4 and D7.

More details about indicator calculation and analysis can be found below in discussion of the individual indicators' Methods sections.

IV. Peer Review

Peer review for the benthic invertebrate indicators was performed in several different venues. The first line of consultation and revision was fellow State of the Estuary contributors April Hennessey and Hildie Spautz (both from the California Department of Fish and Wildlife), as well as Jon Rosenfield and Alison Stover-Weber (both from The Bay Institute). Drafts of the indicator ideas, calculations, and results were presented at State of the Estuary meetings as well as at several California Estuary Monitoring Workgroup meetings, and were discussed in meetings of the the Living Resources section of the California Estuary Monitoring Workgroup. Further discussion on the indicator benchmarks and scoring was conducted with Letitia Grenier and Amy Richey (both of the San Francisco Estuary Institute), as well as with April Hennessey and Hildie Spautz.

In addition, Karen Gerhts (Department of Water Resources) and Jan Thompson (USGS), who have both worked with the EMP benthic data and familiar with the dataset's scope and limitations, were consulted about the indicators' calculation and interpretation. They reviewed drafts of the summary and technical appendix, which were amended accorded to their recommendations.

V. Indicator Rationales, Methods, and Results

A. Indicator 1: Native Diversity

1. Rationale

Diversity is one of the key indicators of a community's health, and tends to be highest in systems that have not experienced as much human alteration and degradation (Butchart *et al*. 2010, Cardinale *et al*. 2012). Native diversity in particular is an important component of measuring ecosystem health, since endemic or rare native species with narrow environmental tolerances and specific developmental or trophic requirements may be lost due to habitat degradation.

In the course of 40 years of monitoring at all of its current and historic sites, the EMP benthic program has identified approximately[1] 397 native species to date (although note that three known cryptogenic species were counted as "native" for this analysis). These species span a salinity gradient that extends from completely fresh water in the Delta to near-marine conditions in the summers of very dry water years in San Pablo Bay. This high benthic invertebrate diversity provides a responsive tool to measure diversity responses to ecosystem health over a relatively long period of record.

2. Methods and Calculations

The native diversity indicator was measured as simple species richness at each site in each year. We had to calculate native diversity differently for the years 1981-1995 (when we took three replicate benthic grabs) with the years 1996-2013 (when we took four replicate benthic grabs). We calculated native diversity for 1981-1995 as:

Equation 1

$$1981 - 1995 \text{ native diversity} = \text{\# of native species identified in a calendar year}$$

For 1981 -1995, data from all three replicate benthic grabs was used, and the native diversity used for calculation of the indicator status and trend was the same as the total number of native species observed in those grabs.

However, for 1996 we used an effort-adjusted measurement of native diversity since an increased number of sampling events increases the total diversity count (assuming that all species were not completely detected by three replicate grabs). Since we had four replicate grabs

[1] The exact number of species is constantly in flux by 5-10 species at any time, as unidentified specimens counted as separate species are determined by taxonomists either to be truly new species or to belong to previously identified species.

(identity numbers were randomly assigned), the calculation process was to repeatedly subsample with replacement:

1. Exclude all data from replicate grab #1 for all sampling events and calculate total native diversity for that site in that year. This diversity = A.
2. Exclude all data from replicate grab #2 for all sampling events and calculate total native diversity for that site in that year. This diversity = B.
3. Exclude all data from replicate grab #3 for all sampling events and calculate total native diversity for that site in that year. This diversity = C.
4. Exclude all data from replicate grab #4 for all sampling events and calculate total native diversity for that site in that year. This diversity = D.

Equation 2

$$1996 - 2013 \text{ native diversity} = \text{Average of } (\mathrm{A}, \mathrm{B}, \mathrm{C}, \mathrm{D})$$

This replicate-adjusted native diversity provided a metric of native diversity that did not inflate total diversity from the increased sampling effort of later years, and was comparable to the 1981-1995 native diversity.

It should also be noted that we took a conservative approach to native vs. non-native designation. Only species that had been specifically denoted as non-native in the database were counted as such, and cryptogenic species or those with uncertain status were counted as native. The findings of this indicator, and indeed all three benthic invertebrate indicators, may therefore be slightly more optimistic with regards to native species presence and abundance than if cryptogenic species were examined separately.

Including the cryptogenic species as natives was done for logistical reasons, because we wanted to count the cryptogenic species in some way, and creating their own category for either indicator or for Indicators 2 and 3 was not feasible. Two cryptogenic species (*Grandofoxus grandis*, an amphipod, and *Macoma* sp. A, a clam) were each seen only a handful of times, in low numbers, while the third (*Macoma petalum*, a clam seen in consistent numbers across the monitoring period in Suisun Bay) was likely a trans-Arctic invasion of Atlantic *Macoma balthica* in the Early Pliocene (Nikula *et al*. 2007). The majority of the "cryptogenic" individuals were therefore more similar to natives than non-natives, and were grouped accordingly.

To find the current status of native diversity, we found the average of the last five years (2009-2013) of native diversity at each site and compared it to the benchmark average diversity of the historic period (1981-86). Native diversity that was equal to or higher than the historical average was counted as "Good", and the upper boundary for "Poor" native diversity was set at one standard deviation below the historical average, with "Fair" all values between these two (Table 2).

Trends in community composition by species were identified by performing a two-sided two-sample t-test comparing the years in the benchmark historic period to the years of the current period. A significant result ($p<0.05$) was counted as a significant trend in native diversity up or down from historic levels. We used this approach rather than a linear regresssion of diversity on year because diversity is not expected to behave in a linear manner and does not meet the assumptions of linear regressions. For example, decreases in biodiversity may dramatically decrease following a catastrophic disturbance, which would be better assessed with to a change-point or step analysis than with a linear regression. A t-test such as the one we used still captures the signal of change, while not assuming a linear rate of change. In addition, each year is not independent of other years, a requirement for linear regression's independent variable; a species' persistence in each year (and thus total biodiversity) is not independent of whether it was found at a site in the previous year.

Results

At all sites, the native diversity is currently at "Good", with no significant trends up or down (Figure 2). The current (2009-2013) native diversity average at D28A (Old River, in the south Delta) was 50.25 species, which was not statistically different from the 1981-86 average of 37.7 species (Figure 2). (Note that 50.25 species is the effort-adjusted species richness; current observed species diversity using all four replicate grabs was 54.2 species). The current native diversity average at D4 (confluence of Sacramento and San Joaquin Rivers) is 32.7 species (effort-adjusted; observed diversity was 36.2 species), which did not differ significantly from the 1981-86 average of 27.3 species. The current native diversity average at D7 (Suisun Bay) is 12.4 species (effort-adjusted; observed diversity was 14 species), which did not differ significantly from the 1981-86 average of 15 species.

The steady maintenance of native diversity at a level close to or slightly above historical levels is an encouraging sign of health in the benthic invertebrate community. Loss of biodiversity is often cited as a cause or correlation with decrease in environmental services and functions (Butchart *et al*. 2010, Cardinale *et al*. 2012). We can conclude that the benthic community has not responded to the stresses and disturbances of the last 30 years with a crisis of native biodiversity loss.

One reason for confidence in these results is that there have been no changes in identification methods, which have been performed in the same way by the same person the same since throughout the length of the monitoring effort. Nor has any real loss of biodiversity been disguised by changes in taxonomic classification, e.g. one original species now identified as two or more; very few of those taxonomic splits have happened with the species in this dataset (Wayne Fields of Hydrozoology, personal communication).

One caveat in interpreting these results is that even though over thirty years of monitoring is often considered to be a respectably long-term dataset, the start of the EMP benthic monitoring used for this analysis happened centuries after the beginning of human influence in the region There may have been much earlier losses to native biodiversity that we do not see in this analysis because of our shifted baseline of comparison. Indeed, considering the scale of alterations to water flow and sediment loading from agriculture, mining, and development that affected the Delta, it would be surprising if there were not early losses to the native diversity. We cannot estimate the size of any earlier decreases in native diversity, but this indicator at least reassures us that decreases are not currently ongoing.

B. Indicator 2: Community composition by species
Indicator 3: Community composition by individuals

1. Rationale

The relative abundances of native and non-native species and individuals are another key component of ecosystem health. Since non-native species may not have the same relationships with other species in the community as natives, the addition of non-native species (and in some cases, their replacement of native species) may affect food web dynamics and overall ecosystem function. While non-native species may increase the total diversity, they are associated with ecosystem disturbance and may actually increase environmental degradation (MacDougall and Turkington 2005, Didham *et al*. 2007), which indicate lower overall ecosystem health.

Community composition by species (Indicator 2) is similar to native diversity (Indicator 1), which both look at status and trends of native species numbers. The difference is that Indicator 2 explictly examines native species diversity in the context of all diversity in each year, which is important since a majority of the species found may not be native to the area, but should be considered when assessing how ecosystem function may have changed.

In addition to examining the relative proportions of native and non-native species, looking at proportions of native and and non-native individuals gives a more nuanced perspective of community composition than either alone. We present two indicators: community composition by species (Indicator 2) measures what proportion of total species diversity consists of native species, while community composition by individuals (Indicator 3) measures what proportion of all the individual organisms belong to native species. Each indicator is analyzed separately for each long-term monitoring site, since the three sites display very different patterns.

2. Methods and Calculations

Note that by "native" species, we are counting all species not designated as "introduced" as native, including cryptogenic species. For the reasoning behind this decision, please see "Methods and Calculations" for Indicator 1.

Community composition by species was calculated as the percentage of native species in the total annual species diversity in each region, for each year. The percentage of non-native species could of course be easily calculated as 100%-percentage of native species.

Equation 3

$$\text{Annual community composition by species } = \frac{\text{\# native species}}{\text{\# of all species}} \times 100$$

Community composition by individuals was calculated as the total number of native individuals as a proportion of all individuals collected, within each region for each year.

Equation 4

$$\text{Annual community composition by individuals } = \frac{\text{\# native individuals}}{\text{\# of all individuals}} \times 100$$

Current (2009-2013) community composition was found in the same way for both species and individuals. The upper boundary of the “Poor” status was set at 50% native for both species and individuals, since an ecosystem with under 50% native species or individuals is generally considered to be in poor ecological health (per 2011 State of the Bay Fish indicators). The lower boundary of “Good” was set at or above 75% native in order to give equally sized intervals to “Good” and “Fair” (Table 2).

Trends in community composition by species were identified by performing a two-sample t-test comparing the years in the benchmark historic period to the years of the current period. A significant result ($p<0.05$) was counted as a significant trend in native diversity up or down from historic levels.

3. Results

The current (2009-2013) community composition by species of site D28A (Old River, in the south Delta) has a status of “Good” with an average of 87.5% native species, with no significant trend from its historic (1981-86) average of 89.5% native species (Figure 3). The community composition by individuals at D28A was “Fair” with 66.5% native individuals, which was was a significant upward trend increase from its historic average of 49.6% native individuals. Most of the numerically dominant species at D28A have remained constant in identity while fluctuating in abundance through the monitoring record. The difference observed between the historic and current period appears to be due largely to a decrease in density in the non-native clam *Corbicula fluminea* from historic highs, and a recent sharp increase of the native amphipod *Americorophium spinicorne*.

At D4 (confluence of Sacramento and San Joaquin Rivers), the current community composition by species is 73.5% native, with a status of "Fair" and no significant difference from the historic community composition of 75% native species. D4 is currently composed of 74.1% native individuals, with a status of "Fair" and not different from its historic composition of 77.6% native individuals. While various species have fluctuated in abundance throughout the period of monitoring, the native amphipod *Americorophium spinicorne* and the native oligochaete worm *Varichaetodrilus angustipenis* have consistently made up much of the total abundance of the community at D4 through time, both in the historic and current time periods.

The current community composition by species at D7 (Suisun Bay) is just under the line for "Poor" at 49.5% native species, which is not significanlty lower than the historic mean of 63.5% native species. The community composition by individuals at D7 is well into the "Poor" category at 4.6% native individuals, a sharp downward trend from the historic average of 59.3% native individuals. The change to a high proportion of non-native individuals is due in large part to the 1986 arrival of the non-native clam *Potamocorbula amurensis* as well as the non-native amphipod *Corophium alienense*, whose rise in numbers at D7 can be dated to the late 1980s and which is especially dominant in dry water years. These two species are by far the most numerically dominant species in the estuary, while formerly dominant native species like the arthropod *Americorophium stimsoni* and the oligochaete worm *Limnodrilus hoffmeisteri* have both declined since the historic period. These dominant non-natives have added massively to the number of non-native individuals, and may have also replaced some of the native individuals through competition for space or other resources.

For many of the species in the benthic community, too little is known about their natural history (either observationally or experimentally) to compare the role of non-native species with the roles of native species.The community composition indicators are therefore not necessarily an indication of lower ecological health in all systems. However, in the Delta, the advent of non-natives, especially clams has been identified as a contributing factor in the Pelagic Organism Decline (Sommer *et al*. 2007), and the dramatic changes seen, particularly in the proportion of native and non-native individuals at site D7, are an effective indicator of major shifts in the community that has had effects on the Delta and Suisun Bay food webs.

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Table 1. Sites used for benthic invertebrate data source

Region	Site	Latitude and longitude	Period of sampling
Suisun Bay	D7	38.1171292 N, 122.0395539 W	1981-present
Delta	D28A	37.9701652N, 121.5741188 W	1981-present
Confluence	D4	38.0581151 N, 121.8193499 W	1981-present

Table 2. Benchmarks and scoring for benthic invertebrate indicators

Indicator	Quantitative reference condition	Evaluation and Interpretation
1. Native diversity	≥ historical period average	"Good"
	< historical period average and > historical period average – 1 standard deviation	"Fair"
	≤ historical period average – 1 standard deviation	"Poor"
2. Community composition (species)	≥ 75 % native species	"Good"
	<75% and >50% native species	"Fair"
	≤50% native species	"Poor"
3. Community composition (individuals)	≥ 75 % native individuals	"Good"
	<75% and >50% native individuals	"Fair"
	≤50% native individuals	"Poor"

Figure 1. Map of benthic monitoring sites used for State of the Estuary analysis

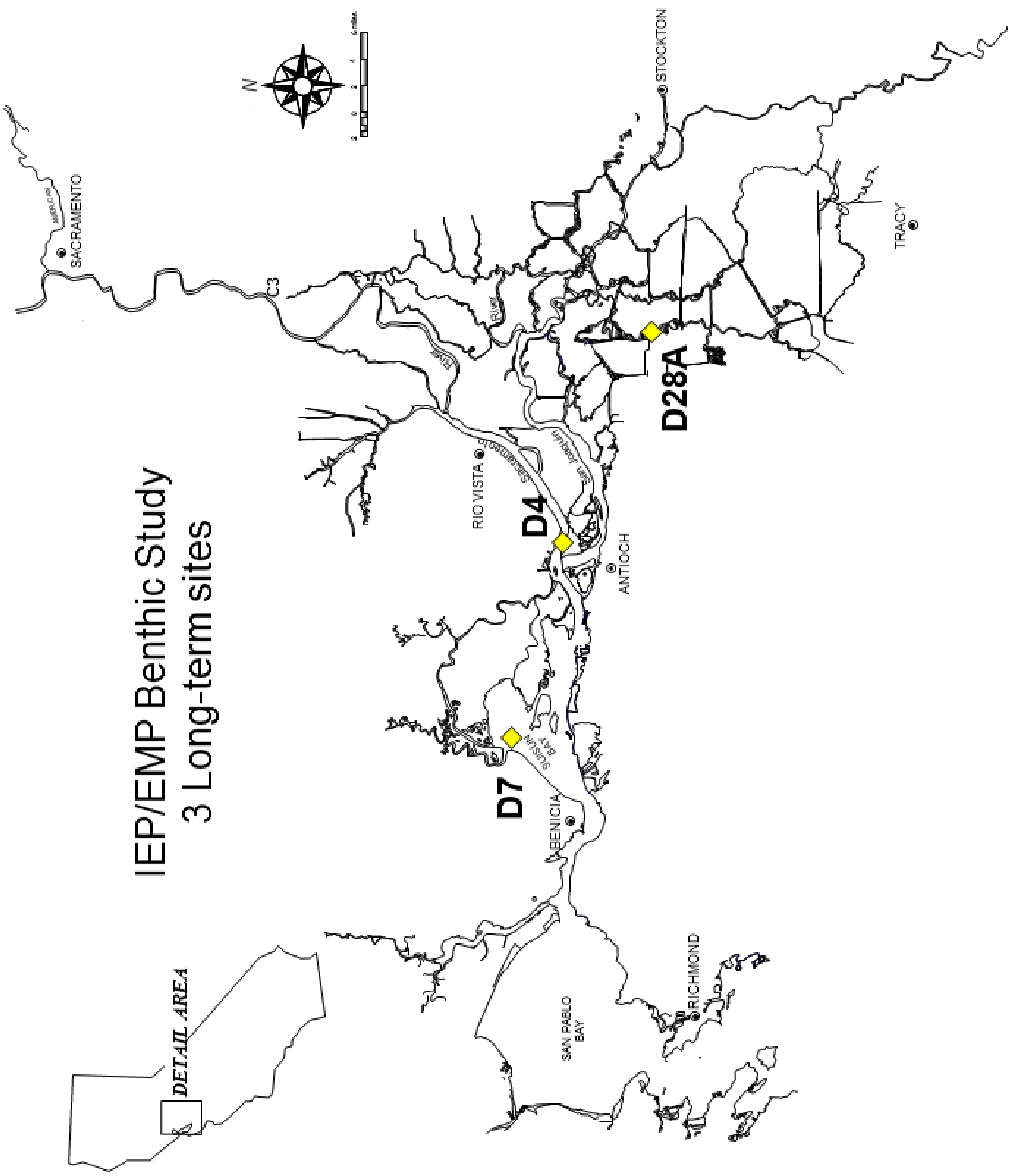


Figure 2. Indicator 1: Native species diversity.

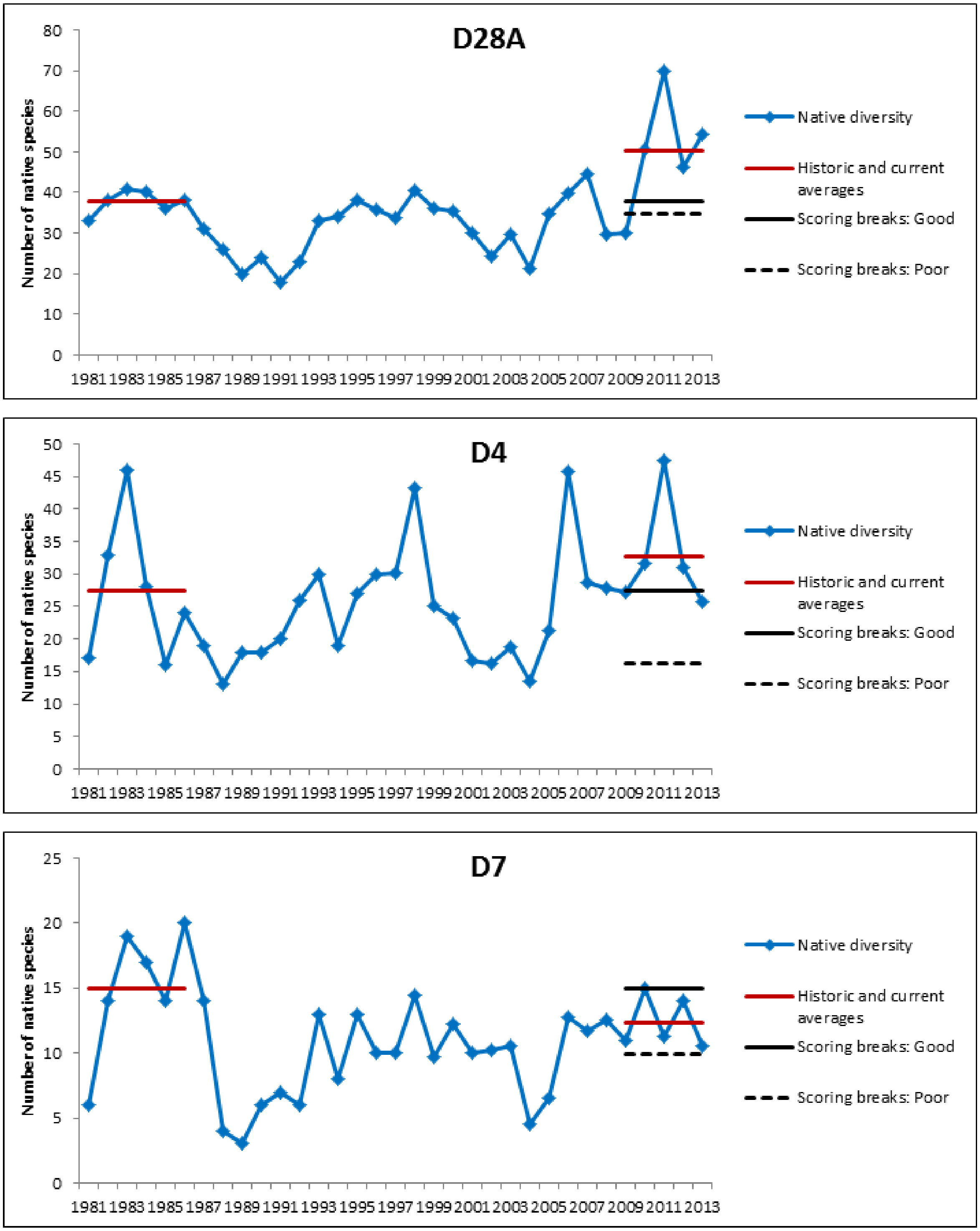
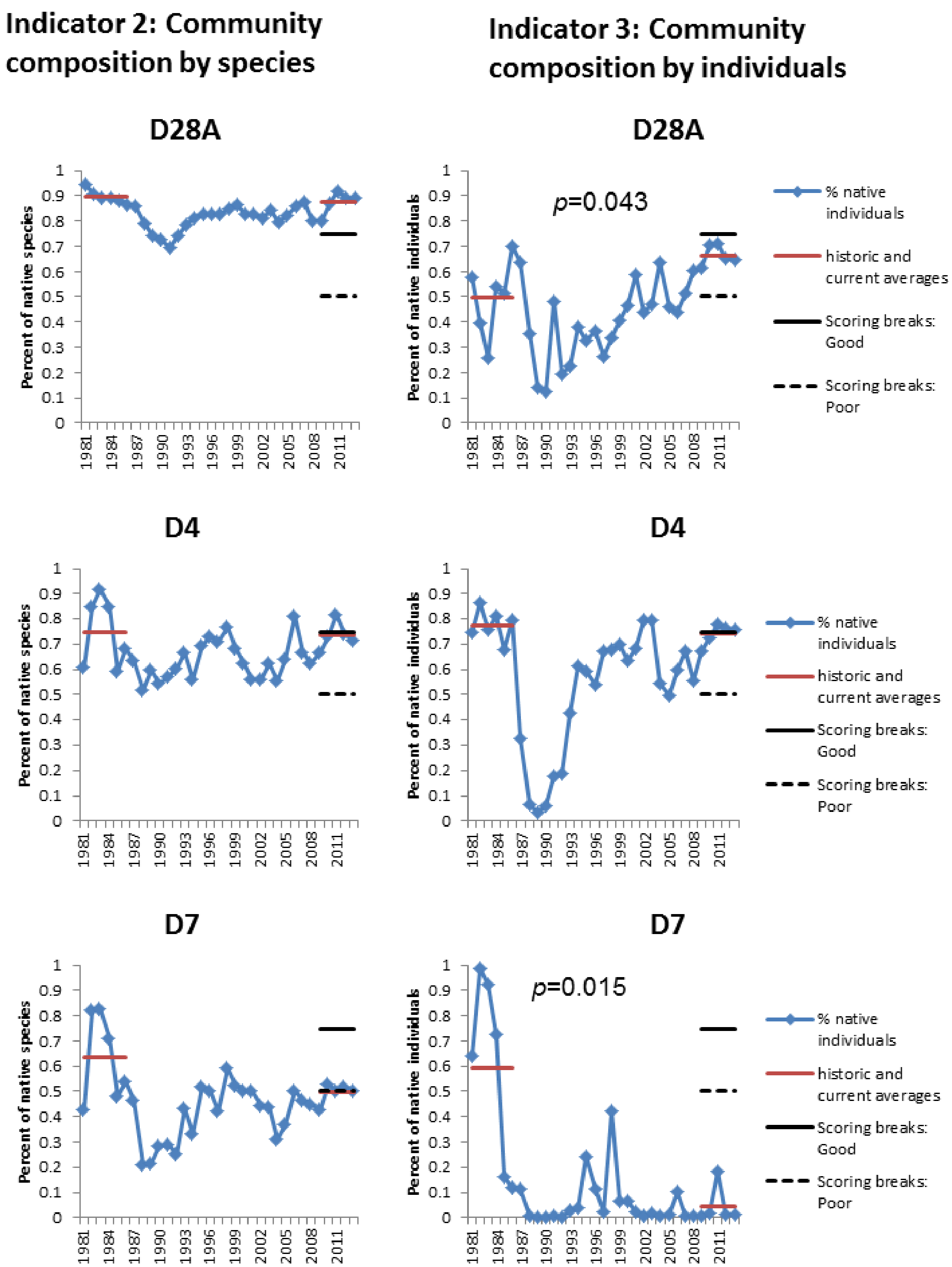


Figure 3. Indicators 2 and 3: Community composition by species, by region. Significant trends are marked with p-values.



State of the Estuary Report 2015

Summary

WILDLIFE – Estuary Fish Summary

Prepared by Christina Swanson, Natural Resources Defense Council; Jonathan Rosenfield, Ali Weber-Stover, The Bay Institute

2015

State of the San Francisco Estuary 2015

WILDLIFE – Estuary Fish Summary

Prepared by Christina Swanson, Natural Resources Defense Council; Council; Jonathan Rosenfield, Ali Weber-Stover, The Bay Institute
April 2015

What are the indicators?

The Bay Fish Index uses ten indicators to measure and evaluate the status and trends of the San Francisco Estuary's fish community in four sub-regions of the estuary; South, Central, San Pablo and Suisun Bays. The indicators are designed to measure and evaluate different attributes of the fish community: abundance (4 indicators for "how many fish"), diversity (2 indicators for "how many different kinds of fish"), species composition (2 indicators for "what kinds of fish"), and distribution (2 indicators for "where are the fish"). The combined result of the indicators in each attribute were aggregated results into a Bay Fish Index, which combines the results of all the indicators into a single metric for each sub-region.

Four indicators measure abundance:

- Pelagic Fish Abundance;
- Northern Anchovy Abundance;
- Demersal Fish Abundance; and
- Sensitive Species Abundance.

Two indicators measure species diversity:

- Native Fish Species Diversity; and
- Estuary-dependent Fish Species Diversity.

Two indicators measure species composition:

- Percent Native Species; and
- Percent Native Fish.

Two indicators measure fish distribution:

- Pelagic Fish Distribution; and
- Demersal Fish Distribution.

Except for the species composition indicators and the Sensitive Species Abundance indicator, all indicators measure only fish species that are native to the San Francisco Estuary and local coastal waters.

To provide a geographically comprehensive view of trends among fishes in the San Francisco Estuary, a smaller set of indicators were developed to reveal conditions in Suisun Marsh, Suisun Bay, and the Sacramento-San Joaquin Delta (collectively, the upper Estuary). The upper

Estuary's aquatic habitat and fish fauna differ from those found in the open waters of the estuary's main embayments and, as a result, different survey programs, using different fish sampling techniques, monitor fish in this area. Indeed, data for indicators in the upper Estuary comes from three different long-term sampling programs, each of which samples a different habitat and region using different gear.

As a result of large amount of data available in the upper Estuary and the heterogeneity of its habitats, only three indicators of fish assemblage health were developed for this region. One measure of abundance (Native Fish Abundance) and two measures of assemblage composition (Percent Native Species and Percent Native Fish) were calculated for the upper Estuary. These indicators were calculated for each sampling program and sub-regions within the upper Estuary and were designed to mirror the approach used for analogous indicators in the Bay Fish Index.

An additional indicator, portraying the fish assemblage's role in the Estuary's food web, was calculated for fishes of the upper Estuary. This indicator is a measure of total fish abundance (introduced and native species combined) in each region and sub-region of the three major habitat types of the upper Estuary. That indicator is described and presented in the Processes section of the 2015 State of the Estuary report.

Table 1.

Attribute	**Indicators**	**Benchmarks**
Living Resources (Bay fish)	Abundance, diversity, species composition and distribution the fish community in four sub-regions of the Bay (South, Central, San Pablo and Suisun Bays)	Benchmarks (or reference conditions) are 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.
Living Resources (Upper Estuary Fish)	Abundance and species composition indicators in Suisun Marsh; subregions of the upper Estuary's Pelagic Zone (Suisun Bay and the West Delta); four subregions of the Delta Beach Zone (littoral habitats)	Primary reference conditions are based on either measured values from early years of the sampling record (1980-1989 for the Suisun Marsh survey and Fall mid-water trawl and 1995-2004 for the Delta Beach Seine), recognized and accepted interpretations of ecological conditions and ecosystem health (e.g., native v non-native species composition), and best professional judgment.

Why is the estuary's fish community important?

San Francisco Bay's estuary is important habitat for more than 100 fish species, including commercially important Chinook salmon and Pacific herring, popular sport fishes like striped bass and white 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 estuary 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.

What are the benchmarks? How were they selected?
The benchmarks (or reference conditions) for the Bay Fish indicators are based on: 1) measured values from the earliest years for which quantitative data were available (1980-1989 for the Bay Study survey); 2) maximum measured values for the estuary or sub-regions; 3) recognized and accepted interpretations of ecological conditions and ecosystem health (e.g., native v non-native species composition); and 4) best professional judgment. The upper Estuary fish indicators mirror this approach for setting benchmarks. The 1980-1989 period was used as baseline for Suisun Marsh (representing the earliest data available) and the Pelagic Zone (data here extend back to 1967); the Delta Beach Seine survey methodology became more consistent in the mid-1990s, so the period 1995-2004 was used as the primary reference condition for those data. Reference conditions for evaluating assemblage composition (native vs. non-native species) were identical to those developed for the Bay Fish index.

What are the status and trends of the indicators and Index?
The conditions and trends of the Bay fish community differ among the four sub-regions of the estuary. Abundance, diversity, species composition and distribution are all highest in Central and South Bays, where overall conditions (meaning the regional Fish Index) were consistently "good", intermediate in San Pablo Bay, where conditions were "good" to occasionally "fair," and lowest in Suisun Bay, the upstream region of the estuary, where over the last 3 decades conditions have declined from "fair" to poor." Overall conditions (the Index) are also declining in South and San Pablo Bay, although the rate of decline is lower than that in Suisun Bay. Declines in n the Fish Index in these regions are driven by substantial declines in the abundance of pelagic (open water) fish species and, in Suisun Bay and San Pablo Bay, declines in species composition (i.e., non-native species are becoming more prevalent) and, in Suisun Bay, declines in distribution (i.e., native species are no longer consistently collected in some areas of the sub-region).

Table 2.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend (long term; 1980-2013)**	**Trend since 1990**	**Current condition** (average for last 10 years)
Pelagic Fish Abundance	Not met in any sub-region	Decline in all sub-regions except Central	Stable at low levels	Fair to Very Poor
Northern Anchovy Abundance	Not met in any sub-region	Decline in San Pablo and Suisun, stable in South and Central	Stable at low levels (Suisun, San Pablo) Declining (South, Central)	Fair to Very poor

Demersal Fish Abundance	Fully met (South and Central) Not met (San Pablo and Suisun)	Decline in Suisun, increase in Central and South, stable in San Pablo	Stable (Suisun) Increasing (South, Central, San Pablo)	Poor (Suisun) Fair to good (South, Central, San Pablo)
Sensitive Species Abundance	Not met on any sub-region	Decline in all sub-regions	Stable at low levels	Poor (all sub-regions)
Native Fish Diversity	Partially met (South) Not met (Central, San Pablo, Suisun)	Decline in San Pablo, increase in Central, stable in other sub-regions	Stable	Poor (Suisun) Fair to good (South, Central, San Pablo)
Estuary-dependent Fish Diversity	Fully met (South, Central) Not met (San Pablo, Suisun)	Decline in South and San Pablo, stable in Central and Suisun	Stable	Poor (Suisun) Fair to good (South, Central, San Pablo)
Percent Native Species	Fully met (South, Central) Not met (San Pablo, Suisun)	Decline in all sub-regions except Central	Stable (South, Central) Declining (San Pablo Suisun)	Good (South, Central) Fair to Poor (San Pablo, Suisun)
Percent Native Fish	Fully met (South, Central, San Pablo) Not met (Suisun)	Decline in Suisun, stable in other sub-regions	Stable	Good (South, Central, San Pablo) Very Poor (Suisun)
Pelagic Fish Distribution	Fully met (South, Central, San Pablo) Partially met (Suisun)	Decline in Suisun, stable in other sub-regions	Stable (South, Central, San Pablo) Declining (Suisun)	Good (South, Central, San Pablo) Fair to Poor (Suisun)
Demersal Fish Distribution	Fully met (South, Central, San Pablo) Partially met (Suisun)	Decline in Suisun, stable in other sub-regions	Stable (South, Central, San Pablo) Declining (Suisun)	Good (South, Central, San Pablo) Fair to Poor (Suisun)
Bay Fish Index	Fully met (Central) Partially met (South) Not met (San Pablo, Suisun)	Decline in all sub-regions except Central	Stable (South, Central, San Pablo) Declining (Suisun)	Good (Central) Fair (South, San Pablo) Poor (Suisun)

Because habitats and sampling programs operating within the upper estuary are substantially different, no synthetic index was calculated for the upper Estuary region. However, it is clear that the fish assemblage in the upper Estuary is in very poor condition (Table 3). Native fish abundance, the percentage of native fish, and the percent of native species are poor or very poor in almost every sub-region of the upper Estuary.

Table 3

Indicator	Region (Sub-region if trends are different)	CCMP Goal Met	Evaluation		Trend
			Reference Period	Short-Term (last five years)	Over the Period of Record
Native Fish Abundance	Suisun Marsh	No	Good	Poor	Decline
	Suisun Bay Pelagic	No	Good	Very Poor	Decline
	Central-West Delta Pelagic	No	Good	Very Poor	Decline

	Delta Beach Zone	No	Poor	Poor	Stable
Percent Native Fish	Suisun Marsh	No	Very Poor	Very Poor	Stable
	Suisun Bay Pelagic	No	Poor	Poor	Stable
	Central-West Delta Pelagic	No	Very Poor	Very Poor	Stable
	Delta Beach Zone	No	Very Poor	Very Poor	Stable
Percent Native Species	Suisun Marsh	No	Poor	Very Poor	Decline
	Suisun Bay Pelagic	No	Fair	Fair	Stable
	Central-West Delta Pelagic	No	Poor	Very Poor	Decline
	Delta Beach Zone	No	Very Poor	Very Poor	Stable

What does it mean? Why do we care?

The condition and trends of the fish community in the San Francisco Bay's estuary are key indicators of the health of the estuary and its function as habitat for resident and migratory fishes. The Bay Fish Index shows that the estuary is in healthy and stable condition in Central Bay, the downstream subregion that is strongly influenced by environmental conditions in the Pacific Ocean. The health of South and San Pablo Bays is fair, but the Bay Fish Index shows that conditions there are declining as well.

In contrast, the both the Bay Fish Index in the Suisun Bay Region and the individual indicators of the different habitats in the upper Estuary confirm that that the health of the upstream region of the estuary, (including Suisun Marsh, Suisun Bay, and the Delta), has declined markedly during the past three decades and is now (and has been for more than 20 years) in poor to very poor condition. During the past twenty years, the upper Estuary has been strongly influenced by fresh water management operations (in the Delta and in Central Valley rivers) that reduce and alter the patterns of freshwater inflows (see Freshwater Inflow Index, Open Water Habitat indicators, and Flood Events indicators).

State of the Estuary Report 2015

Technical Appendix

WILDLIFE– Bay Fish Indicators and Index
Technical Appendix

Prepared by Christina Swanson
Natural Resources Defense Council

June 2015

State of San Francisco Estuary 2015

Wildlife – Bay Fish Indicators and Index
Technical Appendix

Prepared by Christina Swanson
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June 2015

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

Figure 1. Because the an Francisco Bay is so large and its environmental conditions so different in different areas, the Bay Fish Index and each of its component indicators were calculated separately fro four sub-regions in the estuary: South Bay, Central Bay, San Pablo Bay and Suisun Bay and the western Delta.

[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.

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 predominantly captures pelagic fishes that utilize open water habitats. This survey tends to 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.

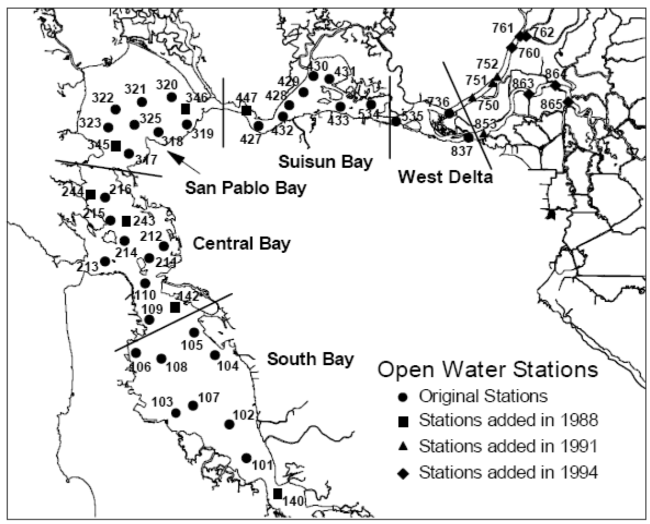

Figure 2. Locations of the sampling stations for the CDFW Bay Study Midwater Trawl and Otter Trawl surveys in the San Francisco Bay. For the Bay Fish Index, only data from the original stations that were sampled continuously from 1980-2013 were used to calculate the indicators for each of the four sub-regions: South Bay, Central Bay, San Pablo Bay and Suisun Bay and the western Delta..

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[2] Information on the CDFG Bay Study is available at www.delta.dfg.ca.gov/baydelta/monitoring/baystudy.asp.
[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.

Table 1. Sampling stations and total number of surveys conducted per year (range for 1980-2013 periods, excludes 1994) by the CDFW Bay Study Survey in each of four sub-regions of the 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 1980-2013 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/Western Delta	427, 428, 429, 430, 431, 432, 433, 534, 535, 736, and 837	87-132 (MWT) 88-132 (OT)

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 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

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[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.

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., 2009-2013). Regression analyses were conducted using continuous results for the entire 34-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.

Table 2. Fish community characteristics and indicators used to calculate the Bay 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

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

[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).

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, 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 (1980-2013, 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:

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

[8] In California, longfin smelt are found in San Francisco Bay, Humboldt Bay, and the estuaries of the Russian, Eel, and Klamath rivers.

The **Northern Anchovy Abundance** indicator was calculated for each year (1980-2013, 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 (1980-2013) 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:

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

The **Sensitive Fish Species Abundance** indicator, the abundance of each of the four species was calculated for each year (1980-2013, 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.

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

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

Table 3. Quantitative reference conditions and associated interpretations for results of the Bay Fish abundance indicators. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Abundance indicators		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>150% of 1980-1989 average	"Excellent," greater than recent historical levels	4
>100% of 1980-1989 average	***"Good," meets CCMP goals***	***3***
>50% of 1980-1989 average	"Fair," below recent historical levels	2
>15% of 1980-1989 average	"Poor," substantially below recent historical levels	1
≤15% of 1980-1989 average	"Very Poor," extreme decline in abundance	0

4. Results

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

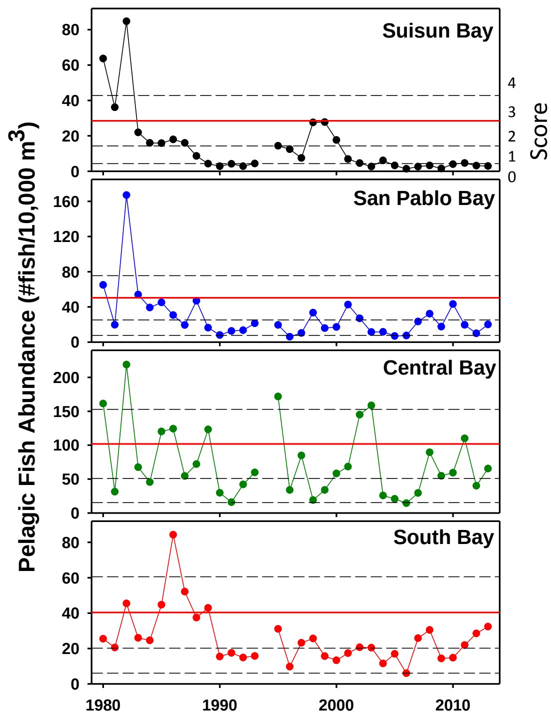

Figure 3. Results for the Pelagic Fish Abundance indicator, expressed as abundance (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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 2013, pelagic fishes were two to three times more abundant in Central Bay (65 fish/10,000m^3) than South (32 fish/10,000m^3) or San Pablo Bays (20 fish/10,000m^3) and more than 20 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 since 1980 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. In the last 10 years, pelagic fish abundance appears to be increasing in South and Central Bays (regression, $p=0.057$ for South Bay and $p=0.064$ for Central 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 and trends in pelagic fish abundance are below the 1980-1989 reference period for most sub-regions of the estuary: average pelagic fish abundance levels for the most recent five years (2009-2013) are "fair" in South Bay (55% of the 1980-1989 average) and Central Bay (65%), "poor" in San Pablo Bay (43%) and "very poor" in Suisun Bay (11%).

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 34 years, northern anchovy have been more abundant in Central Bay (mean: 913 fish/10,000m^3) than all other sub-regions, least abundant in Suisun Bay (16

fish/10,000m^3), and present at intermediate abundance levels in San Pablo (241 fish/10,000m^3) and South Bays (282 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 34 years, abundance of northern anchovy has been variable but roughly stable in South and Central Bays although, in most recent years (2009-2013), Central Bay abundance has averaged about 54% lower than 1980-1989 levels. Northern anchovy abundance has steadily declined in San Pablo Bay (regression: $p<0.001$), falling to 41% of 1980-1989 levels during the most recent five years (2009-2013). The decline was more abrupt in Suisun Bay (regression: $p<0.01$), 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 just 5% 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.

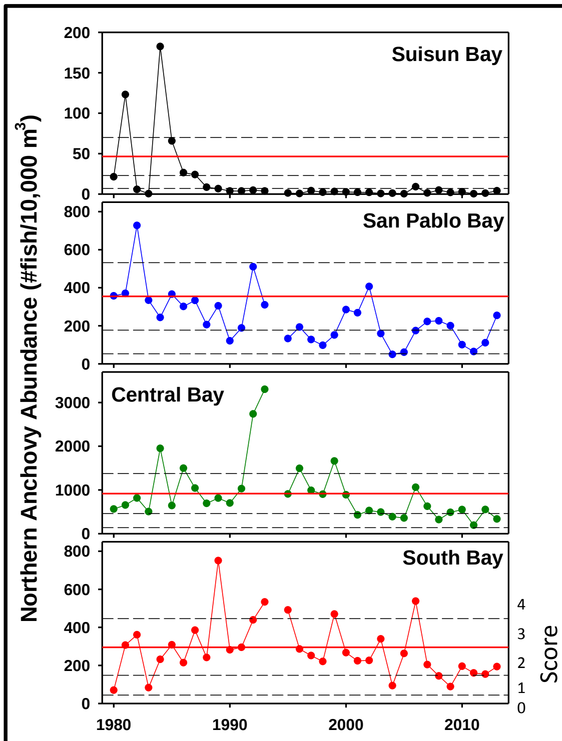

Figure 4. Results for the Northern Anchovy Abundance indicator, expressed as abundance (left Y axis) and score (right Y axis, bottom panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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 significantly throughout the upstream regions of the estuary, San Pablo and Suisun Bays to levels substantially below the 1980-1989 average reference conditions: average northern anchovy abundance in the most recent five years (2009-2013) are "very poor" in Suisun Bay at just 4% of the 1980-1989 average, and "poor" in San Pablo Bay (41%). Although the trends in abundance over the 34-year record, and particularly during the late 1980s and 1990s, are different for Central and South Bays, recent northern anchovy abundance in those regions, "poor" in Central Bay (46%) and "fair" in South Bay, are also too low to meet the CCMP goal. 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.

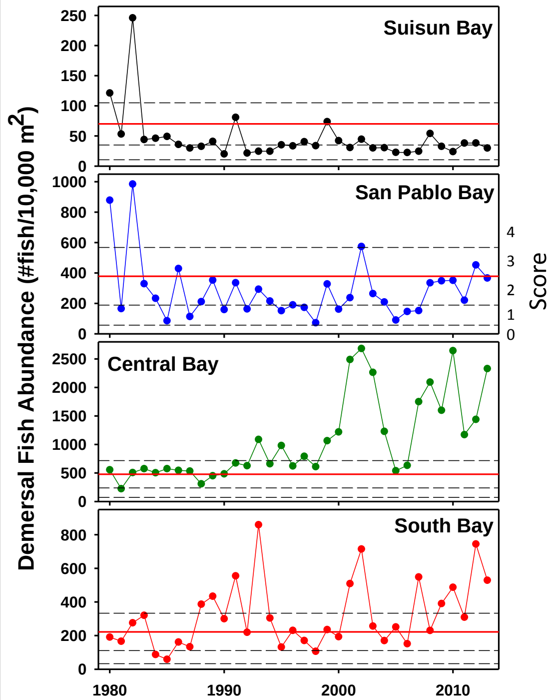

Figure 5. Results for the Demersal Fish Abundance indicator, expressed as abundance (left Y axis) and score (right Y axis, second panel from the top only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

Abundance of demersal fish species differs among the estuary's sub-regions.

Demersal fishes are more abundant in Central Bay (1980-2013 median: 669 fish/10,000m^2) than in all other sub-regions of the estuary and least abundant in Suisun Bay (35 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 (254 fish/10,000m^2) and San Pablo Bays (227 fish/10,000m^2) are comparable. In 2013, demersal fishes were more than four times more abundant in Central Bay (2330 fish/10,000m^2) than South Bay (530 fish/10,000m^2), more than six times more abundant than in San Pablo Bays (367 fish/10,000m^2), and nearly 80 times more abundant than in Suisun Bay (30 fish/10,000m^2).

Abundance of demersal fishes has increased in Central and South Bays but declined in Suisun Bay.

During the past 34 years, abundance of native demersal fishes increased in Central and South Bays (regressions: $p<0.001$ and $p<0.05$, respectively) but declined in Suisun Bay (regression: $p<0.05$). In San Pablo Bays, demersal fish abundance has fluctuated widely but exhibited no significant trend over time. Compared to 1980-1989 levels, recent average abundances (2009-2013) were 53% lower in Suisun, similar in San Pablo Bay (8% lower), and 222% and 384% higher in South and Central Bays, respectively.

Increases in demersal fish abundance in Central and South Bays were driven by multiple species.

In South and Central Bays, increases in demersal fish abundance were largely attributable to high catches of Bay goby and Pacific staghorn sculpin, Bay resident species, and plainfin midshipman and 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 2009-2013 average) and trends in demersal fish abundance were higher or comparable to the 1980-1989 reference period for all sub-regions of the estuary except Suisun Bay, where demersal fish abundance decreased significantly and remain at less than half of recent historical levels. 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 34 years, combined abundance of the four sensitive fish species has declined in all sub-regions of the estuary (regression: $p<0.01$ all sub-regions). For the most recent five-year period (2009-2013), abundance of sensitive fish species abundance San Pablo is just 28% of that sub-region's 1980-1989 average, 30% in Central Bay, 33% in South Bay and 50% 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 South, Central, and San Pablo Bays (regression: $p<0.05$ both tests), striped bass declined in all sub-regions (regression: $p<0.05$ all regions). Pacific herring abundance was variable and did not exhibit significant declines in any sub-region.

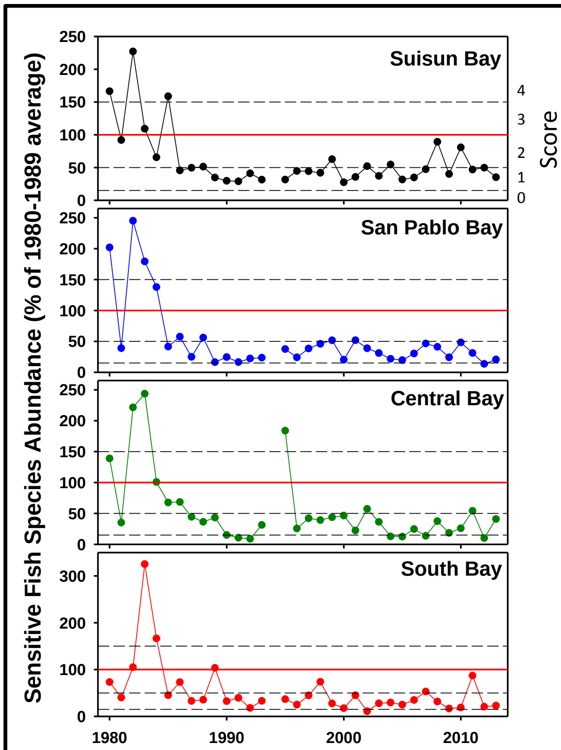

Figure 6. Results for the Sensitive Species Abundance indicator, expressed as abundance (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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 (2009-2013) were significantly lower in only South and Central Bay (t-test or Mann-Whitney, $p<0.05$, both tests). Although recent abundance levels in San Pablo and Suisun Bay were markedly lower than during the 1980-1989 reference period, 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.

Table 4. San Francisco estuary-dependent fish species collected in the CDFW Bay Study 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	

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.

Table 5. Quantitative reference conditions and associated interpretations for results of the Bay Fish diversity indicators. The primary reference condition, which corresponds to “good” conditions, is in bold italics.

Diversity indicators		
Native Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>60% of assemblage present	“Excellent,” greater than 1980-1989 average	4
>50% of assemblage present	***“Good,” meets CCMP goals***	***3***
>40% of assemblage present	“Fair,” below recent historical levels	2
>30% of assemblage present	“Poor,” substantially below recent historical levels	1
≤30% of assemblage present	“Very Poor,” extreme decline in diversity	0
Estuary-dependent Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>85% of assemblage present	“Excellent,” greater than 1980-1989 average	4
>70% of assemblage present	***“Good,” meets CCMP goals***	***3***
>55% of assemblage present	“Fair,” below recent historical levels	2
>40% of assemblage present	“Poor,” substantially below recent historical levels	1
≤40% of assemblage present	“Very Poor,” extreme decline in diversity	0

4. Results

Results of the **Native Fish Species Diversity** indicator are shown in Figure 7.

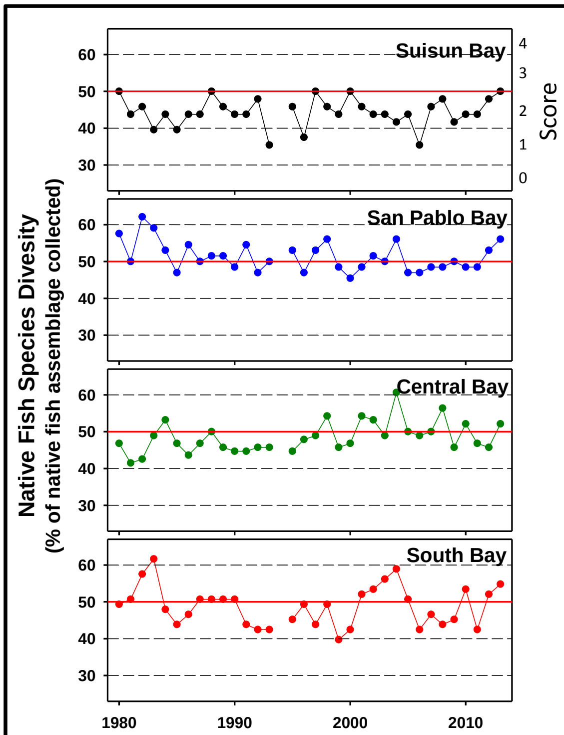

Figure 7. Results for the Native Fish Species Diversity indicator, expressed as percent of assemblage (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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: $p<0.01$).

Trends in native species diversity differ among the sub-regions.

Native species diversity has increased significantly in Central Bay (regression: $p<0.05$) with an average of two 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 two fewer species in the 2009-2013 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 (2009-2013) to that measured during the 1980-1989 period shows no significant differences in any sub-region. Recent diversity levels, 51%, 50%, 49% and 44% in San Pablo, South, Central and Suisun Bays, respectively, have been close to or exceeded the primary reference condition and/or historical conditions for all sub-regions.

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 (83% in Central Bay; 79% in South Bay; and 69% in San Pablo Bay) (ANOVA: $p<0.001$, all pairwise comparisons, $p<0.05$).

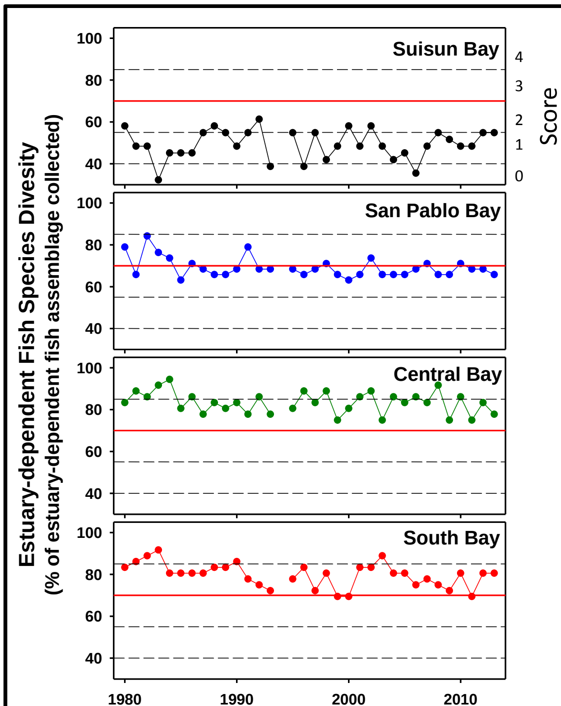

Figure 8. Results for the Estuary-dependent Fish Species Diversity indicator, expressed as percent of assemblage (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

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 1.3 species from the 1980-1989 period to the 2009-2013 period) and South Bay (regression: $p<0.05$, for an average decrease of 2.6 species). In all other regions, estuary-dependent diversity has fluctuated but remained relatively stable over the 34-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.
The percentages of the estuary-dependent fish assemblage that are present, 79%, 77%, 68%, and 52% in Central, South, San Pablo and Suisun Bays, respectively, generally meet or exceed the

primary reference condition in all regions except Suisun Bay, where diversity levels are similar to historical levels.

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.

Table 6. Quantitative reference conditions and associated interpretations for results of the Bay Fish species composition indicators. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Species Composition indicators (Percent Native Species, Percent Native Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>95% native	"Excellent," greater than recent historical levels	4
>85% native	***"Good," meets CCMP goals***	***3***
>70% native	"Fair," below recent historical levels	2
>50% native	"Poor," substantially below recent historical levels	1
≤50% native	"Very Poor," extreme decline in abundance	0

4. Results

Results of the **Percent Native Species** indicator are shown in Figure 9.

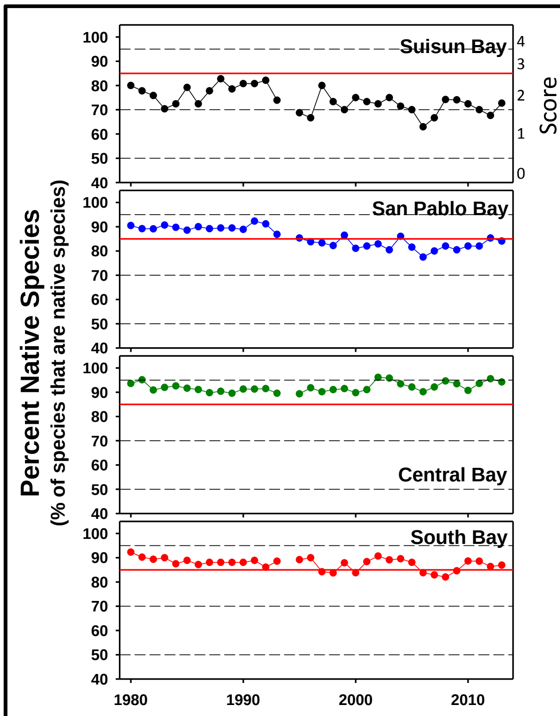

Figure 9. Results for the Percent Native Fish Species indicator, expressed as percent native species (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The percentage of native species in the fish community differs among the four sub-regions of the estuary.

For the past 34 years, non-native species have been most prevalent in Suisun Bay where, on average, 26% of species are non-native (i.e., only 74% of species are native), intermediate in South and San Pablo Bays (12% and 14% non-native, respectively), and the least prevalent in Central Bay (8%) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

The percentage of native species is declining in most sub-regions.

The percentage of native species has declining significantly in all sub-regions of the estuary except Central Bay ($p<0.01$, all tests except Central Bay). In South Bay, the percent native species declined from 89% in the 1980-1989 period to 87% in the most recent five-year period (2009-2013). In San Pablo Bay, the percent native species has declined more sharply, from 90% to 83% and in Suisun Bay from 77% to just 71% native species.

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 34 years, the number of native species in San Pablo Bay declined by an average of 1.6 species and the number of non-native species increased by an average of 2.9 species; in the most recent five years, there 7 non-native species in this sub-region, on average. The number of non-native species collected in Suisun Bay increased by 2.3 species, from 6.6 to 8.8 non-native 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 two 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 primary reference condition, recent measurements (2009-2013) of the percentage of native fish species in the fish community indicate that this characteristic has degraded in both San Pablo Bay (83% native species) and Suisun Bay (71% native species) to levels that do not meet the CCMP goals. In South Bay, the prevalence of native species is also declining but recent levels, 87%, are still "good" and meet CCMP goals.

Results of the **Percent Native Fish** indicators are shown in Figure 10.

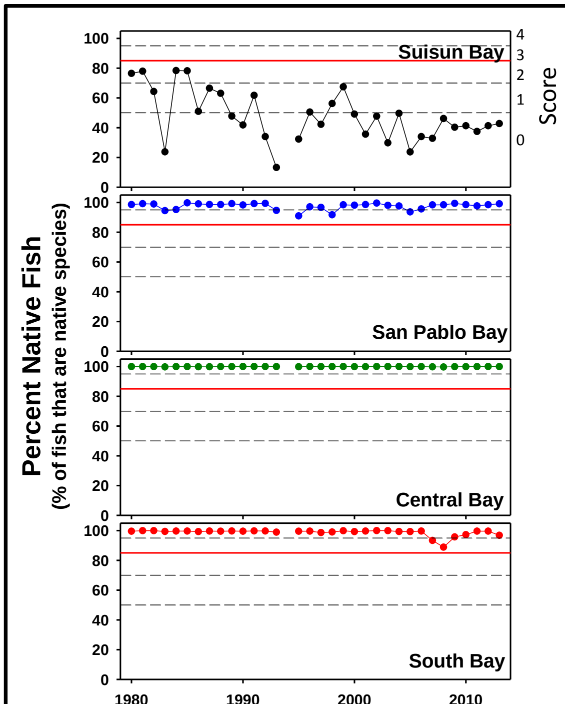

Figure 10. Results for the Percent Native indicator, expressed as percent native fish (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

The percentage of native fish in the fish community differs among the four sub-regions of the estuary.

For the past 34 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: 48%). Non-native fish are rare in the other three sub-regions. Central Bay and South Bay have the lowest prevalence of non-native fishes, 0.1% and 0.4%, respectively, and levels in San Pablo Bay are intermediate at 2.1% (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-regions of the estuary but not in Central or San Pablo Bays (regression, $p<0.5$, both tests). In Suisun Bay, the percent native fish declined from 63% in the 1980-1989 period to just 41% in the most recent five-year period. Percent native fish declined in South Bay from more than 99% to less than 98%. 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 sub-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.

Table 7. Quantitative reference conditions and associated interpretations for results of the Bay Fish distribution indicators. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Distribution indicators (Pelagic Fish, Demersal Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
100% of stations	"Excellent," greater than recent historical levels	4
>80% of stations	***"Good," meets CCMP goals***	***3***
>60% of stations	"Fair," below recent historical levels	2
>40% of stations	"Poor," substantially below recent historical levels	1
≤40% of stations	"Very Poor," extreme decline in abundance	0

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 34 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 1980-2013 period, native fish were present at 97-100% of survey stations in South, Central and San Pablo Bays. In contrast, native fish were present in only an average of 76% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, Suisun v all other sub-regions; $p<0.05$).

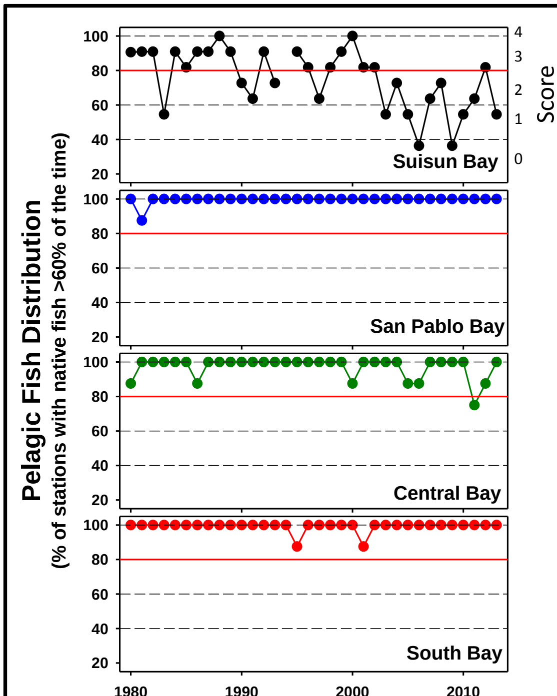

Figure 11. Results for the Pelagic Fish Distribution indicator, expressed as percent of stations (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

Trends in the distribution of native pelagic 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 58% in the most recent five years (2009-2013) (Mann-Whitney Rank Sum test; $p<0.01$; regression: $p<0.01$). This decline in distribution occurred abruptly in 2003; since 2003, native pelagic fish have been consistently present at only 59% of stations, on average, compared to being present at 84% of stations during the first 23 years of the survey. 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 34 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 74% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: p<0.001, Suisun v all other sub-regions; p<0.05).

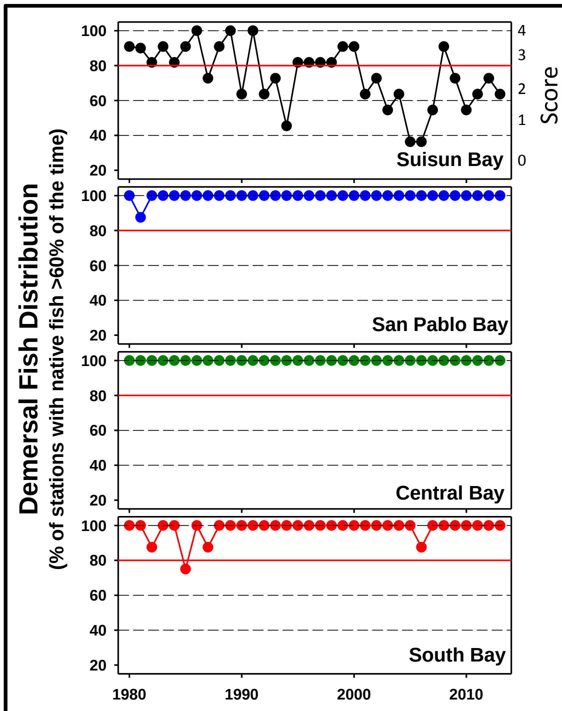

Figure 12. Results for the Demersal Fish Distribution indicator, expressed as percent of stations (left Y axis) and score (right Y axis, top panel only for example), for 1980 to 2013. The horizontal red line shows the primary reference condition. The horizontal dashed lines show the other reference conditions used for evaluation.

Trends in the distribution of native demersal 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 88% of stations (1980-1991) to just 61% of stations (1992-1994), and then recovered to 85% (1995-2000). In 2001, distribution declined again and, even with the relatively high level in one year (2008), it has remained significantly lower since then, 62% on average (t-test, p<0.001 for 1980-2001 v 2002-2013). For the most recent five years (2009-2013), native demersal fish have been present at 65% 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 or equal to 2.5, which reflects at least two community attributes with average scores greater than 3, was interpreted to represent “good” conditions and an index score less than 0.5 was interpreted to represent “very poor” conditions.

B. Results

Results of the four component metrics (Abundance, Diversity, Species Composition, and Distribution) and the Bay Fish Index for each sub-region are shown in Figures 13-16 (following pages).

The Bay Fish Index differs among the four sub-regions of the estuary.
For the 34 year survey period, the Bay Fish Index was equally high in the Central Bay (1980-2013 average: 3.1) and South Bay (3.0), lowest in Suisun Bay (1.5), and intermediate in San Pablo Bays (2.8) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.05$; Central=South>San Pablo>Suisun). For the most recent five years (2009-2013), the pattern among the sub-regions was similar: the average Index was 3.0, 3.0, 2.7, and 1.2 for Central, South, San Pablo and Suisun Bays, respectively. Lower 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 Bay Fish Index differ among the sub-regions.
During the 34 year survey period, the Bay Fish Index has declined significantly in Suisun, San Pablo and South Bays but not in Central Bay (regression 1980-2013: $p<0.05$ 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: 2.2) to consistent “poor” conditions since the 1990s. This decline was driven by significant declines in abundance, species composition and diversity (regression, all test, $p<0.001$). In San Pablo Bay, the Index has declined steadily, from mostly “good” conditions in the early 1980s to “fair” conditions since the 1990s; this decline is largely attributable to significant declines in abundance and diversity (regression, $p<0.05$, both

tests). The decline in the Index in South Bay, while significant, is not as severe, with conditions fluctuating between “good” and “fair.” In Central Bay, the 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 Bay 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 Bay 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, over the 34-year period of record for these indicators, the condition of the fish community in San Pablo and South Bays is declining.
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.

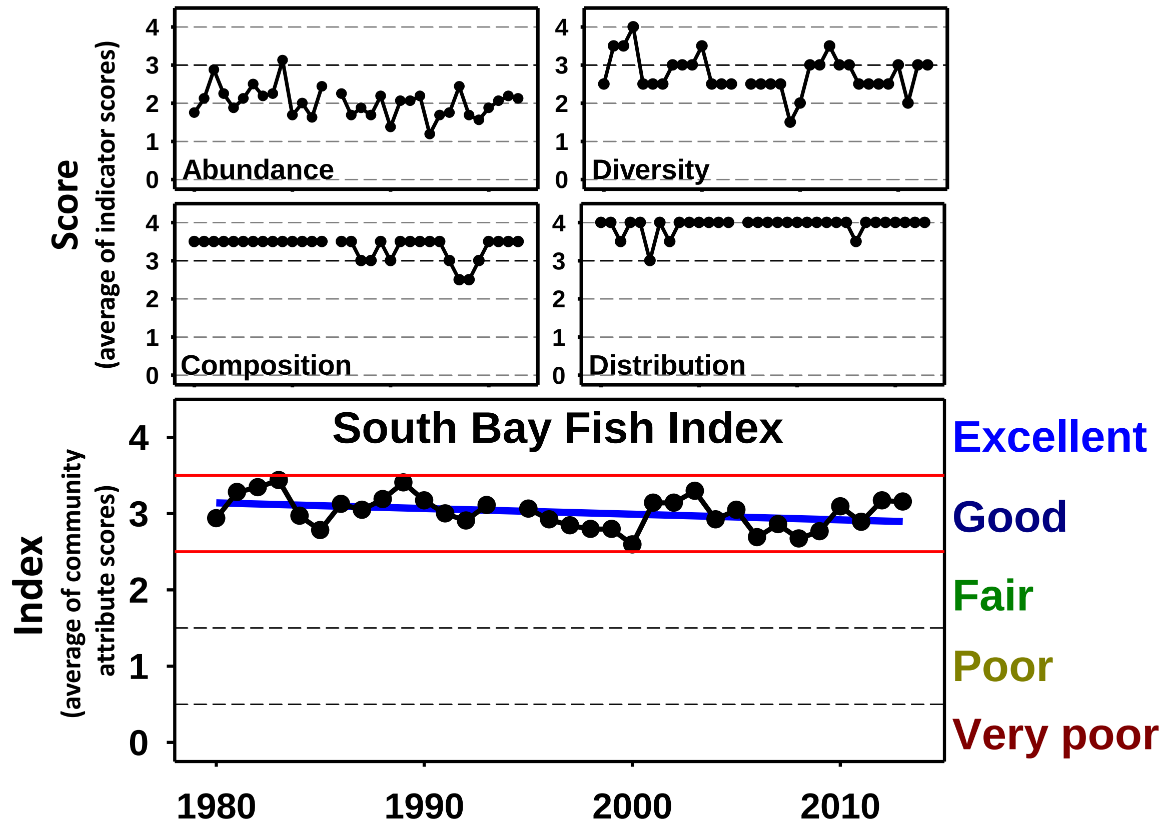

Figure 13. Results for the Bay Fish Index for South Bay for 1980 to 2013. The top four panels show results for the four fish community attributes, expressed as average scores. The large bottom panel shows the Index, calculated as the average of the scores of the four component community attributes. For those results with a significant trend over time, the regression line is shown in blue. For the Index graph, the horizontal red lines and dashed lines show the reference conditions and Index evaluation.

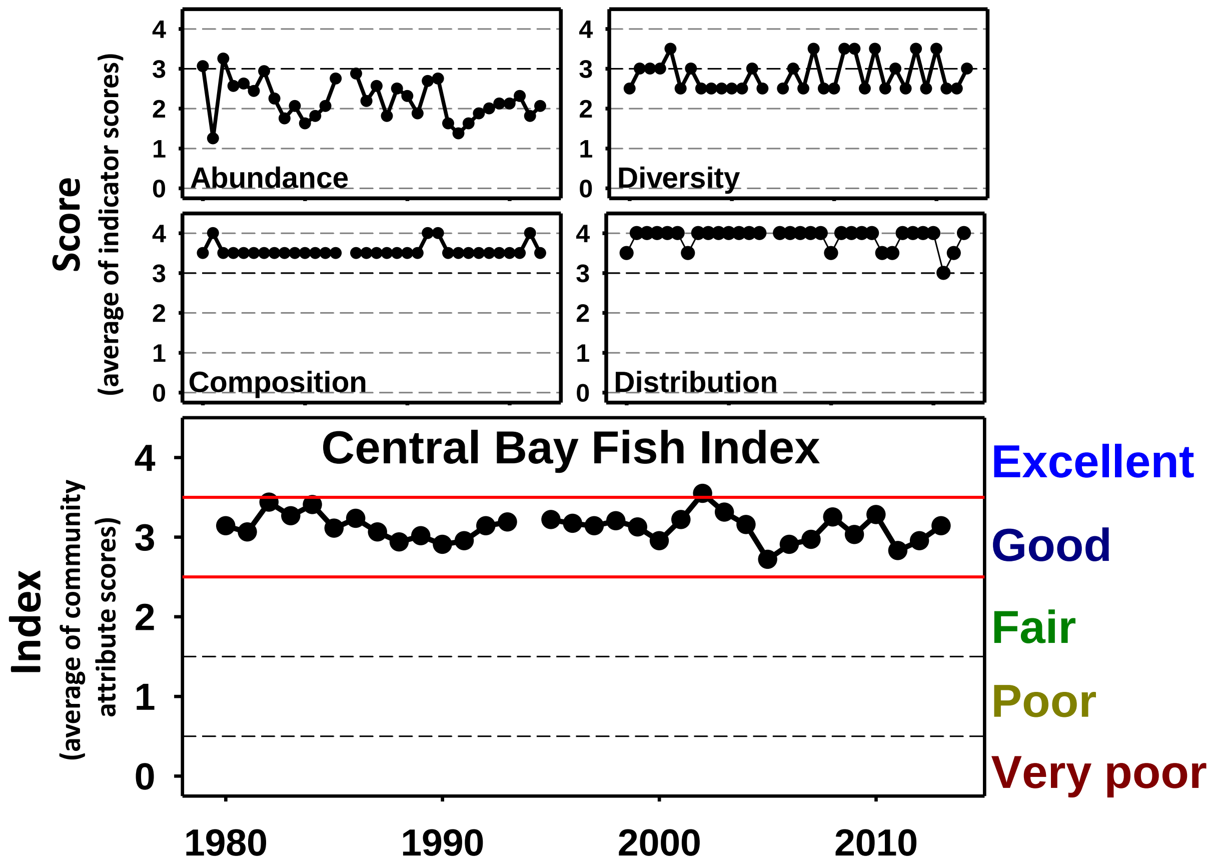

Figure 14. Results for the Bay Fish Index for Central Bay for 1980 to 2013. The top four panels show results for the four fish community attributes, expressed as average scores. The large bottom panel shows the Index, calculated as the average of the scores of the four component community attributes. For those results with a significant trend over time, the regression line is shown in blue. For the Index graph, the horizontal red lines and dashed lines show the reference conditions and Index evaluation..

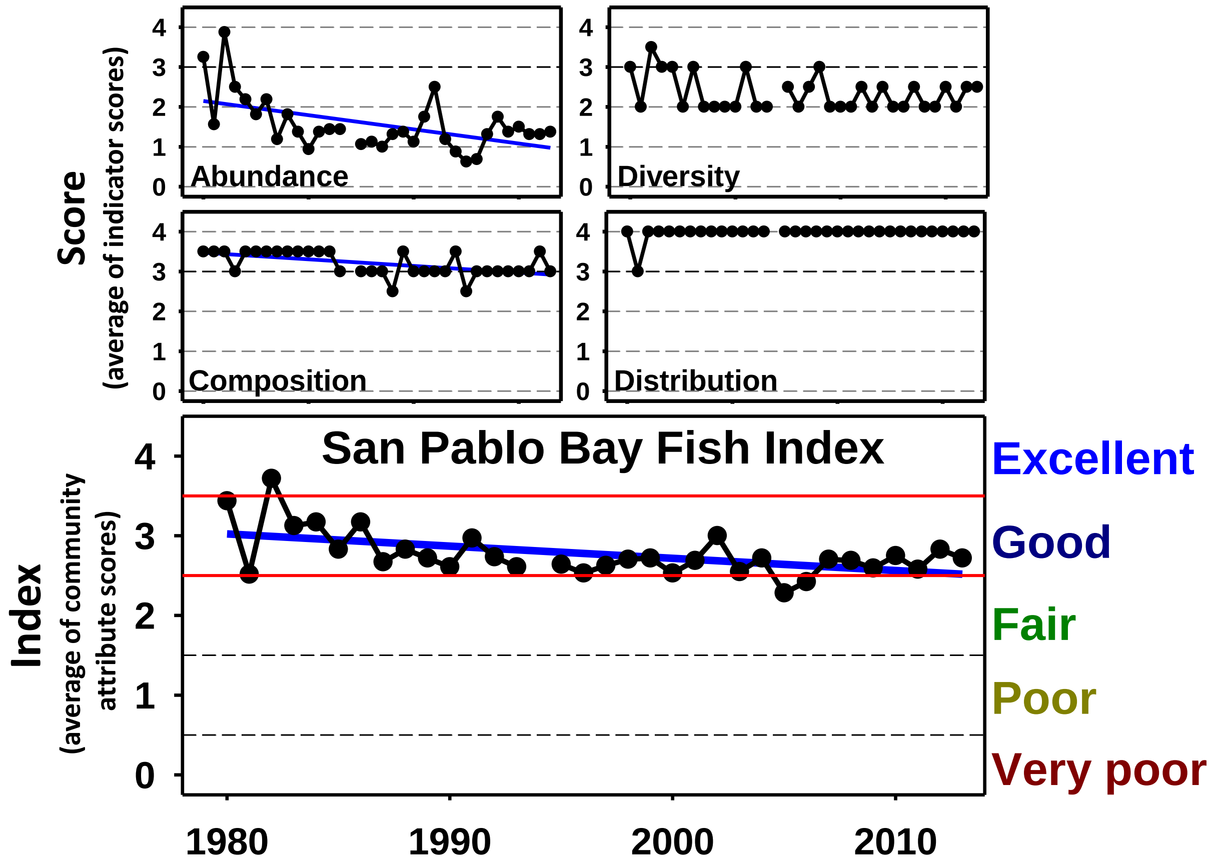

Figure 15. Results for the Bay Fish Index for San Pablo Bay for 1980 to 2013. The top four panels show results for the four fish community attributes, expressed as average scores. The large bottom panel shows the Index, calculated as the average of the scores of the four component community attributes. For those results with a significant trend over time, the regression line is shown in blue. For the Index graph, the horizontal red lines and dashed lines show the reference conditions and Index evaluation.

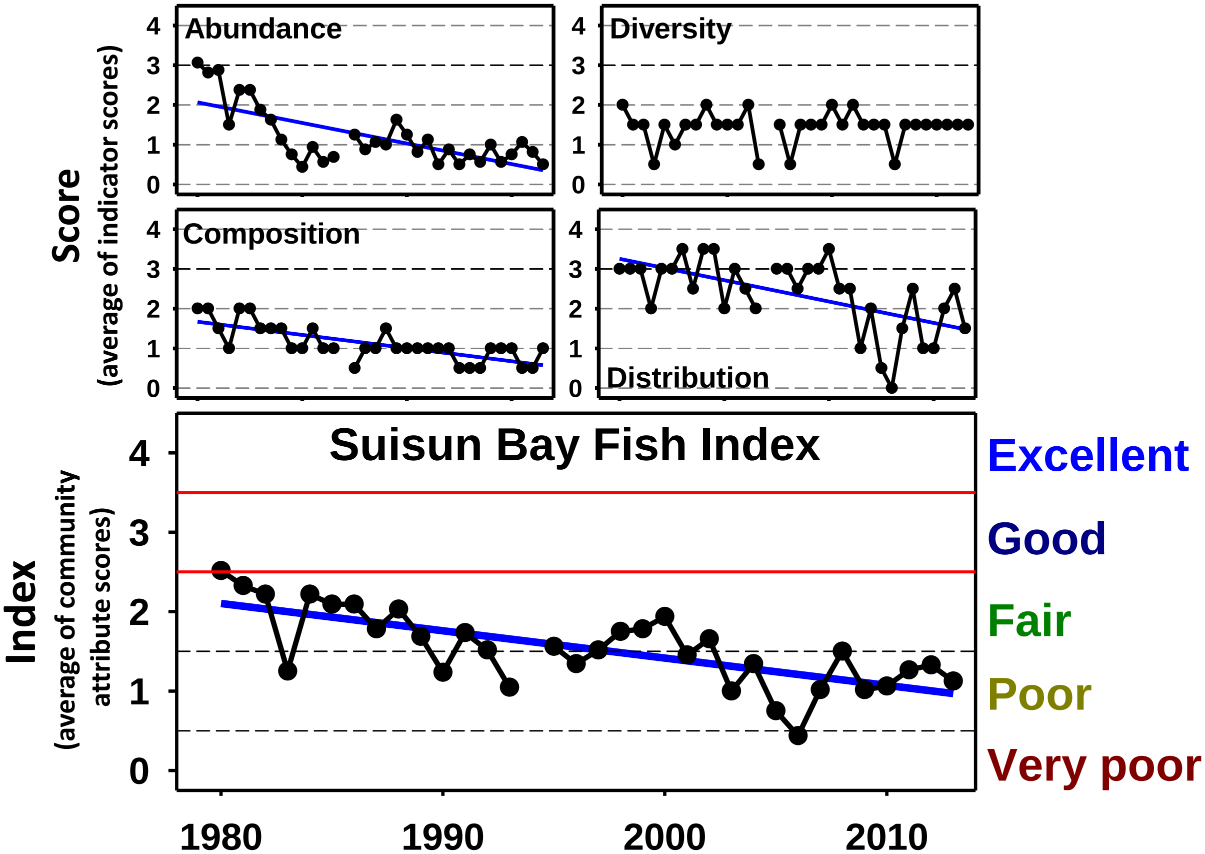

Figure 16. Results for the Bay Fish Index for Suisun Bay for 1980 to 2013. The top four panels show results for the four fish community attributes, expressed as average scores. The large bottom panel shows the Index, calculated as the average of the scores of the four component community attributes. For those results with a significant trend over time, the regression line is shown in blue. For the Index graph, the horizontal red lines and dashed lines show the reference conditions and Index evaluation.

Table 8. Summary of results for the ten Bay Fish indicators.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend since 1990**	**Current condition** (average for last 10 years)
Pelagic Fish Abundance	Not met in any sub-region	Stable at low levels	Fair to Very Poor
Northern Anchovy Abundance	Not met in any sub-region	Stable at low levels (Suisun, San Pablo) Declining (South, Central)	Fair to Very poor
Demersal Fish Abundance	Fully met (South and Central) Not met (San Pablo and Suisun)	Stable (Suisun) Increasing (South, Central, San Pablo)	Poor (Suisun) Fair to good (South, Central, San Pablo)
Sensitive Species Abundance	Not met on any sub-region	Stable at low levels	Poor (all sub-regions) Inflow reduced by 50%
Native Fish Diversity	Partially met (South) Not met (Central, San Pablo, Suisun)	Stable	Poor (Suisun) Fair to good (South, Central, San Pablo)
Estuary-dependent Fish Diversity	Fully met (South, Central) Not met (San Pablo, Suisun)	Stable	Poor (Suisun) Fair to good (South, Central, San Pablo)
Percent Native Species	Fully met (South, Central) Not met (San Pablo, Suisun)	Stable (South, Central) Declining (San Pablo Suisun)	Good (South, Central) Fair to Poor (San Pablo, Suisun)
Percent Native Fish	Fully met (South, Central, San Pablo) Not met (Suisun)	Stable	Good (South, Central, San Pablo) Very Poor (Suisun)
Pelagic Fish Distribution	Fully met (South, Central, San Pablo) Partially met (Suisun)	Stable (South, Central, San Pablo) Declining (Suisun)	Good (South, Central, San Pablo) Fair to Poor (Suisun)
Demersal Fish Distribution	Fully met (South, Central, San Pablo) Partially met (Suisun)	Stable (South, Central, San Pablo) Declining (Suisun)	Good (South, Central, San Pablo) Fair to Poor (Suisun)
Bay Fish Index	Fully met (South, Central and San Pablo) Not met (San Pablo, Suisun)	Stable (South, Central, San Pablo) Declining (Suisun)	Good (South, Central) Fair to Good (San Pablo) Poor (Suisun)

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State of the Estuary Report 2015

Technical Appendix Combined for WILDLIFE: Upper Estuary Fish and PROCESSES: Fish as Food

Fish Assemblage Health Indicators for the Upper San Francisco Bay Estuary, including Suisun Bay, Suisun Marsh, and Delta Technical Appendix

Alison Weber-Stover and Jonathan Rosenfield, Ph.D.
The Bay Institute

July 2015

State of the Estuary Report 2015

Fish Assemblage Health Indicators for the Upper San Francisco Bay Estuary, including Suisun Bay, Suisun Marsh, and Delta
Technical Appendix

Developed for the San Francisco Estuary Partnership

By

Alison Weber-Stover and Jonathan Rosenfield, Ph.D.
The Bay Institute

July 2015

I. BACKGROUND

Evaluations of "health" at the species or population level of biological organization require assessment of different attributes of viability, including abundance, diversity, spatial distribution, and productivity (McElhany et al. 2000). Although these attributes influence each other, they each reveal different and somewhat independent information about a populations' health. Developing conceptual analogs for these species-level attributes of viability can provide insight into the "health" of ecological communities and species assemblages. Tracking changes in and interactions among a suite of these indicators of assemblage health through time can increase understanding of fish assemblage dynamics and the drivers of those dynamics. Several fish-based indices have been developed to assess ecological quality of estuarine systems; indices commonly include species richness (diversity), abundance, fish condition, and nursery function (productivity) as metrics (Perez-Dominguez et al. 2011).

The San Francisco Estuary Partnership's State of the Bay report (2011) developed 10 indicators that reflected the health of the pelagic fish assemblage in the larger San Francisco Bay complex (including San Francisco Bay-proper, South San Francisco Bay, San Pablo Bay and Suisun Bay). Although the State of the Bay report (hereafter, SOTB 2011) developed indicators for Suisun Bay, it did not develop indicators of fish assemblage dynamics for many parts of the upper Estuary. The upper Estuary includes Suisun Marsh, the largest brackish marsh on the west coast of North America (CDWR 2014 – http://www.water.ca.gov/suisun/) and the Sacramento-San Joaquin Delta (hereafter, the "Delta"), a tidal freshwater region east of the confluence of California's two longest rivers. Together Suisun Marsh and the Delta comprise unique habitats in the largest estuary on the west coast of North America and serve as home to more than 55 species of fish. In the past 150 years major changes to the upper Estuary's habitats and patterns of freshwater flow have affected the region's fish assemblages (The Bay Institute 1998), as has introduction and invasion of this area by numerous non-native species (Matern et al. 2002; Light and Marchetti 2007).

SOTB (2011) synthesized pelagic fish sampling data from one long-term survey of the Bay's fish assemblage (the California Department of Fish and Wildlife's Bay Study) to develop indicators that portrayed long-term patterns in fish abundance, diversity, species composition, and spatial distribution from the Golden Gate to Suisun Bay. In addition, SOTB focused on indices of sub-strata of the fish assemblage (e.g., habitat guilds or trophic guilds) to gain further insight into ecological dynamics of the Bay and the forces driving those dynamics.

The Delta, Suisun Bay, and Suisun Marsh (collectively, the upper Estuary) are important habitats for native fish, including those that may inhabit the nearshore ocean, Bay, and/or Central Valley rivers during other parts of their life cycles. Here, indicators of native abundance and species composition (native vs. introduced) for the upper Estuary were developed for three major habitat types in this region – marsh, deep open water, and shallow, unvegetated waters – to compliment the Bay Fish Index from SOTB

(2011). These indicators enable evaluation of broad changes in fish abundance and species composition, two important attributes of the condition of the fish assemblage.

Fish also represent food to many species of birds, mammals, and other fish. Thus, the abundance of fish can be used as an indicator of foodweb productivity and food availability for piscivorous organisms. Here, abundance indices representing all fish (native and introduced) are developed as an indicator of food web productivity and overall ecosystem health.

The State of the Estuary report develops synthetic metrics of population dynamics and diversity (indicators) of the fish assemblage of the entire Estuary, including the embayments of the San Francisco Bay complex. Like its predecessor (SOTB 2011), the State of the Estuary Report presents fish indicators with the expectation that such indicators, correctly designed, can represent multi-species responses to major changes that have occurred in the Estuary and its watershed during the period for which sampling data are available. That said, it is important to recognize that no single indicator is capable of providing a full picture of “health” for ecosystems or even fish assemblages in any region of the Estuary; indeed, factors operating beyond the geographic area of the upper Estuary (e.g. the Central Valley or the nearshore ocean) certainly influence the abundance and diversity patterns described here. Additional indicators, focusing on other attributes of assemblage health, may be needed to relate ecological mechanisms local to the upper Estuary to patterns in the local fish assemblage.

Development of fish assemblage indicators for the upper Estuary was guided by the approach taken in SOTB (2011). Fidelity to that approach (as revised and updated) maximizes the potential to gain a comprehensive understanding of the fish assemblage dynamics across the Estuary as a whole. However, the dominant environments of the upper Estuary are very different physically from the brackish or near marine pelagic environments that dominate much of the San Francisco Bay complex that were the subject of SOTB (2011). The ratio of pelagic habitats to edge (littoral) plus bottom (benthic) habitats is much lower in the upper Estuary than in the San Francisco Bay complex as a whole; for example, the Delta-proper was historically dominated by myriad sloughs (which have now been simplified into a network of channels) that featured extensive shallow water habitat at their edges and productive benthic habitats as well. Because there is interest in restoring shallow, sub-tidal habitats and complex sloughs in the Delta (e.g., the Bay Delta Conservation Plan), measuring the health of the fish assemblage in the Delta should, to the extent possible, be sensitive to fish that specialize in these shallow, edge and bottom habitats. Also, Suisun Marsh, which neighbors the Delta-proper, is: (a) an ecosystem of great significance; (b) not covered by previous Bay indicators; and (c) somewhat representative of the types of habitats that once existed and may be restored in the Delta. Thus, it makes sense to add indicators of fish assemblage dynamics in Suisun Marsh to this section of the State of the Estuary report.

Why were these indicators chosen?

A suite of indicators of the Delta's fish assemblage was considered with the goal of capturing assemblage-level analogs to the species-level attributes of viability defined by McElhany et al. (2000). In order to be regarded as "healthy", fish assemblages in the upper Estuary should reveal good or excellent levels of:

- Abundance (numbers of native fish)
- Inter-specific diversity, including
    - number of species (richness)
    - distribution of abundance across species (diversity)
    - native species richness vs. non-native species richness
- Intra-specific diversity, including
    - life history diversity (e.g. time and size of migration, alternate life history strategies)
    - phenotypic and behavioral diversity
- Spatial distribution
- Productivity, including
    - life-stage specific survival rates
    - condition (weight/length, etc., e.g. Gartz 2005)

Indicators for most of these attributes have not been developed here, but there development in future iterations of this report is recommended.

In addition, we developed a metric of total fish abundance (native plus introduced species) as an indicator of food web productivity.

There are several challenges with interpreting available data for indicators of assemblage health. Several long-term data sets are available for the Delta (Table 1). For the purposes of indicator development, an ideal monitoring program would catch different age classes of all fish species with equal efficiency, over a wide spatial area, year-round, over a long time period, with consistent monitoring methods. No such sampling program exists – each of the existing programs was designed for particular purposes and not to measure or evaluate the health of the entire Delta fish assemblage. All the programs have different sampling biases specific to their respective programs (e.g. associated with sampling gear, detection probabilities, highly mobile species, as well as short- and long- term habitat variation). Even the San Francisco Bay Study (used in the SOTB 2011), which was designed to monitor the health of the entire fish assemblage, did not sample the entire spatial extent of the upper Estuary until recently. Also, this program only samples benthic and pelagic environments. With the exception of preliminary analyses done by the United States Fish and Wildlife Service (USFWS) Delta Juvenile Fishes Program, no monitoring programs have evaluated changes in detection probabilities over time (J. Kirsch, USFWS, personal communication).

To capture the range of different habitats sampled in the upper Estuary across the longest time-series possible, long-term data from three community sampling surveys were analyzed: California Department of Fish and Wildlife's Fall Midwater Trawl (FMWT), the US Fish and Wildlife Service's Juvenile Fishes Program (Beach Seine), and University of California at Davis's Suisun Marsh Fish Survey (Otter trawl). These are not the only sampling programs in the Delta but, taken together, these three sampling programs provide a geographically diverse view of fish assemblage abundance and diversity in a range of habitats over multiple decades (Tables 1 and 2, Figure 1).

Table 1. Comparison of several sampling programs for Upper Estuary Fish Indicators (information adapted from Honey et al. 2004)

Survey	**Period of Record** (colors = new stations added)	**Sampling time during the year**	**Geographic coverage** (colors correspond to "period of record" when new stations added)	**Habitat type sampled**	**Effectively samples body sizes**	**Consistent methods, gear, and locations**	**Sampling effective for:**	**Existing detection probability assessment**	**Other notes**
Fall Mid-water Trawl	1967 1990 1991 2009 2010	Sep-Dec	Western Delta Channels Edge of N. Sac Northern/eastern N. Sac Channel Cache slough	Nearshore channel, open water	>40mm	Generally	**Designed for**: Age-0 Striped Bass **Captures**: Juvenile pelagic	No	Limited to one season, changes in distribution could appear to be abundance changes.
SF Bay Study	1980 1998 1988, 1991, 1994	Year round	Entire estuary, limited sampling in the north, east and south Delta South Suisun Bay San Joaquin River Channel and Delta	Channel, open water & benthic	>40mm	Some sampling missing from late '80s to early 90's	Two gears deployed **Designed for**: Fish and invertebrate assemblage **Captures**: Variety, otter trawl samples demersal fish, in open water	No	Does not sample the northern, eastern and southern Delta well.
Summer Townet	1959 2011 2009	June and then flexible ~August	Southern Delta well, Added channel in north Same as 2011 (2010 skipped)	Benthic	<390 mm Larval fish, juvenile delta smelt	Timing different, gear the same	**Designed for:** age 0 Striped Bass **Captures**: Pelagic, young striped bass	No	Irregular start and end dates, short sampling period in summer.
Salvage	1957 - Tracy 1968 - Skinner	Year round	Two locations South Delta	NA	Juvenile to adult of some species	Yes	**Designed for:** Enumerating entrainment, medium to large fish	No	Single location sampling, dependent on water export, not all fish identified.
Suisun Marsh Fish Survey	1980 1994	Year round	Suisun Marsh eastern Suisun Marsh	Benthic, marsh	Juvenile to adult of some species	Some change in sites, methods and gear relatively consistent	**Designed for:** Marsh habitat, demersal fish **Captures:** May capture pelagic fish in some sloughs	No	Problems with large and small sloughs for pelagic fish.
Delta Juvenile Fish Sampling	1976. 1990's 2002	Year round (more consistent after 1995)	Entire Delta Larger extent Site on the San Joaquin	Littoral zone, floodplain, open water in three locations	<25 mm Juvenile to Adult of some species (smaller fish than 25mm caught, but ID suspect)	Number of locations changed, methods generally consistent	**Designed for:** Salmon fry and cyprinids **Captures:** Most small to medium sized fish (<~150mm) in the littoral zone	Yes (not published)	Year round only since 1992 Boat ramp sites may bias results, problems with inter-annual comparisons of catch trends ID of fish less than 25mm suspect

Table 2. Sampling programs used as data sources for calculation of for Upper Estuary Fish Indicators in different regions and habitats of the Delta and Suisun Marsh.

<table>
<tr><td></td><td colspan="3">Habitats</td></tr>
<tr><td>Region</td><td>Marsh/Demersal</td><td>Pelagic</td><td>Littoral</td></tr>
<tr><td>Suisun Marsh</td><td colspan="2">UC Davis Suisun Marsh Fish Survey (Otter Trawl)</td><td></td></tr>
<tr><td>Suisun Bay</td><td></td><td rowspan="2">CDFW Fall Midwater Trawl</td><td></td></tr>
<tr><td>Central-Western Delta</td><td></td><td rowspan="3">U.S. Fish and Wildlife Service Delta Beach Seine</td></tr>
<tr><td>Northern Delta</td><td></td><td></td></tr>
<tr><td>Southern Delta</td><td></td><td></td></tr>
</table>

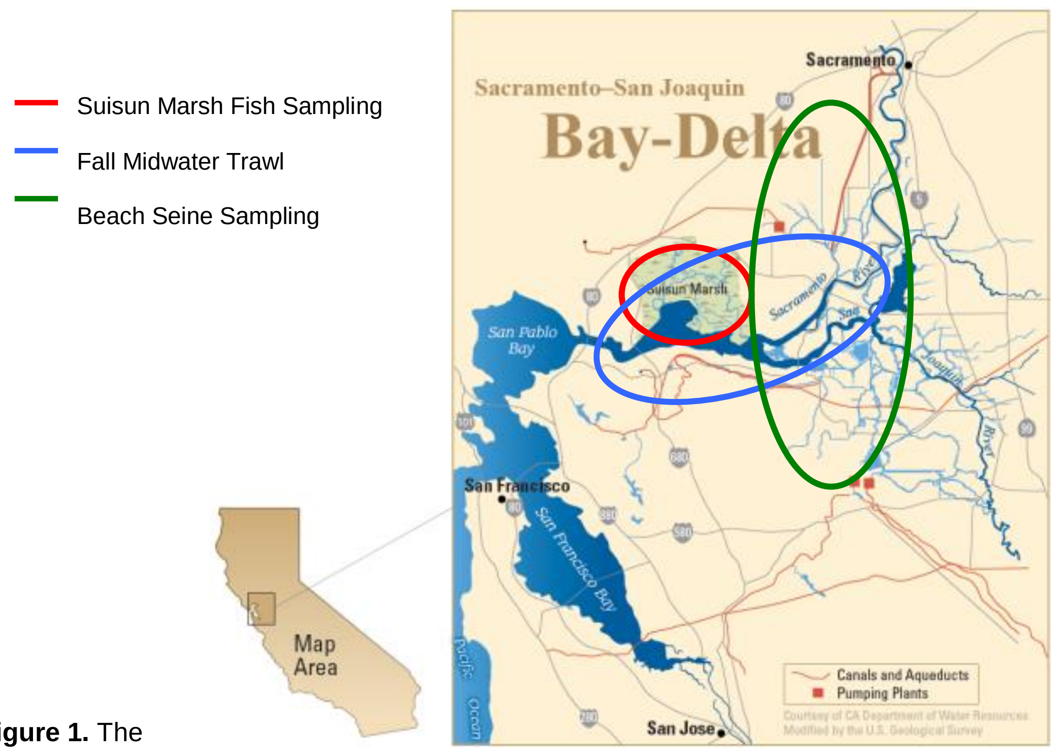


Figure 1. The Sacramento-San Joaquin Bay-Delta is where Central Valley Rivers meet the larger San Francisco Bay Estuary complex. Because the upper estuary is so large and contains a variety of habitats, the indicators of fish assemblage health in this area were calculated from three sampling programs that survey different habitats and regions of the upper estuary (Image accessed 1/12/14 at http://ca.water.usgs.gov/news/2012/SanFranciscoBayDeltaScienceConference.html).

We prioritized development of indicators of fish abundance and community composition for the upper Estuary (Table 6). Future iterations of the SOTER report should incorporate data from other long-term sampling programs. Data from additional sampling programs may help complete and unify the abundance and species composition indices presented here and they are necessary for developing additional indices that can link fish assemblage health in the upper Estuary to local ecosystem processes (e.g., productivity, spatial distribution, guild-specific evaluations, etc.).

The SOTB (2011) provided fish abundance indicators for pelagic, demersal, and sensitive fish species. Additionally, these indicators were measured separately within each of four regions. Here, separate indicators of abundance and assemblage diversity were produced for marsh species, pelagic species of the Delta's open channels, and littoral species in Suisun Marsh and the Delta-proper. Where appropriate, within each sampling program/habitat type, separate indices were produced to characterize sub-regions designated by the Interagency Ecological Program (IEP; Figure 2). Results for the different sub-regions were compared to determine whether data could be combined among regions within a sampling program (i.e. to determine whether regional trends were consistent). Due to the non-overlapping strengths and weaknesses of the different sampling programs available for this analysis (Table 1, Table 2), no effort was made to aggregate all indicators into a single index of fish assemblage health in the upper Estuary.

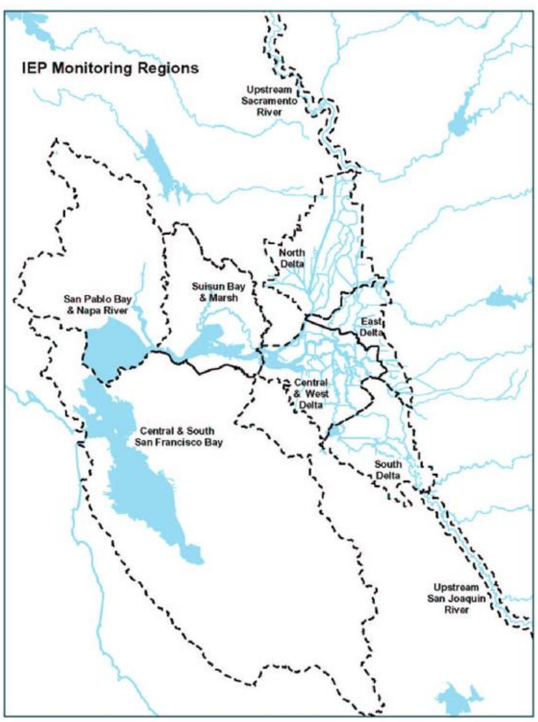


Figure 2. The Interagency Ecological Program's San Francisco Estuary Monitoring Regions (Figure from Honey et al. 2004, p. 6).

How were proposed indicators vetted with experts?
The methods used to calculate indicators of health for the fish assemblage of the upper Estuary were presented to, and sequentially peer-reviewed by, a group of experts in this region's fishes and fish sampling programs. Additional input was received from data administrators for the various sampling programs. A list of reviewers who provided input and direction through small group discussion, one-on-one discussions and written comment is provided below.

Name	**Agency/Organization**
Randall Baxter	California Department of Fish and Wildlife
Matt Dekar	United States Fish and Wildlife Service
Sam Harader	Delta Science Council
Daniel Huang	Delta Science Council
Kristopher Jones	California Department of Water Resources
Joseph Kirsch	United States Fish and Wildlife Service
Teejay O'Rear	University of California, Davis
Ted Sommer	California Department of Water Resources
Jonathon Speegle	United States Fish and Wildlife Service
Hildie Spautz	California Department of Fish and Wildlife
Christina Swanson	Natural Resources Defense Council
Susie Tharatt	United States Fish and Wildlife Service
Darcy Austin	Delta Stewardship Council

II. DATA SOURCES

Suisun Marsh abundance and species composition indicators.
Suisun Marsh Fish Survey (Otter Trawl, UCD).
Suisun Marsh indicators were calculated with data collected by the Suisun Marsh Fish Survey. The survey has been conducted monthly since 1979 in Suisun Marsh, sampling 17 sites consistently since 1980 (Figure 3, Tables 1 and 3); four additional sampling locations (which were not sampled as consistently in early years) were included in the data set as they provided greater spatial coverage, but did not materially affect long-term trends in catch-per-unit-effort data (T. O'Rear, personal communication). An otter trawl was used to sample benthic fish across the spatial extent of the Marsh in large and small sloughs; net tows in large sloughs lasted for 10 minutes and in small sloughs, for 5 minutes (https://watershed.ucdavis.edu/project/suisun-marsh-fish-and-invertebrate-study). Because the size of the net (1m x 2.5m opening) was large relative to the width and depth of some sloughs it samples, the Suisun Marsh Fish Survey may sample most of the water column in some areas – thus, these data provided a relatively good indication of fish occupying open water habitats in smaller Marsh sloughs.

This sampling program provided data from a critically important ecosystem, adjacent to the Delta-proper that is included in many discussions of "Delta" habitat restoration (e.g. the Bay Delta Conservation Plan). The habitats present in the Marsh, though modified, are similar to those that would have existed in the historical Delta and those that may be restored in a future Delta. The Suisun Marsh Fish Survey has been particularly effective at sampling native species that rely on shallow, marsh habitats (e.g., splittail (*Pogonichthys macrolepidotus*), tule perch (*Hysterocarpus traski*) and at detecting new invaders to the estuary ecosystem (Matern et al. 2002). Thus, data from this system are critical to any long-term assessment of the upper Estuary's fish assemblage. On the other hand, the Suisun Marsh Survey did not provide a comprehensive image of the Delta fish assemblage's health because it only sampled in the Marsh and therefore focused on species that are common in marsh slough habitats. Also, like any fish community sampling program, the Suisun Marsh Survey gear and methodology only reliably captured fish within a particular size range (generally ~35mm-250mm).

Figure 3. Locations of stations that have been sampled consistently by UC-Davis' Suisun Marsh Fish Survey. Map created by Amber Manfree. Fish assemblage indicators for Suisun Marsh were calculated from the Suisun Marsh Fish Survey data.

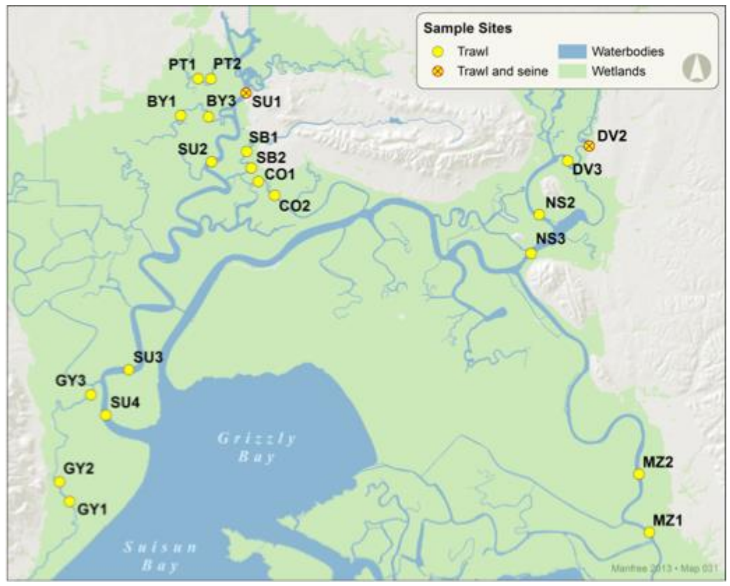


Table 3. Suisun Marsh Fish Survey sampling stations and total numbers of surveys for the 1980-2013 period of record used to calculate indicators (data from UCD Suisun Marsh Fish Survey Otter Trawl; provided by T. O'Rear). Catch per trawl indicators were based on data from 21 sites (despite the fact that only 17 were sampled consistently) following the reporting protocol of the Suisun Marsh Survey. Annual trends in CPUE are not affected by the inclusion of the four sites that were sampled less consistently (T. O'Rear, personal communication).

Region	Sampling Stations	Number of Surveys
Suisun Marsh	BY1, BY3, CO1, CO2, DV2, DV3, GY1, GY2, GY3, NS2, NS3, MZ1, MZ2, PT1, PT2, SB1, SB2, SU1, SU2, SU3, and SU4	8,403

Beach Zone abundance and species composition indicators.

Delta Juvenile Fishes Program (Beach Seine, USFWS).

This survey program sampled littoral habitat throughout the spatial extent of the Delta-proper, throughout the year (Figure 4, Table 1 and 4). Fish were caught in a seine that was 15.2m wide, pulled manually through shallow water (<1.3m) areas that had little bottom vegetation or obstructions (http://www.fws.gov/stockton/jfmp/Docs/Data%20Management/1214/Metadata%20(Upd ated%20September%2009,2014).doc). These habitats, and fish that specialize in them, are usually sampled ineffectively by gear towed behind a boat. Data were collected weekly or bi-weekly since 1976. Because year-round, monthly sampling became consistent in 1995, only data from 1995 onward were used in constructing indicator time trends from this data set. In order to develop a comprehensive image of dynamics in the Delta's fish assemblage, findings from this survey must be considered in the context of other surveys because sampling only occurred in the littoral zone and the gear (like all gear) captured fish efficiently only within a certain (species-specific) body size range (generally ~30mm-200mm).

Figure 4. Sampling station locations of the USFWS Beach Seine Survey used to calculate Delta Beach Zone fish indicators. Only 1995-2013 data from four IEP regions, *North, East, South and Central-West) were used. Map from USFWS Delta Juvenile Fishes Program (http://www.fws.gov/stockton/jfmp/Docs/Data%20Management/12-14/Metadata%20(Updated%20September%2009,2014).doc).

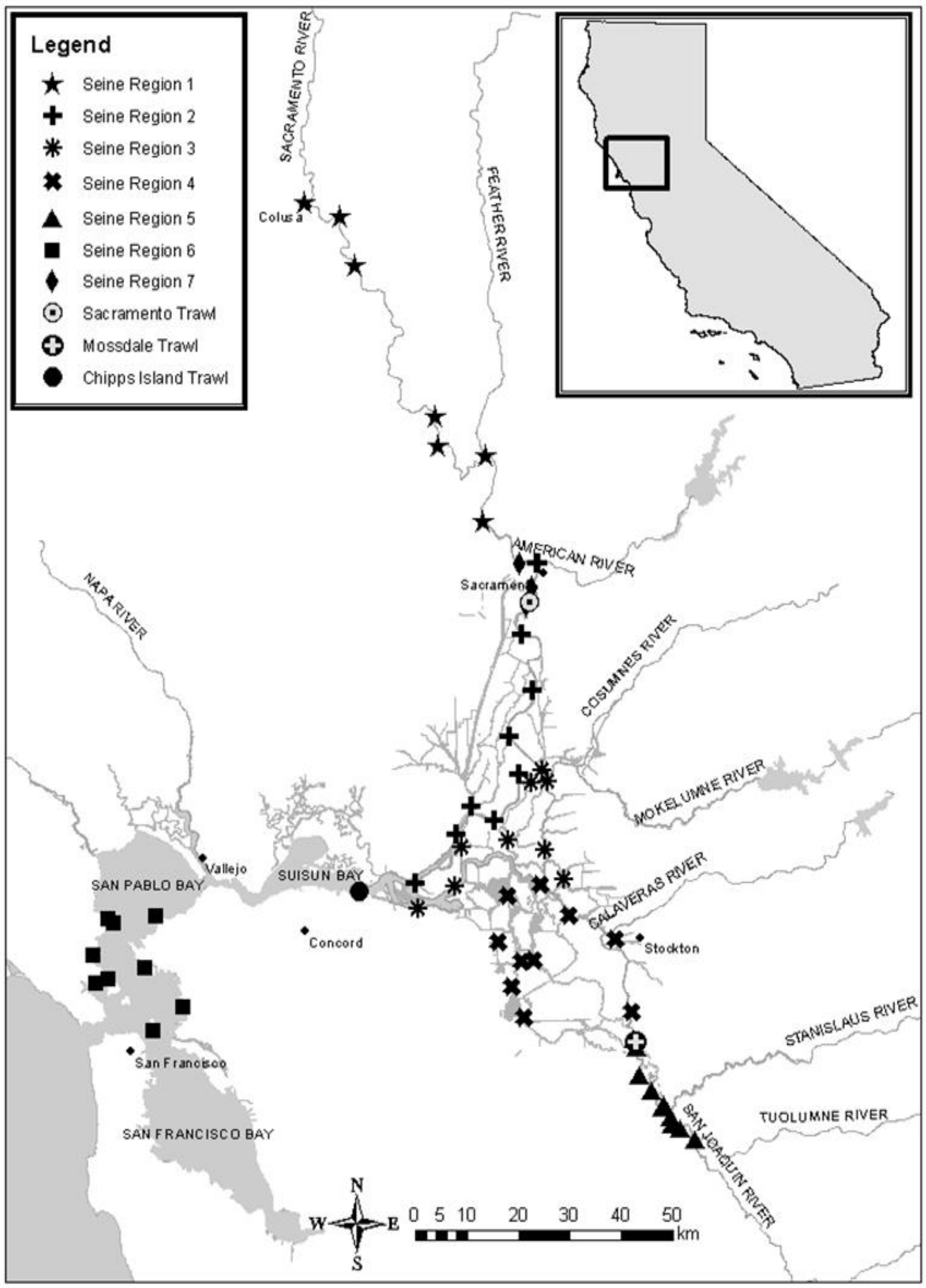


Table 4. Delta Beach Zone sampling stations and total numbers of surveys for the 1995-2013 period of record used to calculate the indicators (USFWS Delta Juvenile Fishes Program, Beach Seine Survey, data provided by J. Speegle). *Indicates that the station is a substitute location for a station that was not accessible at the survey time.

Regions from the Delta Beach Seine Survey	Sampling Stations	Number of Surveys (1995-2013)
North Delta	**SR043W** **SR049E** **SR057E** **SR014W** **SR062E** **SR055E** **SR055A*** **SS011N**	6832
East Delta	**XC001N** **GS010E** **SR017E** **DS002S** **SR024E** **LP003E** **SF014E**	5900
South Delta	**SJ063W** **SJ063E*** **OR014W** **SJ041N** **SJ051E** **SJ068W** **SJ072E*** **SJ070N*** **OR003W** **SJ032S** **SJ026S** **SJ056E** **OR019E** **OR001X*** **SJ074W** **SJ074A*** **OR023E** **WD002W** **WD002E*** **SJ058W** **SJ058A*** **SJ058E***	7951

	MR010W **MR010A*** **SJO56E**	
Central-West Delta	**SJ001S** **MK004W** **TM001N** **SJ005N** **SR012W*** **MS001N** **MS001A*** **SR012E**	5023

Upper Estuary Pelagic Zone abundance and species composition indicators.
*Fall Midwater Trawl (midwater trawl, CDFW)***.**
This survey sampled open-water, pelagic species in the upper Estuary (San Pablo Bay to the western Delta) every month from September through December at fixed sampling locations (Figure 5; Table 1 and Table 5). Methods were relatively consistent over a long time period (since 1967); however, within the upper Estuary, many new sites were added since 1967. In addition, because the Fall Midwater Trawl (FMWT) only sampled during one season and did not sample littoral or benthic habitats that form a relatively large proportion of available space for fish in the upper Estuary, these data did not present a comprehensive picture of the entire fish assemblage in this region. On the other hand, the fact that the FMWT sampled pelagic waters of Suisun Bay and the Central-West Delta for such an extended period means that these data provided an excellent complement to results for Suisun Bay recorded by the Bay Study (e.g., this State of the Estuary Report; SOTB 2011).

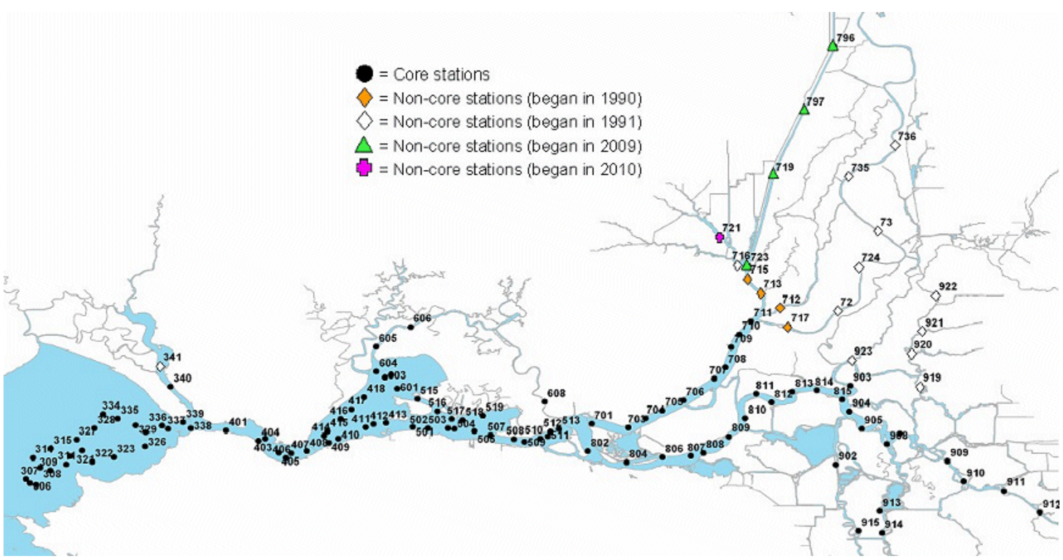


Figure 5. Locations of the sampling stations for the CDFW Fall Midwater Trawl survey used to calculate the Upper Estuary Pelagic Zone Fish Indicators. Only data from core stations, collected 1967-2013, in Suisun Bay and the Central-West Delta were used for calculations (Map from http://www.dfg.ca.gov/delta/data/fmwt/stations.asp).

Table 5. Sampling Stations and total numbers of surveys for the 1967-2013 period of record used to calculate Pelagic Zone Indicators (data from CDFW Fall Midwater Trawl, accessed at ftp://ftp.dfg.ca.gov/).

Regions from Upper Estuary Open Water	Sampling Stations	Number of Surveys (1967-2013)	Years Excluded from Analysis for Partial Sampling
Suisun Bay	401, 403-418, 501-505, 507-513,515-519, 601-606, 608	6376 (1967-2013)	1969-1972 and 1976 (Limited sampling) 1974 and 1979 (no sampling)
Central and West Delta	701, 703-711, 802, 804, 806-815, 902-906, 908-915	5280 (1967-2013)	1969 – 1973, 1975 and 1984 (Limited sampling) 1974 and 1979 (no sampling)

III. INDICATOR EVALUATION

Evaluating indicator trends in ecosystem health requires establishing reference conditions (*what value was the indicator in the past?),* designating thresholds (*what would be considered "good" or "poor"?),* and assessing the significance of any trends (*how does the current condition compare to the established thresholds*; Perez-Dominguez et al. 2011). References conditions may include "primary" reference conditions that reflect indicator status in a known historical period (SOTB 2011) or aspirational objectives – specific, measureable, achievable, relevant, and time-bound (S.M.A.R.T.) articulations of recovery goals. The San Francisco Estuary Partnership's Comprehensive Conservation and Management Plan (CCMP, SFEP 2007) calls for "recovery" and "reversing declines" of estuarine fish and wildlife but does not provide quantitative objectives that would allow for indicators to be referenced to desired outcomes. Thus, the indicators developed here are benchmarked to "primary reference conditions" (SOTB 2011) calculated from historical data. The primary reference conditions provide a scale against which improvement or deterioration can be evaluated. Identification of a primary reference condition does not indicate that such a condition is the desired state for the Estuary's fish assemblage; rather it provides a retrospective baseline with which one can evaluate the direction and relative magnitude of change.

For each indicator, primary reference conditions were established based on the earliest data available for each of the sampling programs studied, maximum measured values for the upper Estuary or sub-region, recognized and accepted interpretations of ecological conditions and ecosystem health (e.g., native versus non-native species composition), and/or best professional judgment. Wherever possible, indicator scoring was accomplished using methods equivalent or parallel to those used in SOTB (2011). In the case of abundance indicators, scores were calibrated to account for differences in absolute values of indicators among the sampling programs or sub-regions. The reference conditions for the assemblage composition indicators were based on the ecological relationship between the prevalence of non-native species and ecosystem and habitat condition (SOTB 2011). For these assemblage composition indicators, the value of the reference condition associated with a particular score (e.g., “good”, “poor”) was maintained in the upper Estuary at the same level as identified in SOTB (2011).

Following SOTB (2011), five intermediate reference conditions were created to provide a scale for assessing deviations from the primary reference condition. In order to ensure that the different levels represented meaningful differences in the measured indicator values, the range of indicator values assigned to each intermediate reference conditions was based on observed levels of variation in the measured indicator values. For each indicator, an assessment of current status was based on indicator trends and the average score of the most recent 5 years of the data set.

IV. INDICATORS

The following indicators were calculated for three regions of the Upper Estuary.

Table 6. Fish community characteristics and indicators calculated.

Fish Community Characteristics	Indicators
Abundance (Natives)	• Suisun Marsh native fish abundance • Pelagic Zone native fish abundance Regions: Central-West Delta and Suisun Bay • Beach Zone native fish abundance Regions: North, South, East, Central-West Delta
Species composition	• Percent Native Fish • Percent Native Species
Food Web Productivity (All fish)	• Suisun Marsh sum of standardized total fish abundance • Pelagic Zone sum of standardized fish abundance Regions: Central-West Delta and Suisun Bay • Beach Zone sum of standardized fish abundance Regions: North, South, East, Central-West Delta

A. Abundance Indicators

1. Rationale

The most obvious measure of fish abundance is a simple index of the number of fish caught. Abundance of native fish can be an indicator of aquatic ecosystem health (see full explanation in the State of the San Francisco Estuary Report Bay Fish Technical Appendix 2015 and Wang and Lyons 2003, Harrison and Whitfield 2004).

Because the Estuary’s fish assemblage is influenced by processes affecting fish production elsewhere (upstream in the Central Valley’s rivers or in the nearshore ocean), caution should be used in relating these abundance indices to local ecosystem processes. Additional indicators (e.g. spatial distribution, survival/productivity) will be useful for connecting trends in fish abundance to ecological drivers occurring within the Delta. For example, we constructed species composition indicators, which highlight the proportion of native to non-native species, to compliment the total abundance indicators. Studying both trends in native fish abundance and assemblage composition may help to reveal ecological changes underlying changes in total abundance. This approach tracks that employed by SOTB (2011) for its abundance indicators.

Limitations and future amendments to the abundance indicators
Catch-per-unit-effort (e.g. fish/trawl, fish/volume) is a measure of fish abundance that standardizes, within sampling programs and habitats, for variation in sampling effort across years. Use of this density metric as an indicator of total abundance relies on numerous assumptions. For example, use of the CPUE metric assumes that the density measured by the sampling program is representative of an "average" density across the region and habitat being sampled; if fish are more or less aggregated around sampling stations than they are throughout the area represented by those sampling stations, the relationship of CPUE to total abundance may be inaccurate. This is especially true if sampling stations are not chosen randomly for each sampling set or across years, as is the case with most fish sampling programs in this estuary. Also, average CPUE for all fish says nothing about the type of fish being caught, nor fish biomass. Because these are synthetic indicators, they also obscure particular relationships and trends that are occurring within sub-sets of the fish assemblage (e.g. individual species trends). Finally, as mentioned above, changes in indicators are not necessarily indicative of mechanistic drivers *within* the region being sampled, as migratory fish species' populations may be responding to conditions elsewhere in their life cycle. However, fish density (and abundance) does represent a snapshot of conditions experienced by fish and other species (e.g. fish predators, anglers, etc.) in the sampling zone at a given time. Therefore, CPUE metrics present a partial picture of system health.

Future iterations of the SOTER should consider creating separate abundance indices for different ecological guilds (e.g., resident, nursery dependent, migratory fish, or sensitive species) to provide a more focused view of population trends within these different ecological groups. Our division of abundance into native vs. non-native species (see Food Web Productivity section) is on example of the additional information to be gained by studying subsets of the entire assemblage. Indicators that would present a more comprehensive view of ecosystem health when combined with abundance and diversity indices should be explored. For example, indicators of within Delta survival and spatial distribution may provide greater insight into local ecosystem processes affecting fish distribution. Also, measuring abundance as biomass would more accurately represent fish productivity and carrying capacity in the sampling zone.

2. Methods and Calculations, Assumptions, and Uncertainties

The SOTB (2011) methodology for constructing fish abundance indicators was applied wherever possible to each of the data sets (representing different sampling programs and major habitats). Differences among the sampling programs required some modification of methods for each sampling program and are explained below.

Suisun Marsh Fish Abundance Indicator
The Suisun Marsh Abundance Indicator was calculated as catch per trawl for each year (1980-2013):

fish/trawl = [native fish caught in year-x]/[trawls year-x]

The monitoring program does not estimate the volume of habitat sampled but has maintained a relatively consistent sampling protocol over the sampling period; thus, standardizing effort by the number of trawls was deemed appropriate (Matern et al 2002; T. O'Rear, personal communication, 2014). Data from sampling locations (n=17-21) that have been sampled throughout all or most of the sampling program (1980-2013) were used here (Table 4). While there are ecological gradients in the Marsh that might affect fish diversity and abundance (and the sampling program distinguishes between small sloughs and large sloughs), we analyzed the Marsh as one ecological unit without sub-regions.

Delta Beach Zone Fish Abundance Indicator
Delta Beach Zone Fish Abundance Indicators were produced for each of four, pre-determined IEP regions in the Delta (Figures 2 and 4). The sampling localities included in each region are identified in Table 4. Within each region, an abundance index was calculated as (1995-2013):

fish/10,000 m^3 = [native fish caught in year-x] / [total volume sampled in year-x] x(10,000)

The volume sampled was calculated as: (seine length x seine width x seine depth)/2 (http://www.fws.gov/stockton/jfmp/Docs/Data%20Management/12-14/Metadata%20(Updated%20September%2009,2014).doc). Because monthly sampling became routine in 1995, we constructed abundance indicators for only 1995-2014 using data from every month of the year. Native fish abundance in each of the Delta Beach Zone regions displayed broadly similar patterns (Figure 9); however, although the scores between regions were mostly well-correlated (Table x); the North Beach Zone patter was only marginally correlated with two other regions. As a result, the Native Fish Abundance Indicator was scored and displayed separately for each region of the Delta.

Upper Estuary Pelagic Zone
Upper Estuary Pelagic Zone Abundance Indicators were calculated using data from the Fall Midwater Trawl program, which samples fixed stations in the upper Estuary from September-December (Figure 5; Stevens 1977). We divided sampling stations into two IEP regions, Suisun Bay and the Central-West Delta and calculated a separate indicator for each region; sampling results from San Pablo Bay were excluded from our analyses. Sampling locations in each region are identified in Table 5. Within each region, an abundance index was calculated as (1967-2013):

fish/10,000 m^3 = [(native fish caught in year-x)/(total trawls in year-x * tow volume m^3)] *(10,000)

Sampling locations in the Delta-proper have been added to the FMWT several times over the program's existence (Table 1; Honey et al. 2004); however, in order to maximize the length of the time series, we restricted the sites used to create our abundance indicators to those that were sampled continuously in the years 1967-2013 ("Core 1" stations). Abundance indicators were not calculated in years where sampling effort (number of trawls) was much less (<68%) than the long-term modal average of trawls. Years included in our calculations are described in Table 5.

Total catch was divided by actual tow volume for 1985-2013 to produce a catch-per-unit-effort value for each year. Tow volume was not measured consistently for years prior to 1985; so, for this earlier sampling period annual catch was divided by the mean tow volume from the 1985-2013 period and, we also displayed annual catch by the 25th and 75th percentiles of 1985-2013 tow volume to bracket our estimated CPUE. Assumptions regarding average tow volume in the time series pre-1985 did not have any effect on scoring of this indicator (see, results section).

Cautions when interpreting results

The abundance indicators described above provide a measure of native fish assemblage health that is easy to understand and explain: *how many fish are caught for a given sampling effort*? However, such an indicator may not reveal the true state of the fish assemblage if the number of fish caught is dominated by one or a few species. In that situation, though the CPUE indicator is still of interest, it may reflect trends in the abundance of one species disproportionately, rather than trends in the assemblage as a whole.

A standardization method (described in the Food Web Productivity Indicator) was conducted for total fish abundance (native plus introduced species) for each data set and for native fish abundance in the Delta Beach Zone. There was no strong indication that one species was driving the trends observed in the Delta Beach Zone for native fish (standardized and raw CPUE values were highly correlated; p values were < 0.0, 0.0, 0.01 and 0.02 for North, East, South and Central-West respectively) or for total fish species in any region (see Food Web Productivity Indicator). Due to time constraints, we did not test whether *native* fish abundance (as opposed to total fish abundance) in Suisun Marsh and the Pelagic Zone was driven by fluctuations in one particular species; this approach is recommended for future iterations of the regional indices. However, there was no indication from the analyses of total fish abundance that one species was driving abundance patterns in those regions.

Reference Conditions

Wherever possible, the 1980-1989 average index value was used as the primary reference condition for abundance indicators. This is consistent with the Bay fish indicators (SOTB 2011). In the SOTB (2011), the 1980-1989 average is considered "good", recognizing that some fish populations were already in decline by the 1980's. A five-tier scale rates annual average CPUE over time from “very poor” to “excellent”. Any individual year in the record may be compared to the reference condition and scored.

Suisun Marsh

The 1980-89 average catch per trawl was established as the primary reference condition for this data set. These were the earliest years for which data was available. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions as described in Table 7.

Table 7. Quantitative reference conditions and associated interpretations for the Suisun Marsh Fish Abundance Indicator. The average score during the primary reference period, which corresponds to “good” conditions, is in bold and all other reference conditions are calculated from that value (e.g. “excellent” is 150% of the 1980-1989 value).

Abundance Indicators Suisun Marsh Catch Per Effort (Data: UCD Suisun Marsh Fish Survey, Otter Trawl)			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
>150% of the 1980-1989 Average	Excellent	>28.71	N/A
>100% of the 1980-1989 Average	**Good**	**>19.1**	**28.7**
>50% of the 1980-1989 Average	Fair	>9.57	19.0
>15% of the 1980-1989 Average	Poor	>2.87	9.56
<15% of the 1980-1989 Average	Very Poor	N/A	<2.87

Delta Beach Zone

The Beach Zone was not consistently sampled year-round until 1995. Thus, average catch per effort from 1995-2004 was established as the primary reference condition for the Delta Beach Seine sampling program. The primary reference condition, during this period was assigned a “poor” score to match the average score of the Suisun Marsh and Pelagic Zone abundance indicators during the same period. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions. Evaluation thresholds for these scores are described in Table 8.

Table 8. Quantitative reference conditions and associated interpretations of the results of the Delta Beach Zone fish abundance indicator. For each region in the Delta, the average of the primary reference condition, which corresponds to "poor" conditions, is in bold. The primary reference condition was rated "poor" to correspond to scores for the Pelagic and Marsh abundance indicators during the 1995-2004 period.

Abundance Indicators Delta Beach Zone Catch Per Effort (Data: USFWS Delta Juvenile Fishes Program, Beach Seine Survey)			
North Delta			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
> 150% of Good	Excellent	> 27976	NA
> (1995-2005 Average / 15%)	Good	> 18650	27976
> 50% of Good	Fair	> 9325	18650
> **1995-2005 Average**	**Poor**	**> 2798**	**9325**
< 1995-2005 Average	Very Poor	< 2798	NA
East Delta			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
> 150% of Good	Excellent	> 27127	NA
> (1995-2005 Average / 15%)	Good	> 18084	27127
> 50% of Good	Fair	> 9042	18084
> **1995-2005 Average**	**Poor**	**> 2713**	**9042**
< 1995-2005 Average	Very Poor	< 2713	NA
South Delta			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
> 150% of Good	Excellent	> 9619	NA
> (1995-2005 Average / 15%)	Good	> 6412	9619
> 50% of Good	Fair	> 3206	6412
> **1995-2005 Average**	**Poor**	**> 962**	**3206**
< 1995-2005 Average	Very Poor	< 962	NA
Central-West Delta			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
> 150% of Good	Excellent	> 19852	NA
> (1995-2005 Average / 15%)	Good	> 13235	19852
> 50% of Good	Fair	> 6617	13235
> **1995-2005 Average**	**Poor**	**> 1985**	**6617**
< 1995-2005 Average	Very Poor	< 1985	NA

Pelagic Zone of the Upper Estuary

The 1980-89 average catch per effort was established as the primary reference condition for this data set. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions as described in Table 9.

Table 9. Quantitative reference conditions and associated interpretations for the results of the Upper Estuary Pelagic Zone Fish Abundance Indicator. The average during the primary reference condition, which corresponds to "good" conditions, is in bold.

Abundance Indicators Pelagic Zone Catch Per Effort (Data: CDFW Fall Midwater Trawl)			
Central-West Delta			
Quantitative Reference Condition	Interpretation	Low End of Range	High End of Range
>150% of the 1980-1989 Average	Excellent	>11.8	NA
>100% of the 1980-1989 Average	**Good**	**>8**	**11.8**
>50% of the 1980-1989 Average	Fair	>4	8
>15% of the 1980-1989 Average	Poor	>1.2	4
<15% of the 1980-1989 Average	Very Poor	NA	<1.2
Suisun Bay			
Quantitative Reference Condition	Interpretation	Low End of Range	High End of Range
>150% of the 1980-1989 Average	Excellent	>155	NA
>100% of the 1980-1989 Average	**Good**	**>103**	**155**
>50% of the 1980-1989 Average	Fair	>52	103
>15% of the 1980-1989 Average	Poor	>15	52
<15% of the 1980-1989 Average	Very Poor	NA	<15

3. Abundance Results

Suisun Marsh

Native fish abundance in Suisun Marsh declined over the period of record (Figure 6). Levels detected in the first few years of the survey were "excellent" or "good", but became consistently "fair" or "poor" during the late 1980's and early 1990's. Over the last five years conditions the indicator was "poor".

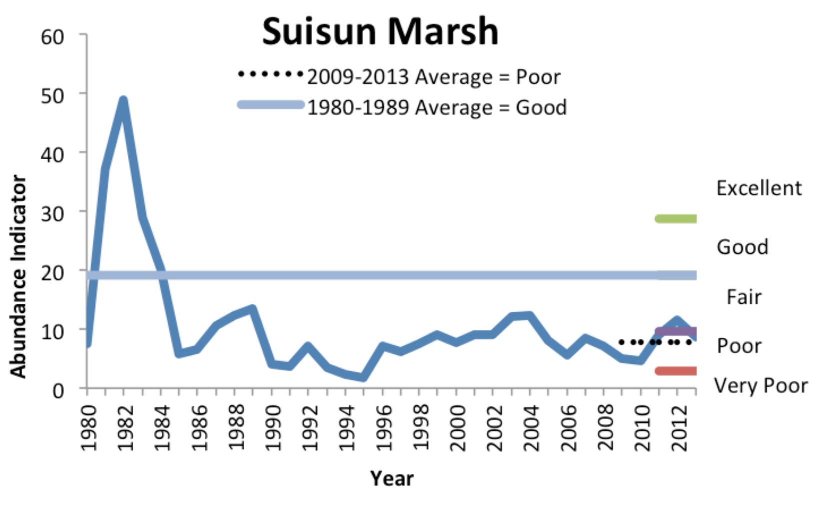


Figure 6. Suisun Marsh Fish Abundance Indicator from 1980-2013. Over the period of record the abundance indicator has declined and the recent five-year average is “poor”. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 7). The primary reference condition (1980-1989 average), indicated by a light blue horizontal line, represented a “good” score. The dotted line, representing the 2009-2013 average, reveals that Suisun Marsh fish abundance is “poor”.

Upper Estuary Pelagic Zone

Native fish abundance in the Pelagic Zone has declined dramatically over time, with recent averages that were very poor. Small differences were detected in the native fish assemblage abundance patterns between the two regions sampled in the Pelagic Zone – Suisun Bay (Figure 7) and the Central-West Delta (Figure 8). Although native fish abundance indicators in both regions declined dramatically, they displayed different patterns of decline. The abundance indicator in Suisun Bay followed a trend that was broadly similar to that seen in Suisun Marsh abundance; abundance of native fish scored “excellent” in the early years of the survey and even in the earliest years of the primary reference period (1980-1989). However, scores declined rapidly just prior to the onset of the 1987-1994 drought. A small rebound in abundance was detected in the late-1990’s, but the indicator declined persistently through the early 2000’s. The average of the last five years indicates that the native fish assemblage in this region/habitat was in “very poor” condition.

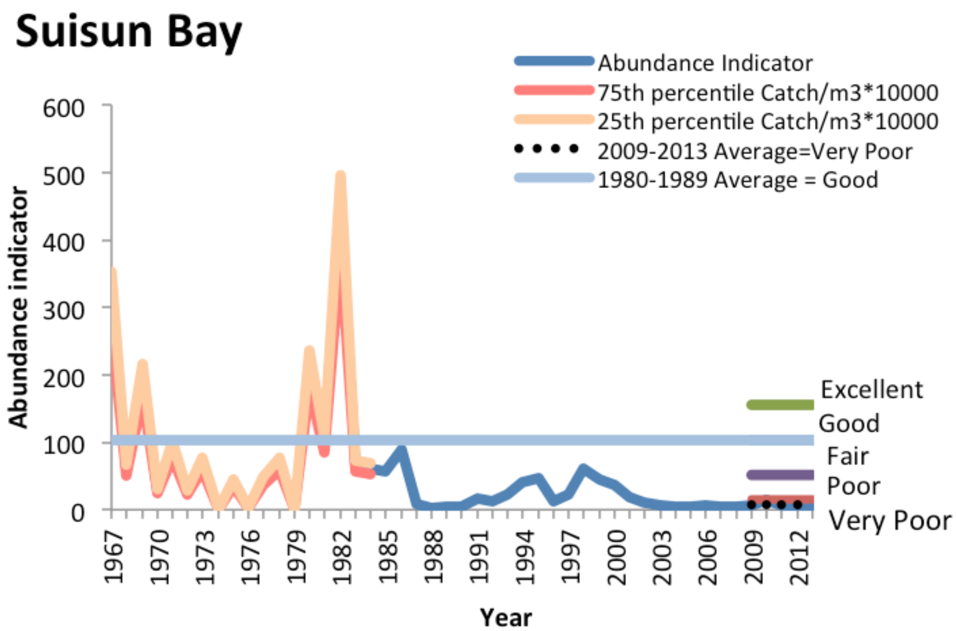


Figure 7. Upper Estuary Pelagic Zone Native Fish Abundance Indicator for the Suisun Bay region from 1967-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 9). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Native fish abundance in the Pelagic Zone of Suisun Bay is "very poor". Volume sampled was not recorded consistently during 1967-1984 period; thus, for this period, volume sampled was estimated as the mean volume from 1985-2013. Catch-per-unit-effort (i.e., per volume) was also estimates using the 25th and 75th percentile values of volume sampled between 1985-2013; the effect of different sampling volume estimates are shown in peach and pink lines respectively.

Abundance trends in the Central-West Delta Pelagic Zone are different in degree from those described for the Suisun Bay Pelagic Zone and Suisun Marsh. Here, the abundance index appeared to be somewhat stable throughout the 1980's and early 1990's. Both, the increase in the late 1990's (to "excellent") and the precipitous decline in abundance after the early 2000's were consistent with patterns seen in Suisun Bay and Suisun Marsh. The average of the most recent five years indicated that the pelagic fish assemblage in this area is in "very poor" condition.

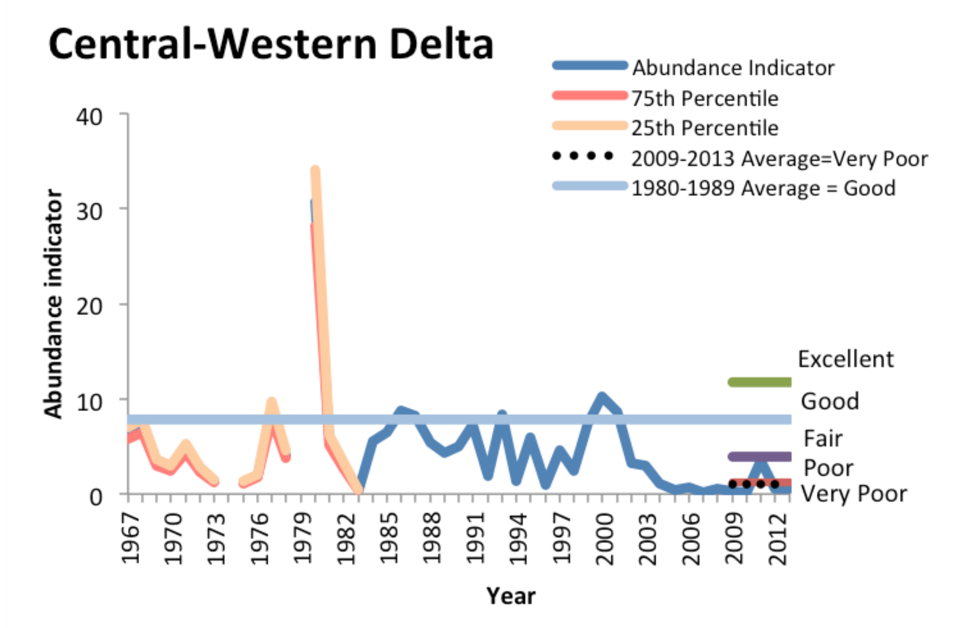


Figure 8. Upper Estuary Pelagic Zone Native Fish Abundance Indicator for the Central-West Delta region from 1967-2013. There has been a rapid decline in native fish abundance since the year 2000. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 9). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average and shows that native fish abundance in the Pelagic Zone of the Central-Western Delta is "very poor".

Delta Beach Zone

Native abundance trends in the Delta Beach Zone were similar in four regions. Trends in native fish abundance in were similar in the North, East, South and Central-West Delta Beach Zone, although the East region displayed greater peaks in abundance (Figure 9). Still, Delta Beach Zone region scores are plotted separately for greater resolution of patterns within the individual regions; a combined score for the Delta Beach Zone as a whole (not shown) produced similar patterns and current scores as the regions considered separately.

Abundance of native fish species remained "poor" in all regions of the Delta Beach Zone for most of the last 20 years (Figure 10) and the current score is "poor". Some regional indicator scores increased briefly in the most recent five years, however, this increase was not sufficient to raise the average score for the last five years above "poor" for any region.

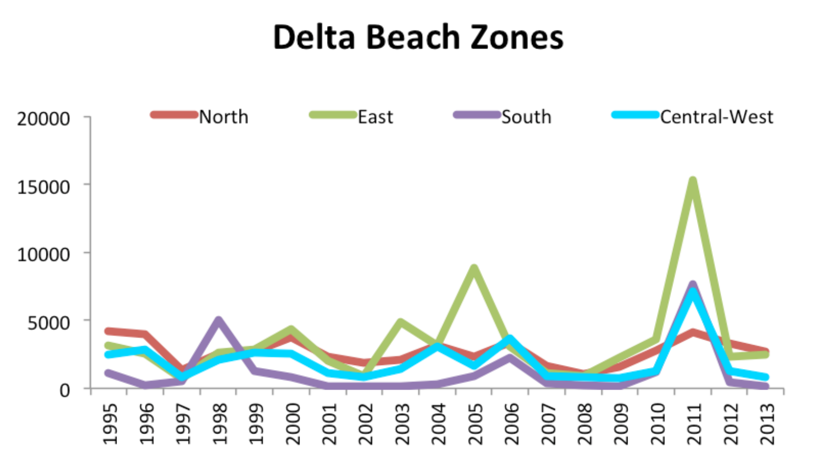
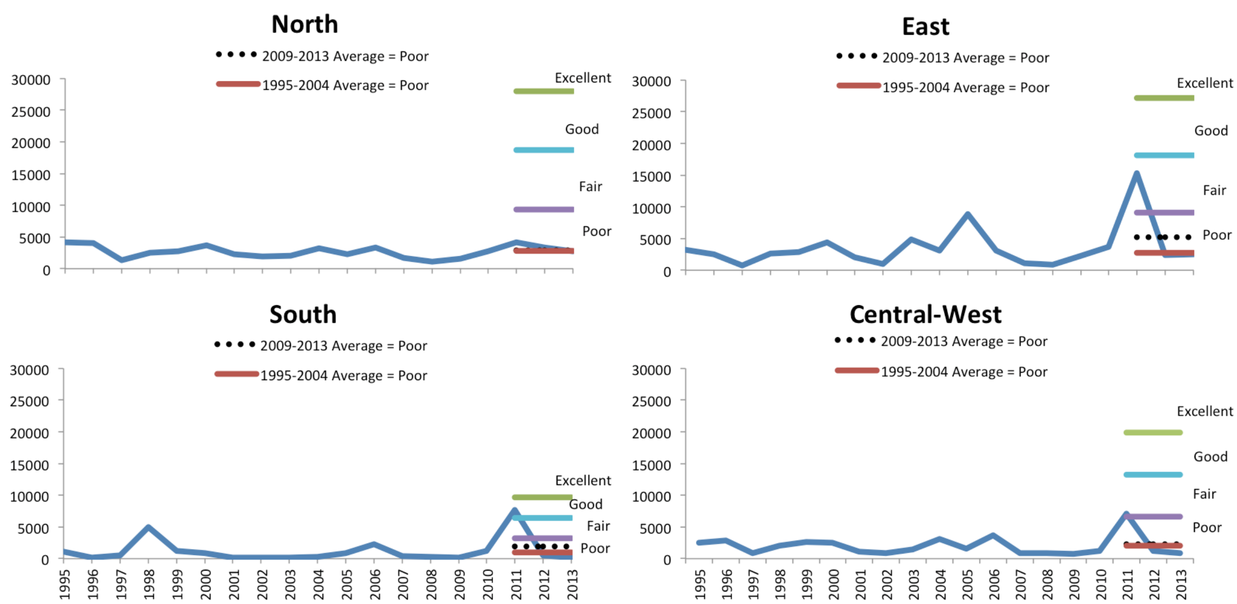


Figure 9. Comparison of native catch per unit effort for four Delta Beach Zone regions. Trends for native fish abundance were similar (see correlation matrix below) for most regions and exhibited different patterns than for total fish abundance (see Food Productivity Indicator).

Table x: Correlation values for comparison of trends between North, East, South and Central-West Delta Beach Zones. Values in red are significant (p <0.00). The correlation between North to East Beach Zones and the North to South Beach Zones were p=0.05 and 0.09 respectively.

Pearson Correlation Matrix	North	East	South	Central -West
North	1.00			
East	0.453	1.00		
South	0.400	0.712	1.00	
Central-West	0.706	0.773	0.787	1.00



Beach Zone regions**.** Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 8). The primary reference condition (1995-2004 average) was considered to be "poor" based on averages calculated from Suisun Marsh data and Pelagic Zone abundance indicators during that same time period. The dotted line represents the 2009-2013 average and shows that native fish abundance in each region of the Delta Beach Zone is currently "poor".

Summary of Beach Zone Abundance and Diversity Trends

Taken together, the Beach Seine data reveal that abundance of fish in the shallow, unvegetated waters of the Delta remained "poor" for the period of record with a peak in 2011. Increases in 2011 were not enough to raise the scoring for 2009-2013 average above "poor".

5. Summary of Abundance Results

Abundance of fishes in the Pelagic Zone and Suisun Marsh decreased substantially since the early 1980's and the decline accelerated in the early part of this century; trends in abundance were remarkably similar between these two habitats. Abundance of native species in the Delta Beach Zone has remained "Poor" during most of the period of record.

Based on abundance, the CCMP goals to recover and reverse declines of estuarine fishes (SFEP 2007) have not been met in the upper Estuary region.

B. Species Composition Indicators:

1. Rationale

An indicator for species composition was developed for the SOTB (2011) based on work by May and Brown (2002) and Meador et al. (2003) who found that the relative proportions of native and non-native species in an ecosystem are important indicators of ecosystem health. The SOTB (2011) states:

> *"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."* [p. 176]

As with the abundance indicators, it is important to note that indicators of assemblage composition are not necessarily tied to local processes as many species in a particular region may have spawned or reared in distant habitats – it is possible that, to some degree, the relative abundance or diversity of non-native species to native species reflects "propagule pressure" from other environments in the Central Valley.

As with the SOTB (2011), two different indicators for species composition were calculated:

- Percent Native Species reflects the species richness of native and non-native fishes in a given region.
- Percent Native Fish reflects the percentage of individual fish collected in each sub-region of the Estuary that were native species.

2. Methods and Calculations, Assumptions, and Uncertainties

In general, the same methodology for constructing species composition indicators was applied to each of the upper Estuary fish data sets (representing different sampling programs and major habitats). Differences among the sampling programs required some modification of methods for each sampling program.

A **Percent Native Species Indicator** was calculated for each year in each sampling program/sub-region as the percentage of fish species collected in the upper Estuary that are native to the Estuary, as follows:

% native species = [native species richness /(native species richness + non-native species richness)] x 100

A **Percent Native Fish Indicator** was calculated in each year in each sampling program/sub-region as the percentage of total individual 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 individuals/(total individual fish caught)] x 100

For each sampling program, the years incorporated into the composition indicators were the same as those described for their respective abundance indicators (see above).

3. Reference Conditions

Primary reference conditions for the assemblage composition indicators were the same as those used in SOTB (2011). These reference condition scores were based on inference from ecological literature and there was no compelling justification to use a different scoring system for the upper Estuary than had been used in the pelagic waters of the lower Estuary. The average percent native fish for the primary reference period, 1980-1989, (~85%) in the lower Estuary, was judged to be “good” (SOTB 2011). Index values where native fish represents less than 50% of total catch were judged to represent highly degraded conditions (SOTB 2011). Suisun Bay was reported to have lower percentages of native fish relative to total catch than other regions of the Bay (SOTB 2011). See Table 10 for quantitative reference conditions used here and in (SOTB 2011).

Table 10. Quantitative reference conditions and associated interpretations for the results of the Fish Species Composition Indicators (Percent Native Fish and Percent Native Species) for Suisun Marsh, Delta Beach Zone and Upper Estuary Pelagic Zone.

Quantitative Reference Condition	Interpretation	Low End of Range	High End of Range
>95%	Excellent	>95	N/A
>85%	Good	>85	95
>70%	Fair	>70	85
>50%	Poor	>50	70
≤50%	Very Poor	N/A	<50

4. Results of Species Composition

Suisun Marsh

The Percent Native Fish indicator is currently “very poor” in Suisun Marsh, a decline from its primary reference condition (1980-1989 average).

The 1980-1989 average percentage of native fish in total catch for the Suisun Marsh Survey was 47.0%. This means that the primary reference condition for Suisun Marsh (the earliest records from regular sampling) was “very poor” (Figure 11, Table 10). In the most recent 5 years, the percentage of native fish has been less than 50% (45. 9%), meaning that Suisun Marsh remains “very poor” for this indicator of assemblage health (Figure 11). Although most of the fish caught in Suisun Marsh are non-native species, it is worth noting that native fish abundance reached an all-time low in 1994 and has rebounded since that point.

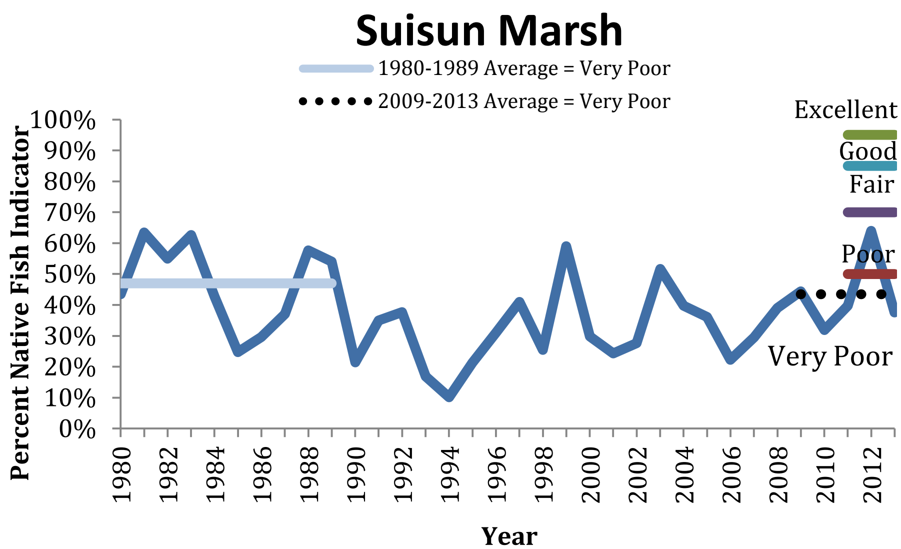


Figure 11. Changes in the relative abundance of native fish (Percent Native Fish Indicator) in Suisun Marsh from 1980-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. The primary reference condition and recent five-year averages are similar (47.0% and 43.5% respectively); both indicate "very poor" health of the local fish assemblage.

The Percent Native Species indicator is currently "very poor" in Suisun Marsh; this index declined from "poor" to "very poor" over the course of the survey. The 1980-1989 average percentage of native species detected in the Suisun Marsh Survey was 51.9%. This means that the baseline conditions for Suisun Marsh (the earliest records from regular sampling) rate "poor" (Figure 12, Table 10). In the most recent five years, the percentage of native fish species was less than 50% (45.9%), meaning that Suisun Marsh scored "very poor" on this index of assemblage health.

In addition to plotting the percent native species, the raw number of native vs. introduced species over the time series was compared (Figure 13) in an effort to assess whether changes in sampling effort (changes in trawl number) across years affected the total number of species detected. Native and non-native species richness was not significantly correlated and did not appear to respond to differences in the number of trawls conducted in the early years of the survey.

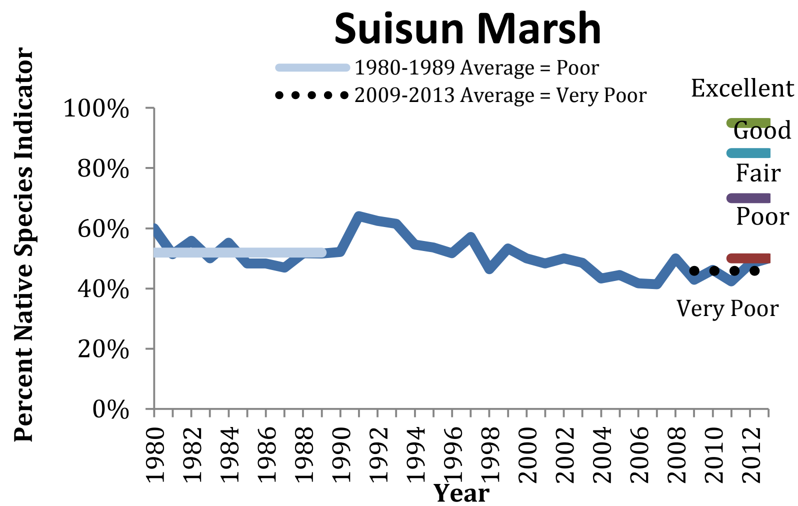


Figure 12. Changes in the Percentage of Natives Species Indicator in Suisun Marsh from 1980-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Reference period averages and recent five-year averages are similar (51.9% and 45.9%, respectively, of species detected in the Suisun Marsh Survey are native). The early reference condition average represented "poor" health and last five-year average indicates that current conditions are "very poor".

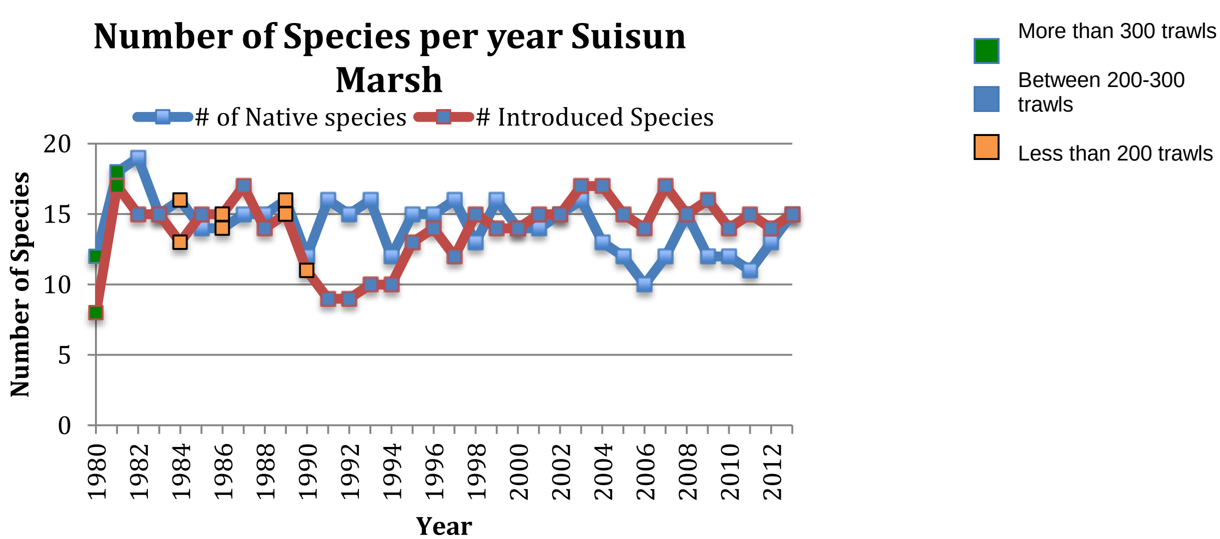


Figure 13. Comparison of native and non-native species richness through time ($r = 0.059$, $p = 0.74$) in Suisun Marsh. Native species richness is declining slowly, whereas non-native species richness has remained stable since the late-1990's. Colored boxes indicate changes in sampling effort (number of trawls) in some years. No relationship between the number of trawls and the richness of native and non-native species or the native/non-native relationship was detected.

Upper Estuary Pelagic Zone

Suisun Bay*. *The percentage of native fish represented in the pelagic assemblage of Suisun Bay was "poor", indicating no change in score between the primary reference condition (1980-1989 average) and the average of the last 5 years. The 1980-1989 average percentage of native fish in total catch of Suisun Bay was 65.6%. This means that the primary reference condition for Suisun Marsh (the earliest records from regular sampling) was "poor" (Table 10). In the most recent 5 years, the percentage of native fish in the total catch declined slightly (to 60.4%), but this too indicates that assemblage health is "poor" (Figure 14). The indicator varied widely over the period of record from "good" to "very poor". Not captured in this comparison is the precipitous decline in the percentage of native fish in the community in the early 1990's and the early 2000's – during those periods, the Percent Native Fish index was "very poor".

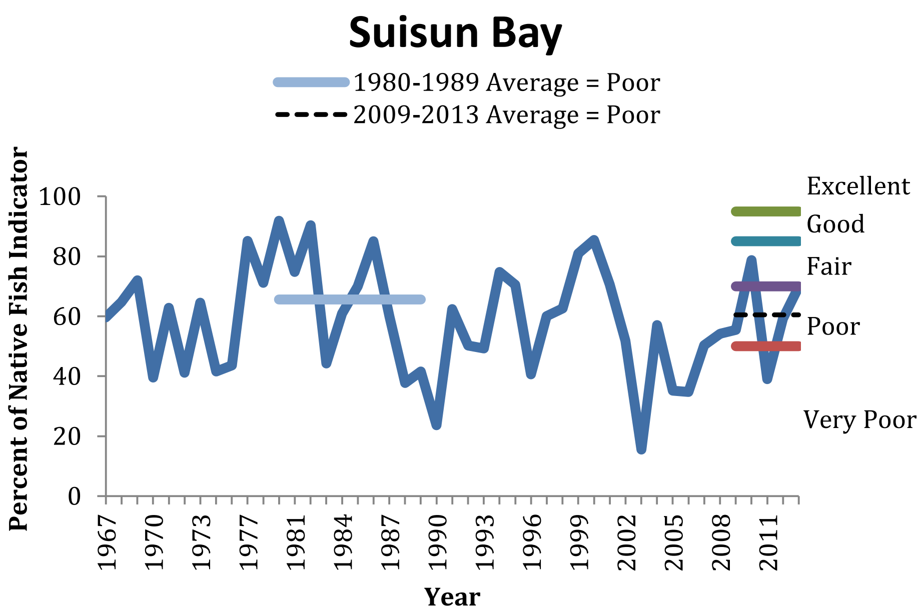


Figure 14. Changes in the relative abundance of native to non-native fish (Percent Native Fish Indicator) for the Pelagic Zone of Suisun Bay from 1967-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Reference period averages and recent five-year averages are similar (65.6% and 60.4% respectively). Both the early reference condition average and last five-year average reflect "poor" health of the fish assemblage in this region of the upper estuary.

The percentage of native species in the pelagic assemblage of Suisun Bay was "fair" representing little change from its primary reference condition (1980-1989). In both the reference period and the last 5 years, slightly less than two-thirds of the species were native (Figure 15). There is no indication that variation in sampling effort in the early years of the program affected total or relative richness scores. Over the period of record the indicator varied between “fair” and “poor”.

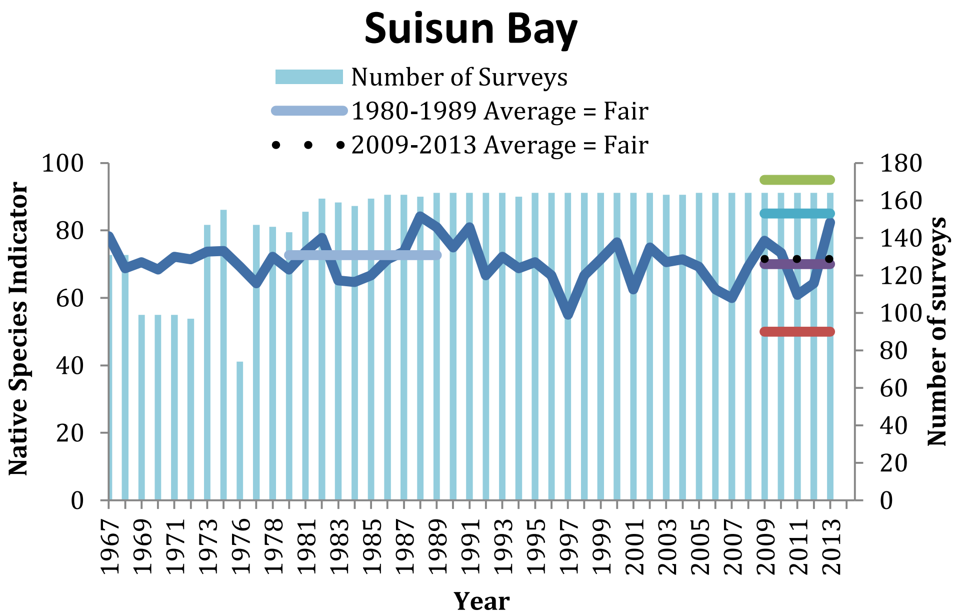


Figure 15. Changes in the Percent Native Species Indicator for the Pelagic Zone of Suisun Bay from 1967-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. The reference period and recent five-year averages are similar (72.67% and 71.55% respectively) indicating that the relative richness of native species remains “fair” in this region of the upper estuary. There was no significant correlation between the number of species detected and the number of surveys conducted (r=-0.007, p=0.96).

Central-West Delta. The percentage of native fish represented in the pelagic assemblage of the Central-Western Delta remained “very poor”. The indicator has remained solidly below 40% throughout the time series (Figure 16). Native species richness reached a peak in 2011, but this increase does not yet constitute a trend as native species richness declined again in the next two years.

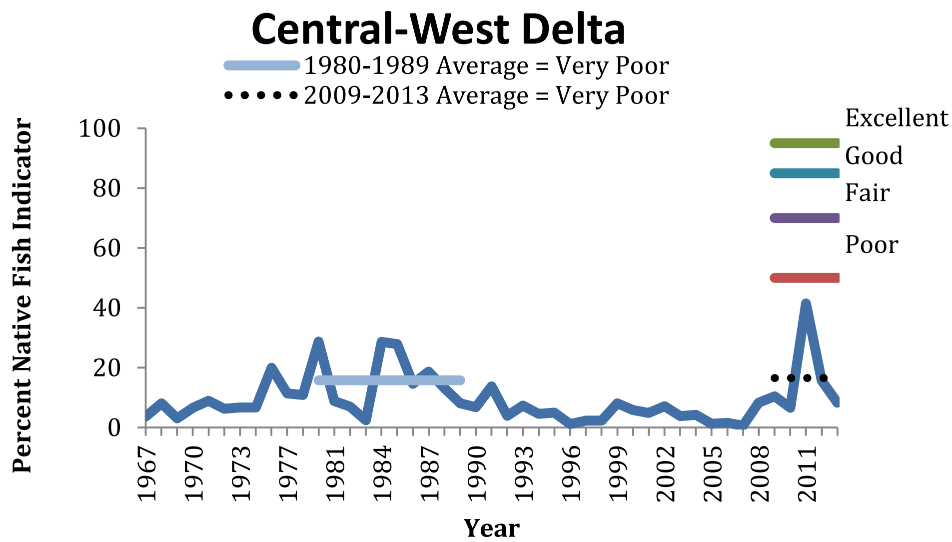


Figure 16. Changes in the Percent Native Fish Indicator for the Pelagic Zone of the Central-West Delta from 1967-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Reference period averages and recent five-year averages are similar (15.78% and 16.51% respectively). Indicating that the relative richness of native species has remained “very poor” in this region of the upper estuary throughout the sampling program time series.

The percentage of native species in the pelagic assemblage of the Central-West Delta declined slowly but persistently following the primary reference period (1980-1989), this indicator was most recently “very poor”. In the reference period native species made up exactly half of the total species caught by the FMWT pelagic sampling program when it sampled in the West Delta (Figure 17). In the last 5 years, that index has decline to less than 40%, on average. In this case, the decrease in relative native species richness came despite an increase in the number of trawls conducted in the western Delta.

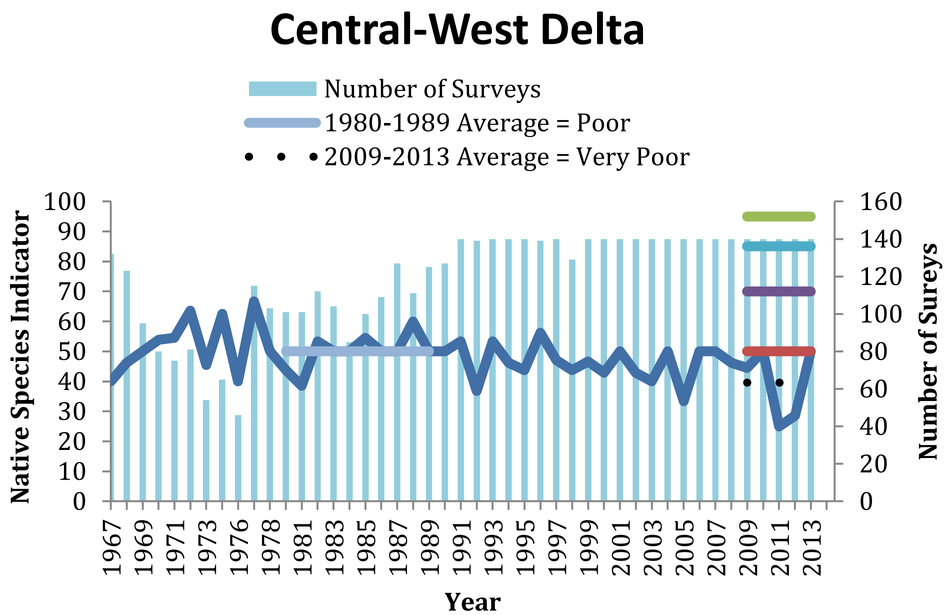


Figure 17. Changes in the Percent Native Species Indicator for the Pelagic Zone of the Central-West Delta from 1967-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Reference period averages and recent five-year averages are different. Conditions in the reference period (50.0% native species) were "poor" but the average of the most recent five years (39.6% native species) was "very poor" There was no significant correlation between the number of species detected and the number of surveys conducted (r=0.10, p=0.51).

Delta Beach Zone.

The percentage of native fish and native species in all regions of the Beach Zone assemblage of the Delta was "very poor" in both the primary reference condition and in recent years. The percentage of native fish caught in the North and East Delta was higher than the South and Central-West and, in 2011, the percentage of native fish increased in all regions, driven largely by high numbers of juvenile Sacramento splittail produced in that year (Figure 18). Native species have accounted for less than 40% of the Beach Zone species assemblage in all Delta regions throughout most of the period of record (Figure 19).

Figure 18. Changes in the relative abundance of native fish (Percent Native Fish Indicator) for the Delta Beach Zones from 1995-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1995-2004 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average.

The primary reference condition for North, East, South and Central-West was “very poor” (37.2%, 41.7%, 5.5%, and 14.5% respectively). The 2009-2013 averages remained “very poor” (34.6%, 45.3%, 10.1% and 17.6% respectively) in all regions of the Delta Beach Zone.

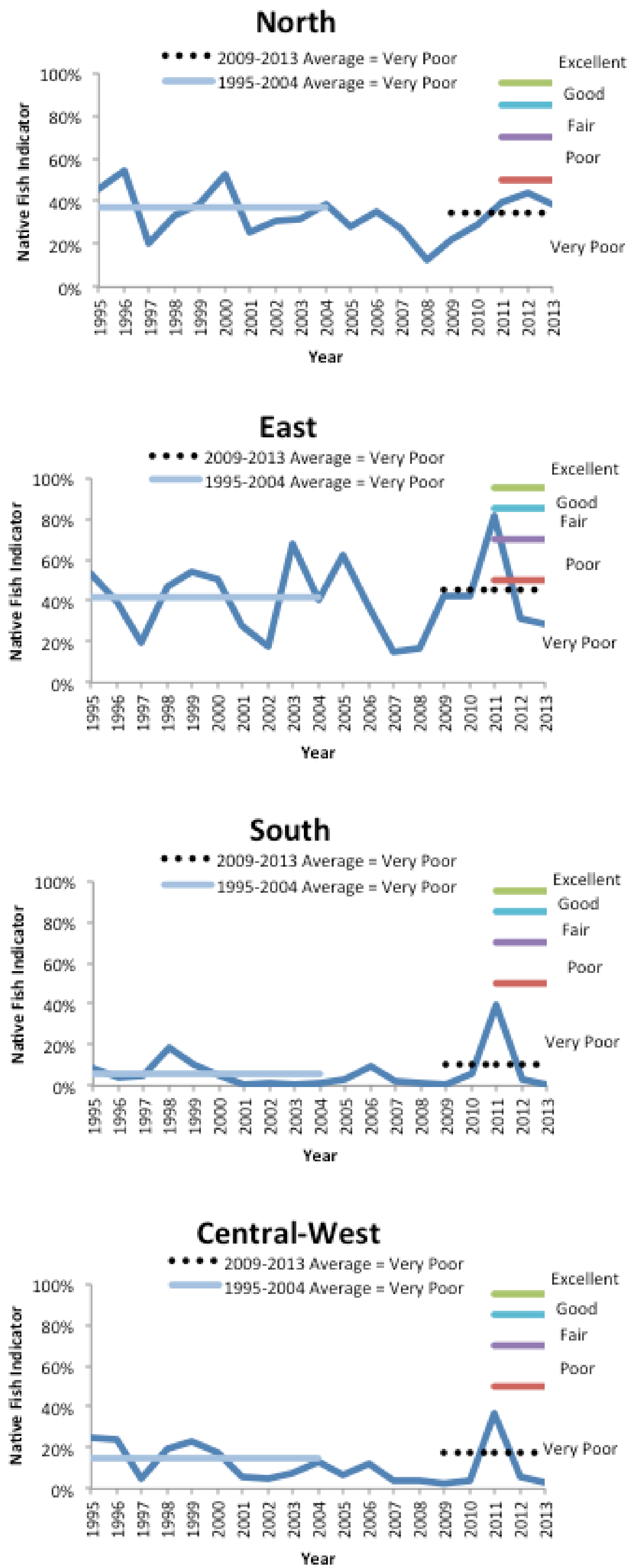
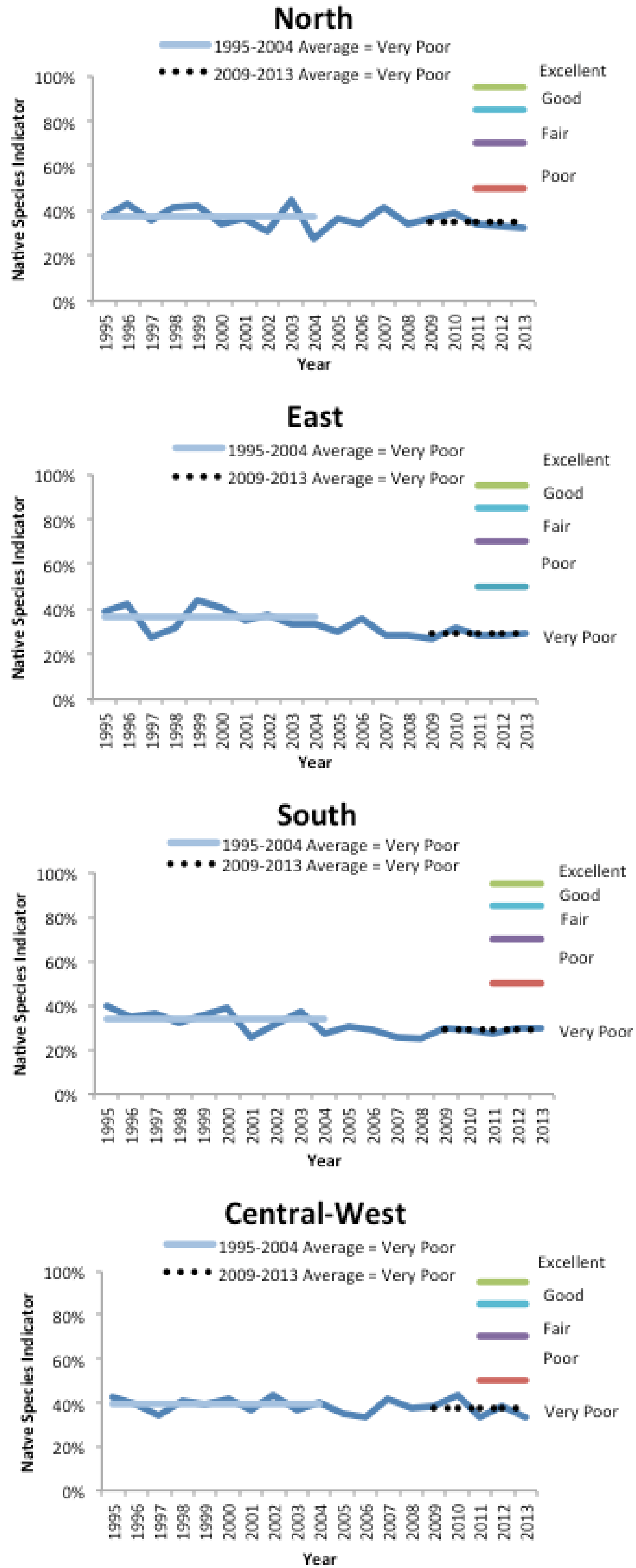


Figure 19. Changes in the Percent Native Species Indicator for the Delta Beach Zones from 1995-2013. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 10). The primary reference condition (1995-2004 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average.

The primary reference condition for native species richness North, East, South and Central-West was "very poor" (37.5%, 36.5%, 33.9%, and 39.4% respectively) and the 2009-2013 averages remained "very poor" (35.1%, 28.9%, 29.1%, and 37.5% respectively).

No significant correlations between the number of species detected and the number of surveys conducted were detected (e.g. in the South Delta; r=0.16, p=0.50).



C. Fish Part of the Food Web Productivity Indicators (Total Fish Abundance):

1. Rationale

The total abundance of fish, native and introduced, represents a snapshot of existing conditions in the ecosystem. Consumers of and competitors with fish may not distinguish between native and introduced fish species; therefore, abundance of all fish is a useful indicator of system productivity at a given time.

For each sampling program and major habitat sampled, we constructed indicators of overall catch (native plus introduced fish abundance), corrected for differences in effort expended catching those fish (catch-per-unit-effort; CPUE). These indices can be studied to determine the relative abundance of fish available (e.g., to fish predators) within different habitats of the Delta through time. Because the habitats and gears used to sample them differ so much across studies, no effort was made to aggregate abundance indicator scores across sampling programs into a single index; however, within sampling programs, if abundance indicators from different sub-regions were highly correlated through time, we combined sub-regions into a single overall indicator of abundance in that sampling program/habitat type.

Also, because different fish species have different value as prey for, competitors with, or consumers of other species, it was important to determine whether indices of total fish abundance reflected variations in the entire fish assemblage or, alternatively, were driven by individual species (see also, Limitations and future amendments to the abundance indicator above). Thus, we compared behavior of a raw abundance index with that of an index of abundance that summed standardized scores of fish abundance within species.

Because the Estuary's fish assemblage is influenced by processes affecting fish production elsewhere (upstream in the Central Valley's rivers or in the nearshore ocean), caution should be used in relating these abundance indices to local ecosystem processes.

2. Methods and Calculations, Assumptions, and Uncertainties

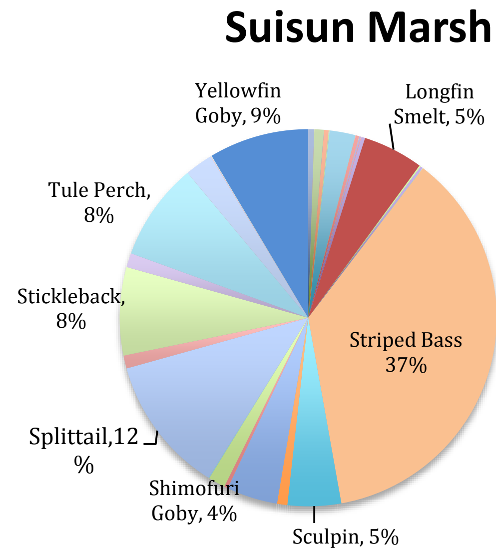
The Food Web Indicator was calculated the same as the abundance indicators, with the exception that both native and introduced fishes were included in the analyses**.** In addition standardized abundance (described below) was added to the analysis.

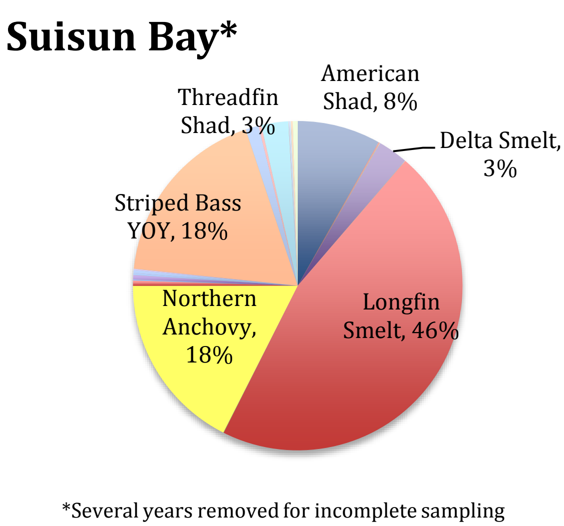
Indicators of Standardized Abundance: Checking for disproportionate effects of single species on annual trends

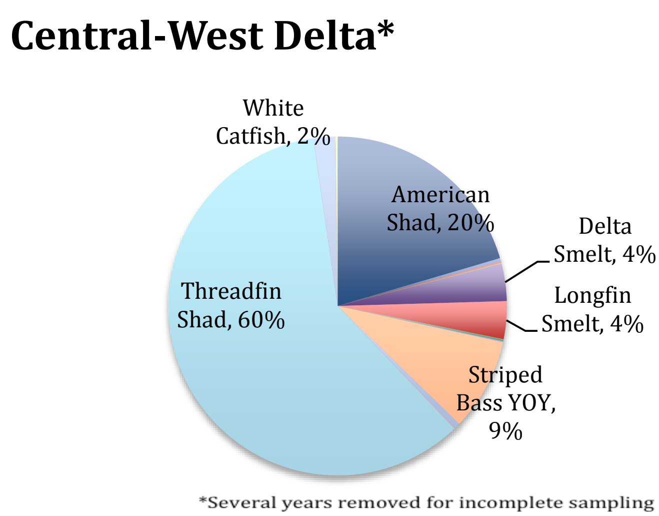
The abundance indicators described above provide a measure of fish assemblage health that is easy to understand and explain: *how many fish are caught for a given sampling effort*? However, such an indicator may not reveal the true state of the fish assemblage if the number of fish caught is dominated by one or a few species. In that situation, though the CPUE indicator is still of interest, it may reflect trends in the abundance of one species disproportionately, rather than trends in the assemblage as a whole.

The Fish Index for the State of the Bay Report (SOTB 2011) created a separate index for Northern Anchovy because, in most years, greater than 80% of the fish caught in the Bay Study were Northern Anchovy (SOTB 2011). Thus, variation in the catch of this one species within a year could mask abundance trends of other species in a combined total catch indicator. In the upper Estuary, a small number of species dominate all others numerically, but the species involved change depending on the habitat sampled. For instance, striped bass represented 37% of the entire catch in Suisun Marsh (Figure 20). In the open waters of Suisun Bay and the Central-West Delta five to six species dominated the catch (Longfin Smelt, Threadfin Shad, American Shad, Striped Bass, Northern Anchovy, and Delta Smelt, Figure 20) and catches of these species displayed high variance across years. In the Delta Beach Zone, Inland Silversides represented a large portion of the catch (62% of the total catch from 1995-2013, Figure 21).

Figure 20. Proportional catch of fish species caught by the Fall Midwater Trawl in Suisun Bay (1967-2013), the Central-West Delta (1967-2013) and Suisun Marsh Fish Survey (1980-2013).







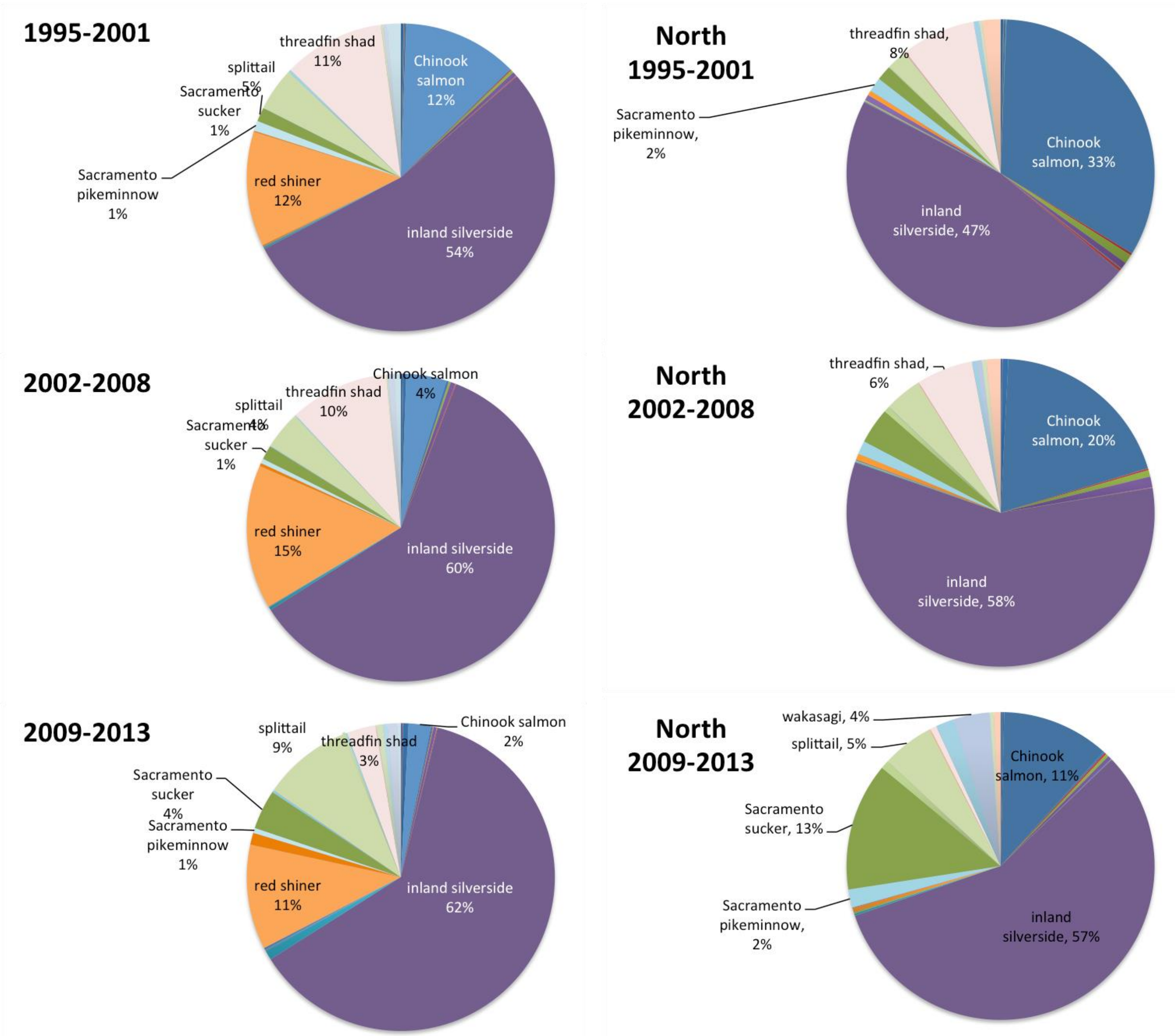


Figure 21. Proportional catch of fish species caught by the USFWS Beach Seine in all regions (left side) and the North subregion of the Delta (right side) during three time periods.

In an effort to create an abundance indicator that reflects abundance changes for all species within habitat-specific fish assemblages measured by the surveys studied here, we created a separate metric – the sum of the standardized abundances of species that were regularly caught in each sampling program. Within each sampling program and sub-region, fish catches were standardized by subtracting the annual catch for each species' by the mean catch for that species over the entire time series; the difference between annual and mean catch was then divided by the species-specific standard deviation in catch over the time series. Thus, for each species an annual catch that equaled the mean long-term mean catch was scored as a 0 and catches one standard deviation above or below the species-specific mean were scored as 1 or -1, respectively. These annual, species-specific standardized scores were then summed for all species that were regularly caught by the sampling program. To avoid undue influence of very rare ("accidental") catches, species that are not well sampled by a given sampling program (either because the methodology or habitat or both) were not included in the sum of standardized scores. For a species to be included in the annual standardized index for Suisun Marsh, a species had to have been caught in more than one quarter of survey years (at least 9 years) and, in years when the species was detected, the mean catch of that species had to be ≥2.0. Species included in and excluded from the standardized abundance indicator for each sampling program and habitat are listed in Tables 11-13.

An example of the standardization calculations, for the Suisun Marsh data set, follows:

1) Exclude any species that was not detected in at least one-quarter of years sampled and for which catch did not average ≥2.0 in years where the species was detected. Species excluded by this filter were deemed to be those for which presence in the sample was accidental (e.g., accidental presence in the habitat or accidental catch by the gear) – in other words, presence of these species in the sample did not necessarily provide any information about local abundance. Steps that follow refer only to species that were not excluded in this manner.

2) Calculate CPUE for each species in each year,
CPUE = catch of species "x"/trawl = î = [Number of individuals of species "x" caught in year "y"]/ [trawls in year "y"].

3) Calculate overall mean CPUE and standard deviation of CPUE for each species over the 1980-2013 Suisun Marsh survey sampling period. For each species,
[Mean catch = ī = sum of CPUE from 1980-2013/years of survey (n=34)]
[Standard deviation =ŝ= $\sqrt{}$(î-ī)^2/(n-1) = square root of the average squared deviation from the mean].

4) Calculate annual standardized score for each species by subtracting its overall mean CPUE from its annual CPUE and dividing the difference by the standard deviation in CPUE for that species [(î-ī)/ŝ]. For example,

Suisun Marsh American Shad standardized $score_{1980}$ = $(AmShadCPUE_{1980} – AverageCPUE_{1980-2013})$/Standard Deviation
(0.0745 – 0.12)/0.16 = -0.304

5) Within each year, sum all the standardized values for each species identified above.

Correlation coefficients between these standardized annual abundance indices and their corresponding total abundance indices were calculated. If trends in the total abundance indicator for any sampling program/region/habitat represent trends across their respective local fish assemblages, then the standardized abundance indicator and the total catch abundance indicator ought to be highly correlated. When these two different metrics were not highly correlated, it indicates that a very small number of species drove trends in the total catch indicator. In that case, the standardized abundance indicator was reported as the measure of health for that sampling program/region instead of the total abundance (mean CPUE) indicator. Figures 22 -23 show the sum of the standardized values for catch/trawl for Suisun Marsh and in the Pelagic Zone of the upper Estuary, respectively. The correlation between the standardized and total abundance indicators ($r = 0.77$, $p< 0.001$ for Suisun Marsh and r=.0.74 and 0.76, $p<0.001$ for pelagic zone of Central-West Delta and Suisun Bay, respectively) indicates that trends in the total catch indices represent trends throughout the assemblage as a whole, rather than changes in one species.

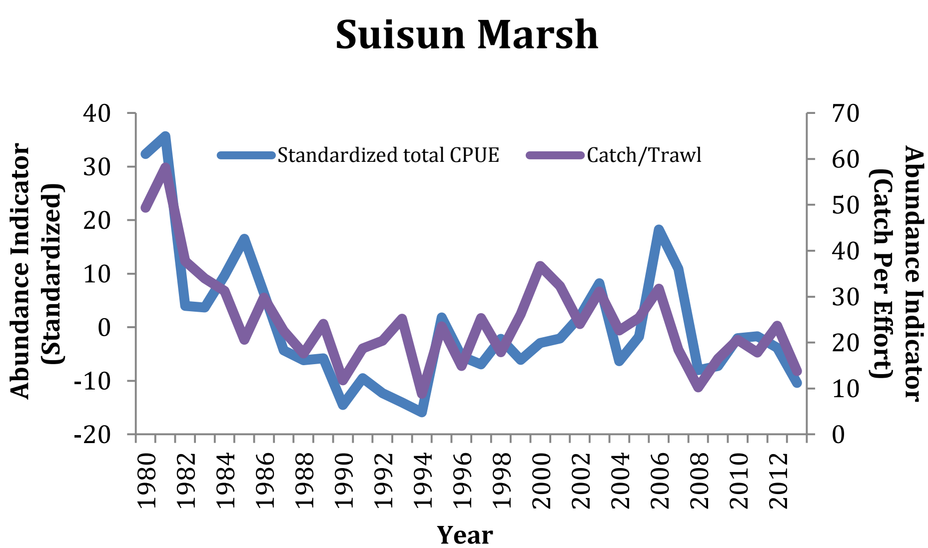


Figure 22. Comparison of raw catch-per-unit-effort (catch-per-trawl) indicator and the standardized catch-per-unit-effort indicator in Suisun Marsh. The correlation between the standardized and raw catch ($r = 0.77$, $p <0.01$) indicates that trends in the raw catch-per-unit-effort indicator represents actual trends throughout the assemblage as a whole, rather than changes in one species.

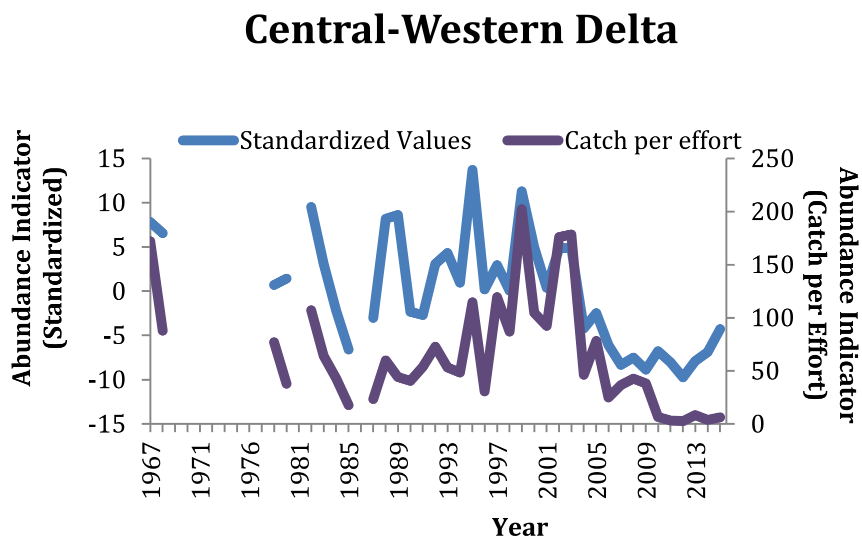


Figure 23. Comparison of raw catch-per-unit-effort indicator and the standardized catch-per-unit-effort indicator in the Pelagic Zone of the Central-West Delta. The correlation between the standardized and raw catch indicators ($r = 0.74$, $p < 0.001$) suggests that trends in the total catch indicator represent actual trends throughout the assemblage as a whole, rather than changes in one species. Similar correlations were detected between raw catch and standardized values for Suisun Bay (not pictured here; $r = 0.76$, $p<0.001$).

The patterns of total fish abundance were more complicated in the Delta Beach Zone data. Regional trends in the CPUE indicator differed, with total abundance increasing through time in throughout Delta Beach Zone regions except for the North Delta regions (Figures 24 and 26). In addition, trends in CPUE were not always reflective of changes in the entire fish assemblage of each region. Standardized abundance scores correlated well with their analogous CPUE abundance indicator in the North Delta ($r=0.72$, $p<0.001$), East Delta ($r=0.77$, $p<0.00$), and Central-West Delta ($r=0.52$, $p=0.02$); but, these indices of relative abundance were not significantly correlated in the South Delta region (Figure 25; $r=0.33$, $p=0.16$). This suggests that the CPUE indicator in the South Delta was likely to reflect trends in abundance of just a few species, not the assemblage as a whole. Indeed, when standardized scores of native species and non-native species were compared within regions of the Delta Beach Zone, it became clear that abundance of non-native species had increased through time in the East, Central-West, and South Delta while long-term abundance trends for native species (as a whole) were less obvious (Figure 26). It is worth noting that the correlation of standardized scores between native and non-native species assemblages was positive in all regions of the Delta Beach Zone (and significantly so, in the North and East Delta, Figure 26). Because of differences between regions and between standardized and raw catch per effort values, Delta Beach Zone results are presented in separate regions in the standardized form.

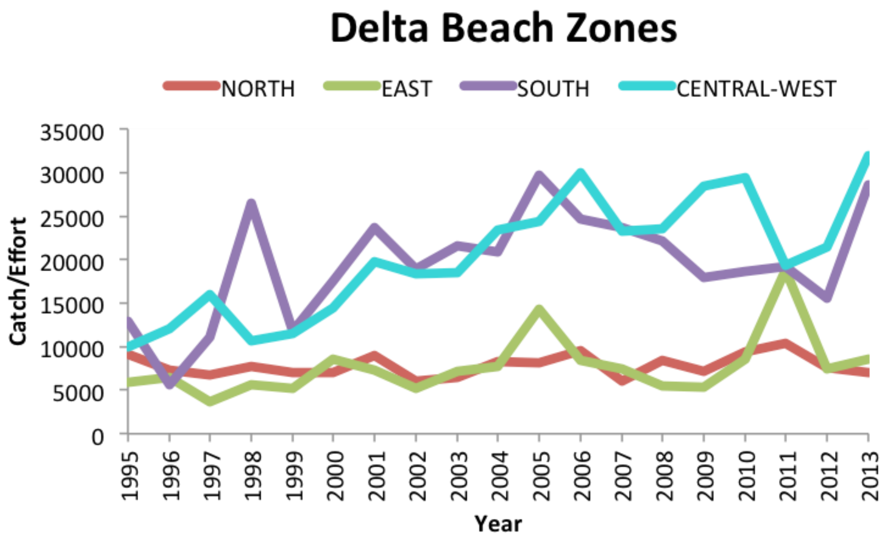


Figure 24. Comparison of Food Web Productivity Indicator (unstandardized, total catch per unit effort) for four Delta Beach Zone regions.

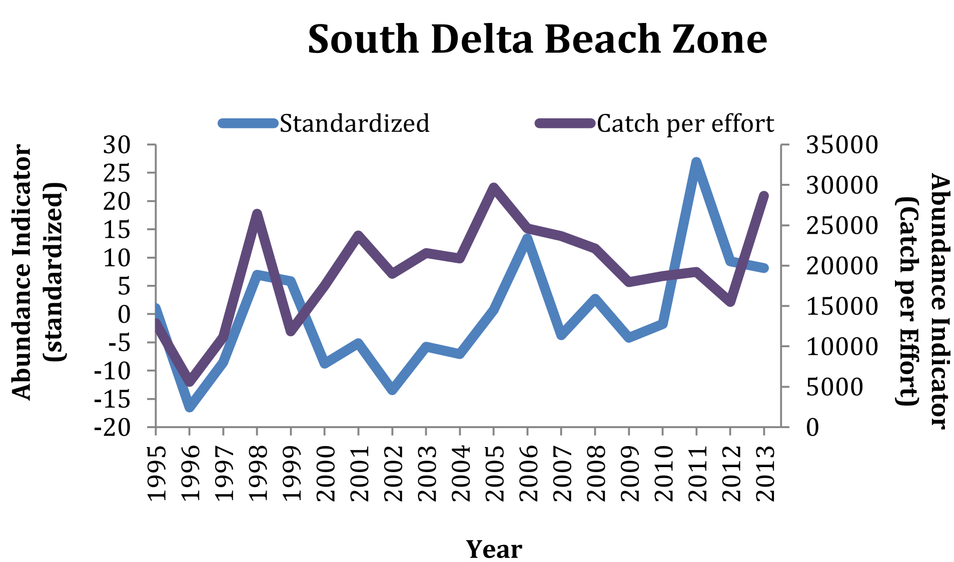


Figure 25. Comparison of raw catch-per-unit-effort indicator and the standardized catch-per-unit-effort indicator in the South Delta Beach Zone. The correlation between the standardized and raw catch indicators ($r = 0.33$, $p = 0.16$) suggests that trends in the total catch indicator may not represent actual trends throughout the assemblage as a whole.

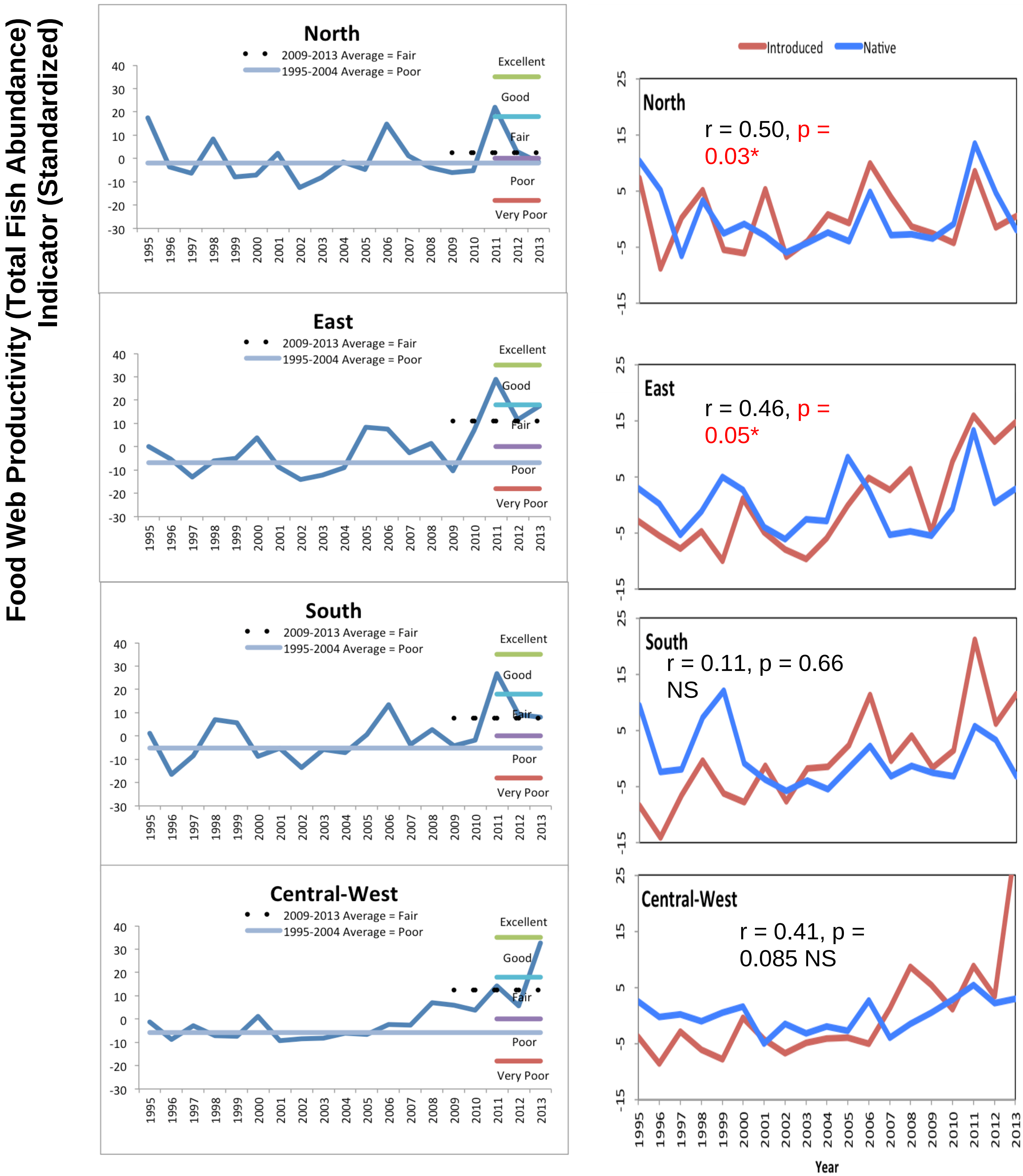


Figure 26. Changes in the four regions of Delta Beach Zone Abundance Indicators through time. *Panels on the left* present the sum of standardized abundance for all species that are well sampled in the regions. Short horizontal colored lines indicate

reference thresholds assigned to this indicator (see Table 15). Scores: above the green line are “excellent”; between the blue and the green lines are “good”; between the blue and light blue lines are “fair”; between the light blue and red lines = “poor”; and below the red line = “very poor”. The primary reference condition (1995-2004 average sum of standardized abundance) is indicated by a light blue horizontal line. For each region, the primary reference condition = 0, indicating that, on average, each species was at its 1995-2014 abundance. The dotted line represents the 2009-2013 average.

Panels on the right show the sum of standardized abundance scores for native (Blue lines) and introduced (red line) fishes. Abundance trends of natives and introduced species are significantly correlated in the North and East Delta. In the South and Central-West regions abundance for the average native and average introduced fish species are not significantly correlated; recent increases in abundance reflect increases in the introduced species assemblage.

Table 11. Species used in the abundance indicator for the Suisun Marsh. Species in bold were used in standardized abundance indicator calculations (Data: UCD Suisun Marsh Juvenile Fishes Sampling Survey Otter Trawl).

Species Used in The Suisun Marsh Abundance Indicator	Native (N) / Introduced (I)	Number of Years Caught	Sum of Catch
American Shad	I	31	1163
Brown Bullhead	I	14	28
Black Crappie	I	27	1850
Sac Blackfish	N	7	24
Bluegill	I	11	19
Black Bullhead	I	27	879
Bigscale Logperch	I	6	17
Bay Pipefish	N	2	2
Channel Catfish	I	24	167
California Halibut	N	3	5
Carp	I	34	5057
Chinook Salmon	N	16	72
Speckled Sandab	N	3	3
Delta Smelt	N	29	659
Fathead Minnow	I	13	36
Goldfish	I	28	298
Green Sturgeon	N	2	3
Green Sunfish	I	4	5
Golden Shiner	I	5	5
Hitch	N	24	114
Hardhead	N	1	1
Inland Silverside	I	34	716
Longfin	N	34	11790
Longjaw mudsucker	N	1	1
Plainfin Midshipman	N	6	11
MosquitoFish	I	10	18
Northern Anchovy	N	15	257
Pacific Herring	N	26	465
Pacific Lamprey	N	13	43
Pacific Sandab	N	2	2
Riffle Sculpin	N	2	2
Rainbow Trout	N	6	7
Rainwater Kilifish	I	14	32
Striped Bass	I	34	83784
Prickly Sculpin	N	34	10460
Strarry Flounder	N	34	2001
Shimofuri Goby	I	28	9974
Shokihaze Goby	I	14	722
Sacramento Sucker	N	34	3331
Shiner Perch	N	4	17

Sacramento Pikeminnow	N	23	148
Surf Smelt	N	3	5
Splittail	N	34	26875
Staghorn sculpin	N	34	2524
Stickleback	N	34	17231
Threadfin shad	I	34	2768
Tule Perch	N	34	19139
Wakasagi smelt	I	5	10
White Catfish	I	33	5453
White Crappie	I	14	112
White Croaker	N	1	1
Warmouth	I	1	1
White Sturgeon	N	26	113
Yellowfin Goby	I	34	19504

Table 12. Species used in the abundance indicators for the Delta Beach Zone. Species in bold were used in standardized abundance indicator calculations. Some species, such as Green Sunfish and Hardhead, were only used in standardized abundance calculations for the regions where they met the minimum requirement for inclusion. These species are indicated with a * (Data: USFWS Delta Juvenile Fishes Program, Beach Seine Survey).

Species Used in The Delta Beach Seine Abundance Index	Native (N) / Introduced (I)	North	Number of years Caught	East	Number of years Caught	South	Number of years Caught	Central-West	Number of years Caught
American Shad	I	464	20	1016	20	626	19	1231	20
Arrow Goby		1	1						
Bass Unknown	NA	18	4	87	3	109	5	81	4
Bigscale Logperch	I	124	18	150	18	1000	20	454	20
Black Bullhead	I	11	2	13	4	2	2	6	3
Black Crappie	I	42	12	27	10	182	14	38	12
Bluegill	I	802	20	975	20	5213	20	770	18
Brown Bullhead	N	1	1	2	2	9	5	15	4
California Roach	N	29	1			1	1	5	1
Chameleon Goby	I	9	3	13	2	1	1	10	5
Channel Catfish	I	13	5	2	2	6	5	2	2
Chinook Salmon	N	68284	20	24286	20	4302	20	20870	20
Common Carp	I	139	9	571	10	2269	18	19	6
Delta Smelt	N	523	20	170	15	38	11	545	19
Fathead Minnow	I	1610	20	89	19	1236	17	84	17
Golden Shiner	I	1793	20	1585	20	2912	20	2145	20
Goldfish	I	12	6	13	6	61	9		
Green Sunfish*	I	10	8	11	5	43	8	8	5
Hardhead*	N	114	11	5	4	28	6	37	6
Hitch	N	310	15	728	11	43	14	420	20
Mississippi Silverside	NA	137153	20	92311	20	544505	20	456867	20
Lamprey Unknown*	N	120	16	27	12	2	2	2	1
Largemouth Bass	N	261	18	2129	20	4298	20	2497	20
Longfin Smelt	I	3	2	8	2	1	1	16	8
Minnow Unknown	NA					2	2	33	2
Pacific Herring	N							18	2
Pacific Lamprey	N	6	4						
Pacific Staghorn Sculpin*	N	12	8	16	6	31	9	857	17
Prickly Sculpin	N	273	18	195	18	489	20	281	17

Rainbow/Steel head Trout*	N	676	19	179	20	2	2	53	9
Rainwater Killifish*	I	8	5	37	9	1123	15	1331	18
Red Shiner	I	1284	19	284	13	254867	20	91	17
Redear Sunfish	I	76	15	2247	19	5810	20	2901	20
Redeye Bass*	I	1	1	19	6				
River lamprey	N	2	2						
Rosyface Shiner	I	1	1	2	1	10	2	4	2
Sacramento Blackfish*	N	18	6	46	4	61	12	21	3
Sacramento Pikeminnow	N	3857	20	2050	20	1473	20	4381	20
Sacramento Sucker	N	13273	20	18281	20	8628	20	1441	20
Sculpin Unknown	NA	1	1			2	1		
Shimorfuri Goby	I	1151	20	464	15	139	11	408	20
Shokahaze Goby	N					1	1	2	1
Smallmouth Bass	I	72	18	182	18	39	14	23	11
Splittail	N	8863	20	42047	20	31578	20	18884	20
Spotted Bass	I	143	11	1342	13	150	13	58	11
Starry Flounder*	N	9	4			3	3	26	11
Striped Bass	I	209	18	214	16	1801	20	1797	20
Striped Mullet	N							2	1
Threadfin Shad	I	11651	20	2958	19	107035	20	32450	20
Threespine Stickleback*	N	56	11	7	7	3	2	1039	20
Tule Perch	N	2083	20	621	19	241	19	3225	20
Unidentified Fish	NA	3	3	7	2	39	1	2	2
Wakasagi Smelt*	I	2932	20	32	7	2	2	293	13
Warmouth	I	2	1	14	8	6	5	3	3
Western Mosquitofish	I	934	20	2524	20	6520	20	4940	20
White Catfish	I	2	2	19	9	22	12	4	4
White Crappie	I	35	11	14	9	29	8	15	6
Yellow Bullhead	I					1	1		
Yellowfin Goby	I	3498	20	3480	20	1870	19	6040	20

Table 13. Species used in the abundance indicator for the upper estuary pelagic zone. Species in bold were used in standardized abundance indicator calculations. A minimum of being caught in 10 years was set for inclusion to the standardized index because this survey is only for four months of the year. Striped bass were summed for all ages. Some species, such as Channel Catfish and Jacksmelt, were only used in standardized abundance calculations for the regions where they met the minimum requirement for inclusion. These species are indicated with a * (Data: CDFW Fall Midwater Trawl).

Species Used in The Fall Midwater Trawl Abundance Index	Native (N) / Introduced (I)	Number of years Caught Suisun	Number of years Caught West Delta	Suisun Raw Catch	West Delta Raw Catch	Sum of Catch
American Shad	I	45	45	20272	36212	56484
Arrow Goby	N	1	0	5	0	5
Bat Ray	N	1	0	1	0	1
Bay Pipefish	N	1	0	1	0	1
Big Skate	N	0	1	0	29	29
Bigscale Logperch	I	0	1	0	1	1
Black Crappie	I	7	9	6	13	19
Bluegill	I	3	9	5	15	20
Brown Bullhead	N	4	1	1	1	2
Brown Smoothhound	N	2	0	2	0	2
Chameleon Goby	I	3	6	5	19	24
Channel Catfish*	I	1	26	1	421	422
Chinook Salmon	N	41	34	303	352	655
Common Carp	I	21	24	103	114	217
Delta Smelt	N	45	45	7343	6290	13633
Diamond Turbot	N	1	0	1	0	1
flatfish (Unid)	NA	1	0	1	0	1
Goldfish	I	0	4	0	5	5
Green Sturgeon	N	7	3	8	3	11
Green Sunfish	I	2	3	2	4	6
Hitch	N	0	3	0	3	3
Jacksmelt*	N	15	0	45	0	45
Largemouth Bass	N	0	3	0	4	4
Longfin Smelt	N	45	45	114523	6370	120893
Inland Silverside	I	17	21	76	125	201
Night Smelt	N	2	0	17	0	17
Northern Anchovy*	N	45	7	43513	168	43681
Pacific Herring*	N	39	6	1292	15	1307
Pacific Lamprey	N	2	1	1	1	2
Pacific Sanddab	N	1	0	3	0	3
Pacific Sardine	N	1	0	1	0	1
Pacific Staghorn Sculpin*	N	34	5	243	7	250
Pacific Tomcod	N	7	0	27	0	27
Plainfin Midshipman*	N	38	0	1023	0	1023
Prickly Sculpin	N	2	2	8	2	10
Rainwater Killifish	I	0	1	0	2	2
Redear Sunfish	I	0	2	0	2	2
River Lamprey	N	4	3	4	8	12
Sacramento Blackfish	N	0	5	0	8	8
Sacramento Perch	N	0	1	0	1	1
Sacramento Pikeminnow	N	3	2	5	2	7
Sacramento Sucker	N	0	1	0	1	1

Shimofuri Goby	I	8	13	14	57	71
Shiner Perch	N	11	1	461	3	464
Shokihaze Goby	I	4	1	7	1	8
Speckled Sanddab	N	2	0	2	0	2
Splittail	N	38	26	733	70	803
Spotted Bass	I	1	0	1	0	1
Starry Flounder	N	39	14	302	29	331
Steelhead	N	10	3	19	5	24
Striped Bass age0	I	45	45	45279	15613	60892
Striped Bass age1	I	45	43	3206	1000	4206
Striped Bass age2plus	I	43	35	666	118	784
Striped Bass age3plus	I	14	9	18	12	30
Surf Smelt	N	2	0	1	0	1
Threadfin Shad	I	45	45	6566	105948	112514
Threespine Stickleback*	N	22	4	48	5	53
Topsmelt*	N	26	0	150	0	150
Tule Perch	N	7	11	6	13	19
Wakasagi	I	4	2	7	4	11
Walleye Surfperch	N	1	0	1	0	1
Western Mosquitofish	I	1	3	1	3	4
White Catfish	I	19	41	331	3612	3943
White Crappie	I	2	7	3	16	19
White Croaker*	N	23	0	88	0	88
White Seaperch	N	2	0	1	0	1
White Sturgeon	N	36	28	390	74	464
Whitebait Smelt	N	2	0	6	0	6
Yellowfin Goby	I	40	29	1228	351	1579

3. Reference Conditions

Wherever possible, the 1980-1989 average index value was used as the primary reference condition for abundance indicators. This is consistent with the Bay fish indicators (SOTB 2011). In the SOTB (2011), the 1980-1989 average is considered “good”, recognizing that some fish populations were already in decline by the 1980’s. A five-tier scale rates annual average CPUE over time from “very poor” to “excellent”. Any individual year in the record may be compared to the reference condition and scored.

Suisun Marsh

The 1980-89 average catch per trawl was established as the primary reference condition for this data set. These were the earliest years for which data was available. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions as described in Table 14.

Table 14. Quantitative reference conditions and associated interpretations for the Suisun Marsh Fish Abundance Indicator. The average score during the primary reference period, which corresponds to “good” conditions, is in bold and all other reference conditions are calculated from that value (e.g. “excellent” is 150% of the 1980-1989 value).

Abundance Indicators Suisun Marsh Catch Per Effort (Data: UCD Suisun Marsh Fish Survey, Otter Trawl)			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
>150% of the 1980-1989 Average	Excellent	>48.78	N/A
>100% of the 1980-1989 Average	**Good**	**>32.52**	**48.78**
>50% of the 1980-1989 Average	Fair	>16.26	32.51
>15% of the 1980-1989 Average	Poor	>4.88	16.25
<15% of the 1980-1989 Average	Very Poor	N/A	<4.88

Delta Beach Zone

The Beach Seine survey was not consistently conducted year-round until 1995. Thus, average catch per effort from 1995-2004 was established as the primary reference condition for this sampling program. The primary reference condition, during this period was scored as “fair” to match the average score of the total fish abundance indicator (native plus introduced species) scores for Suisun Marsh and Pelagic Zone abundance indicators during the same period. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions. Evaluation thresholds for these summed standardized scores are described in Table 15.

Because the Delta Beach Zone Indicator for Food Web Productivity is standardized, cut-offs for different intermediate reference conditions (qualitative scoring categories) were calculated differently than for other sampling programs/habitats. Standardization within each species set each species long-term average abundance to 0 and the standard deviation of abundance to 1. In a given year, if the average species was at its long term average, the sum of all species standardized abundance values would also be 0. The 1995-2013 long term average for all species (cumulative index score = 0) was considered to be “fair” to account for the fact that species abundance had already declined by 1995 and to correspond with averages from Suisun Marsh and Fall

Midwater Trawl during this time period. “Excellent” conditions indicated that the average species was 1 standard deviation above its long term average. Because there were 36 species included in this index, if the average standardized fish species abundance was 1, the cumulative index score would 36 (i.e. standardized score* number of species = 1* 36). “Good” conditions reflected the average species being ½ standard deviation above its long term average (cumulative score = 18), and “poor” conditions reflected that the average species abundance was 0.5 standard deviations below its long term average (cumulative index score = -18). “Very poor” conditions represented that the average species was more than 0.5 standard deviations below its long term average abundance (cumulative index score: <-18).

Table 15. Quantitative reference conditions and associated interpretations for the results of the Delta Beach Zone standardized fish abundance indicator. The average of the primary reference condition, which corresponds to “fair” conditions, is in bold. The primary reference condition was rated “fair” to correspond to scores for the Suisun Marsh and Fall Midwater trawl during the 1995-2009 time period.

Standardized Abundance Indicators Delta Beach Zone (Data: USFWS Delta Juvenile Fishes Program, Beach Seine Survey)			
North Delta, East Delta, South Delta and Central-West Delta			
Quantitative Reference Condition	**Interpretation**	**Low End of Range**	**High End of Range**
>One standard deviation above the 1995-2013 average	Excellent	>36	NA
> One half the standard deviation above the 1995-2013 average	Good	>18	36
>Standard Average of 1995-2013 (0)	**Fair**	**0**	**18**
> One half the standard deviation below the 1995-2013 average	Poor	-18	0
< One half the standard deviation below the 1995-2013 average	Very Poor	NA	<-18

Pelagic Zone of the Upper Estuary

The 1980-89 average catch per effort was established as the primary reference condition for this data set. Following SOTB (2011), the 5-tiered scoring system was developed for other intermediate reference conditions as described in Table 16.

Table 16. Quantitative reference conditions and associated interpretations for the results of the Upper Estuary Pelagic Zone Fish Abundance Indicator. The average during the primary reference condition, which corresponds to "good" conditions, is in bold.

Abundance Indicators Pelagic Zone Catch Per Effort (Data: CDFW Fall Midwater Trawl)			
Central-West Delta			
Quantitative Reference Condition	Interpretation	Low End of Range	High End of Range
>150% of the 1980-1989 Average	Excellent	>75.54	NA
>100% of the 1980-1989 Average	**Good**	**>50**	**75.54**
>50% of the 1980-1989 Average	Fair	>25.18	50
>15% of the 1980-1989 Average	Poor	>7	25.18
<15% of the 1980-1989 Average	Very Poor	NA	>7
Central-West Delta			
Quantitative Reference Condition	Interpretation	Low End of Range	High End of Range
>150% of the 1980-1989 Average	Excellent	>195.85	NA
>100% of the 1980-1989 Average	**Good**	**>131**	**195.85**
>50% of the 1980-1989 Average	Fair	>65.28	131
>15% of the 1980-1989 Average	Poor	>19.58	65.28
<15% of the 1980-1989 Average	Very Poor	NA	<19.58

4. Results of Food Web Productivity Indicator (Total Fish Abundance)

Suisun Marsh

Total Fish abundance in Suisun Marsh declined over the period of record (Figure 27). Levels detected in the first few years of the survey were "excellent" or "good", but became consistently "fair" or "poor" during the late 1980's and early 1990's. A rebound in fish abundance caused the indicator to reach "good" conditions in the year 2000, but since that time, abundance has declined and was "fair" or "poor" (on average, "fair"), over the last five years.

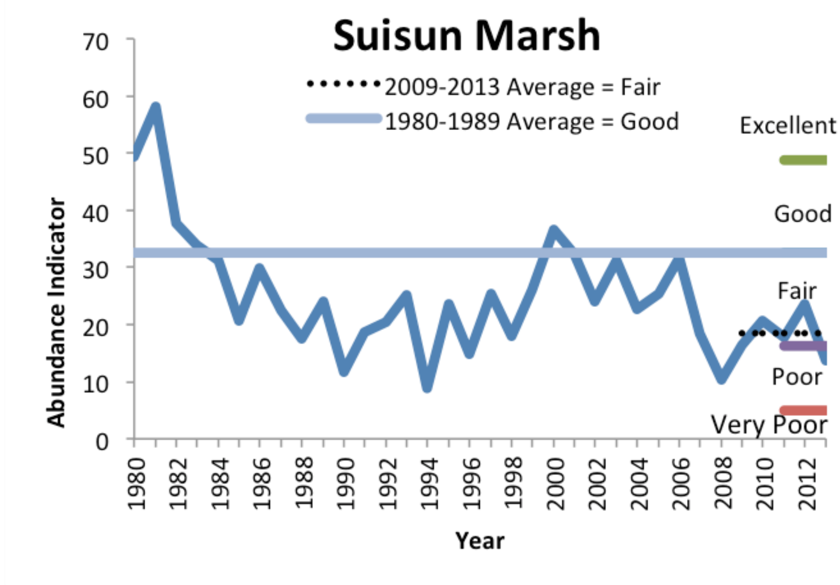


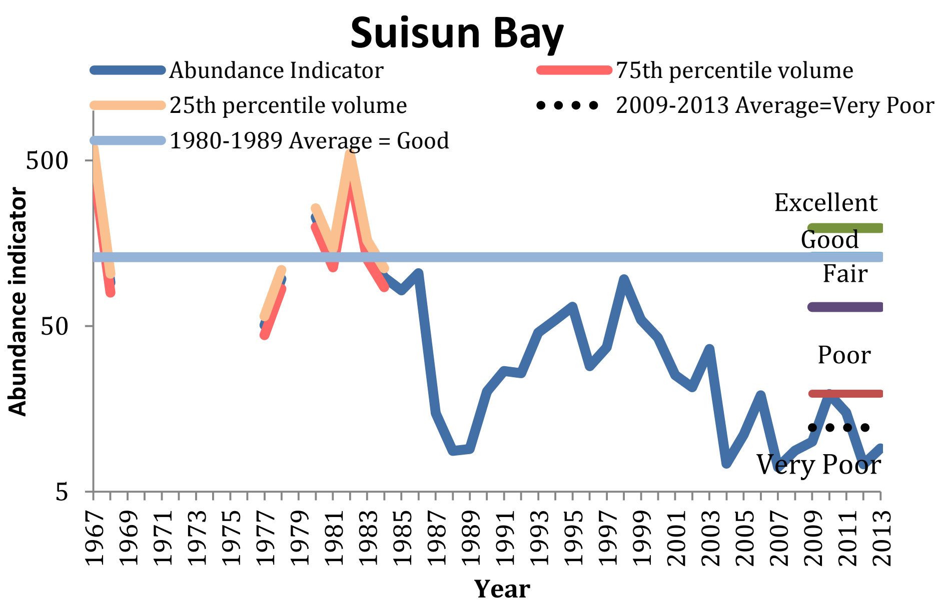
Figure 27. Suisun Marsh Food Web Productivity Indicator (Total Fish Abundance) from 1980-2013. Over the period of record the abundance indicator has declined and the recent five-year average is "fair". Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 14). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line.

Upper Estuary Pelagic Zone

Total fish abundance Indicators in the Pelagic Zone have declined dramatically over time, with recent averages that were "very poor". Small differences were detected in the fish assemblage abundance patterns between the two regions sampled – Suisun Bay (Figure 28) and the Central-West Delta (Figure 29). Although total fish abundance indicators in both regions declined dramatically, they displayed different patterns of decline. The abundance indicator in Suisun Bay followed a trend that was broadly similar to that seen in Suisun Marsh abundance; the abundance indicator was "excellent" in the early years of the survey and even in the earliest years of the primary reference period (1980-1989). However, they declined rapidly just prior to the onset of the 1987-1994 drought. A small rebound in abundance was detected in the late-1990's, but the indicator declined persistently through the early 2000's. The average of the last five years indicates that the fish assemblage in this region/habitat was in "very poor" condition.

Total fish abundance trends in the Central-West Delta Pelagic Zone are different in degree from those described for the Suisun Bay Pelagic Zone and Suisun Marsh. Here, the abundance index appeared to be somewhat stable throughout the 1980's and early 1990's. Both, the increase in the late 1990's (to "excellent") and the precipitous decline in abundance after the early 2000's were consistent with patterns seen in Suisun Bay and Suisun Marsh. The average of the most recent five years indicated that the pelagic fish assemblage in this area is in "very poor" condition.

a. Log-Scale



b. Actual CPUE

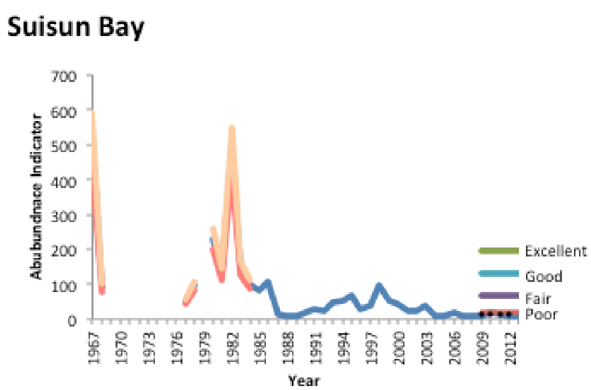
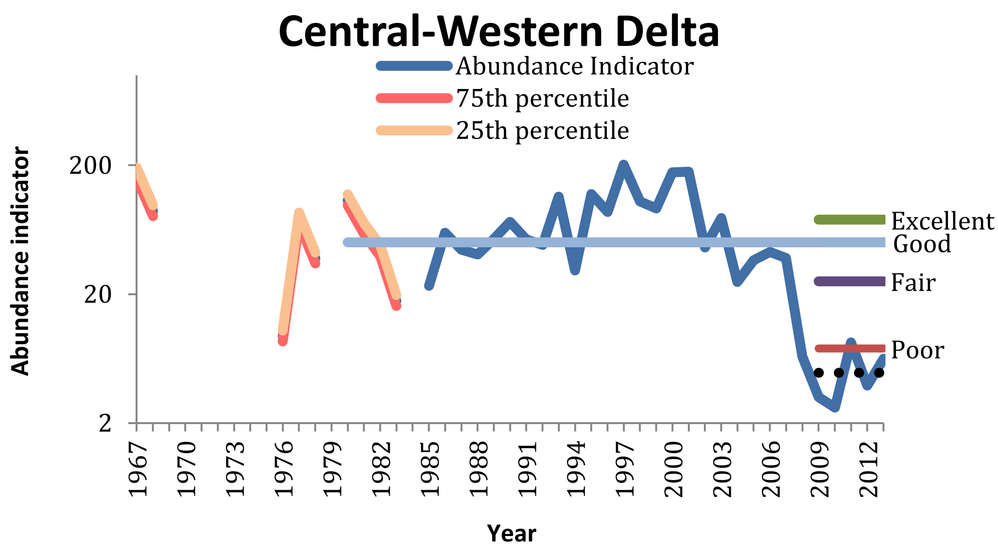


Figure 28. Upper Estuary Pelagic Zone Food Web Productivity Indicator (Total Fish Abundance) for the Suisun Bay region from 1967-2013. In Panel a, the y-axis is log scale; declines appear more pronounced on an untransformed scale (Panel b). Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 12). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average. Fish abundance in the Pelagic Zone of Suisun Bay is “very poor”. Volume sampled was not recorded consistently during 1967-1984 period; thus, for this period, volume sampled was estimated as the mean volume from 1985-2013. Catch-per-unit-effort (i.e., per volume)

was also estimates using the 25th and 75th percentile values of volume sampled between 1985-2013; the effect of different sampling volume estimates are shown in peach and pink lines respectively.

a. Log Scale



b. Actual CPUE

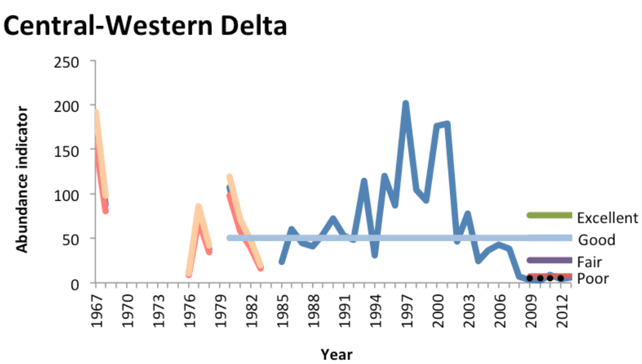


Figure 29. Upper Estuary Pelagic Zone Abundance Indicator for the Central-West Delta region from 1967-2013. In Panel a, the y-axis is log scale; declines appear more pronounced on an untransformed scale (Panel b). In either case, there has been a rapid decline in fish abundance since the year 2000. Short horizontal colored lines indicate scoring thresholds assigned to this indicator (see Table 12). The primary reference condition (1980-1989 average) is indicated by a light blue horizontal line. The dotted line represents the 2009-2013 average and shows that fish abundance in the Pelagic Zone of the Central-Western Delta is “very poor”.

Beach Zone

In most regions of the Delta Beach Zone, total fish abundance has increased over the period of record (Figure 24). Trends in total fish abundance in the South and Central-West Delta Beach Zone but differed from trends in the North Delta Beach Zone (Figure 24). ***In some regions, fluctuations in the raw abundance indices were clearly driven by extreme population changes in abundance of just a few species (Figures 21 and 25).*** Fish that dominate the raw abundance indices (the most common fishes) may or may not represent prey or competitors to other species in the area. As a result, indicators of standardized fish abundance are presented for all species in each region of the Delta and standardized abundance of native and non-native species are compared to determine if different parts of the fish assemblage displayed different abundance trends.

North Delta. Abundance of the average fish species increased slightly from "poor" to "fair" in the Beach Zone of the northern Delta over the last 20 years (Figure 26, left panel). Abundance trends in the North Delta Beach Zone were generally more stable than those in other regions of the Delta Beach Zone (Figure 24). When standardized abundance scores of native species and introduced species were compared (Figure 26, right panel), standardized abundance of the two groups were significantly and positively correlated. This indicates (slight) improvement in the average abundance of the average fish species in the North Delta regardless of whether they were native or non-native.

East Delta. Abundance of the average fish species increased from "poor" to "fair" in the East Delta Beach Zone over the last 20 years (Figure 24; Figure 26, left panel). As with the North Delta Beach Zone, the standardized abundance indicator was "poor" in the early sampling period, hitting a low in the early 2000's. After the early 2000's, the index increased hitting a high for the period of record in 2011 (Figure 24 and 26) -- the average of the last five years is "fair". As in the North Delta, standardized abundance scores of native species and introduced species were significantly correlated (Figure 26, right panel), indicating that native and introduced species are contributing to the improvement in overall abundance.

South Delta. Abundance of the average fish species in the South Delta Beach Zone increased from "poor" to "fair", with some recent years scoring "good" on our scale (Figure 26, left panel); however, standardized abundance scores of native species and introduced species reveal that non-native species accounted for all of the apparent increase in abundance in recent years (Figure 26, right panel). Native species declined in the early years of sampling but the abundance of the average native species remained relatively stable since about the year 2000. The abundance of the average non-native species increased driving an increase in overall fish abundance in this region (see also Figure 24).

Central-West Delta. Abundance of the average fish species in the central and western Delta Beach Zone has increased from "poor" to "fair" (Figure 26, left

panel). Total fish abundance has increased dramatically over the past twenty years in this region of the Delta Beach Zone (Figure 24). As in the South Delta Beach Zone, standardized abundance scores of native species and introduced species were not significantly correlated in this region and introduced species accounted for all of the increase in fish abundance in recent years (Figure 26, right panel).

Summary of Beach Zone Total fish abundance and Diversity Trends
Taken together, the Beach Seine data reveal that abundance of fish in the shallow, unvegetated waters of the Delta increased in recent years, from “poor” to “fair”. Much of this increase was due to a consistent increase across regions in the abundance of introduced species. While our standardized indicator reduces the effect of any one species on the overall pattern for the assemblage, it is worth noting that two non-native species (Inland Silverside and Red Shiner) accounted for the vast majority of all fish caught in the Delta beach seine (Figure 21). Native species abundance increased in concert with introduced species in two regions (North and East Delta) and remained mostly stable in the South and Central-West data.

Analysis of abundance data from this sampling survey reveals important lessons about the construction and application of indicators to measure the health of an assemblage or larger ecosystem. For example, the fact that abundance has increased in all of the four regions of the Delta does not necessarily indicate that the health of the local fish assemblage is improving. Although native species abundance remained stable in some regions and increased in others, most of the change in Delta Beach Zone fish abundance has been due to large increase in abundance of introduced species. Finally, the increases in abundance of native species in some regions of the Delta Beach zone were primarily due to species that spawn predominantly outside of the Delta-proper and then migrate into the sampling area (e.g. Sacramento sucker and Sacramento splittail). These findings reveal the value of evaluating multiple “health” indicators and emphasize the need to dissect trends in synthetic indicators to increase resolution of underlying trends.

5. Summary of Food Web Productivity Indicator (Total Fish Abundance)

Total abundance of fishes in the Pelagic Zone and Suisun Marsh decreased substantially since the early 1980’s and the decline has accelerated since the early part of this century; trends in abundance were remarkably similar between these two habitats. Total fish abundance and abundance of the average fish species in the Delta Beach Zone has increased in recent years; most (but not all) of this change is attributable to increases in abundance of introduced fish species in this habitat/region.

Based on abundance, the CCMP goals to recover and reverse declines of estuarine fishes (SFEP 2007) have not been met in the upper Estuary region.

V. SUMMARY

Collectively the results of fish indicators for the upper Estuary provide insight into a few key attributes of fish assemblage health. **Although no synthetic index of our measures of assemblage health was constructed, it is clear that the fish assemblage in the upper Estuary is in very bad condition (Table 17).** The “good news” is that food web productivity (total fish abundance) indicators in Suisun Marsh (Figure 27) and the Delta Beach Zone (Figure 26) scored “fair” (a decline for the former, but an increase for the latter habitat). Pelagic Zone food web productivity indicators were “very poor” across the upper Estuary (Figures 28 and 29). Total fish abundance in all zones was dominated by introduced fish as most regions scored “very poor” or “poor” for assemblage composition indicators and native fish abundance.

Also, there was no suggestion that introduced species abundance negatively affected indicators of native species abundance, as a whole. In the Delta Beach Zone, the pattern indicated that abundance of native and introduced assemblages were positively correlated or uncorrelated, not negatively correlated as one would expect if introduced species were bad for native species, as a whole. In other habitats, the fraction of fish caught that were native species remained very low throughout the period and were not correlated with the declines in indicators of total fish abundance over the period of record.

Because this coarse metric does not reveal where the fish sampled were produced, a more refined investigation is warranted to determine whether native species and introcued species abundances responded to the same environmental processes and/ or the local operation of those processes. Also, these indicators were designed to reflect trends across a broad range of species and, as a result, they say little about the trends in any one species and the particular forces that drive those changes. Thus, it is possible that certain native fish species are responding to direct or indirect effects of introduced fish species, even though the assemblage-wide trends do not detect a general pattern of this type. Future indicators that assess species distribution and other attributes of health will likely increase our understanding of the health of fishes in upper Estuary and the local mechanisms that contribute to assemblage health.

Table 17. Summary of Results relative to the CCMP goals to “recover” and “reverse” declines of estuarine fishes for the fish indicators in the Upper San Francisco Estuary.

Indicator	Region (Sub-region if trends are different)	CCMP Goal Met	Evaluation		Trend
			Reference Period	Short-Term (last five years)	Over the Period of Record
Native Fish Abundance	Suisun Marsh	No	Good	Poor	Decline
	Suisun Bay Pelagic	No	Good	Very Poor	Decline
	Central-West Delta Pelagic	No	Good	Very Poor	Decline
	Delta Beach Zone	No	Poor	Poor	Stable
Percent Native Fish	Suisun Marsh	No	Very Poor	Very Poor	Stable
	Suisun Bay Pelagic	No	Poor	Poor	Stable
	Central-West Delta Pelagic	No	Very Poor	Very Poor	Stable
	Delta Beach Zone	No	Very Poor	Very Poor	Stable
Percent Native Species	Suisun Marsh	No	Poor	Very Poor	Decline
	Suisun Bay Pelagic	No	Fair	Fair	Stable
	Central-West Delta Pelagic	No	Poor	Very Poor	Decline
	Delta Beach Zone	No	Very Poor	Very Poor	Stable
Food Web Productivity Indicator	Suisun Marsh	NA	Good	Fair	Decline
	Suisun Bay Pelagic	NA	Good	Very Poor	Decline
	Central-West Delta Pelagic	NA	Good	Very Poor	Decline
	Delta Beach Zone – North	NA	Poor	Fair	Increase
	Delta Beach Zone – East	NA	Poor	Fair	Increase
	Delta Beach Zone – South	NA	Poor	Fair	Increase
	Delta Beach Zone – Central-West	NA	Poor	Fair	Increase

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STATE OF THE ESTUARY

HABITAT WILDLIFE PROCESSES PEOPLE

WATER

SAN FRANCISCO BAY & DELTA

State of the Estuary Report 2015

Summary and Technical Appendix

WILDLIFE – Harbor Seals

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Living Resources: “Birds and Mammals”
Revised version, 13 May 2015
Sarah Allen and Denise Greig (reviewed by Nadav Nur)

Harbor Seal Indicator

1. Brief description of indicator and benchmark

Harbor seal abundance in the San Francisco Bay (SFB) estuary, excluding pups, is used as an indicator. The indicator is based on a time series of counts of harbor seals during the breeding season. The range that defines a Fair score is the historical average maximum number counted at select locations from 2000 to 2010 plus or minus one standard deviation. Good is defined as above the upper standard deviation. Poor is defined as below the lower standard deviation. The historical mean from 2000 to 2010 is 328 seals; the range for Fair is between 273 and 382 (i.e., 328 ± 54 [SD]) seals.

2. Indicator status and trend measurements

Fair. In the past three years, one year is within the range for Fair, one year is at the cut-point between Fair and Poor, and one year is in the Poor range. There has been no consistent trend, nor has there been a significant linear trend from 1998-2014 ($P > 0.1$; see Figure 1).

3. Brief write-up of scientific interpretation

Harbor seals are an apex predator within the SFB estuary and along the nearshore of the outer coast. They lead a dual existence in that they rest, molt, and nurse their pups on land at traditional terrestrial haul out sites, and so are easily surveyed, but forage at sea, often in close proximity to haul out sites.

Indicator of estuarine condition

An indicator of harbor seal abundance within the SFB estuary was used to reflect the health of the estuary:

- Number of harbor seals (excluding pups)

Harbor seal abundance is an excellent biological indicator of foraging conditions in the estuary and the outer coast. Seals are opportunistic predators of fish and invertebrate species that are seasonally abundant, and they respond quickly to changes in regional environmental conditions. For example, during El Niño years when many prey species moved away from warmer than usual waters, the number of total seals surveyed at colony sites in central California declined (Allen et al. 1989; Sydeman and Allen 1999). Marine mammals generally are used as sentinels of change in oceanic conditions (Moore 2008), and within SFB estuary harbor seals are the only year round resident marine mammal. Individual seal health has been linked to anthropogenic pollutants in the SFB estuary in several studies since the 1970s: these data have been used to understand health risks to humans using the estuary as well (for example, mercury levels in sport fish).

Seal numbers within the bay are important for understanding the ongoing impacts of humans on the bay ecosystem because the population is vulnerable to disturbance, habitat loss, contaminants and prey availability, as well as the cumulative effects of these factors on seal health and survival.

Benchmark and Scoring

The scoring approach is based on the mean of the annual maximum number of seals counted at two locations from 2000-2010 (excluding 2007 when we did not count one of the two sites), i.e., 328 seals. 1998 and 1999 were excluded from the historic mean because 1998 reflect the strong El Niño effects of that year, and there may have been some residual effects in 1999. From the 2000-2014 time series, a Standard Deviation (SD) about the mean was calculated, i.e., 54 seals. Good and Poor were defined relative to these historical data (i.e., above and below the mean ± 1 SD). A score of "Good" requires three most recent years all to be above the upper SD, i.e., all three years to be above mean plus 1 SD. A score of "Poor" requires three most recent years all fall 1 SD or more below the historic mean. If 1 to 3 of the most recent 3 years fall within the range of Fair, this is scored as Fair.

The indicator is based on counts of seals (excluding pups) during the breeding season. Pups are excluded from the indicator because their numbers are more variable and an occasional year with low pup numbers is not likely to impact the health of the population. Pup sensitivity to short term perturbations is illustrated in the Technical Appendix, where the pup numbers were dramatically decreased during the 1998 El Niño. However, data on pup trends are important ancillary information to interpret seal trends.

Protocols for monitoring harbor seals are well established and have been implemented in the SF estuary since 1998 at two prime locations for breeding harbor seals. This approach provides an index of abundance that can be consistently replicated.

Status and Trend

Status: Fair
We consider the harbor seal population status to be fair: there has not been a substantial drop in numbers, however numbers have not improved substantially since the 1970's as they have for harbor seal populations along the adjacent coast after passage of the Marine Mammal Protection Act (1972). In addition, there remain ongoing concerns regarding their health as a result of pollutants introduced to the bay by humans (oil, mercury, pesticides, and other contaminants) and habitat loss.

Trend: No distinct trend.
The numbers have been variable: numbers were Poor in 2011 and 2012, but back within range in 2014 and nearly so in 2013. There is no significant linear or quadratic trend in the data from 1998 -2014 ($t_{16} = -0.86$, $P > 0.4$ for linear; $t_{15} = -1.25$, $P > 0.2$, for quadratic) analyzing ln-transformed counts), nor was a linear or quadratic trend significant when analyzing just 2000-2014 ($P > 0.1$; $P > 0.9$, respectively), omitting the El Niño year of 1998 and the year following, 1999.

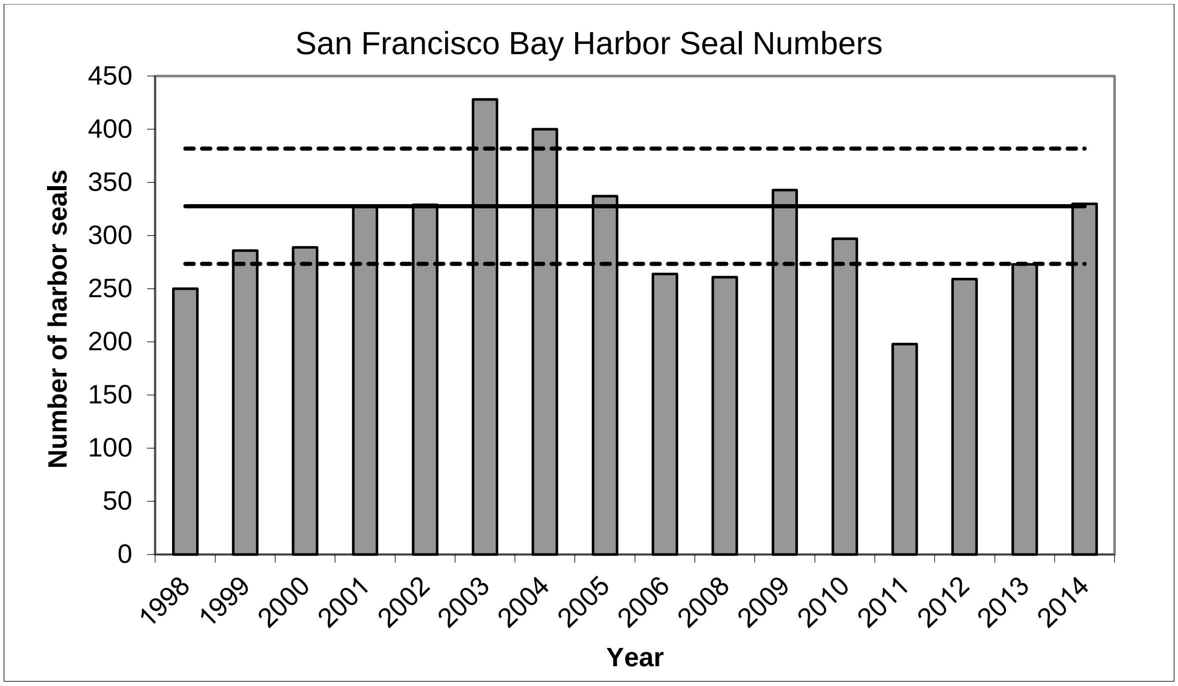
SIGNIFICANCE/INTERPRETATION

While there is no clear trend to the data between 1998 and 2014, seal abundance is a useful indicator for understanding the population of harbor seals within the estuary. Adult harbor seal numbers decreased in 2011, but the pup numbers were not as depressed.
If numbers are consistently depressed below Fair, then this will be a reflection of changes within the estuary that are likely affecting other species as well the seals. If the numbers increase, that would signal that the SFB estuary environment has improved (either more productive or improved habitat).

There is a strong ecological linkage between the SFB estuary and the regional coastal conditions to which the seals within the SFB estuary are responding. The SFB plume of fresh water and sediments extends out of the Golden Gate and drifts north and south, depending upon tides and winds. The plume provides significant nutrients to the coastal waters and contributes to the biological diversity. Conversely, colder and saltier coastal waters extend far up into the SFB estuary contributing equally to the biological diversity within the SFB. When anomalies in weather patterns occur, these linkages are altered as occurs during El Niño years. NOAA documented 2015 as an El Niño year because of unusual warm ocean conditions. The warm conditions are often associated with a breakdown in food webs with less krill and anchovies present in nearshore coastal waters that provide prey to seals and seabirds; however within SFB estuary, resident seals forage more on resident prey species, which might lessen the El Niño effects on the seals within the bay. The intensity and frequency of El Nino events is predicted to increase in the future in response to changes in climate. Monitoring in SFB estuary is important to identify the potential effects of these events on sentinel species such as seals and to provide opportunities to react to these new types of events as they unfold.

4. Related figures

Figure 1. Maximum harbor seal numbers (excluding pups) counted during the breeding season at two locations in the SFB estuary (Castro Rocks and Yerba Buena Island). The solid line is the mean from 2000 to 2010; the dotted lines are 1 SD above and below the mean. The area between the dotted lines is scored as Fair. 2007 is omitted from the figures and the calculation of the mean because YBI was not sampled that year. Estimated linear trend for 2000 to 2014 is -1.4% (SE = 1.0%) per year ($P > 0.1$); trend for 1998-2014 also was not significant. Poor is below the lower dotted line. Fair is between the dotted lines. Good is above the upper dotted line.



5. Technical appendix

Harbor seals are marine carnivores that rest ashore daily, and therefore, seal numbers are indicative of both prey availability and suitable harbor seal habitat within the SFB estuary. Harbor seals have been studied in the SFB estuary since the early seventies when concerns were raised about the connection between pollutants in the bay and premature harbor seals births (Risebrough et al. 1980). Studies have investigated pollutants (Kopec and Harvey 1995, Neale et al 2005, Greig et al. 2011), levels of mercury and selenium (Kopec and Harvey 1995, Brookens et al 2007, McHuron et al 2014), food habits and movements (Harvey and Torok 1994, Nickel 2005, Grigg et al 2009, Gibble and Harvey 2015), disturbance (Allen 1991) and survival (Greig 2011, Manugian 2013). These studies showed that harbor seals do forage within the bay (Harvey and Torok 1994, Nichol 2003, Gibble and Harvey 2015) and the amount of time that they spend on local haul-outs (Green et al. 2006) as well as some of the factors affecting health and survival and therefore seal numbers (Kopec and Harvey 1995, Grigg et al. 2004, Greig 2011, McHuron et al 2014).

The studies above do not, however, provide a consistent, easily replicated indicator for monitoring perturbations in the seal population over time. State and federal agencies do aerial surveys, but not every year (Carretta et al 2013). From 1998 to the present, harbor seals have been counted consistently through the pupping and molt seasons at the same locations by a network of citizen scientists with data managed by the National Park Service. The proposed count data provide the best index of seal numbers and pup production, and thus, indicate the ability of the estuary to support seals and these activities.

- Benchmark
    - Describe the benchmark and why it was chosen.

The scoring is based on the mean of the annual maximum number of seals counted at two haul-out locations from 2000-2010 (excluding 2007 when we did not count one of the two sites), i.e., 328 seals. The Fair scoring range encompasses one standard deviation above or below the mean. This metric was chosen because seals have been consistently counted at two of their larger haul out sites within the estuary and we think the variability around the mean of the historical dataset provides a good reference point for evaluating natural variability and for detecting any deviation away from that mean.

- Discuss any limitations of the benchmark and how it might be improved in the future.

The breadth of the Fair score range is affected by the degree of variability in the dataset from sources not associated with the seal population (such as the observer differences, poor weather and reduced visibility). In addition, only two sites are monitored consistently, so increases or decreases could result from seals moving to or from other locations in the bay, or from the seals using the two index sites differently (for example, increased use of Yerba Buena Island as nursery area). Improvements to the scoring could include additional sites (especially if haul out patterns in the south bay change with restoration efforts) as well as an expanded seal monitoring program within the SF estuary.

- Provide any further related information about goals, reference conditions and targets.

Indicator values below Fair for three consecutive years would be cause for concern and should result in management action. For example, a management action might include an estuary-wide aerial survey program to document if seals moved somewhere else within the estuary or were gone from the estuary's population. An increase above Fair for three years in a row would indicate that the seals and the SFB estuary were healthy and the status of this indicator would then be scored as "Good".

- Data Sources
    - Describe the data used and where they came from.

Data are collected by volunteers for the National Park Service's San Francisco Area Network Inventory and Monitoring Program. The data are curated and validated at Point Reyes National Seashore and published each year in a peer reviewed annual report (Adams et al. 2009).

- Methods
    - Describe the calculation methods.
    - Include a brief description of the assumptions and uncertainties.

Sampling locations consist of two of the largest breeding and resting sites in the estuary: Castro Rocks under the Richmond/San Rafael Bridge and Yerba Buena Island in the middle of the San Francisco/Oakland Bay Bridge. Pupping occurs at other locations and new sites may be colonized by seals, but the sites that make up the indicator have been surveyed consistently since

1998. Note that pup numbers are not included in the harbor seal population index but serve as ancillary data for interpretation of the index.

Breeding season surveys are conducted every other week (coinciding with the low tides) from March through May and every week for the three weeks surrounding peak pupping (late April/early May). The maximum number of non-pups counted on any survey during the season is used as the indicator. Numbers can vary dramatically with weather and disturbance, and this sampling regime is designed to account for that variation, although it is possible that no good counts are acquired in a given season due to weather or other factors.

Because not all locations within the estuary are monitored and because seals spend seasonally varying proportions of their time at sea, these counts are an index of seal abundance rather than an attempt to estimate total numbers of seals in the estuary.

Harbor seal count data from 1998 through 2005 were collected as part of the Richmond Bridge Harbor Seal Survey (Green et al 2006, Grigg et al. 2004). Data from 2006 to the present have been collected by volunteers using the same protocols and datasheets as before, but less effort in terms of time. From 1998 through 2005, paid survey shifts lasted six hours and were distributed across time of day and tidal cycle. Now, volunteers survey from 30 minutes to 2 hours during low tides in concert with the region-wide harbor seal survey (which includes Point Reyes Peninsula and locations along the Sonoma and San Mateo coastline).

- Peer Review
    - Describe how the indicator was vetted with other experts in the community.

See Data Sources section above – the regional count data are reviewed and made public each year (http://www.sfnps.org/harbor_seals/).

Work precedents for this indicator include 30 years of previous monitoring under different programs including Marine Mammal Commission in 1980s, CalTrans and other sources. See references including Risebrough et al. 1980, Allen 1991, Kopec and Harvey 1995, Grigg et al. 2004, and Green et al. 2006).

- Additional Information

Seal mortality is also documented during surveys, therefore an outbreak of disease or other health problem might be detected if carcasses were evident during the survey period. NOAA documents Unusual Mortality Events (UME) of marine mammals and such events are indicators of potential health issues in populations. In the past two decades, two harbor seal mortality events have occurred in the SF Bay Area (Nollens et al. 2010).

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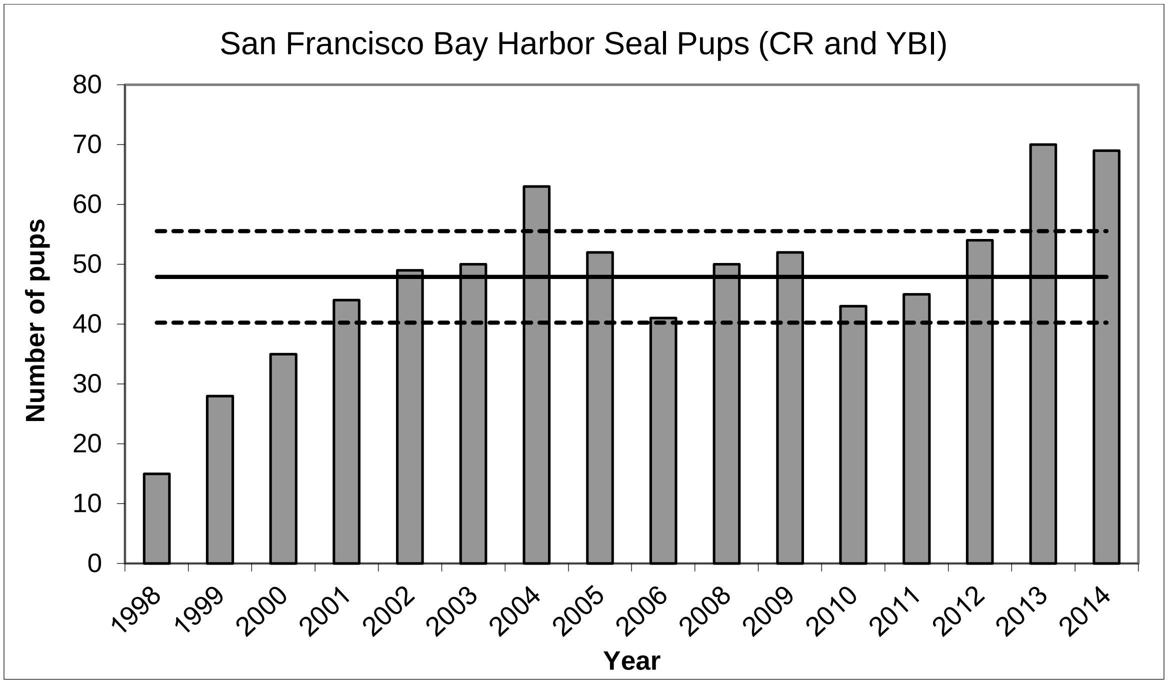
Figure TA-1. Maximum harbor seal numbers (pups only) counted during the breeding season at two locations in the SFB estuary (Castro Rocks and Yerba Buena Island). This metric is provided for additional information and is not used in the calculation of scoring. The solid line is the mean from 2000 to 2010 and the dotted lines are 1 SD above and below the mean. Seals were not counted in 2007.



A degraded pier in Alameda is a harbor seal haul out site and an example of the shortage in SFB of preferable habitat for seals to rest on land near foraging areas. The haul out site, though marginal habitat for seals, will likely be lost due to proposed replacement, which will make the pier inaccessible to the seals. Future restoration of the South Bay will potentially provide high quality habitat for seals where they can give birth and rest on land undisturbed and near feeding areas.

State of the Estuary Report 2015

Summary

WILDLIFE – Wintering Waterfowl

Prepared by

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State of the Estuary 2015: Wildlife

Wintering Waterfowl Population Indicator

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Final, October 2015

1. Brief description of indicator and benchmark; background

The indicator is based on indices of population abundance for two principal groups of waterfowl, dabbling and diving ducks, calculated for three regions of the Estuary: North Bay, Central San Francisco Bay and South San Francisco Bay. The indices here were determined for 1989-2014 based on data from the Midwinter Waterfowl Survey (MWS) conducted in January of each year, an annual survey conducted in the United States since 1955. For more detailed information on the MWS in the San Francisco Bay region, see Richmond et al. (2014).

For dabbling ducks, we used the six most commonly observed species in the dataset: American wigeon, gadwall, green-winged teal, mallard, northern pintail, and northern shoveler. For diving ducks, we used the six most commonly observed "species": bufflehead, canvasback, goldeneye (both Barrow's and common goldeneye), ruddy duck, scoter (black, white-winged, and surf scoter), and scaup (lesser and greater scaup). Henceforth we will refer to the six diving duck taxa as "species." Based on previous studies (Accurso 1992), birds coded as "goldeneye" were assumed to be predominantly common goldeneye, and birds coded as "scoters" were assumed to be predominantly surf scoters. For each group (dabbling ducks, diving ducks), the indicator developed synthesizes population change among all six "species," rather than species by species.

We used the historic period, 1989-1993, as the reference baseline from which we developed our benchmark. The reference period constituted the first five years of the time series available for analysis, and aligned with the published objectives of the San Francisco Bay Joint Venture. The most recent 5-year period (2010-2014) was compared to the reference baseline period. Abundance exceeding the reference value was scored "Good." Abundance 40% or more below the reference baseline was scored "Poor." Thus, Fair was 60% to 100% of the reference value. In addition, we calculated linear and quadratic trends for each group, in each region, during the entire period and also for the last 10 years.

Background:

Because of the long-recognized importance of waterfowl to the mission of the U. S. Fish and Wildlife Service, the MWS has been conducted by this agency, throughout the United States since 1955, in cooperation with state and other federal agencies. The indicator used here for the

San Francisco Estuary draws on a localized subset of the data, gathered as part of a long-standing nation-wide effort. The survey attempts to enumerate all waterfowl, by species, for “high concentration areas” of the estuary. Survey efforts in the San Francisco Estuary target two main habitat types: open bay and salt production/managed ponds. The MWS complements breeding waterfowl surveys (see, in particular, Wildlife: Breeding Waterfowl Index for the Delta and Suisun). Because the MWS has been conducted intermittently in Suisun open bay, here we only present results for North San Francisco Bay (“North Bay”), Central San Francisco Bay (“Central Bay”), and South San Francisco Bay (“South Bay”). Note that North Bay is largely coincident with San Pablo Bay.

Note that most waterfowl wintering in the San Francisco Estuary breed elsewhere in the Pacific or Central Flyways (including Alaska, Canada, the Intermountain-West, as well as additional areas inside and outside of California).

2. Summary of Indicator status and trend measurements

Indicator status was determined for dabbling ducks and for diving ducks, for each of the three regions of the San Francisco Estuary (Table 1). Comparing the recent period with the reference period, the abundance index for dabblers in North Bay increased by 631% and was scored “Good.” Results are displayed in Figure 1A; the index itself is natural log-transformed, but to summarize changes in abundance for the purposes of the Table, index results were back-transformed, providing estimates of percentage change. The abundance index for North Bay divers decreased by 59% and was scored “Poor” (Table 1, Figure 1A).

The abundance index for Central Bay dabblers increased by 274% and was scored “Good” (Table 1, Figure 1B). The abundance index for Central Bay divers decreased by 71% and was scored “Poor.” The abundance index for South Bay dabblers increased by 157% and was scored “Good” (Table 1, Figure 1C). The abundance index for South Bay divers increased by 21%. Because the increase was not statistically significant, we scored this group as “Fair.” Thus, the four “Good” results were all significantly greater than the reference value ($P < 0.01$ or better; Table 1); the two “Poor” results were more than 40% below the reference value and each was significantly different from the baseline (Table 1). The “Fair” result was only 21% higher than the reference value and was not significantly different from the reference value.

Table 1. Summary of Wintering Waterfowl Indicator Results for San Francisco Bay. Percent change in wintering waterfowl abundance index for recent (2010-2014) vs. historic reference (1989-1993) periods for dabbling ducks and diving ducks, by region. Values shown are percent differences in the count index for the two time periods. P values for t tests on the differences between the two time periods are shown as well as the Status Score (see text and Technical Appendix for details).

	Dabbling Ducks			Diving Ducks		
	Percent change	P-value	Status	Percent change	P-value	Status
North Bay	631%	$P < 0.001$	Good	-59%	$P = 0.001$	Poor
Central SF Bay	274%	$P = 0.009$	Good	-71%	$P < 0.001$	Poor
South SF Bay	157%	$P < 0.001$	Good	21%	$P > 0.2$	Fair

Long-term linear trends were assessed over the entire time period, 1989-2014. North Bay and South Bay dabblers demonstrated significantly increasing trends ($P < 0.001$ in both cases; Table 2; Figure 1A, 1C); the trend for Central Bay dabblers was an increase of borderline significance ($P = 0.052$ for the test of the slope differing from zero; Table 2, Figure 1B). North Bay and Central Bay divers exhibited significantly decreasing long-term linear trends ($P < 0.001$ in both cases; Table 2); South Bay divers demonstrated an overall trend that was near zero (<1% change per year), and did not differ significantly from zero ($P > 0.4$).

The long-term linear trend results were predominantly concordant with the benchmark determinations. The only difference was that for Central Bay dabblers, the recent years had a significantly higher abundance than the reference period ($P < 0.001$), but the linear trend was only of borderline significance ($P = 0.052$). The discrepancy was due to the increase for this group only being manifest in the more recent years, rather than during the entire period (Figure 1B).

In addition, non-linear trends were assessed, as were short-term trends; for these, see “Trend Results” below.

3. Brief write-up of scientific results and interpretation

a. What is this indicator?

The indicator is an index of abundance calculated for two groups of ducks (dabbling ducks and diving ducks) during the winter, as determined by the MWS for the period 1989-2014 (Richmond et al. 2014). The index is calculated for each of three regions: North San Francisco Bay, henceforth "North Bay" (which includes San Pablo Bay and adjacent managed ponds), Central Bay, and South Bay (which includes adjacent salt production and managed ponds). Surveys were not conducted in Suisun Bay in many years, hence Suisun Bay surveys have not been analyzed here. Total counts of birds were summed for each region and year, separately for each species. Counts are not adjusted for incomplete coverage of survey areas (for example, open bay transects are spaced more widely than salt production/managed pond transects) nor for imperfect detection. However, the same routes are generally flown year-to-year, allowing for comparisons across years. Further details on the survey methodology are provided by Accurso (1992) and Richmond et al. (2014).

b. Why is it important?

Waterfowl are an important component of the ecosystem of the San Francisco Estuary, and of the aquatic foodweb more specifically. They represent significant energy flow and biomass, consuming both plants and invertebrates. In addition, duck hunting is an important economic and recreational activity.

San Francisco Estuary provides some of the most important wintering habitat for waterfowl, particularly for diving ducks, in the Pacific Flyway (Goals Project 2000, Steere & Schaefer 2001). For some diving duck species, the San Francisco Estuary hosts nearly half of the birds counted on the MWS in the lower Pacific Flyway, which is made up of major waterfowl concentration areas in Washington, Oregon, California, Arizona, Utah and Idaho and portions of New Mexico, Colorado, Montana and Wyoming (Steere & Schaefer 2001). This is in addition to the estuary's value to waterfowl during the breeding season (especially in the 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 2004). The San Francisco Bay Joint Venture has made waterfowl conservation in the Bay area a priority and we follow their population targets (Steere and Schaefer 2001). Waterfowl conservation has also been a prime objective as part of significant restoration projects in the San Francisco Bay Area. There is the potential for waterfowl to be adversely impacted by restoration that converts former salt ponds into tidal marsh (Stralberg et al. 2009). Thus, tracking waterfowl population changes is one important component of assessing overall response to restoration, as well as to management intended to reduce or eliminate adverse impacts.

This indicator tracks two important groups of waterfowl: **dabbling ducks**, which feed at the surface or in shallow water, and **diving ducks**, which forage in the benthos at deeper depths. Under diving ducks we include bay (*Aythya* spp.), sea (Tribe *Mergini)*, and stiff tail (*Oxyura* spp.) duck species.

c. What is the benchmark? How was it selected?

The reference value used to determine the benchmark is the mean MWS abundance index in the period 1989-1993, the first five years of the available time series for which the current survey methodology was used; the San Francisco Bay Joint Venture has chosen a similar time period for setting population goals in the San Francisco Bay region. The most recent 5-year period (2010-2014) was compared to the reference period. All statistical analyses were carried out on natural log-transformed values to stabilize the variance in counts with respect to species and years as is widely recommended (Nur et al. 1999).

Abundance values exceeding the reference were scored "Good," provided that the difference was statistically significant. Abundance 40% or more below the reference was scored "Poor," assuming that the difference was statistically significant. We maintain that a 40% decline over c. 20 years, which represents an average decline of 2.5% per year, is of sufficient magnitude to elicit serious management concerns, and, possibly, management action. In addition, where the difference between current and reference values was not significant, this was scored as Fair.

d. Indicator Results, Status and Trends

i. Calculation of index values and statistical analysis.

Methods are described briefly here and in greater detail in the Technical Appendix. Counts by species were summed for each region and log-transformed in each year. For each region and for each guild (dabblers, divers), we first fit a linear model, with species and year as factors, i.e., categorical variables. We obtained model-predicted values for each year for that region-guild. To obtain these results we used two different approaches: we either weighted each species equally or we weighted each species by overall abundance in that region. Results were similar using the two approaches, but here we present only results weighted by species-specific abundance, since we consider that result to be more meaningful at the ecosystem level. We then estimated the difference between the two time periods (i.e., comparison of the five reference years with the five current years) using a comparable linear model, with model results weighted by species-specific abundance, and tested if the difference between the two time periods was significantly different from zero. Finally, we fit linear and quadratic trends to the data, again controlling for species as a factor. If the quadratic trend was statistically significant, we display that trend; if not, we graph the linear trend. We report both short-term (last 10 years) and long-term (1989-2014) linear trends, whether or not the quadratic trend was significant, to facilitate comparison among guild-by-regions. Note that a linear trend on log-transformed values provides an estimate of a constant proportional change over the period being analyzed (Nur et al. 1999).

ii. Index results and Scores

Annual variation in the natural-log-transformed waterfowl abundance index is depicted in Figures 1A, 1B, and 1C for each region of the San Francisco Bay estuary. In addition, in the Figures we depict the trend over the entire period, either linear or quadratic, choosing the trend of best fit. We depict a linear trend unless a quadratic trend provided statistically significantly better fit than a linear trend.

Comparison of the current period (last 5 years) with the reference period is summarized in Table 1. Dabbling ducks had sizeable and significant increases in all three regions; therefore all regions are scored “Good.” Diving ducks had strong, significant declines in the North Bay and Central Bay; therefore these are scored Poor. South Bay diving ducks had a modest, non-significant increase comparing current to reference periods; they are scored as Fair.

iii. Trend Results

Trends were analyzed for each region-guild, both long-term (entire time series) and short-term (last 10 years). Linear trend results are shown in Table 2. Quadratic trend results are summarized in the text and in Figure 1, but only where they provide superior model fit compared to a linear trend (i.e., the quadratic coefficient was significantly different from zero). Dabbling ducks had increasing trends in all three bay regions, both short- and long-term. For this guild, all trends were significant except North Bay short-term trend ($P > 0.6$) and long-term trend for Central Bay, which was marginally significant ($P = 0.052$). For Central Bay and South Bay dabblers, the quadratic trend was significant, and up-turned (accelerating; Figures 1B, 1C). In the North Bay, there was no significant quadratic curvature.

Diving ducks had long-term significant declines in the North Bay, but not in the short-term (Table 1). In the Central Bay, both long-term and short-term declines were significant. In the South Bay, there were weak, positive but not significant ($P > 0.1$ or greater) increases both long- and short-term.

Central Bay diving ducks had significant downward curvature (i.e., accelerating decline; Figure 1B). North Bay and South Bay diving ducks evidenced no significant curvature.

Table 2. Trends in the San Francisco Estuary Wintering Waterfowl Population Indicator (midwinter waterfowl surveys, USFWS). Long-term (1989 - 2014) and short-term (2005 - 2014) linear trends in the abundance index for two groups of waterfowl. Shown are estimated annual, constant percent changes per year in the abundance index for the two time periods. P values shown for t test of whether slope is different from zero. Analyses control for species-specific differences, weighted by abundance of each species (see text).

	Number of years	Dabbling Ducks Ann Pct Change	P-value	Diving Ducks Ann Pct Change	P-value
North Bay					
Long-term	23	10.2%	P < 0.001	-4.0%	P < 0.001
Short-term	10	1.6%	P > 0.6	-0.3%	P > 0.1
Central SF Bay					
Long-term	24	4.6%	P = 0.052	-5.6%	P < 0.001
Short-term	10	21.2%	P = 0.010	-10.9%	P < 0.001
South SF Bay					
Long-term	24	5.1%	P < 0.001	0.6%	P > 0.4
Short-term	10	7.3%	P < 0.001	2.7%	P > 0.1

Figure 1A. Abundance Index for dabblers and divers, North Bay. Note reference values (mean, 1989-1993) = 6.49 (dabbler); 9.81 (diver). The best fit was a linear trend for both groups, shown. Model-fitted index values (ln-transformed counts) for each year, weighted by species abundance, are shown.

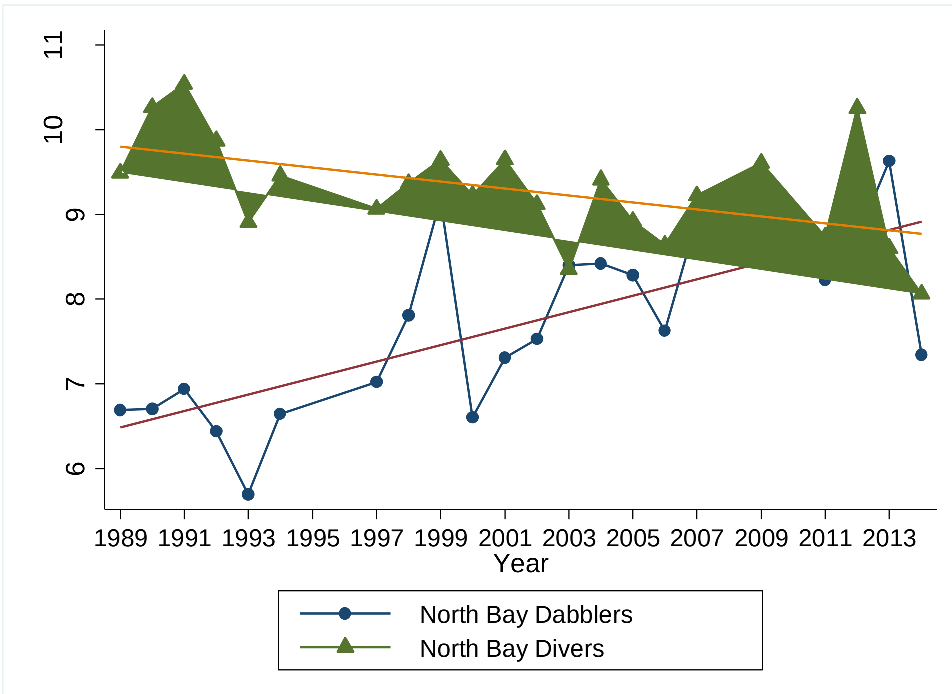


Figure 1 B. Abundance Index for dabblers and divers, Central Bay. Note reference values (mean, 1989-1993) = 2.76 (dabblers); 4.08 (divers). The best fit was a quadratic trend for both groups, shown. Model-fitted index values (ln-transformed counts) for each year, weighted by species abundance, are shown.

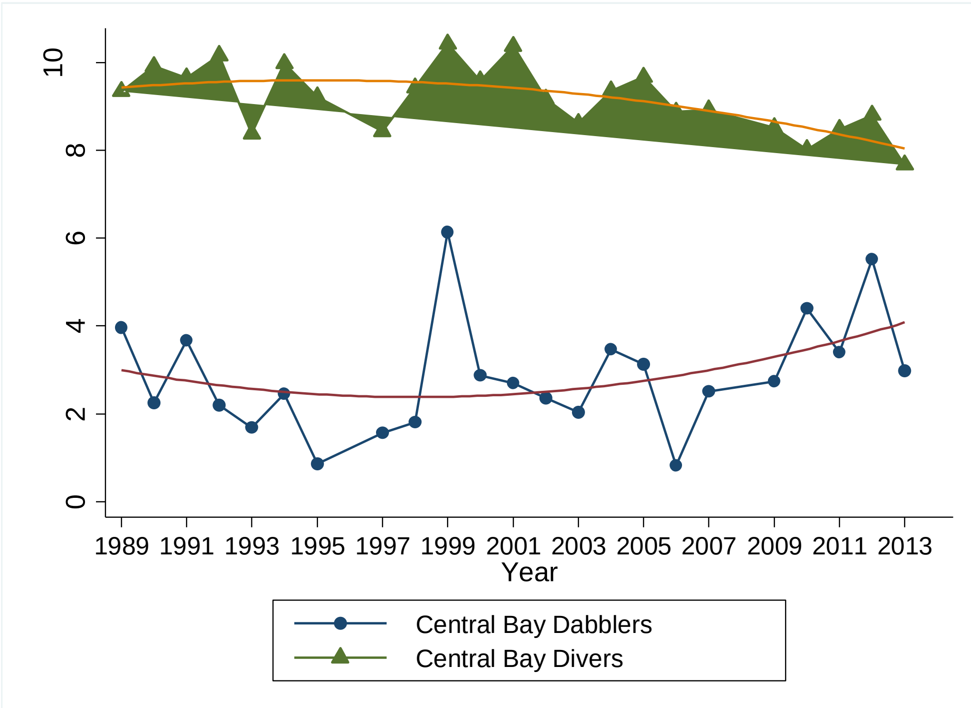
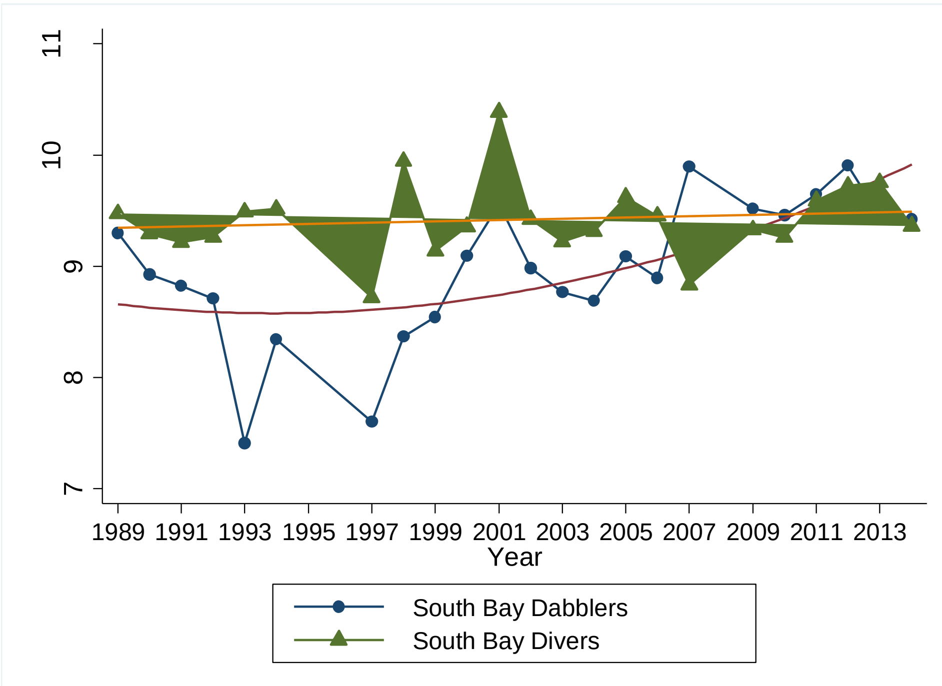


Figure 1C. Abundance Index for dabblers and divers, South Bay. Note reference values (mean, 1989-1993) = 8.63 (dabblers); 9.35 (divers). The best fit was a linear trend for divers and a quadratic trend for dabblers, shown. Model-fitted index values (ln-transformed counts) for each year, weighted by species abundance, are shown.



d. What does it mean?

The dabbler guild demonstrated sharply increasing trends in abundance in the North Bay and South Bay. In fact, species by species, all six dabbler species showed increasing trends in the North Bay, while in the South Bay, five of the six species showed an increasing trend, with only gadwall displaying a non-significant decreasing trend (see Technical Appendix for individual species results). In the Central Bay, dabbler trends were less evident, with no species demonstrating a significant trend, though four out of five were positive; only northern pintail showed a non-significant declining trend (no green-winged teal were present in the Central Bay). Nevertheless, in the Central Bay, recent trends for dabblers were significantly positive, and as a result the comparison of the current period to the reference period was scored as Good, reflecting a 274% increase (P = 0.009).

For **the diving duck guild**, the picture differed depending on bay region. In the North Bay and in the Central Bay, divers demonstrated significant declining trends. Scaup, scoter, and goldeneye all declined significantly in the North Bay; the other three species were weakly positive (ruddy duck) or weakly negative (bufflehead, canvasback). In the Central Bay, scaup and scoter also declined significantly, while bufflehead increased significantly. The other three species either displayed non-significant declines (canvasback, gadwall) or non-significant increases (ruddy duck).

In the South Bay, there was an overall weak, increasing trend for divers, at about 0.6% per year, over the long-term, but 2.7% increase per year in recent years. These trends were not significant (P > 0.1 or greater). Here, results differed most markedly among species. Scaup demonstrated a significant, strong declining trend, whereas ruddy duck, goldeneye and canvasback demonstrated significant positive trends. The other species had non-significant trends either positive (bufflehead) or negative (scoter). Thus, the score of Fair for South Bay divers reflects a mixed picture, i.e., a combination of strong declines for scaup, significant increases for three species, and intermediate, non-significant trends for the other two species.

In summary, dabbling ducks have demonstrated increases (and are scored Good) in all three regions while diving ducks have either declined strongly and significantly (in the North Bay and Central Bay) or demonstrated a mixed picture (in the South Bay) reflecting species-specific differences. In the latter case, the modest overall increase in the diving duck index (compared to reference) is tempered by the strong, significant decline for scaup.

The strong decline in diving ducks is potentially of great concern. However, a better understanding is needed of how wintering distributions of diving ducks are shifting over time in response to climate change and other factors. An open question is whether the diver declines observed in the San Francisco Estuary represent true population declines or shifts in wintering distribution to other areas. A comparison with diver trends for the entire Pacific Flyway provides some insight into this question (see Richmond et al. 2014 for an analysis of trends from 1981-2012), however the MWS does not include the west coast of Canada nor Alaska due to weather restrictions. Thus, the MWS Flyway data may not be adequate for detecting northward shifts in wintering distributions. The difference in outlook for diving versus dabbling ducks likely reflects, in part, differences in food availability, reflecting prey or plant species, as well as availability of foraging locations (dabbling ducks can forage in shallower water; Goals Project 2000, Stralberg et al. 2009). However, a second important factor is the status of breeding populations outside of the San Francisco Estuary, since most wintering waterfowl breed elsewhere. Since Pacific Flyway populations are characterized by declining populations of scaup (Austin et al. 2000; Afton and Anderson 2001; Austin et al. 2006; USFWS 2009) and scoter (Agler et al. 1999) and increasing populations of dabbling ducks (USFWS 2009), such as mallard, it is not surprising that San Francisco Estuary MWS results show a similar picture.

The widely observed increase in dabbling ducks must be considered a positive result. Such a result is consistent with favorable environmental conditions in the San Francisco Estuary during the winter, but it may reflect changes in the wintering distribution of these species. In addition, favorable conditions on the breeding grounds for dabbling ducks, far removed from the San Francisco Estuary, may be contributing to the observed increase. If changes in wintering distribution are leading to reduced abundance of diving ducks, we must consider what may be the underlying causes for such shifts. Possibilities include drought, range contraction due to climate effects, increased development that leads to habitat loss or alteration, and long-term changes in prey resources. For both dabblers and divers, ongoing restoration is of potential concern, since conversion of managed ponds to tidal marsh will likely impact both groups of ducks, but especially diving ducks, because they depend on deeper water to forage, and thus have little opportunity to forage in tidal marsh habitat (Stralberg et al. 2009).

4. Figures. Figures have been inserted into the text above.

State of the Estuary Report 2015

Technical Appendix

WILDLIFE – Wintering Waterfowl

Prepared by
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Technical Appendix.

Background and Rationale

This is described in more detail in section 1, above. The indicator is a multi-species indicator, calculated separately for diving ducks and dabbling ducks and separately for each region of the San Francisco Bay Estuary. For each of the two groups of ducks we analyzed data from the six most abundant species. Two approaches were explored for combining results among species: weighting each species equally or weighting each species in proportion to its overall abundance in the dataset. Note that "combining species" refers to analyzing multiple species in a single model, where waterfowl counts were first natural log-transformed before analysis.

Benchmark

The benchmark chosen was the average index value for the first five years of the time series analyzed, 1989-1993. Note that the MWS began in 1955, but only since 1989 have data been collected in a standardized manner sufficient to allow analysis. The period we have chosen is similar to the period chosen by the San Francisco Bay Joint Venture for their baseline comparisons, 1988 to 1990.

Data Sources and Methods

Data collection for the San Francisco Bay Midwinter Waterfowl Surveys is described in Richmond et al. (2014). The results compiled and analyzed here were collected on surveys led by USFWS in San Francisco Bay, with many collaborators including San Francisco Bay Bird Observatory and USGS. In brief, surveys are conducted on a single day per survey area per year; often several areas are surveyed in a single day. Surveys are conducted from fixed-wing aircraft, as well as from the ground. Open bay and salt ponds are the targeted habitats.

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 (Richmond et al. 2014). Survey effort may be noted, but counts are not adjusted by effort. In theory, one could convert counts into densities by dividing by the area surveyed, but this has not been implemented.

The statistical approach used was to tally the number of individuals counted in each region, by species, separately for each habitat type (open bay or salt pond). Survey numbers are summarized by bay region: North Bay (San Pablo Bay and the northern portion of San Francisco Bay), Central San Francisco Bay, and South San Francisco Bay. For each species in each region, counts were then either summed over the two habitat types (results presented in the main section, above) or analyzed separately (results presented here in the Technical Appendix). The next step was to natural-log-transform all the counts. All analyses were conducted on ln-transformed counts and results presented in the Figures use this index.

The index values were analyzed in a linear model that included species main effects, separately for dabbling and diving ducks and by region. Each analysis included six species of dabbling ducks (American wigeon, gadwall, green-winged teal, mallard, northern pintail, and northern shoveler) and six species of diving ducks (bufflehead, canvasback, goldeneye, ruddy duck, scaup, and scoter). The result was an overall estimate of change over time for each waterfowl group (dabblers and divers). For estimating year to year differences in the index value for each analysis, we used the `margins` command in STATA 13.1 (Stata Corp.). For trend estimation, we estimated the "common slope" across species.

We used three methods to evaluate change over time: (1) long-term trends over time, for the period 1989 to 2014, (2) short-term trends over time, for the most recent 10 yr period (2005-2014), and (3) comparison of the period 2010-2014 with the 5-year reference period, 1989 to 1993. For all analyses, we either weighted each species equally or weighted each species by its mean abundance. Given the large variation in abundance among the twelve species, we chose to present results weighting by abundance.

Assumptions and uncertainties: The number of waterfowl counted during the survey period may be affected by weather conditions and the ability of observers to enumerate and identify individuals to species. In addition survey effort may vary. In some cases transects may be widely spaced and in other cases more tightly spaced. These influences are not incorporated into current analyses.

A second problem is that a comprehensive, systematic, probabilistic sampling frame is not used. Thus, there may be biases in deriving the annual index value because some areas are more likely to be included than others. Tidal marsh habitat, for example, is not sampled. In addition, shallow areas of open bay are less likely to be included. These deficiencies are well known for the MWS and summarized in Richmond et al. (2014). As a result plans are underway to modify the design and analysis of MWS data.

Additional Details Regarding Results

The wintering waterfowl indicator combines data across species for each guild. We maintain that assessing the condition of waterfowl is best accomplished through combining data among multiple species; that said, it is nevertheless also informative to examine species-specific results. Table A1 provides trend results for each species, by bay region. The analysis method is the same as that presented in Table 2 above, except that each species was analyzed separately.

For North Bay dabblers, all species demonstrated increasing trends, and for five of the six species, the trend was significant. In fact, all North Bay dabbler species increased at rates of at least 5% per year.

Central Bay dabblers showed non-significant increasing trends for three species, and a non-significant declining trend for northern pintail. Green-winged teal were not observed in the Central Bay.

South Bay dabblers showed significant increasing trends for two species (American wigeon and Green-winged teal) and marginally significant increasing trends for two species (northern pintail and northern shoveler). The remaining two species displayed non-significant trends, either positive (mallard) or negative (gadwall). Thus, five of six dabblers in the South Bay showed increasing trends, but only two were statistically significant.

For North Bay divers, five out of six species demonstrated declining trends (as did the group overall, Table 2, above). Only the ruddy duck showed an increase among divers, and it did so in all three regions though trends were not significant in the North Bay or Central Bay. Three of the six species showed significant declining trends in the North Bay (all at $P < 0.01$): goldeneye, scaup and scoter. Scoter was the only species (diver or dabbler) to show significant declining trends in all three regions, in all cases exceeding 7% decline per year.

For Central Bay divers the pattern was mixed. Bufflehead increased significantly, but scaup and scoter decreased significantly. The decline in canvasback was estimated to be 10% per year but was borderline significant ($P = 0.050$). The trends for two species were not significant ($P > 0.4$): goldeneye (increasing) and ruddy duck (decreasing).

Among South Bay divers, results were split. Four of the six species had increasing trends; three of these were significant (canvasback, goldeneye, and ruddy duck); bufflehead increased but not significantly. Scoter declined significantly while the decline in scaup was less than 1% per year and not significant. Thus, in the North Bay and Central Bay, either four or five (respectively) of the six species declined, but, in the South Bay, four of six species increased.

Appendix, Table A1: San Francisco Estuary Waterfowl (winter waterfowl surveys, USFWS)
Long-term (1989 to 2014) linear trends for individual species, for two groups of waterfowl
Shown are estimated annual percent changes per year in population index, for open bay and salt ponds combined.

	North Bay			Central Bay			South Bay		
	Coeff	Ann Pct	P-value	Coeff	Ann Pct	P-value	Coeff	Ann Pct	P-value
Dabblers									
American Wigeon	0.1049	11.1%	P < 0.001	0.0606	6.2%	P > 0.15	0.0985	10.4%	P < 0.001
Gadwall	0.1321	14.1%	P < 0.001	0.0128	1.3%	P > 0.7	-0.0124	-1.2%	P > 0.5
Green-winged Teal	0.0560	5.8%	P > 0.2	no observations			0.1094	11.6%	P = 0.019
Mallard	0.1195	12.7%	P = 0.002	0.0617	6.4%	P > 0.2	0.0363	3.7%	P > 0.15
Northern Pintail	0.0787	8.2%	P = 0.024	-0.0111	-1.1%	P > 0.8	0.0478	4.9%	P = 0.052
Northern Shoveler	0.0893	9.3%	P = 0.001	0.0596	6.1%	P > 0.3	0.0317	3.2%	P = 0.10
Divers									
Bufflehead	-0.0232	-2.3%	P > 0.2	0.0794	8.3%	P = 0.008	0.0269	2.7%	P > 0.1
Canvasback	-0.0069	-0.7%	P > 0.7	-0.1054	-10.0%	P = 0.050	0.0228	2.3%	P = 0.027
Goldeneye	-0.1277	-12.0%	P = 0.001	-0.0063	-0.6%	P > 0.9	0.1058	11.2%	P = 0.011
Ruddy Duck	0.0295	3.0%	P > 0.2	0.0262	2.7%	P > 0.4	0.0565	5.8%	P = 0.001
Scaup	-0.0500	-4.9%	P = 0.008	-0.0549	-5.3%	P = 0.003	-0.0076	-0.8%	P > 0.5
Scoter	-0.1042	-9.9%	P = 0.001	-0.0806	-7.7%	P = 0.002	-0.0864	-8.3%	P < 0.001

The wintering waterfowl indicator also combined data across two important habitat types: open bay and salt ponds. For each guild, Table A2 separates trends for open bay from those for salt ponds for the North Bay and South Bay regions (no salt ponds are found in the Central Bay). Table A2 demonstrates that in the North Bay trends differed for dabblers in the salt ponds (significant increase at 12.4% year), compared to the same species in the open bay (non-significant increase of 2.5% per year). However, trends were similar and not statistically distinguishable for divers in the North Bay: divers declined significantly in both habitat types.

Appendix Table A2: San Francisco Estuary Waterfowl (winter waterfowl surveys, USFWS)
Long-term (1989 to 2014) linear trends by habitat, for two groups of waterfowl
Shown are estimated annual percent changes per year in population index

	Dabbling Ducks			**Diving Ducks**		
	Coeff	Ann Pct	P-value	Coeff	Ann Pct	P-value
North Bay						
Open Bay	0.0245	2.5%	P > 0.2	-0.0428	-4.2%	P = 0.002
Salt Ponds	0.1167	12.4%	P < 0.001	-0.0292	-2.9%	P = 0.039
South SF Bay						
Open Bay	0.0610	6.3%	P = 0.006	-0.0208	-2.1%	P > 0.15
Salt Ponds	0.0555	5.7%	P < 0.001	0.0899	9.4%	P < 0.001

The complementary pattern was observed in the South Bay. Here the trend for dabblers was similar in open bay and in the salt ponds (significant increase of about 6% per year). However, for divers trends differed in the two habitats: diving ducks increased significantly in salt ponds but showed non-significant declining trends in open bay habitat in the South Bay.

To summarize, trends for dabbling and diving ducks in some cases differed in salt ponds as compared to open bay, and in other cases did not differ. Habitat-specific trends differed most strongly among North Bay dabblers and South Bay divers.

Acknowledgments

Thank you to the US Fish & Wildlife Service, San Francisco Bay Bird Observatory, San Francisco Bay Joint Venture, and US Geological Survey for support of the Midwinter Waterfowl Survey and analysis of these data.

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State of the Estuary Report 2015

Summary

WILDLIFE – Breeding Waterfowl

Prepared by

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SOTER - Delta & Suisun Marsh Indicator – Breeding Waterfowl

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Reviewers: Nadav Nur PhD (Point Blue), Anitra Pawley, PhD (Department of Water Resources), and Dave Zezulak PhD (California Department of Fish and Wildlife)

Summary Content for Indicator

1. Brief description of indicator and benchmark

Living Resources

Birds

Breeding Waterfowl –

Annual abundance in the Delta of the five most abundant dabbling duck species: Mallard, Gadwall, Green-winged teal, Northern Pintail, and Northern Shoveler.

Annual abundance in Suisun Marsh of the five most abundant dabbling duck species: Mallard, Gadwall, Green-winged teal, Northern Pintail, and Northern Shoveler.

Benchmark – Based on mean of first 10 years of data (1992-2001).

Current status – Based on mean of five most recent years data (2010-2014).

Scoring – proposed scoring standards: > 100% of benchmark = good; > 60% of benchmark = fair; < 60% of benchmark = poor.

2. Indicator status and trend measurements

Status – Fair

- Delta – Fair.
- Suisun – Fair.

Trends – DETERIORATING

- Delta – decreasing.
- Suisun Marsh – decreasing.

Details: Waterfowl (dabbling duck) breeding populations are estimated annually using California Waterfowl Breeding Population Survey data. For the indicator we use data collected 1992-2014.

The current status for the five most abundant species of breeding waterfowl in the Delta and Suisun Marsh is “Fair” and decreasing from baseline.

The trends are different for Mallards relative to the remaining four species, particularly in the Delta. Mallards are the dominant waterfowl species in the Delta (average 92% of total) and Suisun (average 59% of total). Abundance of mallards is decreasing at a rate of 2.3 percent per year in Delta and 2.5 percent per year in Suisun Marsh. For the remaining four most abundant waterfowl species, abundance is *increasing* at a rate of 7.7 percent per year in Delta and decreasing by 2.3 percent per year in Suisun Marsh. This can be compared to Statewide trends, which are also "Fair" and "decreasing": 1.2 percent decrease per year for mallard, and 1.0 percent decrease per year for the other four species.

3. Brief write-up of scientific interpretation

a. What is this indicator?

The indicator is the estimate of abundance of breeding waterfowl: the estimated summed abundance of the five most abundant species of dabbling ducks (Mallard, Gadwall, Cinnamon Teal, Northern Pintail, and Northern Shoveler) in two regions: Suisun Marsh and the Delta. The status and trends are compared to the statewide status and trends for context.

b. Why is it important?

The abundance of breeding dabbling ducks is important for several reasons.

1. Breeding ducks need undisturbed uplands (for nesting) in proximity to water, where they forage during the month they are incubating eggs, and where they bring their brood shortly after hatching. Healthy numbers of breeding ducks are an indicator of the ability of the area to support native wildlife in addition to agriculture and other non-urban land uses. In the Delta the dominant land use is agriculture. Suisun Marsh is dominated by duck clubs primarily managed for hunting waterfowl during the fall and winter. Some of these duck clubs also support breeding waterfowl and are managed for this purpose. Breeding waterfowl numbers are higher in areas with more available habitat, where land use practices are appropriate, and where predation is lower. Populations are also higher during higher rainfall years. As the landscape continues to change in the Delta and Suisun Marsh due to changing agricultural practices, climate change, and habitat restoration, the suitability of habitat available for breeding waterfowl is likely to change. Habitat restoration, enhancement, and protection, particularly of wetlands, is likely to improve habitat for waterfowl, while the changes due to climate change and changing agricultural practices are likely to degrade habitat for waterfowl.

2. Waterfowl hunting during fall and winter is important to a significant sector of the population, primarily in rural areas. Healthy breeding waterfowl populations in California can make a significant contribution to the waterfowl hunting industry. Approximately 20% of the waterfowl individuals found during winter in California's Central Valley also breed in locally in California; the remainder migrated to California after a breeding season elsewhere. Waterfowl hunting limits are based on waterfowl population surveys. The breeding mallard populations in California contribute to the Western Mallard Model, an adaptive harvest management model that sets bag limits for hunting seasons in California. Thus higher breeding populations in California can mean higher hunting bag limits set for California.

c. What is the benchmark? How was it selected?

The benchmark is the average of the first 10 years of the survey: 1992-2001. Status evaluation categories were set up using the formulae in Table 1 (used also for some other SOTER indicators).

Table 1. Status evaluation categories relative to reference condition

Ranking relative to reference condition	**Evaluation & interpretation**
> 100% of historical reference period average	"Good"
> 60% of historical reference period average	"Fair"
< 60% of historical reference period average	"Poor"

d. What is the status and trend for this indicator?

Delta – The breeding waterfowl population in the Delta is "Fair" and decreasing (Figure 2). The current population estimate (7,400 - based on the most five recent years) was 67% of the benchmark (11,000 – based on the first 10 years of the survey). This amounts to an overall decrease of 2% per year.

The trends are different for Mallard relative to the remaining four species. Mallard is the dominant waterfowl species in the Delta (average 92% of total). The Mallard population estimate for 2014 was lowest in the history of the survey (3,826). No Northern Pintail or Gadwall were present in 2014. The 2013 estimate for Northern Shoveler was highest in the history of the survey (1,170)[1]. Mallard is decreasing by 2.3% per year while the other four species (considered together) have been increasing by 7.7% per year.

Suisun Marsh – The breeding waterfowl population in Suisun Marsh is "Fair" and decreasing. The current population estimate (23,000 – based on the most five recent years) was 67% of the benchmark (34,000 – based on the first 10 years of the survey).

Trends are similar for all five species. Mallard is the dominant species in Suisun (average 59% of total). Mallard population estimate for 2014 was third lowest in the history of the survey. No Northern Pintails were present in 2014. Mallard is decreasing by 2.5% per year. The other four species (considered together) are decreasing 2.3% per year.

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[1] Note that wintering Northern Shovelers can migrate north to their breeding grounds later in some years, and that they are counted as part of the breeding population although they don't stay to breed. This could be biasing the data in some years.

e. What does it mean? Why do we care?

Mallard abundances are decreasing at a faster rate in the Delta and Suisun Marsh than they are statewide. The other species (considered together) are decreasing in Suisun Marsh at a faster rate than the overall statewide decrease. Interestingly, these other four species are increasing in the Delta. The implication is that conditions are deteriorating in the Delta and Suisun relative to elsewhere for Mallard. For the four other species, conditions in the Delta are improving, although they are deteriorating in Suisun Marsh. Statewide there are increases in Northern Shoveler and Gadwall, species that tend to nest later, and decreases in species that nest earlier: Mallard and Northern Pintail.

Localized conditions that could be affecting Delta and Suisun Marsh waterfowl populations, their nesting habitat and food availability include the following:

- Agricultural practices
- Refuge management
- Water availability – irrigation ditches, ponds, canals, sloughs
- Predation
- Hunting pressure
- Disease
- Environmental contaminants

There are problems with habitat loss and deterioration outside of the Delta and Suisun Marsh that affect waterfowl populations statewide. There is increased mortality associated with increases in avian botulism caused by reduced water availability in the Klamath Basin, where many of the Central Valley birds go to molt (i.e., shed their feathers and grow a new set, during which time they are flightless).

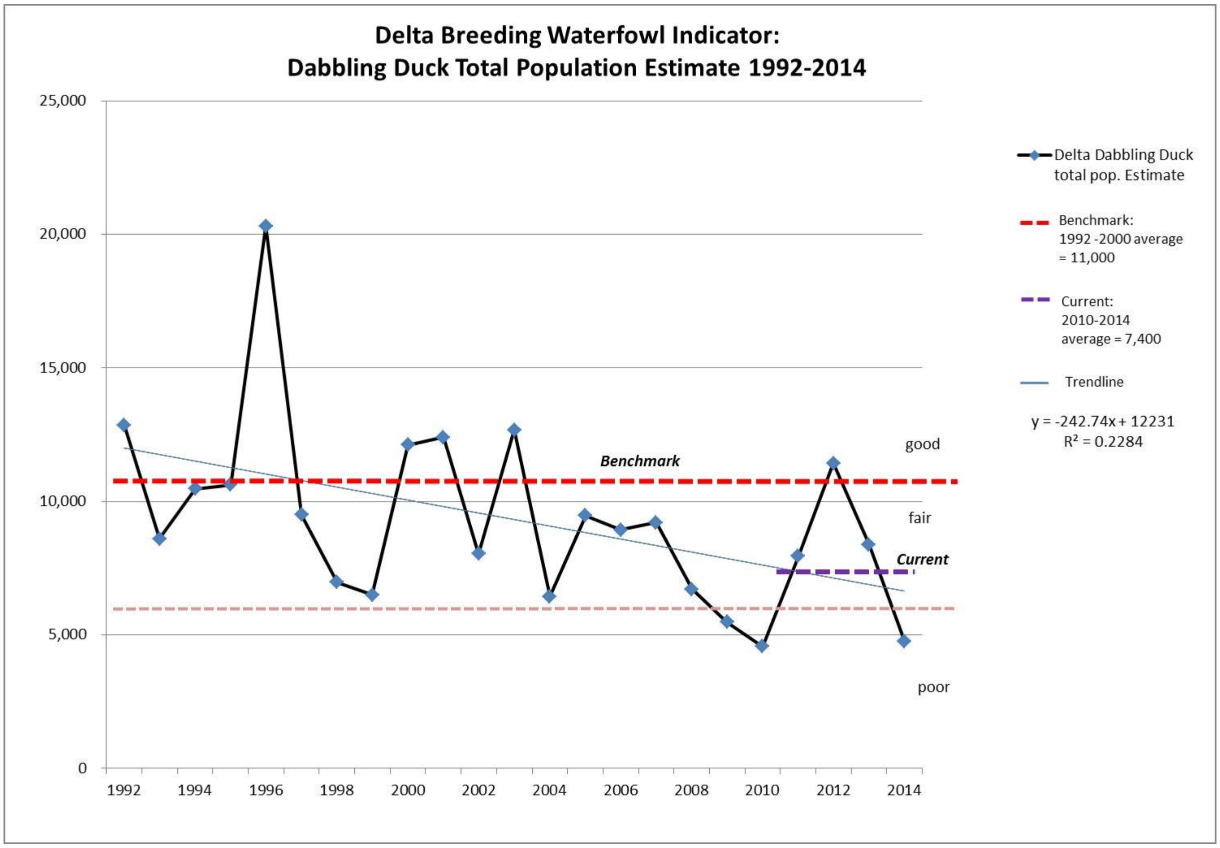
<u>4. Related figures for Report</u>

Graphs (see below; will not include Statewide in SOTER, just in Technical Appendix.)
Photo of Mallard with brood
Photo of typical nesting habitat in the Delta

Figure 1. Mallard pair

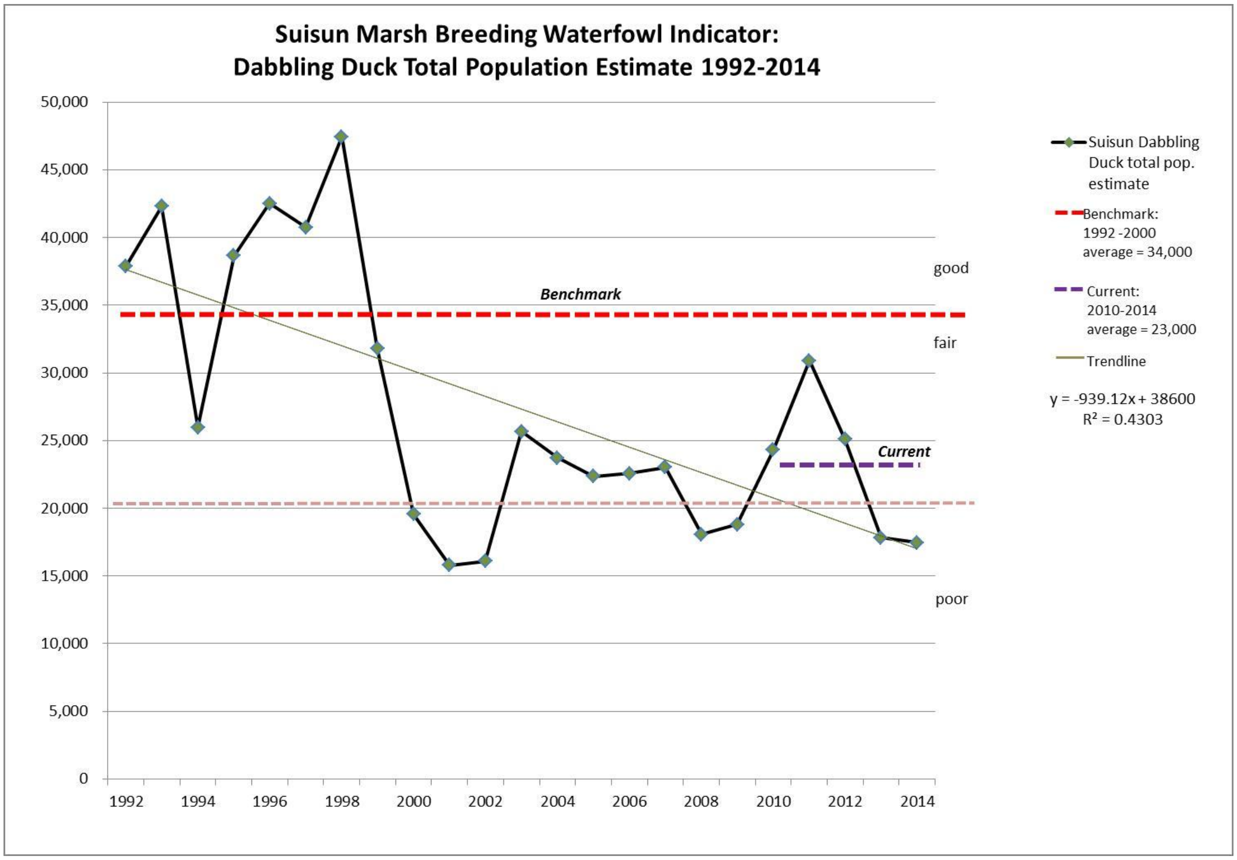
Photo credit: Tom Grey

Figure 2.



Delta Breeding Waterfowl Indicator:
Dabbling Duck Total Population Estimate 1992-2014
25,000
20,000
15,000
10,000
5,000
0
1992
1994
1996
1998
2000
2002
2004
2006
2008
2010
2012
2014
Benchmark
Current
good
fair
poor
Delta Dabbling Duck total pop. Estimate
Benchmark: 1992 -2000 average = 11,000
Current: 2010-2014 average = 7,400
Trendline
y = -242.74x + 12231
R² = 0.2284

Figure 3.



Suisun Marsh Breeding Waterfowl Indicator:
Dabbling Duck Total Population Estimate 1992-2014
50,000
45,000
40,000
35,000
30,000
25,000
20,000
15,000
10,000
5,000
0
1992
1994
1996
1998
2000
2002
2004
2006
2008
2010
2012
2014
Benchmark
Current
good
fair
poor
Suisun Dabbling Duck total pop. estimate
Benchmark: 1992 -2000 average = 34,000
Current: 2010-2014 average = 23,000
Trendline
y = -939.12x + 38600
R² = 0.4303

State of the Estuary Report 2015

Technical Appendix

WILDLIFE – Breeding Waterfowl

Prepared by

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State of the Estuary Report – 2015

Technical Appendix
Delta and Suisun Marsh Breeding Waterfowl Abundance Indicator

June 5, 2015

Authors: Hildie Spautz and Dan Skalos, California Department of Fish and Wildlife

Reviewed by: Nadav Nur PhD (Point Blue), Anitra Pawley, PhD (Department of Water Resources), and Dave Zezulak PhD (California Department of Fish and Wildlife)

A. Background and Rationale

The Breeding Waterfowl Indicator is based on annual estimates of abundance of breeding waterfowl generated by the California Department of Fish and Wildlife Waterfowl Program's Breeding Waterfowl Population Survey (http://www.dfg.ca.gov/wildlife/waterfowl/popassessment.html). The Indicator is a community-level indicator for dabbling ducks, and does not include assessment of individual species. Annual indicator data points are the summed estimated abundance of the five most abundant species of dabbling ducks (Mallard, Gadwall, Cinnamon Teal, Northern Pintail, and Northern Shoveler) presented separately for each of two regions (referred to as "strata" by the Breeding Waterfowl program; Figure 1) in the San Francisco Estuary: Suisun Marsh and the Delta. The status and trends of waterfowl populations in these two regions are compared to statewide status and trends for context. Mallard is the most abundant of these dabbling duck species, while the abundances of the remaining four species relative to mallard tend to be smaller and more variable spatially and temporally. Other species included in the Breeding Waterfowl Population Survey, which were not included in the Indicator, include Canada goose, coots, and additional dabbling duck species that breed in very small numbers in California.

The Breeding Waterfowl Indicator is new for the State of the Estuary Report 2015. The structure of the indicator is similar to the Winter Waterfowl Abundance Indicator reported in the State of the San Francisco Bay 2011 and in the current 2015 Report. With input from others, the authors evaluated extending the Winter Waterfowl indicator to the Delta, but this option was rejected because data collection efforts during the winter are not yet sufficiently standardized in the Delta. This decision should be revisited after the survey protocols are standardized, which is likely to be within the next few years. The Winter Waterfowl indicator includes dabbling duck and diving duck species found in the San Francisco Bay during the wintering period, when large numbers of individuals that breed in other areas migrate to California, and pass through or stay the entire winter before flying back to their breeding areas. It also includes year-round residents: individuals who breed and remain in the area year round. The Breeding Waterfowl Indicator is an indicator of the population size of breeding dabbling ducks only, and explicitly excludes individuals that are not breeding and are likely to be on their way to breeding grounds elsewhere, based on their observed flocking behavior. Diving ducks were not included in the Breeding Waterfowl indicator because this group breeds in small numbers in Northeastern California, but generally not in the Estuary. Geographic coverage also differs between these indicators: the Winter Waterfowl indicator includes data from surveys conducted in San Francisco Bay and San Pablo Bay, while the Breeding Waterfowl indicator includes Suisun Marsh and the Delta. San Francisco Bay and San Pablo Bay are not surveyed for breeding waterfowl. The Breeding Waterfowl survey Napa River stratum was not included as part of the present indicators because much of the Napa River valley is too far removed from the San Francisco Estuary to be considered a valuable indicator of the health of the Estuary (Figure 1).

The abundance of breeding dabbling ducks is an important indicator of the health of the Delta and Suisun Marsh for several reasons:

1. During the breeding season, ducks need undisturbed uplands in proximity to water, where they forage during the month the females are incubating eggs, and where they bring their brood shortly after hatching. Healthy numbers of breeding ducks are an indicator of the ability of the area to support native wildlife in addition to agriculture and other non-urban land uses. In the Delta the dominant land use is agriculture. Suisun Marsh is dominated by duck clubs primarily managed for hunting waterfowl during the fall and winter. Some of these duck clubs also support breeding waterfowl and are managed for this purpose. Breeding waterfowl numbers are higher in areas with more available habitat, where land use practices are appropriate, and where predation is lower. Waterfowl populations are also higher and more productive during higher rainfall years, particularly in Suisun Marsh; associations are less evident in the Delta (CDFW unpublished data). As the landscape continues to change in the Delta and Suisun Marsh due to changing agricultural practices, climate change, and habitat restoration, the suitability of habitat available for breeding waterfowl is likely to change. Habitat restoration, enhancement, and protection, particularly of wetlands, is likely to improve habitat for waterfowl, while the changes due to climate change and changing agricultural practices are likely to degrade habitat for waterfowl (Browne & Dell 2007; Hagy et al 2014).

2. Waterfowl hunting during fall and winter is important to a significant sector of the human population, primarily in rural areas. Waterfowl hunting limits are based on waterfowl population surveys, including the Breeding Waterfowl Survey in California. Healthy breeding waterfowl populations in California can make a significant contribution to the North American waterfowl population and to the waterfowl hunting industry. Approximately 20% of the waterfowl harvested during winter in California's Central Valley each year are locally bred (CVJV 2006); the remainder have migrated to California after a breeding season elsewhere, primarily from northern prairie states, Canada, and Alaska. Higher Mallard breeding populations influence the calculation of bag limits and hunting season length determinations through their contribution to the Western Mallard Model (USFWS 2014), resulting in higher hunting bag limits set for California.

B. Benchmark

The Benchmark for the Breeding Waterfowl indicator was calculated as the mean of the first 10 years of data (1992-2001). Status evaluation categories were set up using the formulae in Table 1, which was also used for some other SOTER indicators. A current population size over the benchmark would be considered "Good". A current population size between 60-100% of the benchmark would be considered “Fair”. Finally, a current population size less than 60% of the benchmark would be considered “Poor”.

We used the default benchmark calculation method recommended for new indicators where there is no alternative historical benchmark, or alternative planning target. Ideally, the Benchmark would be a target healthy population size based on estimates of statewide and regional population long-term viability, or based upon historical population sizes prior to extensive habitat loss, e.g., 100-200 years ago. However, historical estimates are not available, nor have population targets been set based on modeled population viability. Habitat loss most certainly contributed to significant population declines prior to the initiation of standardized surveys. Thus, we chose to classify populations above our benchmark “Good”, while any decrease would be considered “Fair” to “Poor”.

As part of their Implementation Plan, the Central Valley Joint Venture set a statewide target population size for breeding mallards: “maintain, enhance, and restore sufficient habitats to increase mallard populations by 25% over the range of variation observed from 1992-2002” (CVJV 2006). The baseline was between 186,000 to 389,000 statewide. Thus, the mallard target would be 232,000 and 486,000 individuals statewide (CVJV 2006). However, population targets were not set for other species or for specific regions. Instead, they identified general habitat needs in each region, with the plan to revisit and develop more specific habitat targets. The Implementation Plan revision effort is currently in process and it is anticipated that new population targets will be set. When the CVJV develops habitat and/or population targets for breeding waterfowl in Suisun Marsh and the Delta, these targets can be used as alternative benchmarks for the indicator in future iterations of the State of the Estuary Report.

C. Data Sources

Data used for the Breeding Waterfowl Indicator were collected by the California Department of Fish and Wildlife’s Waterfowl Program, which uses annual waterfowl population data to make recommendations for hunting regulations (http://www.dfg.ca.gov/wildlife/waterfowl/popassessment.html).

Breeding Waterfowl Population surveys are conducted nationwide in the spring by the U.S. Fish and Wildlife Service (USFWS), the Canadian Wildlife Service and others, including state wildlife agencies, on the primary breeding areas of waterfowl. These estimates play an important role in establishing annual harvest regulations. Portions of Alaska, Canada, and north-central United States are sampled.

The California Breeding Waterfowl Population survey was modeled after the national effort, using the same methods. Prior to switching to the federal methods in early 1990’s, a less standardized method was used. Non-standardized data collected prior to 1992 were not used to calculate the current Indicator. Surveys are conducted by California Department of Fish and Wildlife and California Waterfowl Association (CWA) biologists. Surveyed areas include wetland and agricultural areas in northeastern California, throughout the Central Valley, the Suisun Marsh, and some coastal valleys (Figure 1a).

D. Methods

Breeding Population Survey methods

Field methods are based on those developed by the USFWS for nationwide breeding population surveys, as described above.

The survey consists of both aerial and ground components. The aerial component uses an airplane flying transects at an altitude of 150 feet at about 105 miles per hour, with two observers, one on each side of the aircraft. Every duck seen within an eighth of a mile of the airplane is counted. The species, sex, and social status (paired or unpaired) is determined. Observers cannot see all ducks on the transect due to vegetation and other visibility issues, so an on-the-ground correction factor is needed. Another set of observers on foot samples a portion of the transects flown by the aerial crew. The difference between what the aerial and ground crews see is used to correct the aerial estimate, minimizing visibility bias (http://www.dfg.ca.gov/wildlife/waterfowl/popassessment.html).

The regions of California where waterfowl breed are broken up into "Strata", within which pre-selected linear transects are flown. (Figure 1a, 1b). There are nine strata statewide. The Northeastern California stratum includes 32 non-contiguous patches, while all the remaining strata are contiguous (Fig 1a). Flight lines include 40 separate segments totaling 1,377 miles. The average length of a segment is 13 miles. In the Delta, there are four separate transects totaling 72 miles, and in Suisun Marsh, there are 2 transects totaling 40 miles (Figure 1b). The strata boundaries do not correspond exactly with other legal boundaries for Suisun Marsh and the Legal Delta. The Legal Delta includes the Delta stratum and portions of other adjacent stratum.

Breeding Population Estimates

The aerial survey transect data and correction factor data are used to estimate the population abundance for each stratum. A program was developed by USFWS to do this based on the survey input, the visual correction factor, transect length, and stratum area. The output includes for each species within each stratum: a population estimate, standard error, and 95% confidence interval.

Breeding Waterfowl Indicator calculation methods

For calculation of the Indicator, we used Breeding Waterfowl survey data collected between 1992 and 2014. Earlier data are available, but survey methods prior to 1992 were not standardized using the USFWS standards and so results may not be comparable.

Each year's indicator datapoint is a sum of the estimated population size of the five most abundant species of dabbling ducks (Mallard, Gadwall, Cinnamon Teal, Northern Pintail, and Northern Shoveler). Indicators were calculated separately for Suisun Marsh, the Delta, and statewide. The statewide indicator is not reported in the 2015 State of the Estuary Report but is included here so that trends in Suisun Marsh and the Delta can be placed in context. The benchmark was calculated as described above (the mean of the first 10 years of data [1992-2001]). The current population size was calculated as the mean of the most recent five years of data: 2011-2015. The mean for each of these periods was used rather than a single year because of the high inter-annual variability in the data.

Annual datapoints were natural log transformed for analysis and were analyzed for changes separately for each region and statewide. Trends in each of the indicators were assessed for statistical significance using a General Linear Model (GLM) in JMP v. 10.0 (2007).

For each region and statewide, three sets of models were run:

- To assess the significance of a linear trend, a linear model was evaluated using population estimates for each year;
    - with year as a continuous variable; and
    - with year as a categorical variable.
- To assess the difference between the Benchmark and Current population size.

Each model included the following main effects: Time (year or period), Species, and Species * Time (an interaction variable). The approach to statistical analysis of the Breeding Waterfowl Indicator data was similar to that used for the Winter Waterfowl indicator. Individual species were excluded if the majority of years had zero counts for that species. Each model included data for four or five species and included a species "main effect." Thus, we allowed for differences in the overall abundance of the four to five species while estimating the trend over time common to the species for the specific region. We report the results of the model with the greatest statistical significance based on its Akaike Information Criteria.

Additional tests for statistical significance of linear trends were conducted separately for each species and region; these are also reported.

E. Peer Review

The Breeding Waterfowl indicator was chosen as the most appropriate bird indicator available for the Delta and Suisun Marsh by a group of experts meeting under the auspices of the Delta Bird Monitoring Network. The evaluation used a suite of criteria including those used for indicator selection for the State of the Bay 2011 Report (SFEIT 2011; Table 2). Criteria include whether the indicator would be meaningful to the public and to decision-makers, whether data were available, and whether those data were of sufficient quality and duration.

The indicator was reviewed by: David S. Zezulak, Ph.D. (CDFW), Anitra Pawley, Ph.D. (Department of Water Resources) and Nadav Nur, Ph.D. (Point Blue Conservation Science).

F. Results

Status – Fair

- Delta – Fair.
- Suisun – Fair.
- Statewide – Fair.

Trends – DETERIORATING

- Delta - decreasing
- Suisun Marsh - decreasing
- Statewide - decreasing

Delta – The current breeding waterfowl population in the Delta is "Fair" and decreasing relative to the benchmark (Figure 2). The current population estimate (7,414, based on the five most recent years) was 67% of the benchmark (11,031, based on the first 10 years of the survey). This amounts to an average decrease of 2% per year. The decrease from the benchmark level to the current five-year average is statistically significant (GLM: $X^2 = 55.46$, d.f. = 4, $p < 0.0001$; Species main effect: $X^2 = 52.46$; df = 3, $p < 0.001$; Period main effect: $X^2 = 6.94$, df = 1, $p = 0.008$). Northern Pintail were counted in 1997 but were absent all other years, so this species could not be included in analyses of trends.

Population trends in the Delta vary by species. Mallard is the dominant waterfowl species in the Delta (averaging 92% of total; Figure 5). In 2014, the Mallard population estimate was lowest in the history of the survey (3,826), and no Northern Pintail or Gadwall were present at all. However, there were recent increases in Northern Shoveler, Gadwall, and Cinnamon Teal. The 2013 estimate for Northern Shoveler was highest in the history of the survey (1,170). The decrease of Mallard is statistically significant (linear regression: $R^2 = 0.30$; $p < 0.006$), an average decrease of 2.3% per year. The increasing trend for Northern Shoveler is statistically significant but the increasing trends for Cinnamon Teal and Gadwall are not (Northern Shoveler- linear regression: $R^2 = 0.28$; $p = 0.01$).

Suisun Marsh – The breeding waterfowl population in Suisun Marsh is "Fair" and decreasing (Figure 3). The current population estimate (23,122, based on the most five recent years) was 67% of the benchmark (34,265, based on the first 10 years of the survey). The decrease from benchmark to current is statistically significant (Generalized linear model, species, period, and species*period main effects: $X^2 = 110.45$, d.f. = 9, $p < 0.001$; Period: $X^2 = 10.74$, d.f. = 1, $p = 0.001$; Species: $X^2 = 105.71$, d.f. = 4, $p < 0.001$; Species*Period: $X^2 = 12.14$, d.f. = 4, $p = 0.0003$).

Trends are similar for all five species: all are decreasing. Mallard is the dominant species in Suisun (average 59% of total; Figure 6). The Mallard population estimate for 2014 was third lowest in the history of the survey. No Northern Pintails were present in 2014. Mallard is decreasing by 2.5% per year. The other four species (considered together) are decreasing 2.3% per year. The decrease is statistically significant for Mallard and Northern Pintail, but not for the other species (Mallard Linear regression: $R^2 = 0.42$; $p = 0.0007$; Northern Pintail linear regression: $R^2 = 0.17$; $p = 0.05$).

Statewide –
The breeding waterfowl population statewide is "Fair" and decreasing (Figure 4). The current population estimate (471,647, based on the most five recent years) was 81% of the benchmark (580,308, based on the first 10 years of the survey). This amounts to an overall decrease of 1% per year. The decrease is statistically significant (Generalized linear model with Year as a continuous variable and Year * Species interactions: $X^2 = 245.82$, d.f. = 9, $p < 0.0001$; Species main effect: $X^2 = 243.60$; df = 4, $p < 0.001$; Year main effect: $X^2 = 4.15$, df = 1, $p = 0.042$; Species*Year: $X^2 = 13.53$; df = 4, $p = 0.009$).

All five species are seeing a downward trend statewide. Mallard is the dominant species statewide (average 68% of total; Figure 7). The Mallard population estimate for 2014 was second lowest in the history of the survey, Cinnamon Teal population estimate was third lowest, and Northern Pintail was fourth lowest. The 2013 estimate for Gadwall was second lowest in the history of the survey. Mallard is decreasing by 1.2% per year. The other four species together are decreasing 1% per year, although the rate of decrease varies by species. The decreasing trend for Northern Pintail is statistically significant but it is not significant for the other species (Northern Pintail linear regression: $R^2 = 0.27$; $p = 0.014$).

G. Discussion

Breeding waterfowl populations are decreasing, and are considered “Fair” in the Delta, Suisun Marsh and statewide. Mallard abundances are decreasing at a faster rate in the Delta and Suisun Marsh than they are statewide. The other species (considered together) are decreasing in Suisun Marsh at a faster rate than the overall statewide decrease. However, there are some increases in the Delta, most notably of Northern Shoveler. The implication is that conditions are deteriorating in the Delta and Suisun relative to elsewhere for Mallard and most other species. For the Northern Shoveler, conditions in the Delta may be improving, although they are deteriorating in Suisun Marsh.

Local conditions that could be affecting waterfowl populations, their nesting habitat and food availability include the following (Hagy et al 2014):

- Conversion of habitat to alternate uses
- Agricultural practices
- Refuge management
- Water availability – irrigation ditches, ponds, canals, sloughs
- Mosquito abatement
- Predation, particularly by mesocarnivores, e.g. skunks and raccoons which do particularly well in human-modified areas.
- Hunting pressure
- Disease
- Environmental contaminants including pesticides

Nationwide, there have been increases in species that tend to nest later (i.e., Northern Shoveler and Gadwall). Mallard and Northern Pintail tend to nest earlier. Northern Pintail in particular has seen decreases nationwide, which may be due to its early nesting behavior, but it’s not clear if this is the case with Mallard. Species that nest in crops, e.g. Mallard, are vulnerable to changes in cropping patterns, and timing of harvest.

Female ducks, particularly Mallards, which have been the most intensively studied, are philopatric, i.e., they will return to their natal area to nest. This can present problems when changing land use makes an area less productive but the females retain their drive to remain in the area to try to nest. Reduced nesting success would contribute to population decreases.

Another factor that may be contributing significantly to population decreases in California is the increase in avian botulism outbreaks on molting grounds. After nesting, ducks may fly north to major inland wetlands, e.g. Klamath Basin, where they remain for the time they molt and regrow flight feathers, when they are flightless. Reductions in water availability in wildlife refuges has resulted in smaller areas of appropriate habitat, and increased waterfowl densities, which has increased the frequency of botulism outbreaks (Yarris 1994).

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Figure 1a. Statewide Waterfowl Breeding Population Survey strata and transects

Fig 1b. Delta and Suisun Marsh Waterfowl Breeding Population Survey strata and transects

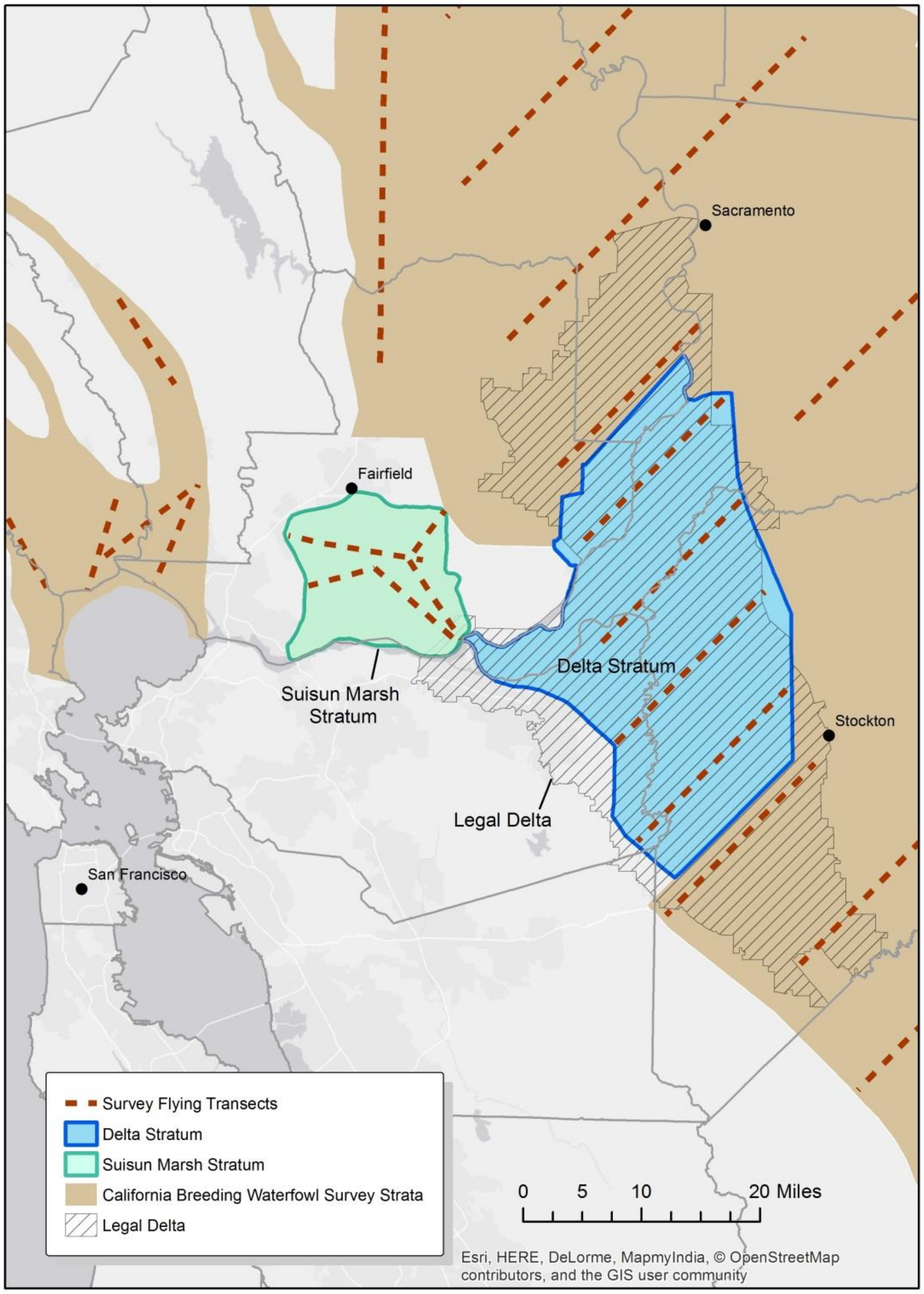


Figure 2. Delta Breeding Waterfowl Indicator – Dabbling Duck Total Population Estimates 1992-2014

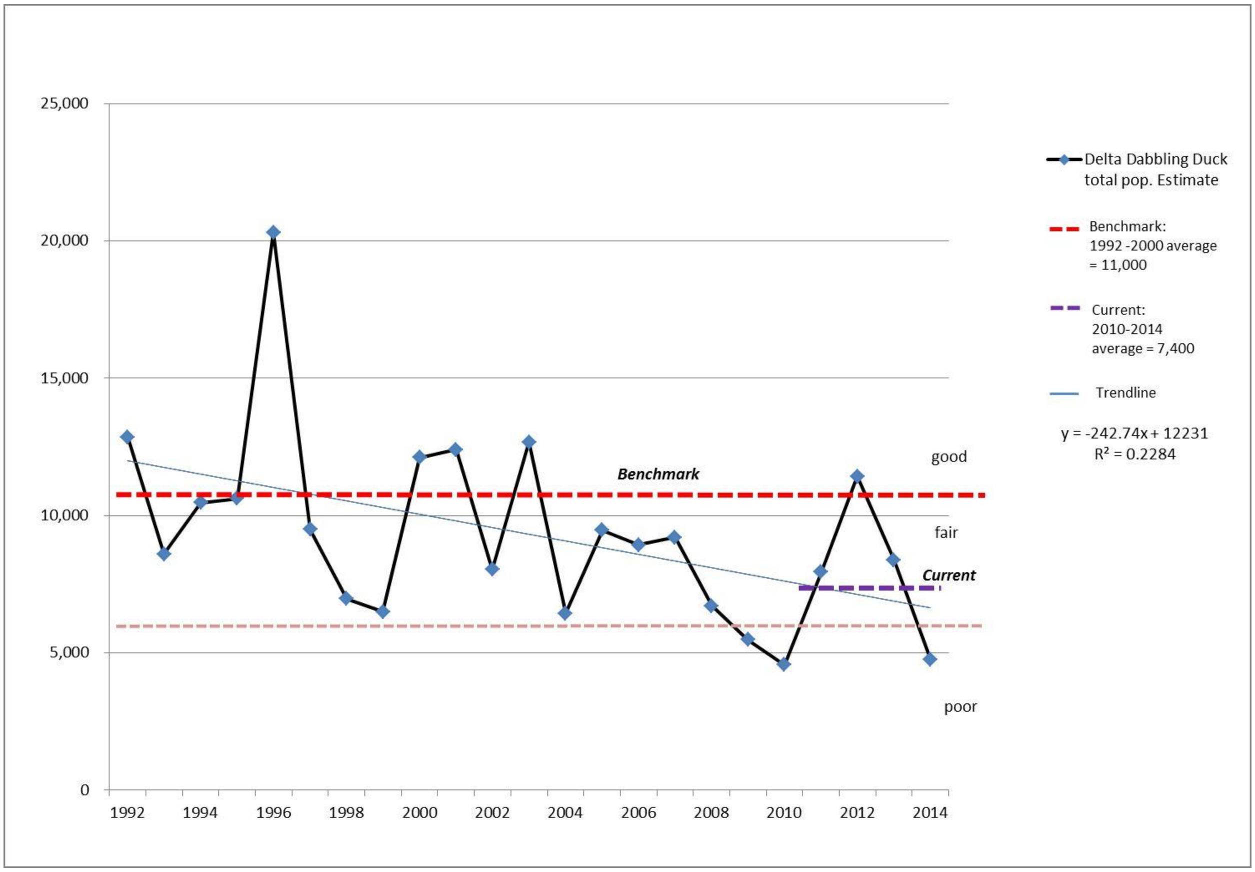


Figure 3. Suisun Marsh Breeding Waterfowl Indicator – Dabbling Duck Total Population Estimates 1992-2014.

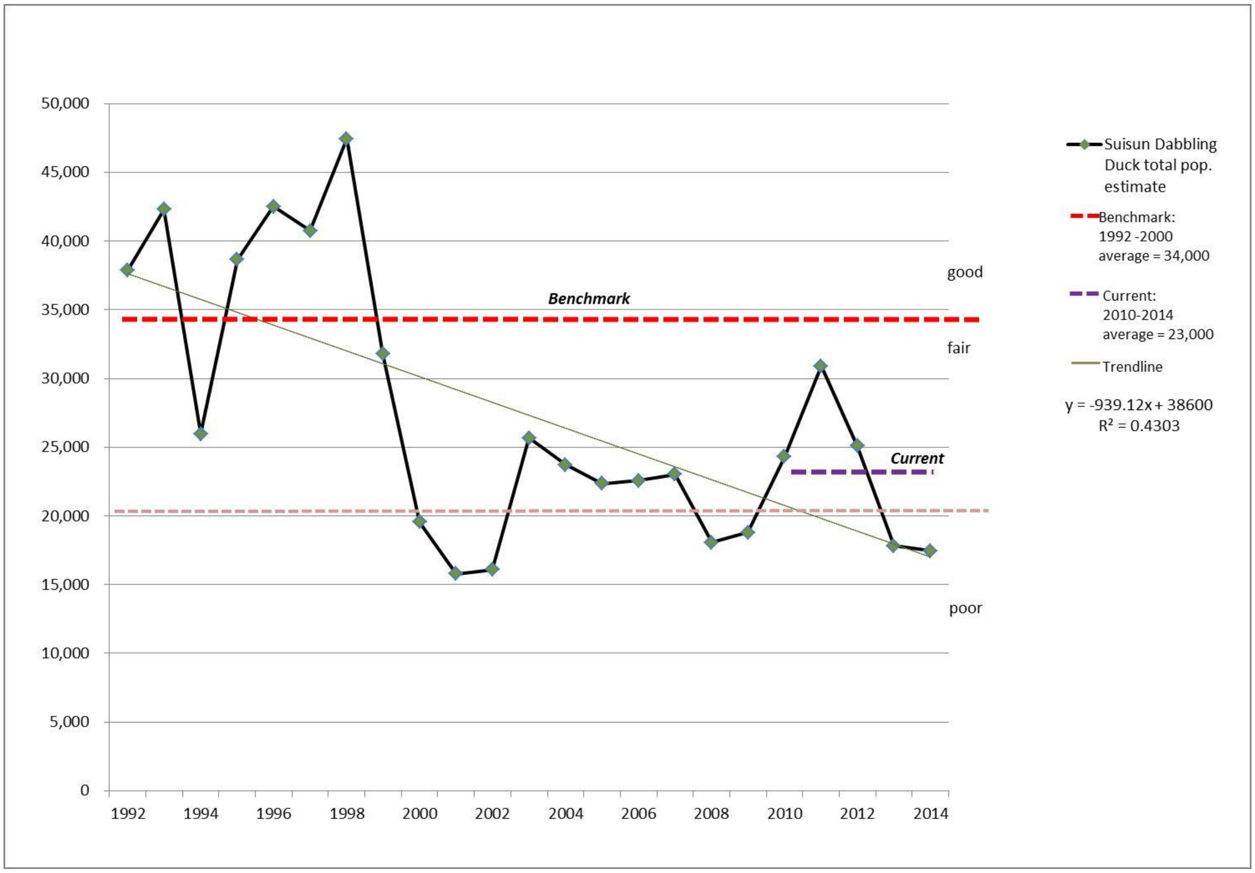


Figure 4. Statewide Breeding Waterfowl Indicator – Dabbling Duck Total Population Estimates 1992-2014.

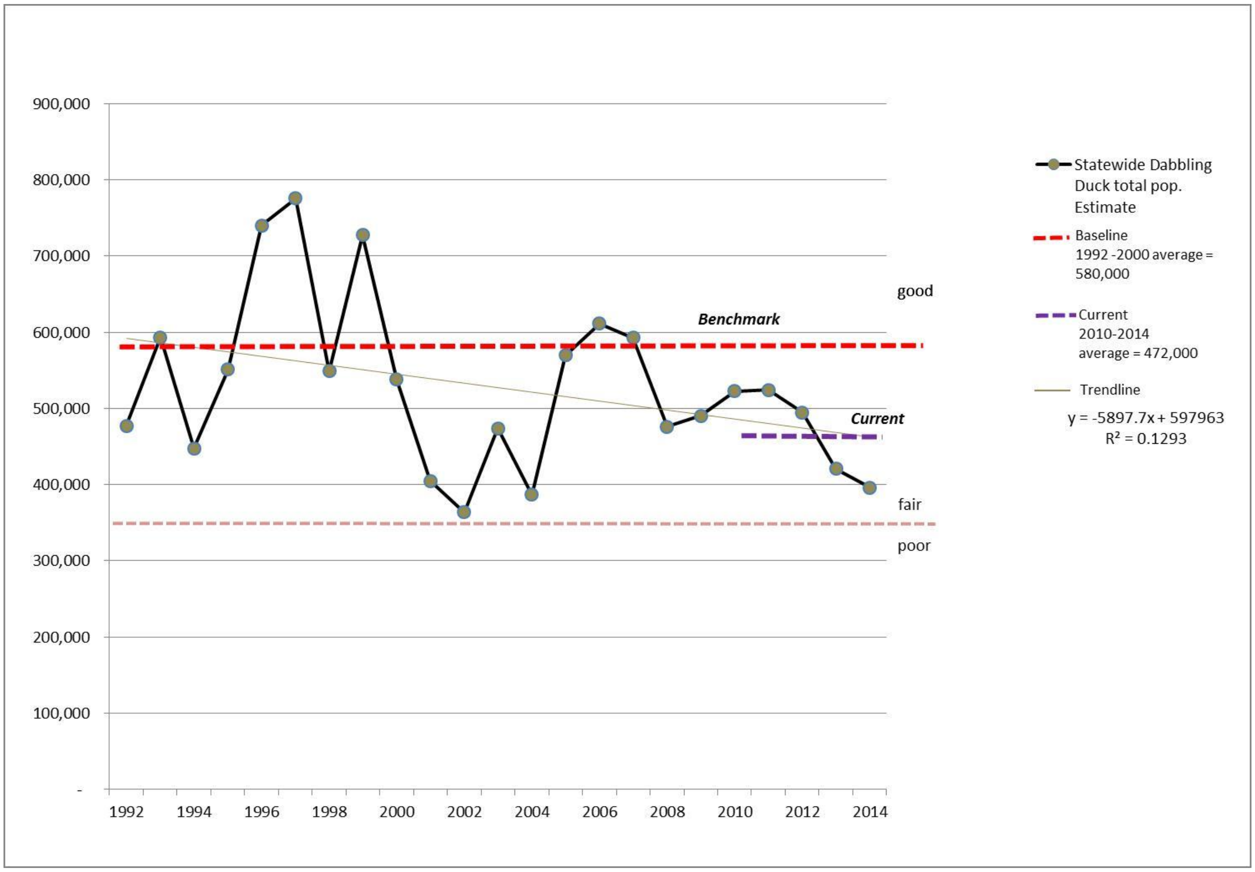


Figure 5. Delta Breeding Waterfowl – Dabbling Duck Community Composition: Population Estimates 1992-2014.

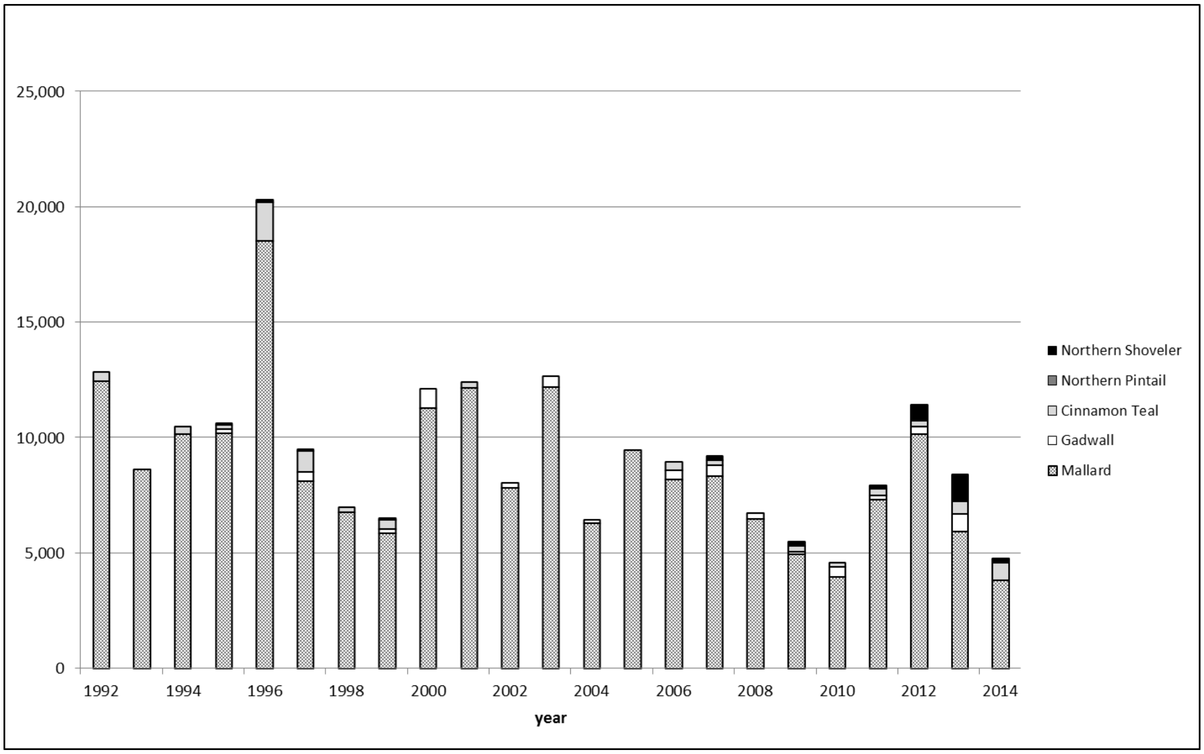


Figure 6. Suisun Marsh Breeding Waterfowl – Dabbling Duck Community Composition: Population Estimates 1992-2014.

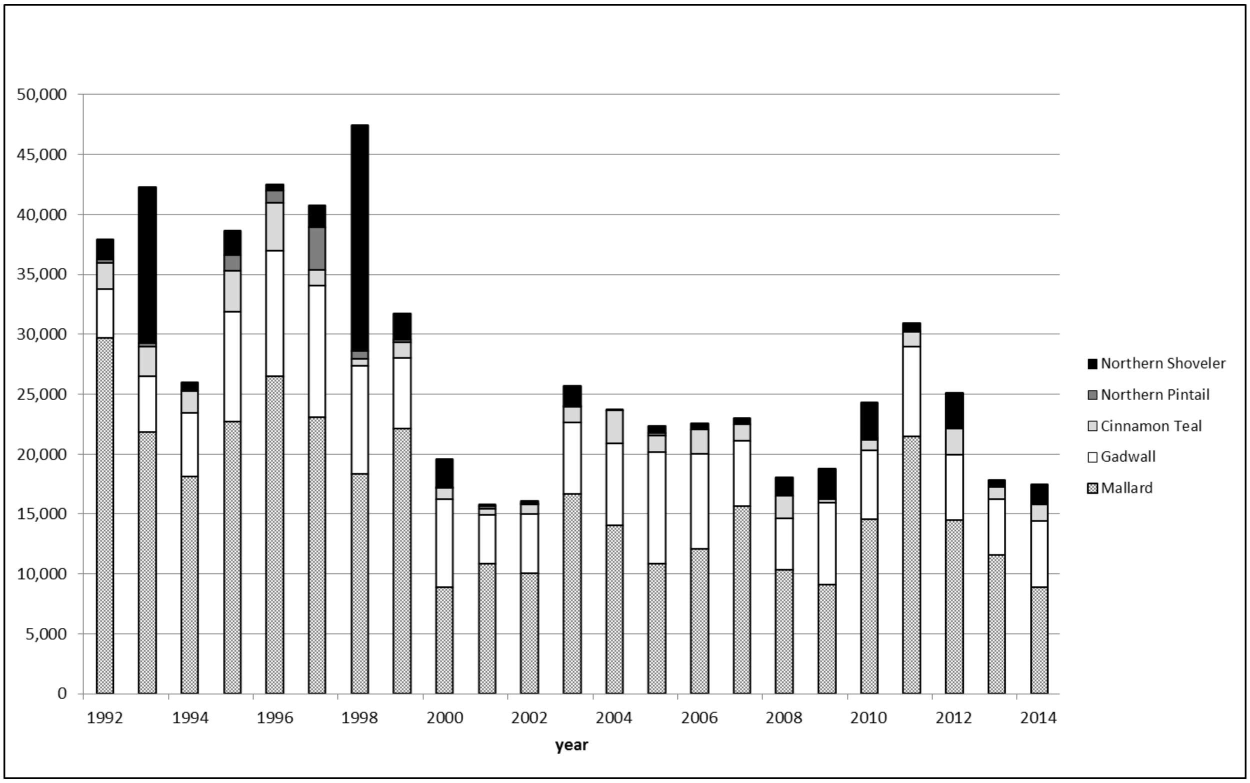


Figure 7. Statewide Breeding Waterfowl – Dabbling Duck Community Composition: Population Estimates 1992-2014.

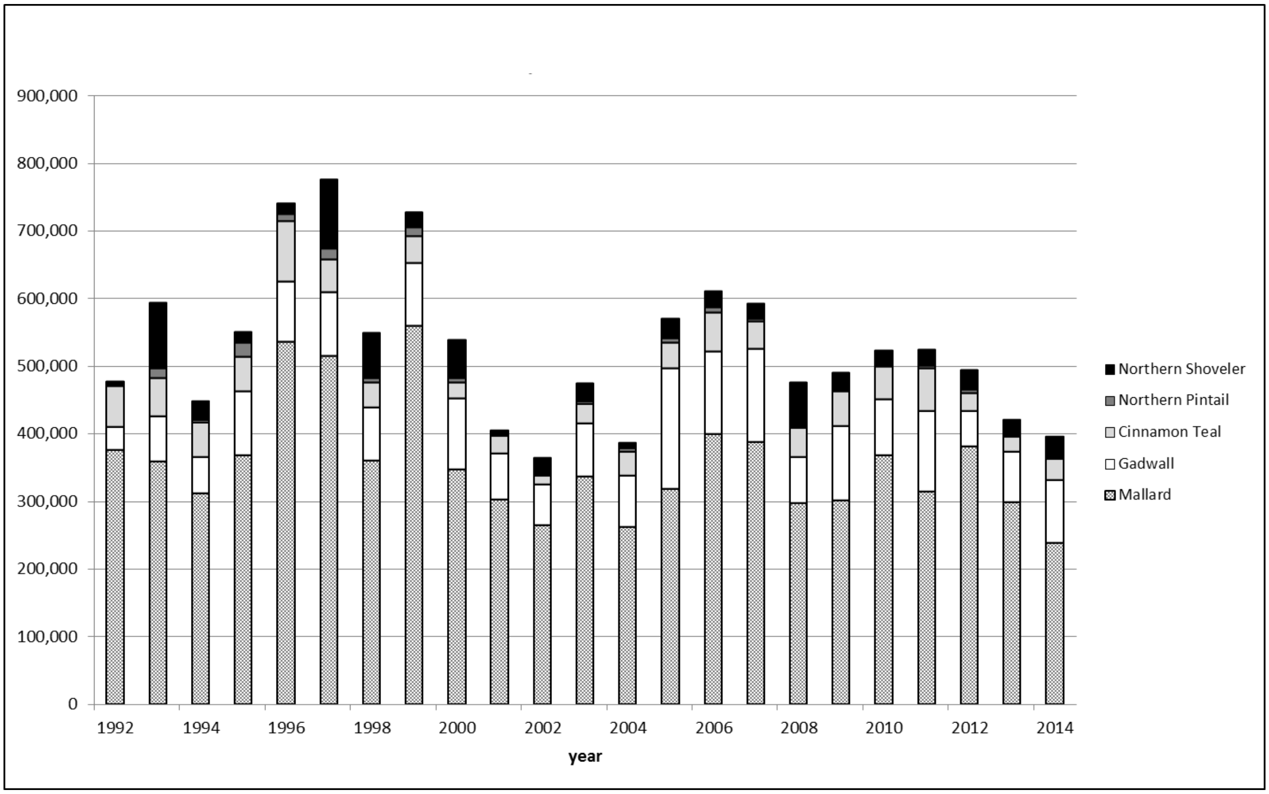


Table 1. Indicator health – status evaluation criteria.

Ranking relative to reference condition	**Evaluation & interpretation**
> 100% of historical reference period average	“Good”
> 60% of historical reference period average	“Fair”
< 60% of historical reference period average	“Poor”

Table 2. New Indicator Selection Criteria – Breeding Waterfowl: Delta and Suisun Marsh.

Delta & Estuary-wide Indicators - Selection Criteria for watershed assessment indicators for the San Francisco Estuary Report 2014.					
Modified from SFEIT (Collins, J. et al). 2011. Assessment Framework as a Tool for Integrating and Communicating Watershed Health Indicators for the San Francisco Estuary. Final Report. Grey cells were added for new Delta indicators for 2014					
Indicator Name:		Breeding Waterfowl - Delta			Draft version January 20, 2015
SotER Attribute Category:		Living Resources	**Metric(s) [3]:**		Breeding waterfowl population size in the Delta: Mallard, Gadwall, Cinnamon Teal, Northern Pintail and Northern Shoveler. Can also include Suisun and Napa River for an Estuary-wide indicator.
WAF Category [1]:		Biotic condition - Species & Populations	**Dataset name(s), Program, Agency [3]:**		CDFW Waterfowl Program
CCMP Goal [2]:		Wildlife Goals: - Optimally manage and monitor the wildlife resources of the Estuary.			
	Result (yes/no)	**Detailed comments, rationale, and Action Items**		**Result (yes/no)**	**Detailed comments, rationale, and Action Items**
Conceptual Relevance I			**Conceptual Relevance II**		
Fits with SotB 2011 indicator for Bay [3]	yes	Indicator of Delta ecosystem health: uses variety of upland habitats to breed, primarily agricultural and grassland habitats. Needs nearby water for foraging and broods.	Fits with Delta Plan or other Delta Stewardship Council documents [3]	yes	Yes. Species may respond to habitat restoration and protection (Delta Plan ER-R2).

Fits with WAF category (ecological function) [1]	yes	Biotic condition - Species & Populations - Population size, Habitat suitability	Fits with FRP/BO and EcoRestore Framework [3]	yes	Not an EcoRestore covered species, but the natural habitats that support breeding waterfowl are covered. No terrestrial species (i.e. birds) addressed in FRP restoration & monitoring framework.
Fits with CCMP (management objectives)	yes		Fits with goals of other plan(s) [3, 4]	Yes	Central Valley Joint Venture Strategic Plan (2006) includes conservation target habitat for Breeding Waterfowl. Updated Strategic Plan will included targets for breeding waterfowl.
Data Availability and Adequacy			**Interpretation (what does it mean?)**		
Data available	yes	Standardized survey data available 1992-2014. Same protocols are used throughout USA.	**Goals, thresholds, reference, and/or triggers available**	no	Propose using 1992-2001 average as a benchmark to correspond with recommended methods for SOTER 2015. CVJV is developing population targets, but none available now.
Data suitable quality	yes	see above.	**Meaningful to public**	yes	Waterfowl hunting is a popular sport in the Delta and Suisun, so waterfowl are important to that constituency. Waterfowl are also a highly visible and charismatic wildlife group.
Data currently or soon to be reported / linked on Estuaries Portal [3]	yes	Static graphs included on Estuaries Portal from 2013. No plan to request web services from CDFW so that data are live.	Meaningful to decisionmakers [3]	yes	see above
Development or application of indicator published / peer reviewed [3]	no	Wintering, not breeding, waterfowl indicator was reported in the State of the Bay 2011. Considered expanding wintering waterfowl indicator to Delta, but survey methods in Delta are not yet sufficiently standardized to the degree they are in the			

		SF Bay.			
Responsiveness (to environmental change)			**Transferability**		
Driver-outcome linkage (describe; and is there a DRERIP model?)	yes	No DRERIP model. However, waterfowl breeding season ecology is well understood in California, particularly for mallard, the most common breeding species of duck in the state. Species requires upland protected nest sites.	**Scalable (spatial / sub-regional, temporal)**	yes	If decide to only report Delta in 2014, could add Napa and Suisun to future reports.
Sensitivity	yes	Highly sensitive to habitat management: cropping type, crop management, vegetation growth, level of inundation, disturbance, contaminants, and predation.	**Transferable to other watersheds[5]**	yes	Because the survey methods are used throughout North America, this indicator could be used in other parts of California or USA.
Response time frame	yes	Likely to respond within a few years to habitat restoration, once the appropriate habitat is available.	**Feasibility**		
Spatial sampling frame	yes	One would expect to see changes in spatial distribution with habitat restoration and other habitat management improvements.	**Feasible for 2015 State of the Estuary Report timetable [3]**	yes	Data are available. Only constraint is availability of CDFW Waterfowl Program staff to prepare indicator, or to review it.
Final decision:		**Selected and Calculated. Will report on Delta and Suisun Marsh.**			
Decision categories:					
Not selected		Selected but not calculated (indicate if deferred to future)			Selected and calculated

Notes:

3) Newly proposed criterion for 2015; was not used as criterion for SOTB 2011

Additional Notes:

1) EPA Watershed Assessment Framework 2002

2) Comprehensive Conservation & Management Plan, SFEP 2009

4) other plans may include Recovery Plans, Permit requirements/Biological Opinions, HCP/NCCP', or other Restoration program documents ERP Conservation Strategy.

5) For the SotB 2011, if this was the only factor to get a "no" answer, it was not used as a reason to remove indicator from consideration.

Mallard account for 80% of breeding waterfowl in CA. Include all top 5 species including mallard.

State of the Estuary Report 2015

Summary

WILDLIFE – Shorebirds

Prepared by

Matthew E. Reiter and Nadav Nur

Point Blue Conservation Science

State of the Estuary 2015 - San Francisco Bay

Final Version, October 2, 2015

Matthew E. Reiter and Nadav Nur, Point Blue Conservation Science

Wildlife: Birds:

Wintering Shorebird Abundance Indicator

1. Brief description of indicator and benchmark

Nine common wintering migratory shorebird species, representing three groups, based on body size (large, medium, and small) and breeding distribution, were selected to be indicators for intertidal mudflats, salt marshes, and saline ponds in the north, central, and south regions of the San Francisco Bay estuary. The indicator, birds detected per ha, is a measure of shorebird abundance during the winter. The benchmark for the wintering shorebird indicator was established as a 10% increase compared to the baseline value, which is the average abundance of each group in each bay region from early winter surveys conducted 2006-2008.

2. Indicator status and trend measurements

We determined whether the current status (2011-2013 average) of the indicator relative to the historical average (2006-2008), that is, the reference value, in each region of the estuary was Poor, Fair or Good, based on whether abundance had decreased, stayed the same, or increased between periods. Status of the indicator differed depending on the **guild** (size class) of shorebird. For **large shorebirds**, status was **Poor** in the central and south regions but **Fair** in the north, and thus they were scored **Poor** overall. **Medium shorebirds** were also **Poor** in the central and south regions, but **Fair** in the north, hence were scored **Poor** overall. **Small shorebirds** were scored **Fair** in each of the three regions and so were scored **Fair** over all. Since Fair and Poor were so evenly split, our overall assessment for shorebirds is "**Fair-to-Poor**".

Benchmark calculation and score assessment:
The benchmark (the break between Fair and Good) was defined as a 10% increase from the historic period (2006-2008) to the current (2011-2013), provided that the 95% confidence interval of the density estimate for the most recent 3 years did not overlap the reference value. Conversely, we defined the break between Fair and Poor to be a 10% decrease from the reference value as well as a 95% confidence interval of the current density estimate for the most recent 3 years that did not overlap the reference value. For each group the parameter estimate (determined in natural log units) was averaged across species in the group and then back-transformed to obtain a density value.

3. Brief write-up of scientific interpretation

The San Francisco Bay estuary is a site of hemispheric importance for non-breeding migratory shorebirds (Order: Charadriiformes; sub-order: Scolopaci, Charadrii) (Page et al. 1999, Stenzel et al. 2002). Over 1 million shorebirds use the intertidal mudflats, marshes, and saline ponds of the estuary each year (>300,000 birds in winter). The species of shorebirds using the estuary in the non-breeding season vary greatly in body size and abundance, as well as in their migratory pathways and the location of their breeding grounds. While some breed as close as San Francisco Bay and the Central Valley, others nest as far away as the tundra in northern Alaska. The importance of San Francisco Bay for non-breeding shorebird populations representing different species and different migratory traits makes shorebirds in the winter a good indicator of the condition of San Francisco Bay's intertidal wetlands and saline ponds.

Change in shorebird densities between the reference period 2006-2008 and the most recent years available, 2011-2013, were summarized as a Wildlife Indicator of the State of the Estuary for San Francisco Bay. The benchmark for shorebird density was established for each of three regions of the bay (North Bay, Central Bay, and South Bay) and three groupings of nine total shorebird species (based on body size and migratory pathways). The reference value used for comparison was the average density observed on early winter surveys from 2006 to 2008. The benchmark and score was then based on the magnitude of the difference in density relative to the reference value and the degree of certainty in density estimates. Shorebird species included were:

- American Avocet (*Recurvirostra americana*), Willet (*Tringa semipalmata*), and Marbled Godwit (*Limosa fedoa*) to represent **large-bodied,** generally **temperate breeding** birds;
- Black-bellied Plover (*Pluvialis squatorola*) and Short- and Long-billed dowitchers (*Lindronomous griseus*, *L. scolopaceus*) to represent **medium-bodied, mid- to high-latitude breeding** birds;
- Three species of the genus *Calidris* (Dunlin [*Calidris alpina*], Western Sandpiper [*C. mauri*], Least Sandpiper [*C. minutilla*]) to represent **small-bodied, high-latitude breeding** birds.

We selected 2006-2008 for the reference period as it represents the state of shorebird populations just prior to a period of substantial change in wetlands in San Francisco Bay from large-scale restoration of saline ponds to tidal marshes. Within each year, we selected the early winter to measure the indicator as it is a time of stability in shorebird populations (no migration) resulting in relatively lower year to year variation in population counts. Furthermore early winter surveys of the same locations, completed annually in 2011 to 2013 as part of the Pacific Flyway Shorebird Survey (www.pointblue.org/pfss), provide an opportunity to measure change between the reference period and more recent surveys.

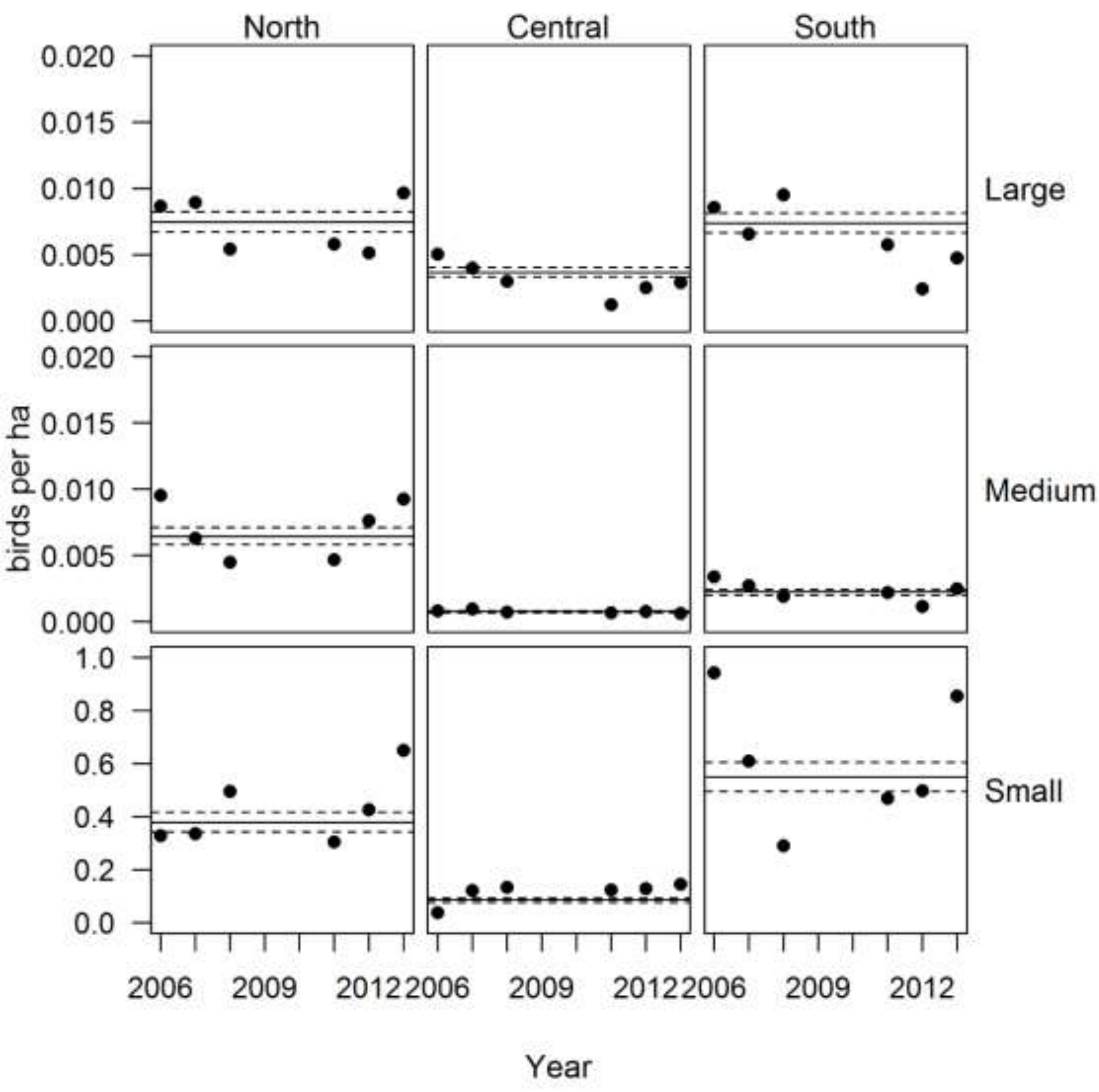
Overall, indicator densities of small shorebirds were the highest among the three shorebird size groups (small, medium, and large) (Fig. 1). Large and medium shorebirds had roughly equivalent densities in the north and central regions, but large shorebirds had higher densities than medium shorebirds in the south region. For all groups, indicator densities were higher in the north and south regions compared to the central region.

Large shorebirds were below the reference values across all regions (-20% in the north bay, -59% in central bay, and -52% in south bay). They were scored poor in the central and south regions as the 95% CI of density estimates did not include the reference value. The north region was considered fair because its 95% CI did include the reference value. The overall score for this group was poor. **Medium shorebirds** were below the reference values in both the central and south regions (-32% and -35%, respectively), but only 5% below the reference value in the north. This group received a status of poor in the south and central region but fair in the north as the 95% CI of the current density estimate overlapped the reference value in that region. Overall, we score the status of medium birds as poor. The average density of **small shorebirds** from 2011–2013 was 3%, 4% and 37% higher than the 2006–

2008 average in the south, north and central regions, respectively. However the 95% CI of current density estimates in all regions overlapped the reference values indicating a status of fair.

Non-breeding shorebird populations of different species and size groups are changing in different ways in abundance and perhaps in distribution within San Francisco Bay. Though some promising trends are evident, none of the three groups achieved the benchmark. Small shorebirds display variability but generally appear stable. Large and medium shorebirds are in decline across the estuary but particularly so in the central and south regions. There has been a large amount of change in wetlands in San Francisco Bay particularly in the south region. Whether declines in medium and large shorebirds in south region are related to these changes in wetlands requires additional research. Ongoing annual monitoring of randomly selected sites and periodic (every 5-8 years) bay-wide comprehensive surveys are needed to better understand the year-to-year variation in shorebirds and to establish whether the changes observed represent changes in wintering shorebird abundance or shifts in bird distribution since the 2006-2008 reference value was established.

Figure 1. Density (birds counted/ha) of large, medium, and small shorebirds within three regions of San Francisco Bay in early winter 2006-2013. The solid horizontal line represents the reference value set as the mean density of the 2006-2008 surveys. The dashed horizontal lines represent ±10% of the reference value. Densities >10% larger than the reference value were considered good and those >10% smaller were considered poor, provided that their respective 95% CIs (not shown) did not overlap the reference value; otherwise, they were considered fair. Densities within 10% of the reference value (between the dashed horizontal lines) were also considered fair.



State of the Estuary Report 2015

Technical Appendix

Prepared by

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Technical Appendix

State of the Estuary: Shorebird Population Indicator

Matthew Reiter and Nadav Nur, Point Blue Conservation Science

Background

The San Francisco Bay estuary is a site of hemispheric importance for non-breeding migratory shorebirds (Order: Charadriiformes; sub-order: Scolopaci, Charadrii) (Page et al. 1999, Stenzel et al. 2002). Over 1 million shorebirds use the intertidal mudflats, marshes and saline ponds of the Bay each year (>300,000 in winter). Non-breeding shorebirds species using San Francisco Bay vary greatly in body size and abundance, as well as in the location of their breeding grounds. Some breed as nearby as the Central Valley while other nest on the Arctic tundra in northern Alaska. Given the importance of San Francisco Bay for shorebirds, and that different shorebird species there use different migratory pathways and breeding grounds, make them a good indicator of the condition of San Francisco Bay's intertidal wetlands.

We summarized year-to-year variation in shorebird populations from surveys in early winter between 2006 and 2013 to develop an indicator of the State of the Estuary for shorebird abundance. Specifically we established a reference value based on the average density of shorebirds observed on surveys between 2006 and 2008 in three regions of the bay (north, central, and south) and for three groupings of shorebird species based on size and breeding distribution. We selected this time period (2006-2008) for the reference period as it represents the state of shorebird populations just prior to a period of change in wetlands in San Francisco Bay from large scale tidal marsh restoration. Within each year, we selected the early winter as it is a time a stability in shorebird populations (no migration) allowing for relatively lower year to year variation in counts compared to migration surveys. Furthermore annual surveys of the same locations were completed again from 2011 to 2013 and are ongoing as part of the Pacific Flyway Shorebird Survey (www.pointblue.org/pfss). The 2011-2013 surveys were compared to the 2006-2008 data to assess change in San Francisco Bay shorebird populations.

We selected nine common shorebird species in San Francisco Bay representing three general groups based on body size and breeding distribution. First, we identified three species of large-bodied shorebirds (American Avocet [*Recurvirostra Americana];* Willet [*Tringa semipalmata*]; Marbled Godwit [*Limosa fedoa*]) that breed in California, the Great Basin and/or the Prairie Pothole region of the north-central United States and south-central Canada (Gratto-Trevor 2000, Lowther et al. 2001, Robinson et al. 1997). Second, we selected the Black-bellied Plover (*Pluvialis squatarola*) and Short- and Long-billed Dowitchers (*Limnodromus griseus*, *L*. *scolopaceus*) combined to represent medium-bodied shorebirds that breed in the mid- to high-latitudes in the arctic (Paulson 1995, Takekawa and Warnock 2000, Jehl et al. 2001). Lastly, three species of the genus *Calidris* (*C. alpina*, *C. mauri*, *C. minutilla*) were combined to represent small bodied shorebirds that are generally high-latitude arctic breeders (Warnock and Gill 1996, Nebel and Cooper 2008, Franks et al. 2014). These nine species composed 96% of the total shorebirds counted in baywide surveys from 2006-2008 (Wood et al. 2010). We selected relatively abundant species to ensure adequate sample sizes of detections. Further, by choosing these different groups of species as indicators, we are better poised to understand whether changes to shorebird populations reflect changes in the condition of intertidal wetlands in San Francisco Bay or are driven by changes on the breeding grounds or along migratory pathways. For example, if all migratory shorebird species show similar trends in abundance through time we are more likely to conclude this has something to do with the condition of San Francisco Bay wetlands than if declines were observed in only birds that breed in the high arctic, which may suggest conditions on the breeding grounds or along the migratory pathway are changing.

Methods

We used November–December high-tide shorebird survey data from 114 randomly selected survey areas around San Francisco Bay (see Wood et al. 2010 and Reiter et al. 2011 for full description of the sampling design and survey methodology) from 2006-2008 and 2011-2013 to estimate the shorebird density (birds per ha) for each of the species in each of the three groups for each year. We established the reference value for each species as the average density across the 2006-2008 surveys and measured change by comparing to the 2011-2013 average density as an indicator of the State of the Estuary of intertidal wetlands, particularly tidal flats and saline ponds. We estimated densities and change of each species for each of three regions of San Francisco Bay (north, central, and south bay) as defined by Wood et al. (2010). We used generalized linear mixed models to estimate mean densities by year while accounting for overdispersion driven by autocorrelation associated with repeated surveys at specific survey areas and across years (Gelman and Hill 2007). We also fit a model that compared the 2006-2008 average density with the 2011-2013 average. We report the percent change between the modeled average density from these two three-year time periods and indicate whether this change was statistically significant for each species in each region. To account for survey areas that varied in size, we included the natural logarithm of the survey area size (ha) as an offset term in all models. We included a random effect of survey area to account for correlation among counts from the same location and a random effect for year to account for among year variation within the 3-year period.

For group density estimates by region (large-sized and medium-sized birds only) and time period (2006-2008 and 2011-2013), we calculated an average of the species-specific density estimates, as well as the average change comparing current and reference periods. For each group, the density parameter estimate (determined in natural log units) was averaged across species in the group and then back-transformed to get density. We estimated the 95% confidence interval (CI) of the group density estimates using a Monte Carlo simulation which randomly sampled the parameter estimates based on their mean and SE and calculated 10,000 estimates of the average. We applied the percentile method and used the 250th and 9750th sorted value to determine the 95% CI. Challenges with identifying Calidris shorebirds to species in the field and their tendency to occur in large roosting flocks (>5,000 - 10,000 individuals), resulted in many observations attributed to an unknown mix of Dunlin (*C. alpina*), Western Sandpiper (*C. mauri*) and Least Sandpiper (*C. minutilla*). We pooled these three species of Calidris spp., whether identified to species or in mixed flocks, into a single analysis.

Species-specific percent change from the reference value was estimated from the modeled change parameter (β) as:

$$(e^{\beta} - 1) * 100$$

The percentage change in the pooled current group average compared to the reference value was estimated as:

$$\frac{current\ value - reference\ value}{reference\ value} * 100$$

We evaluated the status of the indicator by establishing a benchmark value for each shorebird group, as determined for each region of the San Francisco Bay. The benchmark (i.e., the break between status of Fair and Good) was achieved with at least a 10% increase from the reference value (2006-2008) to the current value (2011-2013), provided that the 95% confidence interval of the density estimate for the current value did not overlap the reference value. Conversely, we considered an indicator status to be Poor when there has been at least a 10% decrease from the reference value, provided that the 95% confidence interval of the current value did not overlap the reference value.

Results

There has been variation and change in shorebird populations across San Francisco Bay since 2006-2008. Overall densities were highest for small shorebirds compared to large and medium shorebirds. Additionally all species groups experienced their highest densities in the north and south regions compared to the central bay.

North region: Large birds were scored as fair (though they exhibited -20% change) comparing current to the reference value in the north region. Species-specific changes from the reference value suggest the American Avocet (10%, z = 0.57, P = 0.57) and Willet (1%, z = 0.02, P = 0.90) were stable, whereas Marbled Godwit was pulling down the overall change with a significant decline (-38%, z = -2.01, P = 0.04) for the large shorebird group. Overall, medium shorebirds were scored as fair (-5%) in the north region. Dowitchers were increasing in the north region (55%, z = 1.02, P = 0.30), whereas Black-bellied Plovers were declining (-26%, z = -1.15, P = 0.25) though neither change was significant. Small shorebirds increased by 4% in the north region though the change was not statistically significant (z = 0.63, P = 0.53), so their status was considered fair.

Central region: Large shorebird density declined relative to the reference value in the central region (-59%). Significant declines of Willet in the central region (-68%, z = -2.62, P = 0.01) drove the negative trends observed in large birds, though American Avocet (-54%, z = -1.37, P = 0.17) and Marbled Godwit (-53%, z = -1.68, P = 0.09) experienced non-significant declines as well. For medium shorebirds, Dowitchers declined (-42%, z = -1.79, P = 0.07) while Black-bellied plover increased (16%, z = 0.55, P = 0.58) however overall change (-32%) indicated poor status. Small shorebirds increased, albeit non-significantly in the central region (37%; z = 1.21, P = 0.22).

South region: Large shorebirds declined in the south region (-52%), driven largely by a significant decline in Willet (-69%, z = -4.72, P <0.01). However, American Avocet (-17%, z = -1.56, P = 0.12) and Marbled Godwits also declined (-53%, z = -1.48, P = 0.40). Medium shorebirds were scored poor in the south region (-35% change compared to reference value). Both dowitchers (-27%, z = -1.12, P = 0.26) and Black-bellied Plover (-31%, z = -1.29, P = 0.20) declined though these changes were not statistically significant. Small shorebirds were relatively stable in south region compared to large and medium birds, with evidence of only a small, non-significant, change from the reference (6%; z = 0.67, P = 0.86).

Summary

Non-breeding shorebird populations of different species and size groups within San Francisco Bay are changing in different ways in abundance and perhaps in distribution. None of the shorebirds have achieved their benchmark. Overall, large and medium shorebirds declined across all bay regions and significantly so in the central and south regions. However, small shorebirds were stable across all regions compared to the reference period. Significant year to year variation in abundance for some species groups made estimates of change quite imprecise with only six years of data thus limiting inference. In many cases, observed changes exhibited large declines and increases in point estimates, but were not statistically significant. Ongoing annual monitoring of randomly selected sites and periodic bay-wide comprehensive surveys are needed to better understand the year-to-year variation observed and whether the estimated trends are real.

Peer Review

This indicator was reviewed by shorebird ecologists at Point Blue including W. David Shuford and Gary Page.

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State of the Estuary Report 2015

Summary

WILDLIFE – Herons and Egrets

Prepared by

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State of the San Francisco Estuary 2015
Wildlife – Heron and Egret Nest Density and Nest Survival Indicators

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What are the indicators?

As top wetland predators that operate over large areas of the San Francisco Estuary, herons and egrets depend on extensive tidal marshes, seasonal wetlands, and associated freshwater systems. The State of the Estuary Report uses two indicators based on the status of nesting herons and egrets to assess ecological conditions across broad wetland landscapes. The Heron and Egret Nest Density Indicator provides an index of regional heron and egret population sizes. The Heron and Egret Nest Success Indicator is based on nest survival through the breeding cycle (not on the productivity of successful nests) and is used to assess the dynamics of nest-predator populations, human disturbance, and changes in human land use that can affect the size and distribution heron and egrets nesting colonies. The chapter on "Processes-Feeding Chicks," in the 2015 State of the Estuary Report, summarizes the Heron and Egret Brood Size Indicator, which uses the number of young produced in successful nests to index conditions that affect the availability of food, the productivity of estuarine food webs, and the quality of wetland feeding areas. For details, see State of the Estuary 2015: Processes – Heron and Egret Brood Size Indicator document.

Attribute	Indicator	Benchmark
Wildlife: Birds	Great Blue Heron/Great Egret Nest Density	The benchmark for nest density is the average nest density observed from 1991-2000, for each region: Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all three areas combined.
	Great Blue Heron/Great Egret Nest Survival	The benchmark for nest success is the average nest survival from 1994-2000, for each region: Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all three areas combined.

How are the current indicator conditions measured?

Heron and Egret Nest Density Indicator
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, 2008; Kelly and Condeso 2014). The data, which reflect repeated annual nest counts at all known colony sites, provide intensive and extensive measurements of nest abundance and an index of regional breeding population sizes. Results

are provided for each year (1991-2014), for all known nesting colonies in each of three northern subregions (Central San Francisco Bay, San Pablo Bay, Suisun Bay, and the combined area of all three subregions.

The Nest Density Indicator is calculated as the geometric mean of annual nest densities for two species, Great Blue Heron (*Ardea herodias*) and Great Egret (*Ardea alba*). The Heron and Egret Nest Density Indicator was also calculated separately for Great Egrets and Great Blue Herons (see Technical Appendix). Nest density estimates are based the peak number of active nests at each of 40-50 active colony sites each year, summed within and across subregions, based on four (monthly) visits per year to each site within foraging range (10 km) of the historic tidal wetland boundary (ca.1770–1820; San Francisco Estuary Institute 1999). Density is calculated as the number of nests per 100 km^2, within the region or subregion, excluding the extensive open water areas of the San Francisco Estuary.

For analysis, we calculated the percent change in the mean indicator values during recent years, 2009-2014, relative to the ten-year baseline period, 1991-2000. In addition, patterns of proportional change over time were modeled as linear or quadratic trends over the 24 years of monitoring, 1991-2014. The trends were estimated using quadratic models, with increasing or decreasing slopes, *if and only if* the quadratic term was significant ($P<0.05$); otherwise changes over time were estimated as linear trends.

Heron and Egret Nest Survival Indicator

Audubon Canyon Ranch has monitored the survival of focal Great Blue Heron and Great Egret nests (proportion of nest attempts that fledge at least one young) across nesting colonies throughout the northern San Francisco Estuary, annually, since 1994 (Kelly et al. 2007, Kelly and Condeso 2014).

The Heron and Egret Nest Survival Indicator, calculated as the annual, arithmetic mean of apparent nest success, between species, for Great Egret and Great Blue Heron, is calculated within and across the three major subregions of northern San Francisco Bay (Central San Francisco Bay, San Pablo Bay, and Suisun Bay; Indicator values were also calculated separately for Great Egrets and Great Blue Herons; see Technical Appendix). 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 in 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.

For analysis, we calculated the percent change in the mean Nest Survival Indicator between recent years, 2009-2014, and the seven-year baseline period, 1994-2000. In addition, changes over time were estimated using linear or quadratic models over 21 years, 1994-2014. As with the Nest Density Indicator, above, changes were modeled as quadratic trends *if and only if* the quadratic term was significant ($P<0.05$); otherwise they were estimated as linear trends. We converted estimates of proportional change to percent annual change over the entire monitoring period, or before/after years with minimum/maximum values.

What are the benchmarks for these indicators, and how were they selected?

Heron and Egret Nest Density Indicator

The benchmark for the Heron and Egret Nest Density Indicator is the geometric mean indicator value (back-transformed, $\log_e$ mean) during the first ten years of regional monitoring, 1991-2000. This period was selected because it reflected a period of relatively lower annual variation in nest density for both of the two study species, relative to subsequent years.

Heron and Egret Nest Survival Indicator

The benchmark for the Heron and Egret Nest Survival Indicator is mean annual proportional nest survival during the first seven years of regional monitoring, 1994-2000. This period was selected to be consistent with the benchmark selected for the Heron and Egret Nest Density Indicator but reduced in length because of nest survival data were not available for 1991-1993.

What is the status and trend of each indicator in each area?

Heron and Egret Nest Density

Heron and Egret Nest Density (Figure 1, Table 1) increased by 1.1% annually across all areas (log_e density increase: 0.012 ±0.005 [SE] per year; P=0.046). In recent years, 2009-2014, Heron and Egret Nest Density

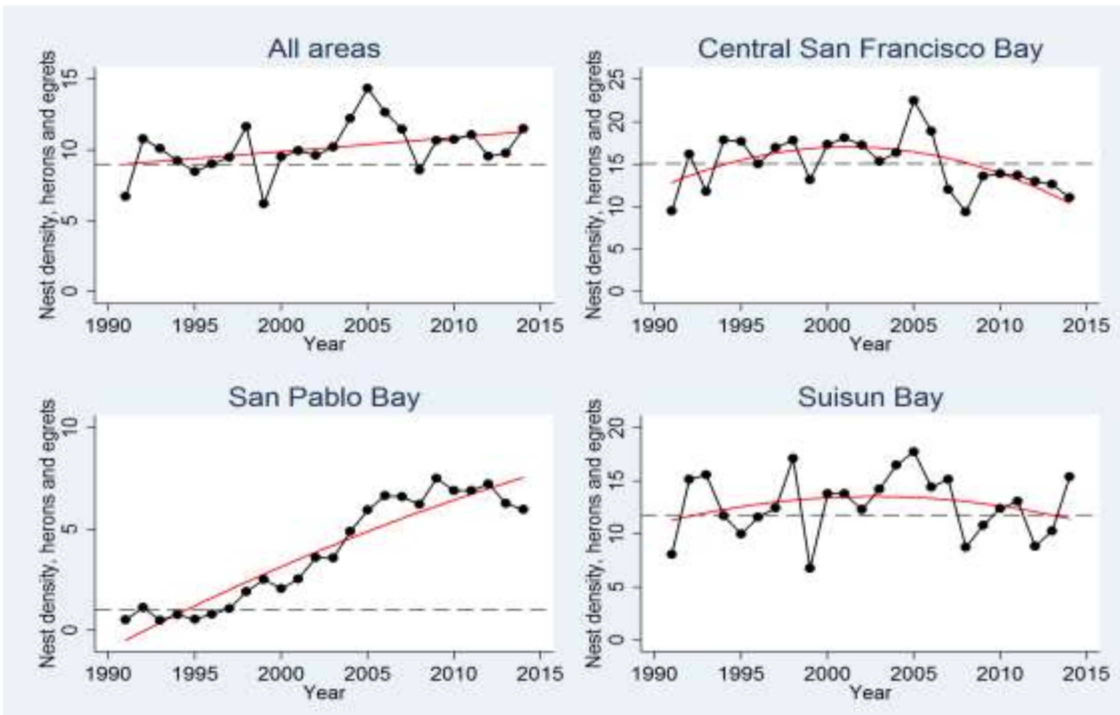

Figure 1. Annual Heron and Egret Nest Density Indicator and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

was 17% greater, on average, than during the baseline period of 1991-2000 ($F_{1,14}$=3.6, P<0.07), although the difference was only marginally significant (Table 1).

Heron and Egret Nest Density in Central San Francisco Bay exhibited an increasing, quadratic trend that leveled off in 2001 ($F_{2,21}$=5.82, P<0.01) at 17.4 nests per 100 km^2, 13% above the baseline. After 2001, nest density declined, on average, by 3.5% annually (log_e decline: 0.04±0.01; P=0.01). Heron and egret nest density appeared to be 14% lower in 2009-2014 than during the baseline period, although the significance was marginal ($F_{1,14}$= 2.71, P=0.12; Table 1).

In San Pablo Bay, the Heron and Egret Nest Density Index increased by 13.5% annually since 1991 (log_e increase: 0.126±0.012 per year, P<0.001), but leveled off after 2010. Heron and Egret Nest Density was 570% greater in San Pablo Bay, on average, in 2009-2014 than during the baseline period ($F_{1,14}$=59.3, P<0.001; Table 1). In Suisun Bay, Heron and Egret Nest Density was dynamic across years but showed no significant trend ($F_{2,21}$=0.66, P=0.53) or difference between recent years and the baseline period, 1991-2000 ($F_{1,14}$=0.01, P=0.91; Table 1).

Table 1. Heron and Egret Nest Density Indicator (species combined) results including current and baseline means (nests/100 km^2), 95% confidence intervals (CI), and percent change comparing the "current" period of recent years, 2009-2014, relative to the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the *F*-value and significance (*P*) of the change; all results are back-transformed from natural-log values.

Area	Current (2009-2014)	95% CI	Baseline (1991-2000)	95% CI	Percent change	$F_{1,\ 14}$	*P*
All areas combined	10.6	9.8 - 11.4	9.0	7.9 - 10.3	**17.0**	**3.6**	**0.07**
Central San Francisco Bay	12.9	11.8 - 14.1	15.1	12.9 - 17.5	-14.1	2.7	0.12
San Pablo Bay	6.8	6.2 - 7.4	1.0	0.7 - 1.7	**570.0**	**59.3**	**<0.001**
Suisun Bay	11.6	9.5- 14.3	11.8	9.5 - 14.5	-1.4	0.1	0.91

Heron and Egret Nest Survival

Mean Heron and Egret Nest Survival (Figure 2, Table 2) across northern San Francisco Bay was dynamic but stable, exhibiting no significant trend ($F_{2,18}$=.80, P=0.46) and no significant difference between recent years (2009-2014) and the baseline period (1994-2000; $F_{1,14}$=0.9, P=0.35; Table 2). However,

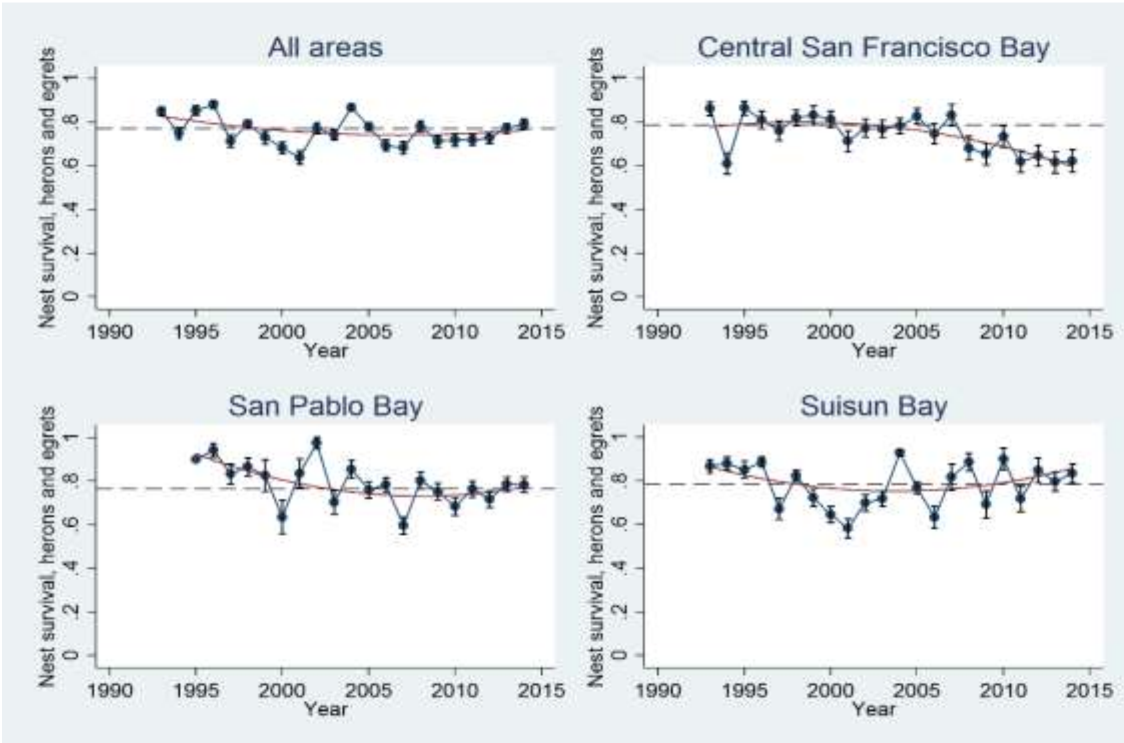

Figure 2. Annual Heron and Egret Nest Survival Indicator and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1994-2000.

within Central San Francisco Bay, nest survival began to decline in 1998, at an average rate of 1.8% per year ($\log_e$ rate: -0.018±0.0003; $F_{1,15}$=35.2, P<0.001), dropping from 78% nest survival in 1994-2000 to 65% in 2009-2014 ($F_{1,14}$=13.1, P=0.004). In San Pablo Bay, Heron and egret nest survival declined by 1.46% per year from 1995 to 2008, then leveled out through 2014 ($F_{2,17}$=3.78, P=0.04). Mean nest survival was relatively stable in Suisun Bay ($F_{2,18}$=0.89, P=0.42).

Table 2. Heron and Egret Nest Survival Indicator (species combined) results, including the mean and standard error (SE) of annual percent nest survival, weighted equally among years, l during the "current" period of recent years, 2009-2014, and the baseline period, 1994-2000, the mean percent change between current and baseline periods, and the F-value and significance (P) of the change relative to variation among years.

Area	Current (2009-2014)	SE	Baseline (1994-2000)	SE	Percent change	$F_{1, 11}$	P
All areas combined	74.3	2.44	77.5	2.22	-4.1	0.9	0.35

Central San Francisco Bay	64.7	2.91	78.8	2.59	**-17.9**	**13.1**	**0.004**
San Pablo Bay	75.0	4.21	76.3	6.29	-1.7	0.03	0.87
Suisun Bay	79.3	5.19	78.7	3.04	0.7	0.01	0.93

In general, what do the results mean and why are they important?

Heron and Egret Nest Density

The nesting densities of herons and egrets are stable or increasing in the northern San Francisco Bay region. This suggests improvements in wetland condition associated with the extent or quality of suitable foraging or nesting areas, or with the supply or availability of fish or other suitable prey. In San Pablo Bay, substantial increases in heron and egret nesting density may be associated with wetland restoration efforts; the apparent, recent leveling off of nest densities in San Pablo Bay suggests that regional heron and egret distributions have stabilized after the colonization of new wetland feeding areas. A relatively steep, declining trend in nest density in Central San Francisco Bay may be of some concern, with regard to the management of several islands used for nesting, including the potential disturbance by ravens or other nest predators.

Heron and Egret Nest Survival

Heron and Egret nest survival is stable when measured across all areas of northern San Francisco Bay. This is consistentwith the localized scale of disturbance to heronries that accounts for most of the variation in nest survival (Kelly et al. 2007). The declining trend in nest survival in Central San Francisco Bay is consistent with the parallel decline in nest density. This suggests that localized disturbances by nest predators or humans, which typically account for most heron and egret nest failures, could be reducing the number of nesting herons and egrets in Central San Francisco Bay.

How do the indicators relate to the ecological health of the estuary?

Heron and Egret Nest Density Indicator

Heron and egret nest abundance is recognized as a valuable metric for assessing biotic condition in estuarine and wetland ecosystems (Parnell et al. 1988, Kushlan 1993, Fasola et al. 2010, Kushlan and Hancock 2005, 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).

Heron and Egret Nest Survival Indicator

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, Rothenbach and Kelly 2012). 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 (Kelly et al. 2008), and are therefore likely to differentially affect nesting colonies of herons and egrets. Nest survival is not a strong indicator of food availability; processes affecting food web conditions are more clearly monitored by the Heron and Egret Brood Size Indicator (see chapter on Processes).

State of the Estuary Report 2015

Technical Appendix

WILDLIFE – Herons and Egrets

Prepared by
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Technical Appendix

Great Blue Heron Nest Density.

The evaluation of Great Blue Heron nest density (Figure 3, Table 3) across all areas revealed an increasing, quadratic trend that peaked in 2005 ($F_{2,21}$=3025 *P*=0.058), with an estimated maximum density of 6.3, 17.2% above the baseline average. After 2005, nest density declined, although non-significantly, by 2.2% per year ($F_{1,7}$=2.0, *P*=0.20). Mean Great Blue heron nest density in 2009-2014 did not differ from the baseline period ($F_{1,14}$=1.58, *P*=0.23; Table 3).

In Central San Francisco Bay, Great Blue Heron nest density increased by 4.1% annually ($\log_e$ trend: 0.040±0.006 [SE], *P*<0.001), until 2011 reaching a an estimated maximum density of 12.9 birds per 100 km^2, 67.8% above the baseline average ($F_{2,21}$=37.52 *P*<0.001). In 2009-2014, Great Blue Heron nest density in Central San Francisco Bay was 66.2% greater, on average, than during the baseline period ($F_{1,24}$=24.1; *P*<0.001; Table 3).

Great Blue Heron nest density in San Pablo Bay exhibited a marginally significant linear increase of 1.3% ($\log_e$ increase of .013±007 per year; *P*=0.08), leading in 2009-2014 to nest density 25.9% greater, on average, than during the baseline period ($F_{1,14}$=5.0; *P*=0.04; Table 3).

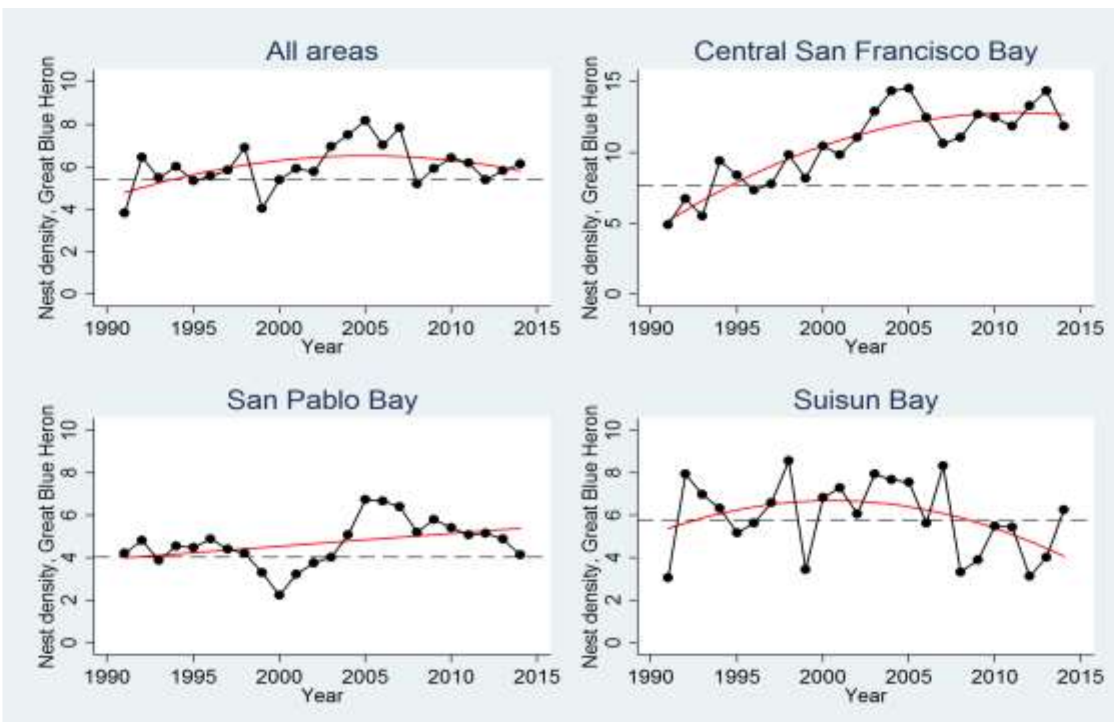

Figure 3. Annual Great Blue Heron nest density and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

Table 3. Great Blue Heron Nest Density Indicator results, including current and baseline means (nests/100 km2), 95% confidence intervals (CI), and percent change during the "current" period of recent years, 2009-2014, relative to the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the F-value and significance (*P*) of the change; all results are back-transformed from natural-log values.

Area	Current (2009-2014)	95% CI	Baseline (1991-2000)	95% CI	Percent change	$F_{1, 14}$	*P*
All areas combined	6.0	5.6 - 6.4	5.4	4.7 - 6.2	10.5	1.6	0.23
Central San Francisco Bay	12.8	11.8 - 13.8	7.7	6.4 - 9.1	**62.2**	**24.1**	**<0.001**
San Pablo Bay	5.1	4.5 - 5.7	4.0	3.4 - 4.8	**25.9**	**5.0**	**0.04**
Suisun Bay	4.6	3.5 - 6.1	5.8	4.6 - 7.4	-20.8	2.3	0.15

In Suisun Bay, Great Blue Heron nest density exhibited a quadratic trend with a peak density of 15.7% above the baseline in 2003, but the strength of the quadratic coefficient (b = -0.015±0.008, P<0.07) and the overall trend was marginal ($F_{2,21}$=2.43 P=0.11). Mean Great Blue Heron nest density, in Suisun Bay, 2009-2014, was 20.8% below the baseline period, suggesting a decline in Suisun Bay, but the difference was not significant ($F_{1,14}$=2.1, P=0.17; Table 3).

Great Egret Nest Density

Great Egret nest density (Figure 4, Table 4) increased by 1.3% annually across all areas ($\log_e$ increase: 0.013±0.006 per year; P=0.03). Great Egret nest density was 24.6% greater, on average, in 2009-2014 than during the baseline period ($F_{1,14}$=5.1, P=0.04).

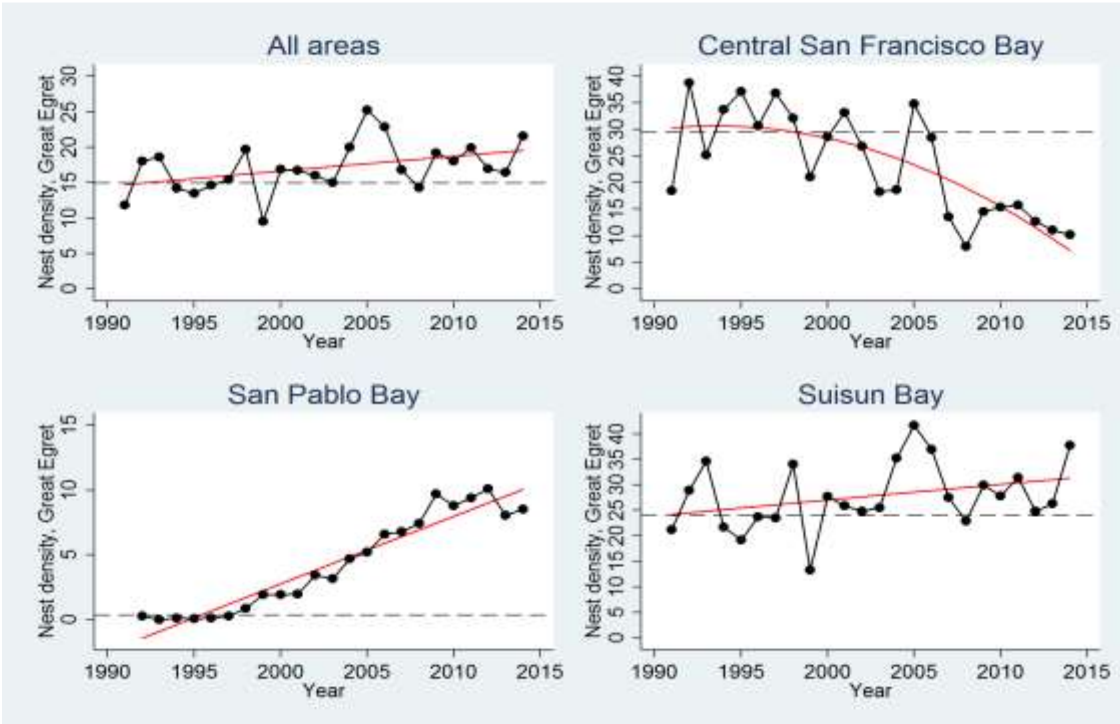

Figure 4. Annual Great Egret nest density and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

In Central San Francisco Bay, an accelerating decline in Great Egret nest density began in 1995 ($F_{2,21}$=15.1 P<0.001), by 5.6% annually ($\log_e$ decline: -.0638573±0.009, P<0.001), leading to nest densities in 2009-2014 that were 55.5% lower, on average, than in 1991-2000 ($F_{1,14}$=47.7; P<0.001; Table 4).

In San Pablo Bay, Great Egret nest density increased by 13.5% annually since 1991($\log_e$ increase: 0.224±0.024; P<0.001) and, by 2009-2014, Greg Egret nest densities were 2452% greater than the low average only 0.36 nests per 100 km^2 in 1991-2000 ($\log_e$ mean: -1.03±0.45; $F_{1,14}$=37.7, P<0.001; Table 4).

Great Egret nest densities in Suisun Bay suggested a marginally significant linear trend, increasing by 1.2% annually ($\log_e$ increase: 0.012±0.007, P=0.10). However, the 22.7% increase in mean nest densities in 2009-2014 over the 1991-2000 baseline period was not significant ($F_{1,14}$=2.6, P=0.13; Table 4).

Table 4. Great Egret Nest Density Indicator results, including current and baseline means (nests/100 km2), 95% confidence intervals (CI), and percent change during the "current" period of recent years, 2009-2014, relative to the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the *F*-value and significance (*P*) of the change; all results are back-transformed from natural-log values.

Area	Current (2009-2014)	95% CI	Baseline (1991-2000)	95% CI	Percent change	$F_{1,14}$	P
All areas combined	18.6	16.7 - 20.7	14.9	12.7 - 17.5	**24.6**	**5.1**	**0.04**

Central San Francisco Bay	13.1	10.9 - 15.8	29.5	24.7 - 35.3	**-55.5**	**47.7**	**<0.001**
San Pablo Bay	9.1	8.3 - 9.9	0.4	0.1 - 1.0	**2452.0**	**37.7**	**<0.001**
Suisun Bay	29.4	25.2 - 34.4	24.0	19.5 - 19.4	22.7	2.6	0.13

Great Blue Heron Nest Survival

Mean annual Great Blue Heron nest survival (Figure 5, Table 5) was relatively stable, with no long-term trends in the Central Bay, San Pablo Bay, Suisun Bay, or all areas combined, and no significant differences in annual nest survival between recent years (2009-2014) and the 1994-2000 baseline period (P>0.05).

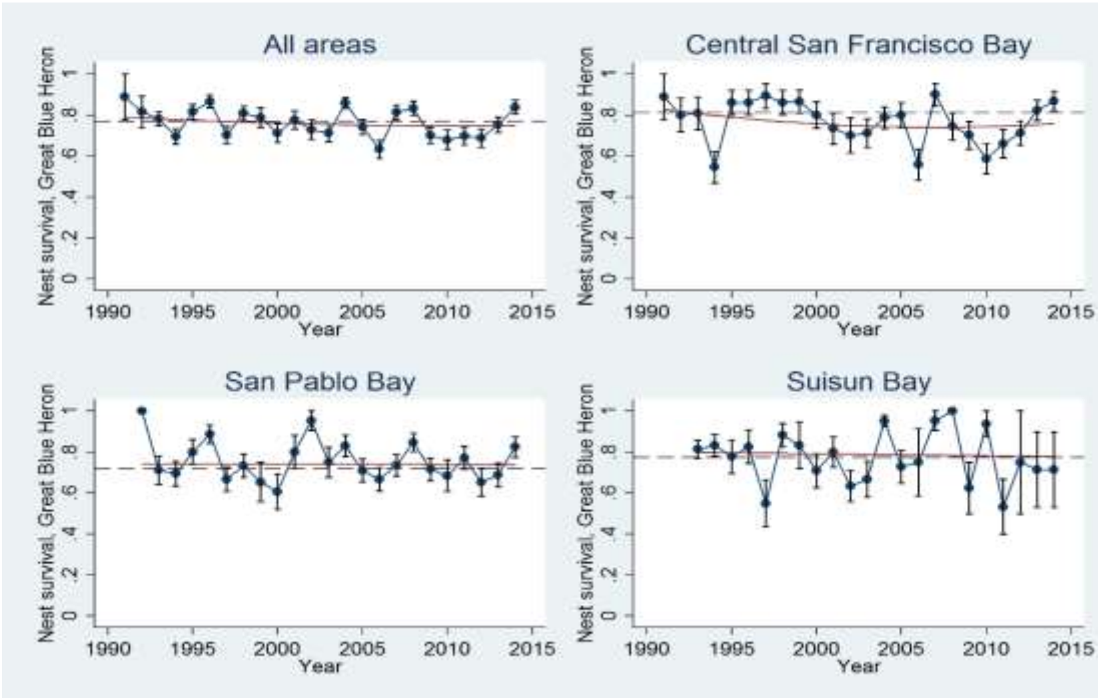

Figure 5. Annual Great Blue Heron nest survival and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1994-2000.

Table 5. Great Blue Heron Nest Survival Indicator results, including the mean and standard error (SE) of annual percent nest survival, weighted equally among years, during the "current" period of recent years, 2009-2014, and the baseline period, 1994-2000, the mean percent change between current and baseline periods, and the *t*-value and significance (*P*) of the change.

Area	Current (2009-2014)	SE	Baseline (1994-2000)	SE	Percent change	$F_{1, 11}$	*P*
All areas combined	72.8	2.65	77.3	2.41	-5.7	1.5	0.24
Central San Francisco Bay	72.8	4.74	81.0	4.22	-10.1	1.7	0.22
San Pablo Bay	72.2	2.47	72.6	3.69	-0.6	0.0	0.93
Suisun Bay	69.8	5.92	79.2	3.46	-11.8	1.8	0.20

Great Egret Nest Survival
Great Egret Nest Survival (Figure 6, Table 6) exhibited no significant trends when evaluated across all subregions combined ($F_{2,19}$=1.2, P=0.31). However, in Central San Francisco Bay, a significantly negative trend began in 1999 ($F_{2,19}$=8.7, P=0.002), with nest survival declining by 3.9% per year, on average,

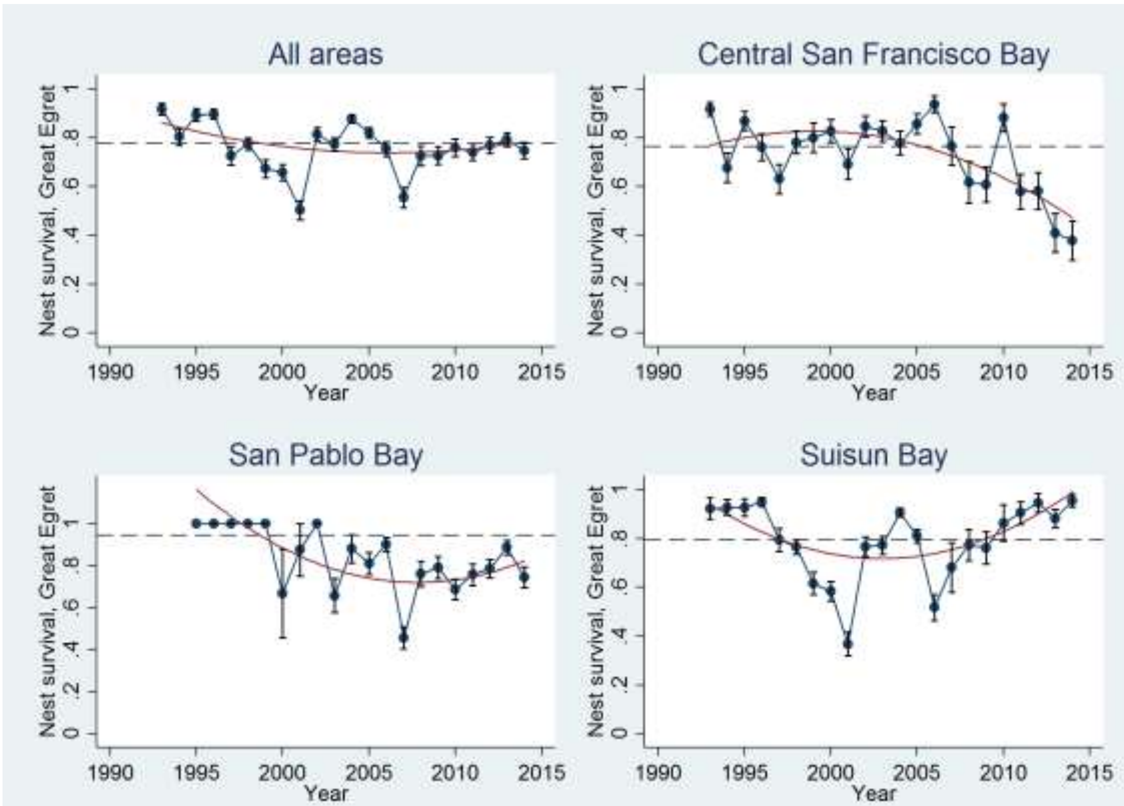

Figure 6. Annual Great Egret nest survival and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1994-2000.

through 2014. As a result, average nest survival in recent years (2009-2014) was significantly lower than during the reference period, averaging 57.3±7.34% survival compared to a baseline of 76.4±3.16% ($F_{1,12}$=9.0, P=0.01; Table 6).

In San Pablo Bay, Great Egret nest survival declined from a relatively high average of 94.4±5.6% during 1995-2000 (based on relatively small samples of only 15.6±4.9 nests per year) to a low of 72.0% survival in 2008, followed by an apparent recovery to near baseline levels by 2014; however, survival rates varied substantially and the trend was not statistically significant ($F_{2,17}$=1.43, P=0.27). In addition, recent nest survival in 2009-2014, averaging 77.5±2.71%, did not differ significantly from the baseline level ($F_{1,10}$=0.27, P=0.61).

In Suisun Bay, a marginally significant (quadratic) trend in Great Egret nest survival suggested a decline survival in the late 1990s, followed by a recovery through 2014 ($F_{2,18}$=2.7, P=0.09). In recent years (2009-2014), average nest survival (88.6±2.87%) did not differ, on average, from the baseline period ($F_{1,12}$=0.02, P=0.90).

Table 6. Great Egret Nest Survival Indicator results, including the mean and standard error (SE) of annual percent nest survival, weighted equally among years, during the "current" period of recent years, 2009-2014, and the baseline period, 1994-2000, the mean percent change between current and baseline periods, and the *t*-value and significance (*P*) of the change.

Area	Current (2009-2014)	SE	Baseline (1994-2000)	SE	Percent change	$F_{1, 11}$	*P*
All areas combined	75.7	2.99	77.6	2.71	-2.5	0.2	0.64
Central San Francisco Bay	56.5	5.39	76.6	4.80	**-26.2**	**7.7**	**0.02**
San Pablo Bay	77.8	7.67	80.0	11.47	-2.8	0.0	0.88
Suisun Bay	88.7	7.43	78.2	4.35	13.3	1.5	0.25

What are the historical uses of these indicators and current programs to evaluate them?

Audubon Canyon Ranch (ACR) has monitored Great Blue Heron and Great Egret nest abundance at all known nesting colonies (40-50 sites) in the northern San Francisco Estuary, annually, since 1991. ACR continues to sustain this effort on an ongoing basis, and to produce regular reports based on this information (e.g., Kelly et al. 1993, 2005, 2006, 2007, 2008; Kelly and Rothenbach 2012; Kelly and Condeso 2014).

What is the suitability of the reference conditions and targets?

Heron and Egret Indicators are suitable targets for monitoring wetland conditions at landscape scales (Kelly et al. 2008). Nest densities during 1991-2000 represent a relatively stable period. Inter-year variation in water levels, weather and climate may challenge identification of reference conditions and targets. Nest densities may be affected by inter-year movements of individuals to or from the Central Valley.

What are the data sources?

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, 2008; Kelly and Condeso 2014). 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.

What assumptions and uncertainties are involved?

Heron and Egret Nest Density Indicator

The Nest Density Indicator assumes that most or all of the colony sites are known and monitored, that hidden, concealed nests are rare, and that the intraseasonal peak nest abundance is documented accurately. The conspicuousness of heron and egret colonies and nests facilitates the successful use of this indicator.

Heron and Egret Nest Survival Indicator
The Nest Survival Indicator assumes (1) that nestling ages in successful nests are accurately estimated, based on repeated nest monitoring and physical and behavior correlates of nestling development, and (2) that nestlings do not fledge before they are 7 weeks old, post-hatch (Great Egret) or 8 weeks old (Great Blue Heron). Uncertainties are related to early failures during incubation, unobserved nest failures followed by and renesting between observations. The conspicuousness of heron and egret nests facilitates the successful use of this indicator.

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State of the Estuary Report 2015

Summary

WILDLIFE – Tidal Marsh Birds

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State of the Estuary 2015: Wildlife

Tidal Marsh Bird Population Indicator

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Final, October 2015

1. Brief description of indicator and benchmark

The Tidal Marsh Bird Population Indicator provides a measure of the breeding season population abundance of three tidal marsh-dependent bird species in the San Francisco Bay Estuary: the Black Rail (*Laterallus jamaicensis*), Song Sparrow (*Melospiza melodia*), and Common Yellowthroat (*Geothlypis trichas*). All three species are represented in the estuary by subspecies that display unique adaptations to tidal marsh habitat, but are, at the same time, of conservation concern (respectively: *L. j. coturniculus*; *M. m. pusillula*, *M. m. samuelis*, *M. m. maxillaris*; *G. t. sinuousa*). Standardized surveys of these species have been conducted in tidal marsh habitat since 1996, and from this an index of population density was constructed, combining results for the three species; the indicator was included in the 2011 State of the Estuary Report.

We evaluate the densities of the three species, comparing the most recent four years (2011-2014) with respect to the benchmark values. The benchmark for distinguishing Good from Fair is the 75^{th} percentile of density as determined with reference to surveyed marshes during the benchmark period (1996-2008). The benchmark for distinguishing Poor from Fair is 25% reduction for the most recent 4 years, on average, when compared to the mean value during the benchmark period. The indicator and benchmark values all refer to results combined across the three species.

2. Indicator status and trend metrics

The mean value of the index over the last four years is 1.09 birds per hectare, below the criterion for Good (1.29 birds/ha) and above the criterion for Poor (0.77 birds/ha). Hence the status is Fair. In one of those four years, the density index was in the region demarcated “Good”; in the other three years, the annual density index was in the “Fair” region. This result, that three out of four years were in the “Fair” region, supports the overall score of “Fair” for this indicator.

The trend during the entire period of study, 1996 to 2014, is significantly positive ($P = 0.002$), increasing at 2.7% per year. Thus, progress is good, and gives indication of moving towards the desired goal.

3. Brief write-up of scientific interpretation

<u>What is the indicator?</u>

The indicator provides an index of population density of three tidal marsh-dependent species in the San Francisco Bay estuary during the breeding season. As part of Point Blue's tidal marsh bird monitoring project begun in 1996 (Nur et al. 1997, Spautz et al. 2006), standardized surveys are conducted every year for all bird species in tidal marsh habitat. Three species are included in this indicator, as they are valuable indicators of tidal marsh ecosystem condition. Each species is year-round resident and is represented by subspecies that are dependent on, or strongly associated with, tidal marsh habitat (Goals Project 2000). As part of the surveys, individuals are identified and enumerated within 50 m of an observer, surveys conducted at about ten survey stations per marsh. Ten marshes were regularly surveyed during the period 1996-2014 and included here. The indicator is based on the estimated value in each year for each species, statistically adjusted for variation in abundance among survey sites (not all sites are surveyed in each year), and then combined across all three species. We stress that it is an index of density but does not measure absolute density. Changes in the density index, we believe, reflect changes in the absolute density, which are translated into variation in underlying population abundance.

<u>Why is it important?</u>

San Francisco Estuary tidal marsh habitat has been dramatically altered in the past two centuries. Over 80% 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 (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. 2012). For these reasons, many of the species that depend on tidal marsh habitat are currently listed as Federally- or State- threatened or endangered, such as the Black Rail, or are designated as California Species of Special Concern (Shuford & Gardali 2008). As a result of the significant loss of habitat quantity and quality, current management and restoration by agencies and non-governmental organizations has been directed at recovering depleted populations or ensuring their stability. The tidal marsh bird index provides a measure of current condition of three tidal marsh-dependent species, as well as providing insight into success at recovering or maintaining these threatened populations.

<u>What is the benchmark? How was it selected?</u>

We expect increased density of tidal marsh breeding birds, reflecting population recovery and improvement in habitat quality (e.g., due to reduction of threats, maturation of restored habitat). The benchmark value for Good is the upper quartile value of population density observed for mature tidal marsh, combined across the three target species. The justification for using the upper quartile for the benchmark for Good is that it represents the median of the highest 50% with respect to density. The value determined for the baseline period, 1996 to 2008 was determined specifically with respect to the set of marshes that have been regularly surveyed during the entire period (1996 – 2014). The minimum value for Good was calculated to be an index value of 1.29 birds/ha.

For demarcating Poor vs Fair, we used a value that was 25% below the mean as determined for the benchmark period. Thus, if the mean index value for the most recent 4 years was below 0.77 birds/ha, the indicator was scored Poor. Given that the goal of tidal marsh management and restoration is to increase populations of tidal marsh dependent species, all three of which are either State-Threatened or California Species of Special Concern, we consider a 25% or greater decrease compared to the benchmark period to merit a score of Poor.

<u>What is the status and trend for this indicator?</u>

The three species Tidal Marsh Bird Population Index varied from 0.93 birds/ha to 1.32 birds/ha during the four years 2011 to 2014, with a mean value of 1.09 (Figure 1). Thus, the indicator was scored Fair. We note that three of the four years were in the region corresponding to Fair; one year (2012) was in the Good region.

The Tidal Marsh Bird Index demonstrated a significant, increasing trend over the entire time period (Figure 1), of 2.78% per year (S.E. = 0.73%; $P < 0.002$). An average annual growth rate of 2.78% over the course of 18 years (i.e., 1996-2014), translates into a total estimated increase of 64%. However, not all species exhibited increases over this period. Black Rail and Common Yellowthroat significantly increased over the entire period ($P = 0.012$ and $P = 0.016$, respectively). Song Sparrow exhibited a weak increase, but this was not significant. However, the overall increase in trend reflects an early increase (1996-2005) followed by no overall increase during the latter period (2005-2014), as demonstrated in Figure 1. The best estimate of the trend in the first 9 years is 5.1% per year, whereas in the last 9 years, the trend is indistinguishable from 0% change.

<u>What does it mean? Why do we care?</u>

The Tidal Marsh Bird Population Indicator provides an insight into two important aspects of the tidal marsh ecosystem. First, it reflects the condition of three species of conservation concern that depend on tidal marsh habitat, with respect to their population status and how that has changed since 1996. Second, the index also bears on the apparent efficacy of activities designed to stabilize or increase populations of these three species.

The 2015 Tidal Marsh Bird Population Indicator reflects, overall, a mixed picture. On the one hand, the current status is Fair; the benchmark value of Good has not yet been reached. While there is a general increase in density of the three-species-index since 1996, no clear increase is evident in the more recent years (2005 to 2014). Furthermore, only two out of the three species demonstrate an increase in density over the entire period, 1996 to 2014. That said, there is no evidence of a decline in density during the entire time period, nor a portion of the time series, nor do any of the three species demonstrate a decline. This is in contrast to findings for the Ridgway's Rail Population Indicator which has demonstrated a recent decline in the South Bay, from which the population has yet to recover. In contrast, the rebound observed for Ridgway's Rail in the North Bay since 2005 (the first year of the Ridgway's Rail Population Index), is consistent with the increase observed for Black Rails, which are almost entirely confined to the North Bay, for the period 2005-2014 as well as during the entire period (Evens and Nur 2000).

Our conclusion is that habitat suitability is currently sufficient to maintain populations at their current density, and possibly is sufficient to support increase in density, at least for rail species and for the Common Yellowthroat. Furthermore, an increase in density is expected to be manifest as young restored marshes become more mature, and thus increase in their ability to support growing populations of tidal marsh bird species. The prognosis for the near future is encourage.

4. Related Figures

Figure 1 displays results of the Indicator. The Technical Appendix includes index results by species as well.

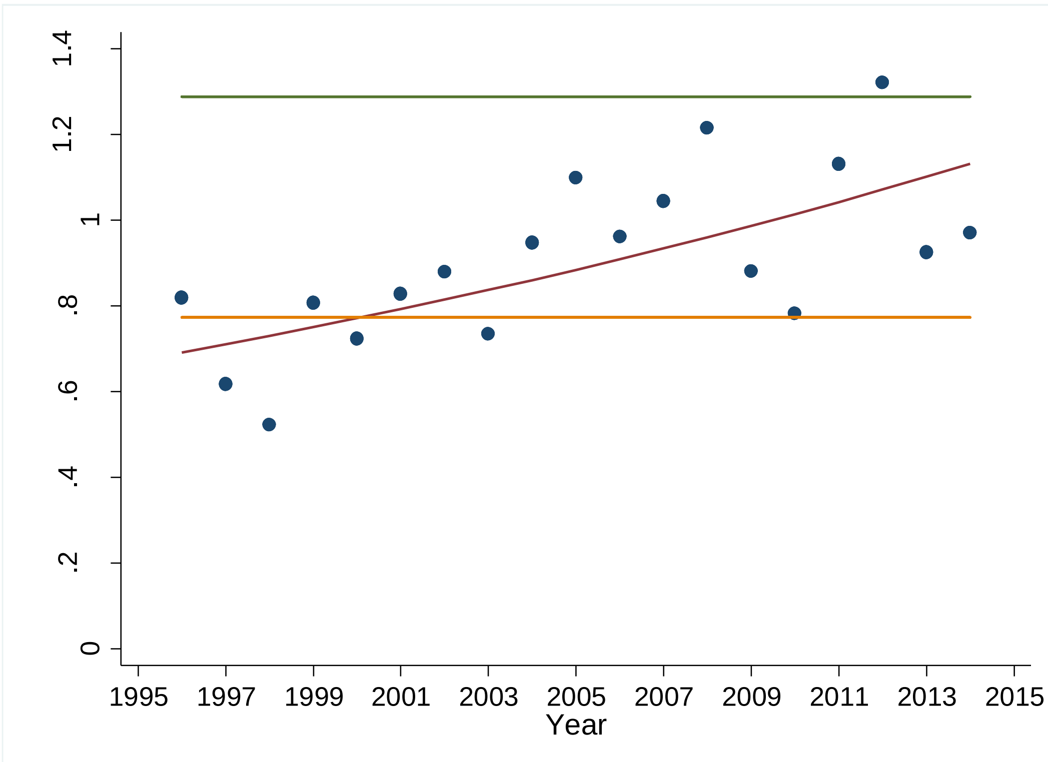

Figure 1. Density index values, combined over the 3 species, are displayed, as well as the line of best fit (shown in brown), a significantly increasing trend. All analyses were carried out on natural-log-transformed counts per unit area; results have been back-transformed for display purposes. Note that the regression line is calculated using the back-transformed values. Also shown are the criterion distinguishing Good from Fair (i.e., benchmark value), shown in olive green, and the criterion distinguishing Poor from Fair (i.e., scoring criterion for Poor), shown in orange. See text for further details.

State of the Estuary Report 2015

Technical Appendix

WILDLIFE – Tidal Marsh Birds

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5. Technical Appendix

Background and Rationale

Population abundance of tidal marsh-dependent species has been used as an indicator of population health of sensitive species and thus of the condition of tidal marsh ecosystem in the San Francisco Estuary, dating back to the State of the Estuary 2011 Report and earlier (e.g., Goals Project 2000). Density is a particularly suitable metric, because it tallies the number of individuals detected or estimated to be present in the survey area, in relation to the area surveyed. Because in this study the same sites are sampled repeatedly over time, the statistical analysis of change in the density over time is facilitated. Tidal marsh bird density has been evaluated at multiple tidal marsh sites every year since 1996 by Point Blue Conservation Science (Spautz et al. 2006).

The three species comprising the tidal marsh bird indicator are represented in the estuary by subspecies that display unique adaptations to tidal marsh habitat, but are, at the same time, of conservation concern, being either State-Threatened or California Species of Special Concern. A fourth species, the Ridgway's Rail, is tracked with its own indicator in the State of the Estuary Report because standardized results for this species are only available since 2005, whereas the three species used here have been monitored since 1996.

Benchmark

The benchmark value chosen is a density for tidal marsh birds that we consider to be a desirable target. The value chosen is the 75th percentile among all marsh sites studied during the period 1996-2008. The 75th percentile value corresponds to the median value for the upper half of all marsh sites surveyed during the reference period. We then used the most recent four years (2011-2014) to provide an assessment of current condition relative to the benchmark value.

The rationale for choosing a target density for the benchmark is that density of tidal marsh bird species reflects, in part, habitat suitability and may also reflect efficacy of management actions designed to support healthy tidal marsh populations. We therefore expect that improvements in habitat suitability, including maturation of newly restored tidal marsh habitat, should be reflected in an increase in the density of tidal marsh bird species.

Data Sources and Methods

Data for the indicator are from standardized, avian tidal marsh surveys conducted by Point Blue Conservation Science since 1996 (Nur et al. 1997, Spautz et al. 2006, Stralberg et al. 2010, Wood et al. 2012). Field methods for avian surveys in tidal marsh habitat are described in the above-listed references.

Briefly, multiple survey stations have been established within each marsh site surveyed. All individuals detected within 50 m of the observer are enumerated and identified to species. Two surveys are conducted per station during the course of the breeding season. The number of individuals detected per species is averaged over the two survey visits in each year. The analysis is then conducted at the individual survey station level or, instead, the number of individuals are averaged over all stations in a marsh and average density per marsh station per year is analyzed.

The latter approach is used here. Note that we divide the number of individuals detected by the area surveyed, thus yielding a density estimate.

We analyze each species separately, specifically the number of detections per unit area, natural-log transformed, for each marsh in each year. We statistically modeled the variation in density among years for all three species. Our models included “marsh site” as a main effect. That is, we estimated year to year change in tidal marsh bird density (fitting a model in which “year” was a categorical variable, or factor), while also statistically adjusting for marsh site, treated as a fixed effect. Adjusting for variation in density among marsh sites improves our ability to estimate the annual variation in density common to all sites. We used the `margins` command in STATA 13.1 (StataCorp.) to obtain the estimates of annual change in the density index, derived from the statistical model.

To obtain a three-species combined metric we first calculated the geometric mean of density across the three species, which is the back-transformed value with respect to the mean of the ln-transformed, model-derived values across the three species. We then multiplied the geometric mean by three to represent the total estimated number of individuals per hectare among the three focal species (Black Rail, Common Yellowthroat, and Song Sparrow).

Trends across the time period, 1996 to 2014, were calculated using the ln-transformed density values; we analyzed each species by itself, and also analyzed the combined (geometric-mean based) three-species metric, calculated as described above. The magnitude and statistical significance of the trend are reported with respect to the analysis of ln-transformed density values (see Table A1, below). However, for illustrative purposes, the trend line shown in Figure 1 is based on the geometric mean, three-species density index values, rather than the ln-transformed values. Thus, results displayed in Figure 1 are scaled in terms of birds/ha rather than in log units.

Assumptions and Uncertainties.

The two areas of greatest concern are: 1) We were not able to estimate detection probability independently of abundance. Hierarchical statistical models to separately estimate detection probability and abundance could not also incorporate marsh-specific variation in abundance. Hence we use the number of individuals detected as a proxy for abundance, and thus, density. Thus, we do not know if some of the annual variation in density is due to variation in detection probability. That said, we have no evidence that this is the case. 2) The sample of marshes systematically surveyed is small: 10 marshes were included in this analysis. We assume that the ten surveyed marshes are representative of the larger set of tidal marshes in the San Francisco Estuary, but that has not been confirmed. A greater sample size of marshes is needed, which requires sufficient funding to accomplish that objective.

Results

The three-species density index results are described above, section 3. Here we describe results for the individual species in more detail.

Species-specific trends are summarized in Table A1. Black Rail and Common Yellowthroat displayed significantly increasing tends in density, exceeding 3% per year for both species (P = 0.012, P = 0.016, respectively). The observed trend for Song Sparrows was positive but represented a very modest increase (1.1% per year), and was not statistically significant. However, the Common Yellowthroat trend exhibited significant down-turning (i.e., the quadratic coefficient was significantly negative, P = 0.021). Thus, the increasing trend has not continued in recent years.

Appendix, Table A1: San Francisco Estuary Tidal Marsh Bird Index
Trends 1996 to 2014 for three tidal marsh bird species
Estimated annual rates of change in the index and significance of the trend shown

	Trend Coefficient	S.E.	Annual Pct Change	P-value
Species				
Black Rail	0.0386	0.0137	3.94%	P = 0.012
Common Yellowthroat	0.0328	0.0123	3.33%	P = 0.016
Song Sparrow	0.0108	0.0078	1.09%	P = 0.18
Three Species, Combined Index	0.0274	0.0072	2.78%	P < 0.002

Peer-review and Acknowledgments

Methods for field collection and analysis used here have been published in several peer-reviewed scientific publications such as Spautz et al. (2006) and Wood et al. (2012). We thank many field staff for data collection over the years especially M. Elrod, L. Liu, and H. Spautz. Funding for analysis of the indicator presented here was provided by the Richard Grand Foundation. We thank the many agencies and land-owners for allowing access to tidal marsh study sites.

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State of the Estuary Report 2015

Summary

WILDLIFE – Ridgway's Rail

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State of the Estuary 2015: Wildlife
Ridgway's Rail Population Indicator

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Final October 2015

1. Background and Description of Indicator and Benchmark

The 2015 State of the Estuary includes two indicators reflecting the condition of biotic resources that specifically focus on tidal marsh habitat: Tidal Marsh Bird Population Indicator (which was included in the 2011 Report) and this Indicator, Ridgway's Rail Population Indicator.

a) Background and Indicator:

Extensive loss of tidal marsh habitat was well-documented in the Goals Project (1999). In addition, there has been substantial alteration of habitat, reflected in changes in salinity and channelization (e.g., reduction of sinuous, dendritic channels characteristic of mature tidal marsh habitat); habitat fragmentation; invasive plant and animal species; contaminants (e.g., mercury), and loss of natural transitional habitat bordering tidal marsh. These stressors are well-documented in the Tidal Marsh Recovery Plan (TMRP; USFWS, 2013) and the Bayland Ecosystem Habitat Goals Technical Update Report (2015; see especially Case Study by Overton and Wood 2014).

While all species relying on tidal marsh habitat are affected by habitat loss, alteration, and stressors, California Ridgway's Rail, formerly California Clapper Rail, is an especially sensitive subspecies because of their use of low marsh for foraging, their dependence on channelized marshes, and their use of mid-marsh and upper marsh areas for nesting and as refugia from predators, especially during extreme tides (Overton and Wood 2014). Furthermore, the California Ridgway's Rail (*Rallus obsoletus obsoletus*) in the San Francisco Estuary is of much reduced population size compared to the 1970's, and its numbers may currently be in the vicinity of 1000 individuals (Liu et al. 2012), thus subjecting this subspecies to stochastic fluctuations which may lead to local extirpation or inhibit recovery.

At the same time, the California Ridgway's Rail, as a Federally Endangered subspecies, has been the focus of extensive tidal marsh restoration efforts throughout the San Francisco Estuary (see TMRP), as well as a number of management activities. Hence, the Ridgway's Rail Population Indicator serves an important function, reflecting both the condition of the suite of species that depend on tidal marsh habitat, as well as being an indicator of a species of intrinsic significance with respect to its current viability and how this has changed, and will change, over time with respect to restoration and management activities.

The specific indicator metric is the density of birds per hectare as determined from comprehensive, standardized breeding season surveys conducted throughout the San Francisco

Estuary since 2005 (Liu et al. 2012). Surveys have been conducted by the Invasive Spartina Project (of the State Coastal Conservancy), Point Blue Conservation Science, USFWS, USGS, CDFW, EBRPD, and others. The most recent, state-of-the-art analysis is that conducted by Point Blue for surveys carried out in 2005-2013, and is presented here.

b) Benchmark, Scoring Breaks and their justification:

We established a benchmark breeding population density separately for the North Bay and the South Bay populations, using results from the first three years that comprehensive standardized surveys for Ridgway's Rail were available, 2005-2007 (Liu et al. 2012). The 3-year mean density for each region (North Bay or South Bay) is used as the Benchmark. On this basis we then scored the index as Good, Fair, or Poor.

For distinguishing Good from Fair we use a 10% increase in density over the benchmark value. Thus, Good represents a modest improvement over the average density observed in 2005-2007. Considerable management effort is currently directed at improving habitat quality, including reduction of invasive species, reduction of predation and disturbance; this is in addition to the expectation that habitat quality for Ridgway's Rail will increase as recently restored tidal marshes (say about 20 years before the present) become more mature, and thus of increased suitability for Ridgway's Rail. Therefore, a modest increase in density can reasonably be expected as habitat conditions improve in the future relative to those of 2005-2007. Note that the TMRP sets a goal of increasing the total population per region by five-fold over 50 years; hence the target set here (10% increase in density) is relatively modest.

Benchmark means were 0.54 and 0.49, respectively for North and South Bay regions. Thus, the criterion of Good is a density of at least 0.60 birds/ha for the North Bay and 0.54 for the South Bay. Note that this criterion was reached in 2005 for the North Bay as a whole and in 2006 for the South Bay as a whole (Figure 1), which underlines the point that Good is a feasible target to achieve.

For distinguishing Poor from Fair we chose a value that represented a meaningful decrement relative to the same benchmark, i.e., the mean density for 2005-2007, calculated for each bay region. The value we use is 20% below the mean. A 20% decrease over 4 to 8 years (comparing 2011-2013 to 2005-2007) is of some concern especially if such a trend were to continue. Thus, the criterion of Poor is a density below 0.43 birds/ha in the North Bay and below 0.39 birds/ha in the South Bay. In summary, the range for Fair extends from 0.43 up to, but not including 0.60 in the North Bay, and from 0.39 up to, but not including, 0.54 in the South Bay (see Table 1).

Relative to the benchmark value, the two cut-points (Good vs Fair, and Fair vs Poor) are not symmetric. That is because a 10% decrease in density does not represent reason for concern, whereas a 10% increase in density represents a substantial achievement, one that is only attainable once management actions (e.g., predator reduction) have been implemented and/or newly restored habitat has become sufficiently mature to support relatively high densities of Ridgway's Rail.

2. Scoring Assessment and Trends:

In the **North Bay**, the mean density for the 3 most recent years is 0.47 birds/ha, and therefore this indicator is scored Fair (Table 1). Note that the mean for 2011-2013 is 13% below that of 2005-2007. Over the entire period 2005-2013 there has been on average a slight decline of 2.4% per year (NS, $P > 0.5$). However, there has been a significant "bottoming out" during this period: the quadratic curvature is significant ($P = 0.016$), demonstrating an upturn starting in about 2009. In other words, most of the decline in density observed between 2005 and 2008 has been reversed.

In the **South Bay**, however, the mean density for the 3 most recent years is 0.23, and therefore this indicator is scored Poor. Note that the mean for 2011-2013 is 53% below that of 2005-2007. Over the entire period 2005-2013, there has been a significant decline of 10.8% per year, on average ($P = 0.025$). During this period, the quadratic trend was not significant ($P > 0.14$). Thus, there was little statistical evidence that the declining trend had been reversed as of 2013.

To summarize, the density of Ridgway's Rail declined in the North Bay between 2005 and 2008, but since then the trend has mostly been reversed. In the South Bay, density declined between 2006 and 2008, and apparently into 2009. Though no further decline has been seen since 2009, neither is there clear evidence of a reversal.

It is important to note that these results are only through 2013. A full analysis of surveys completed in 2014 and 2015 is needed, but note that 2015 was characterized by reduced survey effort. A full survey effort in 2016 and comprehensive analysis of 2014-2016 survey data are needed.

3. Brief Write-Up of Scientific Interpretation:

The goal of the Tidal Marsh Recovery Plan (USFWS 2013) is to increase the current population size of Ridgway's Rail in the San Francisco Bay Estuary to approximately 5500 individuals over a 50-year period. The best recent estimate (for the period 2009-2011) for this region is fewer than 1200 individuals (Liu et al. 2012). To meet the TMRP's ambitious goal will require both an increase in the overall density of Ridgway's Rail in current tidal marsh habitat as well as an increase in the total amount of tidal marsh habitat, as a result of restoration of habitat and tidal action (as described in the current NFWF Business Plan for Tidal Marsh. The Ridgway's Rail Population Indicator reflects the first component: density of breeding populations. The SOTE tidal marsh habitat Indicator reflects the second component.

Changes in density reflect several factors, including (1) habitat suitability for this species, in particular suitability for nesting, foraging, and refugia from extreme tides, and (2) the impact of stressors such as invasive species, disturbance from humans, excess predation, inundation of habitat, and contaminants. One aspect of habitat quality to highlight is vegetation structure, which plays an important role for Ridgway's Rail, providing cover, refugia from predators, and suitable locations for successful nesting. This is one reason for current management actions and activities (conducted by SCC and USFWS) directed at planting or maintaining important plant species, in particular gumplant (*Grindelia stricta*) and native cordgrass (*Spartina foliosa*).

The time series presented here indicates two distinct trends in Ridgway's Rail density. For the North Bay, a decline in density was observed up to 2009, followed by an increase in recent years, back to a level comparable to that observed in 2006-2007. For the South Bay, a decline began in 2007, was especially steep between 2007 and 2008, with no further decline since 2009, but also no appreciable recovery observed as of 2013.

The declines observed in both regions reflect multiple causes. An important contribution to the decline, especially in the South Bay, was loss of intact vegetation structure as a consequence of the large-scale removal of invasive *Spartina* (especially the *alterniflora x foliosa* hybrid) during the period 2006 to 2010. The peak of non-native *Spartina* infestation was observed in 2005 through 2007, with high levels observed in 2005 and 2006 and, to a lesser extent, 2007. In some cases where hybrid Spartina invaded mudflats and converted the open tidal flats to a monoculture, cordgrass marsh, the removal of invasive *Spartina* also removed all available Ridgway's Rail habitat. For those areas below Mean Higher High Water, where tidal mudflat is the native condition, recovery to 2005-2007 levels may not be realistic (J. McBroom, pers. comm.). Instead, an increase in bay-wide rail numbers to 2005 – 2007 levels will require time for the current habitat restoration efforts to provide mature native tidal marsh, in addition to management actions targeting Ridgway's Rails.

While South Bay populations demonstrated a sharp decline, coincident with invasive *Spartina* eradication efforts, in the North Bay there was also a substantial decline, though there was much less eradication in the North Bay, due to reduced infestation by invasive *Spartina*. Thus, the large bay-wide decline in California rail detections in 2008, including areas unaffected by invasive *Spartina* eradication, highlights the sensitivity of this species to annual variation in ecological conditions.

The partial recovery of density in recent years in the North Bay merits "fair" status, indicating that further improvement is needed and can be expected as the impact of stressors on the population are reduced (USFWS 2013). The as-yet lack of recovery of density in the South Bay highlights the need for management actions to improve habitat suitability and population viability. One example of such actions is the planting of important species for Ridgway's Rail, *Grindelia stricta* and *Spartina foliosa*; another area to address is provision of refugia from high tides, which is also being pursued by SCC, USGS, and others. There is also a current focus on improvement of the transition zone between marsh and upland habitat. Reduction of predation (e.g., due to cats) is yet another action under consideration. We can expect that implementation of such management actions will result in an increase of the Ridgway's Rail population density in future years, and that this will be tracked by the Ridgway's Rail population Indicator.

In addition to specific management actions, as discussed above, further information is needed regarding survival and reproductive rates of Ridgway's Rail, and the factors that directly influence these demographic rates (Overton and Wood 2014). Completion of full survey efforts in 2016 (after a reduced effort in 2015) is needed as well as a complete analysis of survey data since 2013.

Table 1.

Benchmark and Indicator Results for Ridgway's Rail Population Indicator

Region		Explanation	Value	Score
North Bay				
	Benchmark	2005-2007 Mean for region	0.54	
	Cutpoint Good vs Fair	10% above benchmark	0.60	
	Cutpoint Fair vs Poor	20% below benchmark	0.43	
	Observed value	2011-2013 Results	0.47	Fair
South Bay				
	Benchmark	2005-2007 Mean for region	0.49	
	Cutpoint Good vs Fair	10% above benchmark	0.54	
	Cutpoint Fair vs Poor	20% below benchmark	0.39	
	Observed value	2011-2013 Results	0.23	Poor

4. Related Figures.

Indicator results are displayed in Figure 1.

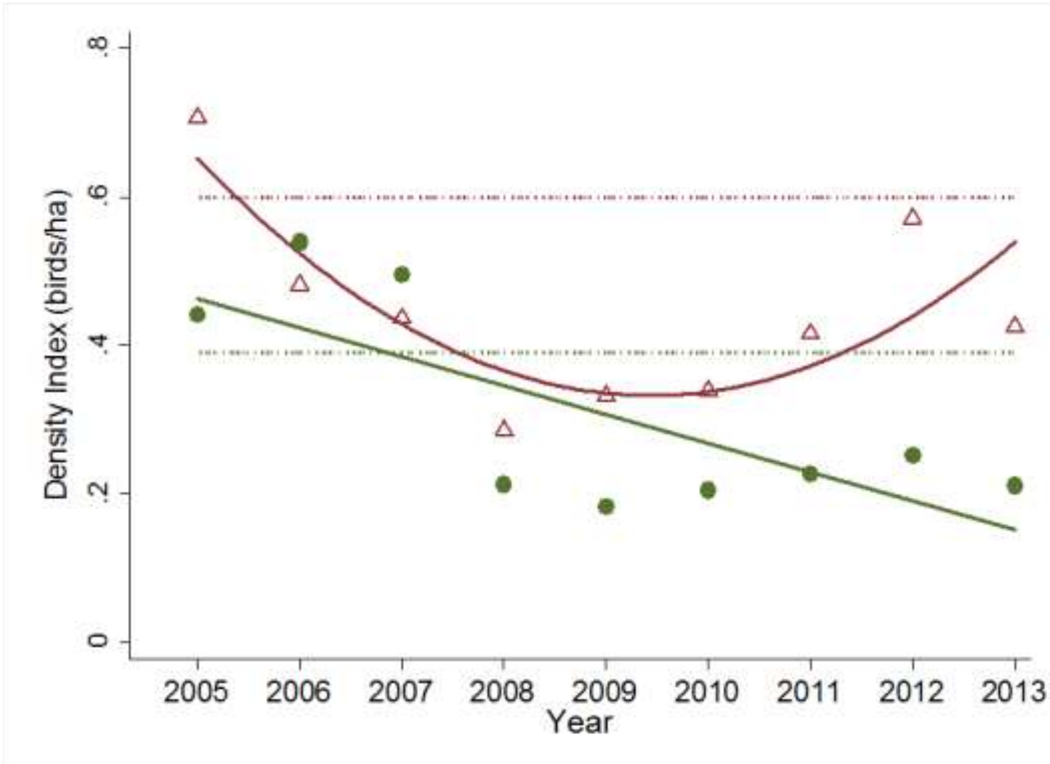

Figure 1. Density Index of Ridgway's Rail, from comprehensive, standardized surveys, conducted by multiple partners throughout San Francisco Estuary. The methodology is a modification of Liu et al. (2012), as well as the addition of results from 2012 and 2013 surveys (see Technical Appendix). North Bay estimates (triangles) and trend of best fit (quadratic) shown in maroon; cutpoint "Good/Fair" for North Bay is shown as dotted maroon line. South Bay estimates (green) and trend of best fit (linear) shown in green; cutpoint for "Fair/Poor" for South Bay is shown as dotted green line.

State of the Estuary Report 2015

Technical Appendix

WILDLIFE – Ridgway's Rail

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5. Technical appendix

Background and Rationale

Background and Rationale for the choice and interpretation of this indicator is provided above, in section 1.

Benchmark

The benchmark, its calculation, and its justification are described in detail above, see section 1.b. In brief, the benchmark chosen was a 10% increase in density relative to the 2005-2007 period, which is the earliest period available for comparison. For scoring Poor vs. Fair, the scoring break is a 20% decrease.

The primary basis for choosing a 10% increase in density is that the Tidal Marsh Recovery Plan has set ambitious goals for the recovery of the California Ridgway's Rail (USFWS 2013). Specifically, the TMRP goals are an approximate five-fold increase in total population size over 50 years. Achieving these goals will require both an increase in density as well as an increase in the total amount of habitat available for the subspecies through habitat restoration. A 10% increase relative to 2005-2007 may not appear to be a substantial increase, but note that density levels dropped by about 50% in the North Bay and South Bay regions, comparing 2008-2009 to 2005-2007 (the reference period). Hence, achieving a 10% increase above the 2005-2007 levels represents a significant accomplishment and thus merits a score of "Good."

- **Limitations of the benchmark and possible improvements**.

Density alone does not provide a complete measure of condition. Compiling information on reproductive success and/or survival will be essential in the future. The criterion of 10% increase relative to 2005-2007 should be scaled to the time frame. Thus, for the 2019 State of the Estuary Report, a new criterion may need to be used, e.g., by choosing a greater percent increase for the current period relative to 2005-2007.

Methods and Data Sources

The data and methods used in calculating indicator values are for the most part described in Liu et al. (2012). The data used here updates Liu et al. (2012) by including results from all available surveys in 2012 and 2013. Data were provided by Point Blue collaborators, including Invasive Spartina Project, USFWS, and others (Liu et al. 2012).

In addition, the methods are slightly modified as described herein. In this analysis, we analyzed call count data during the Ridgway's Rail breeding season period (15 January to 15 April) from 123 "transects," where a transect generally corresponded to a local marsh site, and consists of multiple survey stations (often six to 10, depending on marsh size).

The number of survey visits to each survey station was usually three (sometimes four or five) per breeding season. Because of the multiple visits, we were able estimate detection probability in relation to date (i.e., Day of Year), fitting a quadratic effect, see below (detection probability peaks in February, and then declines).

For the analysis, we excluded those transects where Ridgway's Rails was never detected in any year, at any of the survey stations for a transect. Such transects were assumed to be in either unsuitable habitat for Ridgway's Rail, or outside the current range. We fit hierarchical imperfect detection models, allowing for failure to detect a Ridgway's Rail, given that one was present, and examined covariates that may be affecting detection probability (see Liu et al. 2012 for details).

We compared multiple competing models and chose the best model based on comparison of AIC values, evaluation of negative log-likelihood values, and model coefficients. The best model for the Ridgway's Rail analysis included quadratic effects of Julian day and difference in time since sunrise or sunset for the detection function. For the abundance function (a component of the hierarchical model), the model chosen had only year and Bay effects (and their interaction). That is we allowed year to year differences to be estimated independently for the North Bay and South Bay regions. Estimates of density and associated confidence intervals were calculated by profiling the values of the detection model estimates and calculating the mean of the distribution of the abundance parameter (i.e., density), for each combination of year and bay, incorporating random effects (Royle and Dorazio 2008).

Assumptions and uncertainties.

The strength of our approach is that we statistically estimate two components determining number of Ridgway's Rails detected per survey: detection probability and the true, underlying abundance. Parameters affecting detection probability included date of the survey as well as time before or after sunrise/sunset. However, one difficulty, and thus source of uncertainty, is that any errors in estimating detection probability will also affect our estimates of abundance.

The second issue to consider is the representativeness of our sample. However, because the sample of sites surveyed was large in almost all years, lack of representativeness is not likely a major problem. The exception was 2013; due to budgetary constraints the sample of sites surveyed in 2013 was relatively small. Thus, it is difficult to ascertain how 2013 may have differed from 2011 and 2012, or how similar it may have been.

However, additional surveys were conducted in 2014 and 2015 (not analyzed here) and an extensive survey program is planned for early 2016. Analyses of 2014-2016 will go far to clarify the current status of Ridgway's Rails.

Peer Review

We thank Jen McBroom and Cory Overton for providing helpful review of the Indicator and this Summary and Technical Appendix.

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State of the Estuary Report 2015

Sidebar Source Material

WILDLIFE – Three Delta Birds of Concern

Tricolored Blackbird: prepared by Hildie Spautz and Neil Clipperton, California Department of Fish and Wildlife; Bob Meese, University of California, Davis

Black Rail: prepared by Hildie Spautz, California Department of Fish and Wildlife

Sandhill Crane: prepared by Hildie Spautz, Julie Garcia, California Department of Fish and Wildlife; Gary Ivey, International Crane Foundation

State of the Estuary Report 2015 - Delta Birds of Concern Sidebar – Tricolored Blackbird

Authors: Hildie Spautz (CDFW), Bob Meese (UC Davis), and Neil Clipperton (CDFW).
Peer input: Randi Logsdon (CDFW) and Rob Doster (USFWS)

Sidebar Text

1. **Why is this topic important to ecological health of the Estuary?**

The Tricolored Blackbird (*Agelaius tricolor*), a species found almost entirely in California, was given emergency protection as an Endangered species under the California Endangered Species Act in 2014. Protection was triggered by an extremely low statewide population estimate of 145,000 birds - a 63% decline since 2008. Recent threats to the Tricolored Blackbird population include loss of foraging habitats in the San Joaquin Valley, catastrophic nesting failures associated with the harvest of agricultural field nesting substrates, and high predator populations.

The Tricolored Blackbird is a colonial breeder and an indicator of the availability of its key habitat requirements and appropriate agricultural practices. During the breeding season, it requires nesting vegetation (primarily wetlands, grain fields, Himalayan blackberry, or weedy fields) adjacent to highly productive foraging areas that furnish the insect resources to support large breeding colonies, and water for drinking and bathing. The species appears to have evolved to take advantage of ephemeral resources, including insect outbreaks and recently disturbed wetlands that exhibit vigorous new growth for nesting. Due to its colonial nesting habit (with 100's to 1000's of pairs breeding closely together in a single field), crop harvest or weed abatement activities that affect the largest colonies may significantly reduce the annual reproductive output for the entire species.

Records of Tricolored Blackbirds breeding in the Delta are few; although, a few relatively large colonies have been documented in Suisun Marsh and around the periphery of the Delta in the recent past. Delta sites documented include West Sacramento, near the Port of Sacramento, the Yolo Bypass Wildlife Area, and Tracy. There were no breeding colonies found in the Delta during the 2014 Statewide Survey of the species. Given what we know about their habitat affinities, the species was likely abundant in portions of the vast freshwater wetlands of the historical Delta that were adjacent to productive uplands (i.e., primarily around the periphery of the Delta).

The species is most abundant in the Delta in winter, when it forms huge foraging flocks with other blackbird species and eats primarily grains found on agricultural fields or provisioned for livestock. The importance of the Delta to wintering populations, relative to other parts of California, and the effects of blackbird control efforts to limit grain harvest losses on the population are poorly documented and need further study.

For more information about the Tricolored Blackbird, its habitat requirements, and conservation status, see The Tricolored Blackbird Portal.

2. **Why is it not being developed as a quantitative indicator in 2015?**

It is not being developed as a quantitative indicator for the Delta because there are few recent records of nesting colonies in the Delta, although there are long-term records of nesting on the periphery of the Delta. The species has recently been intensively surveyed statewide, targeting known nesting locations, so the interior of the Delta was not intensively surveyed. Currently, the species is most common as a winter resident and the Delta may be a core wintering area.

3. **What is likely to happen regarding this topic in the future – is it likely to become an indicator? If not, than what are the next steps?**

The Tricolored Blackbird is a covered species under several existing and proposed regional Habitat Conservation Plans and may be a rare Delta breeder due to poor vegetation management. As Delta wetlands are restored and managed specifically for Tricolored Blackbirds, the numbers may increase in the Delta as required resources are provided. Management actions should stress the importance of young, lush, rapidly-growing wetland vegetation for nesting birds and provide wetlands with adjacent uplands for foraging. Additional research is needed to assess the importance of the Delta to wintering birds, and to assess whether a wintering population indicator would be appropriate.

California Black Rail – Delta Region –Sidebar content - State of the Estuary Report 2015
Main Author: Hildie Spautz, CDFW
Reviewers: Danika Tsao, Daniel Burmester, Nadav Nur, Juliet Lamont, Sarah Estrella, Randi Logsdon, Jules Evens, Orien Richmond, and Anitra Pawley.

FINAL – April 3, 2015

- **Why is this topic important to ecological health of estuary?**
  California black rail (*Laterallus jamaicensis coturniculus*) is a small (house sparrow-sized), secretive bird species found in wetlands in widely disparate portions of California and the southwestern U.S. It is listed as Threatened under the California Endangered Species Act due to population declines caused by habitat loss.

  The largest black rail population in California is in the SF Estuary where it is found primarily in San Pablo and Suisun Bay tidal marshes (see SF Bay Tidal Marsh Bird Indicator) with a smaller outlying sub-population inhabiting tidal and non-tidal wetlands in the Delta. There are also small populations in tidal wetlands at Tomales Bay, Morro Bay, and Bodega Bay, along the Colorado River, in the Imperial Valley, and in marshes associated with seeps and irrigation ditches in the Sierra Nevada foothills.

  The black rail is an indicator of the availability of good quality emergent wetlands, including tidal marsh habitat with an adjacent upland transition zone. Black rails require wet areas with shallow water (generally < 3 cm) and dense vegetation close to the ground for nesting. In tidal marshes, the species also requires adjacent higher elevation vegetated areas (high marsh and upland transition zone) for refuge during high tides when they are most vulnerable to predation.

  Breeding habitat requirements in the Delta are poorly understood, and are based on locations of black rails in a few relatively small patches of remaining tidal and non-tidal wetlands with emergent vegetation including tules and cattails. They were likely present in the historical Delta at the upper edges of tidal marshes and in other perennial wetlands with shallow water. During recent Delta surveys, black rails have been documented using the largest available mid-channel islands with mature tidal marsh, non-tidal marshes, and most recently, along tidal channels at newly-restored Lindsey Slough (Hastings Tract, Solano County). Because of their elusive and cryptic natures, and apparent ability to colonize new areas quickly, they may more abundant in the Delta than we currently think.

  We need to know more about black rails' habitat needs, particularly those associated with successful breeding, to ensure that wetland restoration projects can be appropriately designed to benefit the species. In the future, the black rail is likely to be an important indicator of successful tidal marsh habitat restoration in the Delta, as it is in San Francisco Bay.

- **Why is it not being developed as a quantitative indicator in 2015?**
  California black rail is not being developed as a quantitative indicator in 2015 due to lack of consistent and recent survey data. Surveys have been conducted in portions of the Delta at various times since 1970. In 1992 – 1993 and in 2009 – 2011, limited areas of the Delta were surveyed, but not in a systematic fashion; moreover, the density and size of the Delta population has never been calculated. We need at least four years of data to estimate current population status and trends.

- **What is likely to happen regarding this topic in the future – is it likely to become an indicator? If not, than what are the next steps**?
  To develop a California black rail indicator for the Delta, baseline surveys need to be conducted with the express purpose of establishing density and distribution and total population size. As habitat restoration proceeds, black rail surveys should be repeated at regular intervals to determine appropriate restoration design criteria and to establish trends.

Fig 1 California Black Rail in pickleweed. San Pablo Bay. (photo credit: D. Tsao, USGS)

SOTER 2015 - Delta Sidebar – Sandhill Crane

Authors: Hildie Spautz, Julie Garcia, Gary Ivey.
Peer input: Randi Logsdon, Daniel Huang, Anitra Pawley, Daniel Burmester, Nadav Nur, Juliet Lamont, Kristal Davis-Fadke.

Sidebar Text

1. **Why is this topic important to ecological health of the Estuary?**

A classic example of "charismatic megafauna", the sandhill crane ("crane", *Grus canadensis*) attracts a myriad of visitors to the Sacramento-San Joaquin Delta each year. The Delta supports approximately one-third of the cranes wintering in California, including both the greater sandhill crane (*G. c. tabida*), listed as Threatened under the California Endangered Species Act, and lesser sandhill crane (*G. c. canadensis*), a California Species of Special Concern.

Historically, cranes were distributed much more widely throughout the San Francisco Estuary than they are now: they used coastal tidal marsh in the SF Bay, and shallowly flooded wetlands throughout the Delta and Central Valley, wherever water levels were right and vegetation was short - likely seasonally inundated wetlands, and edges of tidal marsh. Most of this historical habitat has been lost, and disturbance makes much of the remaining habitat unsuitable. Today, we know that cranes require shallowly flooded, undisturbed night roost sites (usually protected wetlands or flooded croplands) and forage in adjacent agricultural fields, primarily post-harvest corn and rice. Conversion from seasonal row crops to incompatible uses (e.g. vineyards, orchards, and residential areas) has resulted in a loss of valuable foraging habitat for cranes. Due to concern over habitat loss, agencies and conservation groups have acquired, protected, and enhanced Delta lands specifically for use by cranes.

2. **Why is it not being developed as a quantitative indicator in 2015?**

The data currently available for sandhill crane populations in the Delta are not sufficient to assess trends required for a quantitative indicator.

In the winter of 2007-2008 there was a comprehensive survey of cranes wintering in the Delta, which estimated a maximum of 27,213.

The state-wide Mid-Winter Waterfowl Survey has produced estimates of Delta cranes since the 1950s, but because the survey was not designed to assess cranes, these estimates are not sufficiently accurate, and do not distinguish subspecies. Winter crane surveys have been periodically conducted at Staten Island and other reserves in the Delta; however, these data are insufficient to evaluate crane population trends on a Delta-wide scale.

3. **What is likely to happen regarding this topic in the future – is it likely to become an indicator? If not, than what are the next steps?**

Two documents released in 2014: *Conservation Priorities and Best Management Practices for Wintering Sandhill Cranes in the Central Valley of California* and the *Coastal California Waterbird Conservation Plan* include recommendations to develop survey methods to estimate winter populations of both subspecies.

Several local Habitat Conservation Plans, approved and in progress, propose to restore and preserve habitat for the Greater sandhill crane in the Delta, and their proposed monitoring strategies focus on the crane habitat specifically preserved or enhanced by those conservation plans.

A Delta wintering crane indicator will be possible when there are regular sandhill crane population surveys throughout the entire Delta, using methods specifically designed to accurately assess cranes.

State of the Estuary Report 2015

Sidebar Source Material

The Ocean Connection – Warm Anomaly in Ocean 2014-2015 (aka "the blob")

Prepared by John Largier, University of California, Davis Bodega Marine Laboratory; Nadav Nur, Point Blue Conservation Science; Sarah Allen, Ocean and Coastal Resources Program, National Park Service.

Warm Anomaly in Ocean 2014-2015 (aka “the blob”)

SOTER 2015 Sidebar

The surface water of the northern Pacific has been anomalously warm for over a year – developing in concert with California’s drought. Far offshore from Oregon, Washington, British Columbia, and the Gulf of Alaska, this warm water mass is up to 2^{o}C above normal temperatures and extends to 100m depth. Affectionately known as “the blob”, it is thought to result from milder winters and less cooling due to a persistent high-pressure atmospheric ridge in the region. The anomalous conditions are very widespread, with related features in the remote Bering Sea and also off southern California and Baja California. In general, the coastal waters of central and northern California have been less affected than offshore waters due to wind-driven upwelling. But not so in late 2014 and early 2015 – although normal cold temperatures and upwelling were observed off San Francisco Bay until June 2014, in July there was a sudden increase in temperatures and a shut down of upwelling winds. With this, the coastal currents turned northward and even warmer water and plankton were transported into the region from the south. From July to December 2014 water over the shelf was 3^{o}C warmer than normal (and the anomaly peaked at 4^{o}C last September – exceeding even the strongest El Niño events). Warm water anomalies of 1-2^{o}C persisted through winter, until April 2015, when coastal upwelling returned with enough strength to bring temperatures back to normal (for now).

Given that cold upwelled waters supply the nutrients for plankton productivity, which supports the continental shelf ecosystem off San Francisco Bay, this warm anomaly represents a major disruption of the food web. Also, there is anomalous northward water transport and air temperatures have remained high – specifically, February air and sea surface temperatures at the Southeast Farallon Islands were higher in 2015 than in any of the previous 45 years. Ecological changes that are likely associated with this anomaly include the disappearance of krill and juvenile rockfish from seabird diets, species occurring north of their typical range, and starving sea lions and fur seals. Species normally associated with sub-tropical waters moved north off central California, with common dolphins being seen off Bodega Bay and starving Guadalupe and northern fur seals off Point Reyes. During the 2014/15 winter on the Farallon Islands, seals and sea lions had difficulty reproducing and finding food, locally breeding seabirds had low colony attendance, and two tropical species of seabird showed up on the island.

With the return of upwelling in April, Cassin’s Auklets and other seabird nesting species were laying eggs as they do in spring each year. However, poor salmon catches off San Francisco and Bodega Bay may signal that the ecosystem has not fully returned to normal. Further, while anomalous conditions may have departed from the coastal zone for the 2015 upwelling season, the blob persists offshore – both north towards the Gulf of Alaska and south off Baja and southern California. It is likely that late summer and fall will see the return of anomalous conditions, with the prospect of conditions being exacerbated by a concurrent El Nino that is developing in the equatorial Pacific. Again in late 2015 and through the subsequent winter, we may see a severe squeezing of the cool habitat for temperate marine species that normally occur off central/northern California, likely resulting in mortality of juveniles and possibly adults.

State of the Estuary Report 2015

Summary

PROCESSES – Migration Space

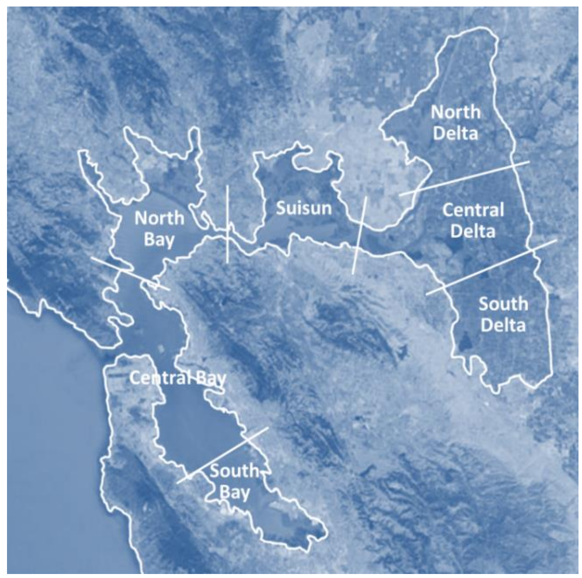
Prepared by
Josh Collins
San Francisco Estuary Institute

Migration Space Indicator (V3 June 30 2015)

Brief Description of Indicator and Benchmark

Migration space is the upland area between the present-day shoreline of the Estuary and a higher, future shoreline resulting from sea level rise. This report considers two alternative migration spaces, based on the assumption that the Estuary rises either two feet or five feet. Both of these rises in sea level are possible during this century. Migration space excludes all existing tidal areas as well as any reclaimed areas, such as salt ponds in South San Francisco Bay or diked farmlands in the Delta that would be flooded without their dikes or levees. However, migration space includes all areas of landfill within the historical limits of the Estuary that are above the future shorelines. The total area of migration space is due mainly to the slope of the land immediately adjacent to the Estuary. The space is widest across broad, gently sloping valleys and plains.

This indicator measures the current percentage of undeveloped space, and the percentage of that space that is protected from development. This indicator is based on the need to protect and restore the zone of natural transition from estuarine habitats to terrestrial habitats that is critically important for the ecological and economic health of the Estuary. The indicator has been estimated for each major sub-region of the Estuary.



Sub-regions of the Estuary.

There are no existing benchmarks for migration space. The benchmarks are arbitrarily 50% of the total migration space in each sub-region being undeveloped, and 75% of that undeveloped space being protected. The scoring break between fair and poor scores is arbitrarily set at 40% and 50%, respectively.

Indicator Status and Trend Measurements

Much of the commercial, industrial, and cultural resources of the Estuary are associated with its shore. Shorefront businesses contribute great wealth to the region. The shoreline adjoins the airports, railroads, and highways that are vital to domestic and international commerce.

These uses of the shore have historically overridden concerns for the natural benefits provided by its undeveloped areas. But, there is a growing appreciation that the natural transition zone beautifies the Estuary, supports much of its ecological diversity, and provides abundant recreation. It contributes substantially to the quality of life in the region.

While appreciable amounts of undeveloped migration space exist in some sub-regions, most of the space around the Estuary has been developed, and only a small percentage of the undeveloped space is protected from future development. For the Estuary as a whole, the existing transition zone is not well protected, and opportunities to restore the transition zone are not abundant. Given that much less than half the total migration space is undeveloped, and that less than half the undeveloped space is protected, the overall condition of the migration space is considered poor.

Scientific Interpretation

The migration space indicator represents the ability for the shallow habitats of the Estuary, principally the tidal marshes and mudflats, as well as the associated terrestrial habitats, such as grasslands and forested hillsides, to migrate inland as sea level rises. The shallow estuarine habitats help protect the shore against erosion and flooding due to storm surges or erosive waves generated by high winds. Without protected, undeveloped migration space, the Estuary will rise against the developed landscape, compressing the natural shore into a narrow band of vulnerable habitats with minimal cultural, economic, or ecological value.

The migration space indicator also represents the opportunity for native populations of plants and animals to track appropriate habitat conditions that are also migrating inland and upstream. The rising sea will cause saline conditions in the Estuary to move upstream in local watersheds and toward the Delta. Areas of healthy transition zone are needed in every sub-region of the Estuary to allow the associated plants and animals to migrate along with their required salinities.

The migration space indicator has never been calculated before. There are no data to quantify a trend in the percent of undeveloped migration space that is protected. The overall patterns of development in the region suggest that much of the migration space was developed during the latter half of the last century, before the advent of environmental regulations. Since then the rate of development of the migration space has likely lessened, although the quality of the remaining undeveloped space may be subject to continuing decline due to pollution, over use, biological invasion, and ecological isolation. Furthermore, there is generally more undeveloped space for the two-foot rise in sea level than for the five-foot rise. This reflects the pattern of urban encroachment toward the shoreline. It suggests that there will be less undeveloped space in the future than there is now. For either a two-foot or five-foot rise in sea level, very little of the undeveloped space is protected.

The challenge for the future is to protect the existing undeveloped space, create more of it if possible, and protect it from future development. There are opportunities to meet this challenge in every sub-region of the Estuary. It's noteworthy, however, that Suisun Bay has the most undeveloped migration space that is unprotected.

Further development of the migration space indicator should be guided by regional experts in land use, sea level rise and its landscape effects, and landscape ecology. There is a critical need to determine the geodetic elevation of the MHHW contour for the Delta. There is also a need to estimate the full extent of the transition zone around the Estuary, and to determine what migration space is needed to conserve the transition zone under different sea level rise scenarios. Scientifically sound criteria will be needed to identify and prioritize opportunities to conserve and restore the transition zone.

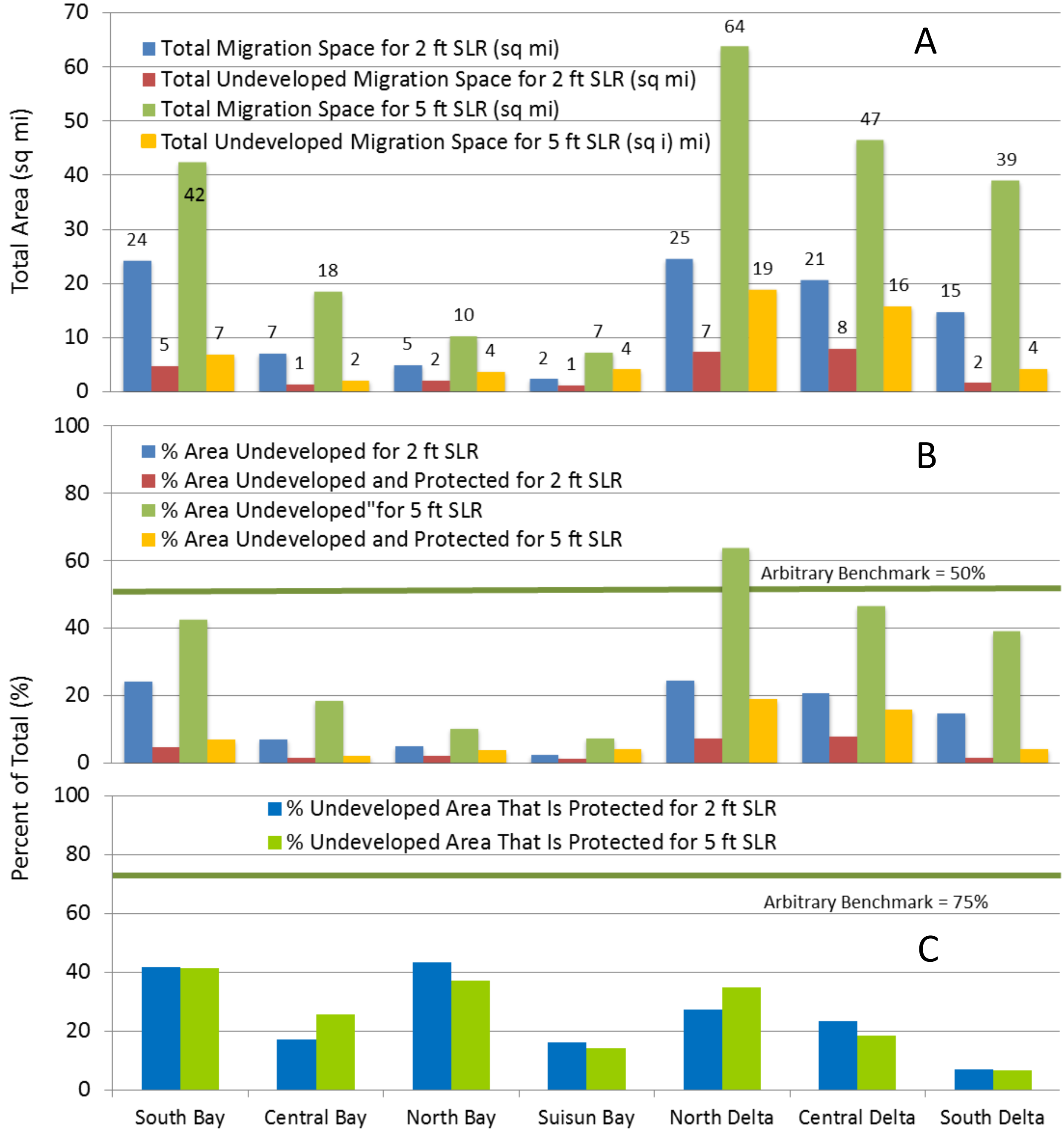


Figure 2. Sub-regional distribution of developed and undeveloped migration space for sea level rise (SLR) of 2-ft and 5-ft, showing (A) total migration space (sq mi); (B) percentage of total migration space undeveloped and protected, showing an arbitrary target value of 50%; and (C) the migration space indicator (i.e., the percentage of the total undeveloped migration space that is protected), showing an arbitrary benchmark of 75%.

State of the Estuary Report 2015

Technical Appendix

PROCESSES – Migration Space

Prepared by
Josh Collins
San Francisco Estuary Institute

Estuary Migration Space
Technical Appendix
Version 3.0
July 20, 2015

Background and Rationale

The purpose of this indicator is to provide an initial assessment of the scale of opportunity to conserve the natural ecosystem services of the estuarine-terrestrial transition zone by identifying undeveloped areas of migration space around the San Francisco Estuary (the Estuary) into which the transition zone can be allowed to evolve as sea level rises.

Definitions

The shorelines of estuaries have great ecological, economic, and cultural importance (Daily et al. 1997, NOAA 1999, NOAA 2008, BCDC 2011). They have been studied in detail from a variety of perspectives, resulting in particular terminology regarding their natural processes, functions, forms, and structures. The following terms are relevant to this report.

Accommodation Space. For an estuary with an unobstructed connection to the sea, the volume of space between two sea levels is its accommodation space (Jervey 1998, Posamentier and Allen 1999). As sea level rises in an estuary, it fills the accommodation space with sediment and tidal water. Changes in accommodation space are the result of one or more of three processes:

- Rise or fall in global sea level,
- Net sedimentation in the estuary, and
- Tectonic or seismic rise or fall in the floor of the estuary.

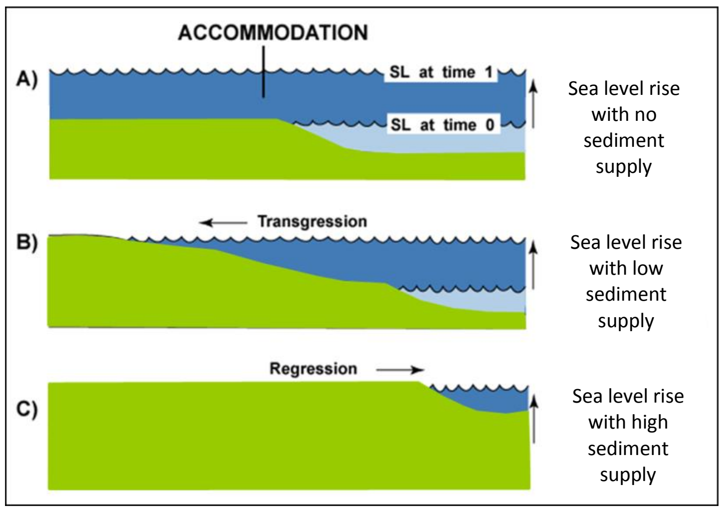


Figure 1. Three possible scenarios for natural changes in accommodation space due to interactions between sea level rise and sediment supply, showing (A) the space filled with water due to sea level rise without sediment input, (B) the accommodation space being filled by estuarine water and sediment (the same figure pertains to the estuary transgressing across former upland as sea level rises), and (C) an abundance of sediment causing the estuary to regress (after Posamentier and Allen 1999).

Interactions among these processes determine whether accommodation space increases, decreases, or remains the same. Earthquakes that raise or lower the floor of an estuary can suddenly and substantially alter its accommodation space (e.g., Gilbert 1907, Byrne et al. 2005). In general, however, the interactions between sea level rise and sediment accumulation regulate accommodation space (Figure 1).

Estuarine Transgression. This occurs when sea level rises relative to the land, causing the shoreline to move inland, and causing the head-of-tide (i.e., the upstream boundary of tidal effects in a river or stream) to move upstream (Pethnic 2000).

Estuarine Regression. This is the opposite of transgression. It occurs as sea level falls relative to the land, causing the estuary shoreline to regress or retreat (World Earth Science. 2003). Regression is common where rivers build deltas into estuaries. Artificial regression results from areas of an estuary that are reclaimed, causing its shoreline to move seaward. Reclamation in the Estuary has reduced its tidal area by about 98% in the Delta (SFEI-ASC 2014) and nearly 85% between the Delta and the Golden Gate (Goals Project 1999).

Transition Zone. The transition zone is defined as the spatial limits of the interactions between terrestrial processes, including runoff, and tidal processes that result in assemblages of plants and animals that are distinct from the adjoining estuarine, riverine, or terrestrial ecosystems (BEHGU 2015). The transition zone varies in width depending on topographic slope and land use constraints. At any given location and time, the width of the transition zone also varies with its function. It tends to be wider for ecological functions, such as support for wildlife, than for physical functions, such as shoreline erosion control.

Migration Space. For the purposes of this indicator, migration space is defined as the upland area landward of the historical shoreline of the estuary that would be flooded by the Estuary due to sea level rise in the absence of levees, dikes, or other water control structures. Migration space is therefore the landward component of accommodation space. The size of the space varies as sea level rises, and is affected by topographic slope and land use constraints. It does not include any existing tidal areas of the Estuary. Nor does it include any reclaimed areas, such as salt ponds in South San Francisco Bay (south Bay) or diked farmlands in the Delta, which would be flooded by tidal waters under the present-day sea level, if their levees or dikes were beached. However, it includes any areas of artificial fill within the historical limits of the Estuary that would remain above tidal waters for a specified sea level rise. This definition of migration space is consistent with the concept of marsh migration zone used elsewhere (Heberger et al 2009, Klausmeyer et al. 2013)

Why is migration space important?

The migration space is an area of great ecological, economic, and cultural importance. Under modest sea level rise, it encompasses many of the important historical landmarks and archeological sites of the Bay-Delta region, housing for more than 300,000 people, shorefront businesses of great monetary value, and ports that are vital for the economic health of the State. It also delimits the possible extent of the future transition zone. Industry, cultural heritage, and ecological diversity are concentrated along the shoreline, and are directly threatened by accelerated sea level rise (Gleick and Maurer 1990, Heberger 2012, Rodgers et al 2015, BEHGU 2015). Efforts to plan for sea level rise are largely focused on the migration space.

As sea level rises, the built landscape along the shoreline will need to be protected in place, or intentionally moved out of the way through managed retreat (sensu Townsend and Pethick. 2002). The ecological functions of the shoreline cannot stay in-place. They depend on the natural interplay among estuarine, terrestrial, and riverine processes that will move inland as sea level rises. They must be allowed to move into undeveloped areas of the migration space though natural migration or managed realignment (sensu Rupp-Armstrong and Nicholls 2007). The ecological health of the Estuary depends on providing adequate areas of undeveloped migration space to sustain an ecologically healthy transition zone (BEHGU 2015).

Overview of the Migration Space Indicator

The migration space indicator is a fundamental step toward a Bay-Delta regional tool to conserve and restore the natural ecosystem services of the transition zone. Efforts to plan for sea level rise are just beginning (BCDC 2011, ULI 2015) and the economic, technological, engineering, and scientific aspects are evolving rapidly. At this early stage of planning, three technical recommendations about migration space present themselves:

- Improve models for predicting estuarine flooding including extreme flood events;
- Model the landward extent of the transition zone for its various services; and
- Use the models of future estuarine flooding and transition zone extent, plus maps of land cover, habitat, and infrastructure to identify opportunities to conserve and restore the ecosystem services of the transition zone.

Implementing the latter recommendation involves overlaying maps of migration space onto maps of land cover. Mapping the migration space can be complicated by many factors, including local variations in sea level through space and over time (e.g., Knowles 2010, Holleman 2013), the technical difficulty of relating tide height to inland elevation (e.g., NOAA 2010, Kenny et al. 2011), the influence of topographic relief on flood pathways (e.g., Pelletier et al. 2005), and land use change that affects flooding patterns. Furthermore, the existing landscape will change as sea level rises. Developing models to predict estuarine flooding that account for landscape changes caused by the flooding will be an additional challenge.

A variety of scientific and technological efforts are underway to assess regional sea level rise (Table 1 below). Local analyses of migration space are also emerging (TNC 2013, Riordan Seville 2014). These efforts and others will evolve as the need for them becomes clearer and the related science and technology continue to advance. There are also efforts to coordinate these activities (e.g., Adapting to Rising Tides www.adaptingtorisingtides.org/; Lifting the Fog, http://coastaladaptation.org/liftingthefog/; Bay Area Ecosystems Climate Change Consortium, www.baeccc.org; Surging Seas http://sealevel.climatecentral.org/responses/plans).

As realistic models of future local estuarine flooding are being developed, and even afterward, more basic models will be useful to inform regional and local planning. At this time, none of the modeling efforts extends throughout the Estuary, although plans for that are pending. The migration space indicator presented here is an early step toward a Bay-Delta regional planning tool. There are important next steps that must be taken for the tool to better meet the need for planning and tracking migration space health (see Assumptions and Uncertainties below).

Table 1. Prominent efforts to model sea level rise or estuarine hydrodynamics for the San Francisco Estuary entirely or in part. Main source: Related Tools Comparison – California; http://sealevel.climatecentral.org/ssrf/related-tools-comparison-CA.

	Climate Central - Surging Seas Risk Finder	NOAA Coastal Services Center - Sea Level Rise and Coastal Flooding Impacts Viewer	Pacific Institute - The Impacts of Sea Level Rise on California's Coast	Cal-Adapt - Exploring California's Climate	Our Coast, Our Future	SUNTANS
URL	http://sealevel.climatecentral.org/	http://coast.noaa.gov/slr/?redirect=301ocm	http://pacinst.org/publication/the-impacts-of-sea-level-rise-on-the-california-coast/	http://cal-adapt.org/sealevel/	http://www.pointblue.org/outage.html	http://sourceforge.net/projects/suntans/
Purpose/ Description	Provides public multi-part web tool to help communities, planners, and leaders conduct a screening-level analysis of sea level rise and coastal flood risks, using 1) detailed searchable maps; 2) analysis of over 100 variables for 1000s of communities; 3) community comparisons; and 4) local sea level and flood risk projections.	A visualization tool for coastal communities showing potential impacts from sea level rise and coastal flooding as well as a planning level tool.	Provides access to sea-level rise scenarios generated by the Pacific Institute, ESA PWA and the U.S. Geological Survey as part of the CA Energy Commission's Public Interest Energy Research Program (PIER). The tool shows the threat of coastal erosion and inundation due to flooding over three depths based on a 100 year flood scenario.	Provides access to sea-level rise flooding scenarios generated by the Pacific Institute, ESA PWA and the U.S. Geological Survey as part of the CA Energy Commission's Public Interest Energy Research Program (PIER). The tool shows the threat of inundation due to flooding over three depths based on a 100 year flood scenario.	A collaborative, user-driven project focused on providing San Francisco Bay Area coastal resource and land use managers and planners locally relevant, online maps and tools to help understand, visualize, and anticipate vulnerabilities to sea level rise and storms.	The Stanford unstructured-grid, non-hydrostatic, parallel coastal ocean model. For simulation of non-hydrostatic flows at high resolution in estuaries and coastal seas. Requires a grid generator and ParMETIS (if run in parallel).
Scope	National	National	California	California	Bodega Head to Half Moon Bay and SF Bay	Adaptable

	Climate Central - Surging Seas Risk Finder	NOAA Coastal Services Center - Sea Level Rise and Coastal Flooding Impacts Viewer	Pacific Institute - The Impacts of Sea Level Rise on California's Coast	Cal-Adapt - Exploring California's Climate	Our Coast, Our Future	SUNTANS
Release Yr	Rolling: 2013 -2014	2012 (West Coast of US)	2009	2011	2013 Half Moon Bay to Bodega; 2014 SF Bay	Ongoing
Organization/ Sponsor	Climate Central	NOAA Coastal Services Center	California Energy Commission, California Environmental Protection Agency, Metropolitan Transportation Commission, California Department of Transportation, and the California Ocean Protection Council	California Energy Commission; UC-Berkeley Geospatial Innovation Facility	Point Blue Conservation Science; USGS; Gulf of the Farallones National Marine Sanctuary; Coravai LCC	Stanford University Environmental Fluid Mechanics and Hydrology Program
Sea Level Rise Scenarios	Up to 10 feet in 1-foot intervals above local high tide line (Mean Higher High Water)	Up to 6 feet in 1-foot intervals above local high tide line (Mean Higher High Water)	Current water levels, 19", 39" and 55" inundation	Current water levels, 19", 39" and 55" inundation	Total of 40 combinations of sea level rise and storm scenarios that include 0-2 m SLR in 25 cm increments plus a 5 m extreme, and 4 storm scenarios: no storm, annual, 20 year, and 100 year	

	Climate Central - Surging Seas Risk Finder	NOAA Coastal Services Center - Sea Level Rise and Coastal Flooding Impacts Viewer	Pacific Institute - The Impacts of Sea Level Rise on California's Coast	Cal-Adapt - Exploring California's Climate	Our Coast, Our Future	SUNTANS
Inundation Model	Modified bathtub approach, modeling hydrologic connectivity and locally adjusted Mean Higher High Water levels.	Modified bathtub approach, modeling hydraulic connectivity and locally adjusted Mean Higher High Water levels.	Bathtub approach	Bathtub approach	USGS Coastal Storm Modeling System (CoSMoS)	
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Methods and Data Sources

This approach to develop the migration space indicator delineates the boundaries of alternative migration spaces for the entire Estuary based on two future sea level rise scenarios, and quantifies areas within the two spaces that could be dedicated to migration of the natural estuarine-terrestrial transition zone. The approach is very similar to that taken in other efforts for sub-regions or selected locations within the Estuary (e.g., CLN 1.0). The basic details of the methodology, including the sources of data used in the indicator, are presented below.

Seaward Boundary.

For the Bay Area, a modern shoreline was created that ignores the levees of reclaimed estuarine areas that would be flooded under existing sea level if these levees were breached. The shoreline was derived from the Modern Baylands layer of the Bay Area Aquatic Resource Inventory (BAARI) by dissolving the bay, channel, diked baylands, tidal flats and tidal marshes (http://www.sfei.org/sites/default/files/SFEI%20MAPPING%20STANDARDS_08092011_v8_0.pdf). The resulting shoreline is essentially the historical (pre-settlement) shoreline updated to account for sea level rise over the last two centuries, and to account for artificial fill other than levees that is above the selected future sea levels. It is assumed that this shoreline corresponds to local Mean Higher High Water tidal datum (MHHW). This boundary corresponds well to the MHHW contour plus the diked "low-lying areas" derived by NOAA (NOAA 2010).

For the Delta, a modern shoreline was created that ignores the levees of reclaimed estuarine areas following a simple multi-step process. A line was derived from the historical Delta tidal habitats layer (SFEI-ASC 2014) of the CA Aquatic Resources Inventory (www.ecoatlas.org/data/#cari) by dissolving the water and tidal features. The resulting shoreline is essentially the historical (pre-settlement) shoreline. It is assumed to correspond to present-day local MHHW, although it has not been adjusted for historical sea level rise.

Landward Boundary

The landward limit of the migration space was estimated throughout the Estuary for two future sea levels, +2 ft and +5 ft above present-day MHHW. These heights are generally consistent with the heights recently used to explore sea level rise effects on Bay Area intertidal habitats (BEHGU 2015)[1].

Generally, the process used to estimate future landward boundaries of estuarine flooding can be described as a bathtub approach or linear superposition method (NOAA 2010, Marcy et al. 2011). For many reasons, sea level varies in height along the shoreline of an estuary, relative to a common geodetic datum. To represent this variation, it should be modeled as a spatially variable water surface. In addition, the elevations of this surface must be referenced to the same vertical datum as the land surfaces (i.e., NAVD88). There are currently two primary ways this surface can be created. The first and simpler approach is to covert the MMHW tidal datum derived for well-gauged tide stations to NAVD88, and then interpolate the surface between the stations. The second and more accurate approach is to use NOAA's vertical datum conversion software, VDatum (http://vdatum.noaa.gov/) for a dense array of points along the shoreline. Both approaches were incorporated into the migration space indicator.

[1] The future sea level values used in the Baylands Ecosystem habitat Goals Update (BEHGU 2015) are 52 cm (1.7 ft) and 165 cm (5.4 ft). These values were rounded to the nearest whole foot because other data incorporated into the migration space indicator do not support the spatial resolution denoted by increments of elevation less than about one foot. For example, sea level data provided by NOAA is based on 1-ft increments of sea level rise.

For the Bay Area, the areas denoted as "high confidence" in the NOAA Sea Level Rise and Coastal Flooding Impacts Viewer (http://coast.noaa.gov/slr/) for the +2-ft and +5-ft sea levels were adopted (http://coast.noaa.gov/digitalcoast/tools/slr). According to the documentation for the viewer (Marcy et al. 2011), where VDatum was available, it was used to covert MHHW into elevations relative to NAVD88. A linear superposition method was used to raise the resulting grid of elevation points in 1-ft increments of sea level rise up to 6 ft above present-day MHHW. Because tidal datum transformations in VDatum extend only slightly beyond the present-day MHHW shoreline, interpolation and extraction routines to extend the MHHW surface inland were done according to methods suggested in NOAA (2010). Where VDatum was not available, methods outlined in NOAA (2007) were used to interpolate between NOAA tide gages for which the relationship between MHHW and NAVD88 had been resolved.

A much simpler and less accurate method was used in the Delta. The NOAA sea level rise data have not yet been developed for the Delta, and insufficient data were available to apply the NOAA method of VDatum conversion from tidal to geodetic elevations. The elevation of the seaward boundary (see above), which was assumed to correspond to the local MHHW, was further assumed to have a tidal elevation of 6.4 ft NAVD88, based on reckonings reported for a single station at Cache Slough by the CA Department of Water Resources (DWR) http://baydeltaconservationplan.com/Libraries/Dynamic_Document_Library/2D_Hydrodynamic_Modeling_of_the_Fremont_Weir_Diversion_Structure_with_average_Westside_tributary_flows.sflb.ashx. In other words, the sea level surface was assumed to have one elevation relative to NAVD88 throughout the Delta.

Land Cover

All assessments of land cover depended on the 2011 National Land Cover Database of USGS (NLCD 2011; http://www.mrlc.gov/nlcd2011.php) and the California Protected Areas Database of the Green Info Networks (CPAD; http://www.calands.org/).

These two datasets (NLCD 2011 and CPAD) were the basis for deciding areas of migration space that could be devoted to the conservation of the estuarine-terrestrial transition zone. The decisions involved value judgements about the relative likelihood of different land covers being left undeveloped or being converted from developed to undeveloped status (Table 1). It is assumed that any land coves categorized as undeveloped can be devoted to the transition zone. All areas designated as protected in the CPAD are assumed to be undeveloped. The assignment of land covers to these categories can be revised at any time.

Table 1. Classification of NLCD land cover types as developed or undeveloped.

Categorization of NLCD Land Cover Types as Developed or Undeveloped	
Developed	***Undeveloped***
Developed - Low Intensity	Water - Open Water
Developed - Medium Intensity	Developed - Open Space
Developed - High Intensity	Barren - Barren Land
Planted/Cultivated - Cultivated Crops	Forest - Deciduous Forest
	Forest - Evergreen Forest
	Forest - Mixed Forest
	Shrubland - Shrub/Scrub

	Herbaceous - Grassland/Herbaceous
	Wetlands - Woody Wetlands
	Wetlands - Emergent Herbaceous Wetlands

The areas of developed, undeveloped, and protected lands were quantified for each major sub-region of the Estuary: South Bay (South San Francisco Bay), Central Bay (Central San Francisco Bay) North Bay (San Pablo Bay and the western portion of Carquinez Straight), Suisun Bay (Suisun Bay and the eastern portion of Carquinez Strait), North Delta, Central Delta, and South Delta (Figure 2). Bay sub-regions are based on the Baylands Goals Project (Goals Project 1999). The Delta sub-regions are generally based on patterns of subsidence that distinguish the Central Delta from its northern and southern areas, with the Central Data being more subsided (DWR 1995).

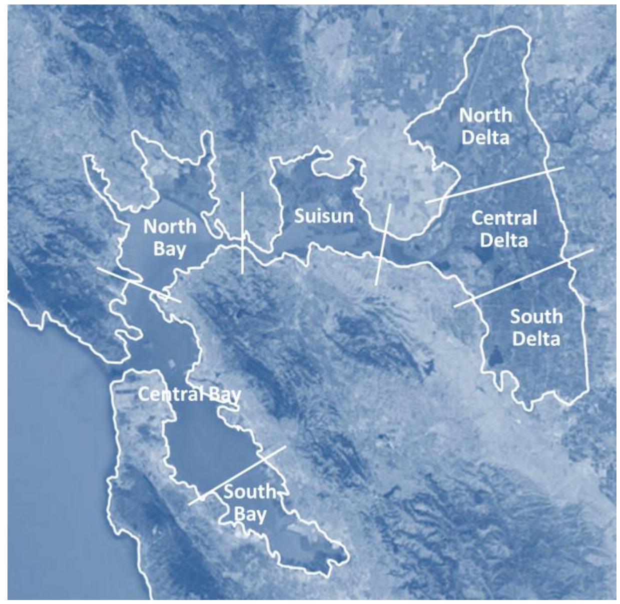

Figure 2. Sub-regions of the Estuary.

Assumptions and Uncertainties

Many assumptions underlie the reported measures of migration space. The most important assumptions are discussed below, along with recommendations for either eliminating them or testing their validity.

Seaward Boundary Location

The seaward boundary of the migration space is assumed to be the MHHW contour. Local MHHW cannot be exactly reckoned without an adequate series of site-specific tide height records. Without such records, the contour must be modelled (e.g., NOAA 2010) or derived from inexact ecological field indicators (Harvey et al 1978).

For the Bay Area and Delta, the seaward boundary (i.e., local MHHW) is assumed to be the historical upland limit of the tides as derived from the historical wetlands datasets of the Bay Area and Delta versions of the California Aquatic Resource Inventory (CARI; the Bay Area version is called BAARI), areas of fill above the projected sea levels. This boundary is based on many collaborating historical records (Collins et al. 1998, Beller et al. 2013, Whipple et al, 2012, SFEI 2014) plus limited local ground-truthing. The assumption that this boundary corresponds to local MHHW is probably conservative. That is, the historical boundary as depicted in BAARI might be slightly higher than the MHHW contour. How much higher depends on the local tide range and the accuracy of the historical records, both of which vary around the Estuary. It might be expected that the depicted boundary is less than 2 ft higher than the actual MHHW contour (Harvey et al 1978, Collins et al. 1998). For existing, low-gradient remnants of the historical, non-diked shoreline, the boundary derived from CARI corresponds closely to the boundary provided by NOAA, plus one foot of seas level rise (Figure 3). Based on their imprecision, and given the purposes of the migration space indicator, the boundaries derived by NOAA and based on CARI are comparable. The boundary provided by CARI is preferable because of its local documentation.

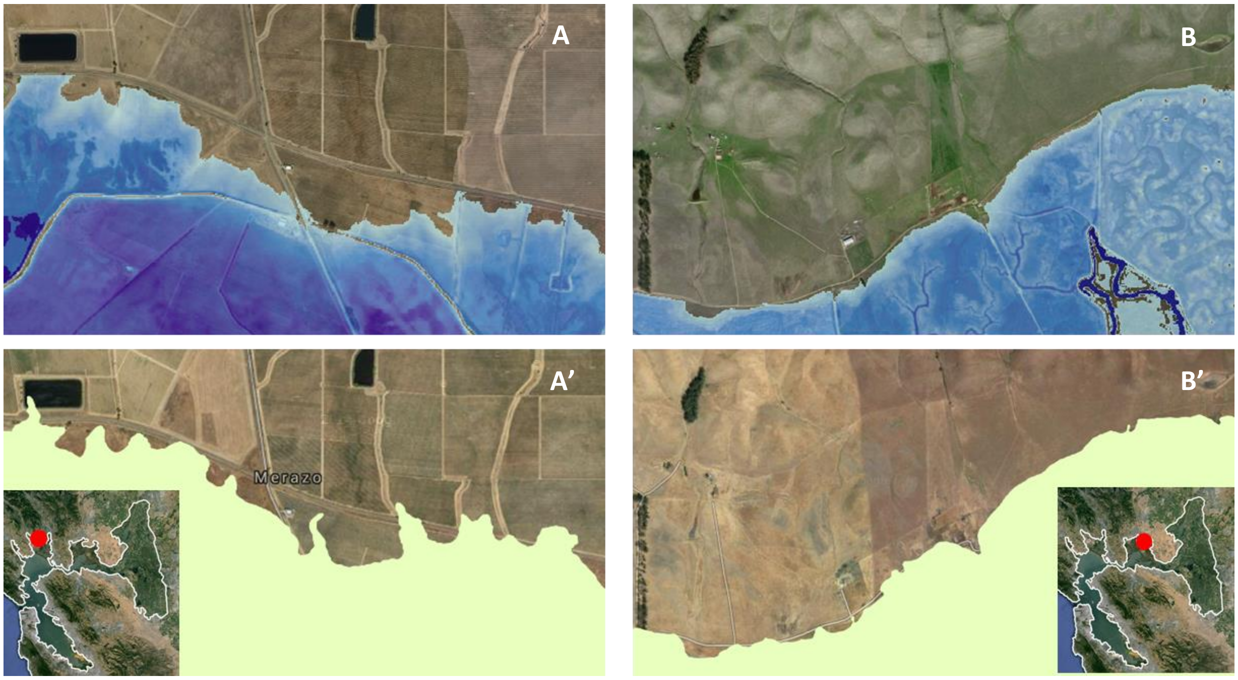

Figure 3. Comparison of seaward boundaries of migration space as represented by the NOAA MHHW contour plus 1 ft of sea level rise, and as derived from the historical tidal wetland boundary provided the CARI (A' and B'), for areas of remnant natural shoreline in North Bay (A, A') and Suisun Bay (B, B').

Seaward Boundary Elevation

For the Bay Area (South Bay, Central Bay, North Bay, and Suisun Bay), the geodetic elevation of the MHHW contour as depicted in BAARI was assumed to be the same as the MHHW contour derived by NOAA (NOAA 2010, Marcy et al. 2011). In other words, the local NAVD88 elevations determined by NOAA for its estimated MHHW contour were transferred to the MHHW contour provided by CARI. This is reasonable, given the close correspondence between the two boundaries (Figure 3).

For the Delta, NOAA has not yet developed an estimated MHHW contour. As discussed above, the MHHW contour was reasonably assumed to be the historical tidal wetland boundary. The geodetic elevation of the MHHW contour has also not been determined, accept for a few locations. The Sacramento-San Joaquin Delta is one notable area along the California coast where VDATUM has not been calibrated (OLS 2012). The best documented reckoning of local MHHW and its conversion to NAVD88 for the Delta or its immediate vicinity pertain to Cache Slough and Sacramento in the North Delta (NAVD88 elevations for MHHW equal 6.4 ft and 7.96 respectively), and Port Chicago in Suisun Bay (MHHW equals 6.04 ft). The Sacramento tide station is near the head of tide on the Sacramento River, and geodetic elevation of MHHW at this station is therefore especially high due to the large and immediate influence of the high river flows. In contrast, the value for Port Chicago is much more removed from such influences because it is outside the Delta. In general, MHHW increases in geodetic elevation with distance upstream through Suisun Bay (DWR 2004). The value for Cache Slough was therefore assumed to be the most representative of the Delta overall. In other words, sea level was assumed to be the same relative to the land surface throughout the Delta, and the MHHW contour was assumed to have the same elevation relative to NAVD888 as reported by DWR for Cache Slough.

This is a large assumption with uncertain effects on the estimates of migration space around the Delta. The difference in NAVD88 elevation of MHHW for the three stations referenced above is less than two feet, which agrees with differences in elevations for MMHW relative to Mean Sea Level of 1929 (NGVD29) reported elsewhere for the Delta (Simenstad et al. 2000, OLS 2012). The reported geodetic elevation for Cache slough is probably within one foot of actual local geodetic elevations, which is comparable to the expected error of estimated MHHW contour in the Bay Area.

The difference in migration space width caused by any error in reckoning the geodetic elevation of the MHHW contour depends mainly on the topographic slope of the lands between the present-day and future tidal boundaries. Based on the DEMS for the Bay Area and Delta, a reckoning error of 2 ft could represent nearly 1,000 ft in migration space width for the most gently sloping areas. However, for most of the Estuary, a reckoning error of 2 ft represents much less than 200 ft of migration space width.

Landward Boundary

The landward boundaries are projected contours of MHHW for selected future sea levels. How fast the sea will rise to the selected levels is unknown. Furthermore, it is expected that the rate of sea level rise will generally decrease from the deeper areas the Estuary to it shoreline, and with distance upstream from the Golden Gate. The future differences in the rate of sea level rise around the Estuary are also unknown.

The landward boundary of the migration space does not correspond to the landward boundary of the associated transition zone. The transition zone extends seaward and landward of the MHHW contour. Under natural conditions, the landward extent of the transition zone depends directly on the slope of the land. For any given slope, the landward extent of the zone also varies with its physical and ecological services. It is generally wider for ecological services, such as wildlife support, than for physical services, such as shoreline protection (BEHGU 2015). This means that the migration space defined by the 5-ft rise in sea level might, for some services, be needed to conserve the transition zone associated with the 2-ft rise. For the purpose of conserving the transition zone, there is a need to visualize its full extent around the Estuary for all its essential services, and to determine what migration space is needed to conserve the services under different sea level rise scenarios.

Systematic error in the measurement of migration space begins with the uncertainty in reckoning existing sea levels, as discussed immediately above in relation to seaward boundaries. The error can be increased by the uncertainty of sea level rise projections (Reilly et al. 2001, Guttorp et al. 2014). These uncertainties can seem large (Church et al. 2013). However, it is useful to develop local scenarios of sea level rise, conduct vulnerability assessments based on the scenarios, and start to consider suitable adaptation policies (IPCC 2011). The migration space indicator is consistent with this guidance provided by the International Panel on Climate Change (IPCC).

The error in migration space measurement is also affected by omissions and inaccuracies in the digital elevation models (DEMs) used to resolve topographic relief and estuarine flooding pathways. The DEMS for the Bay Area and Delta do not incorporate such details as culverts and ditches that can significantly influence flooding extent. This can be remedied in the future by using DEMs that are based on high-resolution LiDAR and ground-truthed through local flood control agencies.

NOAA (NOAA 2010) provides guidelines for the accuracy of estuarine flood mapping due to sea level rise. Considering that a 1-ft contour map has a Root Mean Square Error (RMSE) of 0.3 ft, the 95% confidence level of the estuarine flood map would be 0.6 ft. The minimum sea level rise that can be confidently mapped is twice (1.96 x) that of the 95% confidence interval, and is therefore about one foot (1.19 ft) for a 1-ft contour map. The DEMs used in the migration space indicator therefore support the estimates of migration space for the selected sea level rises of two and five feet.

Land Cover

As reported here, the migration space indicator assumes that croplands (i.e., lands used for truck crops, vineyards, orchards, and hay) are not available to accommodate the landward migration of the transition zone. This assumption is based on the subjective decision that croplands are as valuable as the built environment and might be subject to same degree of protection from sea level rise. Two aspects of this assumption are worth noting. First, the maps of croplands was published in 2011 (NLCD 2011) and might not reflect more recent land use change. Second, there is some uncertainty about the future dedication of these lands to agriculture. Salt water intrusion due to sea level rise, the cost of building and maintaining levees, and an increased frequency of extreme flood events could eventually render these lands physically or economically unsuited for agriculture (Lund et al. 2008, Madani and Lund 2011, NRC 2012). Adding these croplands into the category of undeveloped lands could significantly increase the estimated amount of space potentially available to accommodate transition zone migration.

The migration space indicator assumes that reclaimed areas of the Estuary that have not been filled more than 2 ft or 5 ft above present-day MHHW are not part of the migration space. However, this assumes that these areas will not be filled to these elevations or higher in the future. The migration space indicator could accommodate scenarios of filling diked areas of the Estuary to create migration space by adjusting the DEMs to reflect the future fill elevations. In this way, the indicator could be used to assess the effects of intentional modifications of the shoreline, such as the creation of "horizontal levees", (Lowe et al 2013) on the amount of undeveloped migration space.

The migration space indicator involves no analyses or decisions about which areas of the Estuary most need the transition zone restored or conserved. The indicator as presently configured assumes that all existing transition zone areas should be conserved and that all suitable migration space should be dedicated to the transition zone of the future. This first generation of the migration space indicator can serve to begin prioritizing the opportunities that are identified.

Landscape Response

The data for future estuarine flooding do not consider how natural processes, such as erosion and marsh migration, will be affected by future sea level rise. The effects of changes in estuarine depth on tidal velocities are also not considered. Ongoing changes in the depth profile of the Estuary, including especially increases in the extent of shallow water, are likely to cause the rate of sea level rise to vary along the shoreline. Sea level is unlikely to rise at the same rate throughout The Estuary (Holleman 2013). Large scale levee breaches in Suisun Bay and the Delta could increase the rates of sea level rise in those sub-regions (DWR 2002, Jack R. Benjamin & Associates 2005), although they would likely be lesser than the rates further downstream toward the Golden Gate. Failing to address these processes is a significant limitation of the estimates of migration space. Overcoming this limitation will be difficult because it requires new understanding of the likely interactions between sea level rise and its landscape effects Important efforts to achieve this understanding though simulation modeling have begun (e.g., Morris et al 2002, Stralberg et al. 2011, Kirwan and Megonigal 2013), and are likely to continue. When the models are suitably developed, they should inform transition zone conservation and restoration efforts.

Benchmark and Scoring

The migration space benchmark is 50% of the total migration space in each sub-region being undeveloped, and 75% of that undeveloped space being protected. The scoring break between fair and poor scores is 40% and 50%, respectively. This benchmark and the threshold scores are arbitrary. They are not based on any ecological or economic analysis. An alternative benchmark could easily be incorporated into the indicator at any time. Ideally, the benchmark should reflect collaborative decisions by the responsible agencies about how much transition zone is needed, where it is needed, and why. Such decisions should reflect the new transition zone typology from the Baylands Goals Science Update (Goals Project 2015), the ecosystem services of the types, the costs and likely success of any necessary land use conversion or realignment, and the contribution of each future area of transition zone to the overall health of the Estuary. The scoring thresholds should be based on empirical relationships between the scores and levels of selected ecosystem service.

Peer Review

Comments on a draft of this technical report were solicited from Donna Ball of Save the Bay, Susan De La Cruz and Karen Thorne of the US Geological Survey, Matt Gerhart of the State Coastal Conservancy, Brenda Goeden of the Bay Conservation and Development Commission, Kirk Klausmeyer of The nature Conservancy, Andy Gunther of the Center for Ecosystem Management and Restoration, Hildie Spautz of the CA Department of Fish and Wildlife, Luisa Valiela of US Environmental Protection Agency, and Sam Veloz of Point Blue.

Results

The results of applying the migration space indicators for the San Francisco Estuary are summarized in the text below and the following figure (see Figure 4).

Total Migration Space

Total migration space includes all lands, developed or not, within the area delimited by the present-day MHHW contour and the likely landward extent of MHHW for either a 2-ft or 5-ft rise in sea level (Figure 4A). There is no relationship between total migration space and land cover. The amount of migration space is directly related to the tidal elevation and topographic slope of lands immediately joining the existing MHHW contour and draining toward the Estuary. In the Bay Area, the total amount of migration space increases from Suisun Bay through North Bay and Central Bay to South Bay. The relatively large amount of migration space in South Bay, relative to other Bay Area sub-regions, is due to the extensive lowlands of Santa Clara Valley adjoining the Estuary that have subsided below sea level due to groundwater extraction (Polland and Ireland 1988). The migration space of the other Bay Area sub-regions does not involve subsidence and is much more constrained by more steeply sloping lands. The three sub-regions of the Delta have nearly as much or more migration space than South Bay, due to the extensive low gradient lowlands of the Central Valley. For the Delta, the total migration space deceases from North Delta through Central Delta to South Delta. These patterns are not obvious at small scale (i.e., when either the Estuary or any one of its sub-regions is viewed in its entirety) because the migration space is seldom more than a few hundred feet wide, although it can exceed a thousand feet in some locations. The migration space corresponding to a rise in sea level of two feet is uniformly about half the size of the space corresponding to a sea level rise of five feet. This is indicative of the fairly uniform topographic slope of both the 2-ft and 5-ft migration spaces.

A separate analysis of total migration space not reported here compared the historical (pre-settlement) migration space to the modeled future spaces. The total migration space has decreased since historical times. This is due to the purposeful filling of diked estuarine areas to elevations above the selected future sea levels. The filling has effectively moved the MHHW contour seaward and thereby increased the amount of migration space. Most of the fill has been developed and therefore has not increased the space for the future transition zone.

Undeveloped Migration Space

In the Bay Area, for a 2-ft rise in sea level, the amount of undeveloped migration space is greatest in South Bay. There is perhaps twice as much in North Bay than in Central Bay or Suisun, but the amounts are very small everywhere outside South Bay. For a 5-ft rise in sea level, the amount of undeveloped space is still greatest in South Bay, but it is approximately equal in North Bay and Suisun, and least in Central Bay. The percent increase in space between a 2-ft rise in sea level and a 5-ft rise is least for South Bay and greatest for Suisun.

In the Delta, for a 2-ft rise in sea level, there are comparable amounts of undeveloped space in North and Central Delta, and substantially less in South Delta. For a 5-ft rise, the amount of undeveloped space decreases markedly from North Delta through Central Delta to South Delta. The percent increase in space between a 2-ft rise in sea level and a 5-ft rise is by far greatest for North Delta.

These patterns reflect complex spatial relationships between topography and land use. For the Bay Area, the sub-region with the most undeveloped migration space is South Bay. Although this sub-region is densely urbanized, it also relatively flat and low-lying, with relatively numerous areas of protected open space along the shoreline. There are larger undeveloped areas in North Bay and Suisun, but they are in general much steeper. Central Bay has the least amount of total migration space and undeveloped migration space because the lands adjoining the shore are relatively steep, densely developed, and have less undeveloped area. For the Delta, where the topography of lands adjoining the Estuary is more uniformly flat and low-lying (western extent of the Central Delta notwithstanding), differences in migration space among the sub-regions largely reflect differences in land use and the distribution of people. Population density and land development along the shoreline of the Delta increase from North Delta through Central Delta to South Delta.

Undeveloped and Protected Migration Space

Much less than half of the undeveloped migration space is protected from development (Figure 4C). Nearly twice as much of the undeveloped migration space has been protected in South Bay and North Bay than in the other sub-regions, with the exception of the North Delta. This is mainly due to shoreline parks and other public open space in urbanized environments. The relatively large areas of protected migration space in North Bay, North Delta, and Central Delta are due in large part to state and federal wildlife refuges that adjoin the historical MHHW contour. A noteworthy finding is that very little of the undeveloped migration space in Suisun and South Delta is protected. While there is probably a need to explore opportunities in every region to conserve and restore the transition zone, the need might be greatest in these sub-regions of the Estuary.

Recommended Next Steps

Further development of the migration space indicator should be guided by regional experts in land use, sea level rise and its geomorphic effects, and GIS. There is a need to assure that the indicator always utilizes the best available data. There is a critical need to develop the Vdatum tool for the Delta. There is also a need to estimate the full extent of the transition zone around the Estuary for all its essential services, and to determine what migration space is needed to conserve the services under different sea level rise scenarios. Capabilities for online mapping and visualization should be developed to support analyses of alternative scenarios for transition zone conservation and restoration. These scenarios will need to be guided by scientifically sound criteria for identifying and prioritizing restoration and conservation opportunities.

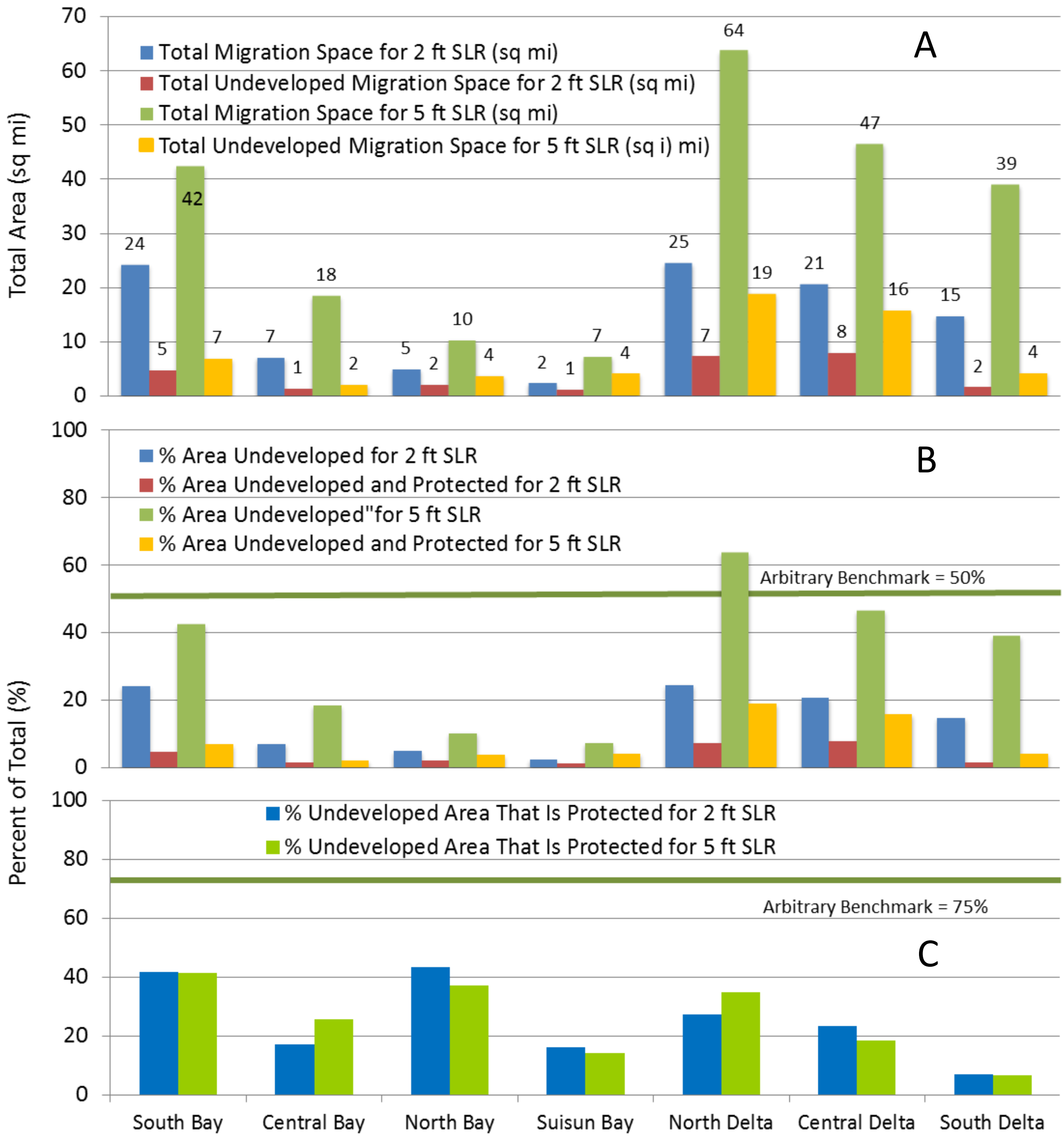

Figure 2. Sub-regional distribution of developed and undeveloped migration space for sea level rise (SLR) of 2-ft and 5-ft, showing (A) total migration space (sq mi); (B) percentage of total migration space undeveloped and protected, showing an arbitrary target value of 50%; and (C) the migration space indicator (i.e., the percentage of the total undeveloped migration space that is protected), showing an arbitrary benchmark ~~target~~ of 75%.

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State of the Estuary Report 2015

Sidebar Source Material

PROCESSES – Sediment on the Move

Prepared by
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Sediment on the Move

Prior to European colonization of the Estuary's watershed, high river and stream flows that occurred every winter and spring would mobilize sediments from streambeds and eroding riverbanks and slopes upstream. This sediment flowed downstream to the estuary where it replenished marsh and other parts of the shore. Long-term monitoring suggests there is now much less suspended sediment in the Estuary's waters than there was for more than 100 years between the Gold Rush and the late 1990s. The estuary's sediment balance is changing because dams now control the flow of water that would historically have transported sediments, rivers and streams have been armored and otherwise engineered to limit erosion, and the large volume of sediment created by Gold Rush mining has finally petered out. Reduced sediment availability promises to alter several important ecological processes in the Estuary.

Estuarine waters that are murky from suspended sediments can be good for native ecosystems. Suspended sediment is critical for marsh building processes, for example. A marsh with sufficient sediment supply can capture enough material to rise in elevation and keep pace with rapid sea level rise. Suspended fine sediment also reduces the penetration of sunlight into the water, limiting the potential for invasive weed species, toxic algal blooms, and negative effects from excess nutrients introduced by outflow from sewage treatment plants, agricultural runoff, and urban stormwater runoff. The muddy water also provides cover for small fish and zooplankton.

In addition, coarser sediments now trapped behind dams once fueled habitat-forming processes upstream. Currently, rivers and streams below dams are starved of gravels that salmon need in order to build nests for their eggs. At the same time, accumulation of these materials behind dams also reduces their storage capacity.

Human activities over the past 200 years have altered sediment supply to the Estuary. From the early 1800s into the 20th century, intensive ranching and farming, hydraulic mining in the Sierra, and urbanization caused chronic erosion of stream channels, resulting in large increases in the sediment load carried by winter and spring flows. Today, land surface erosion is better managed, the large bulk of sediment created by historic land uses upstream has moved through the system, stream flows are highly regulated (lacking the high flows that move large amounts of sediment downstream), and reservoirs, dredges, and sand mining remove sediment from the system, resulting in an Estuary starved of sediment.

With less sediment in circulation, critical ecological processes are jeopardized. Projections of marsh-building processes show a strong likelihood that there will not be enough sediment for marshes to keep up with rapid sea level rise in the latter half of this century.. Water quality experts are planning for how to address the effect of clearing waters on nutrient impacts (eutrophication) and . Marsh restoration planners are concerned that there won't be enough sediment for marshes to keep up with rising seas in the decades to come. Water quality regulators seekhow to avoid potential impacts to human and fish health from toxic algal blooms that are facilitated by the lack of suspended sediment. Fish biologists worry that native fish species suffer greater exposure to predators whenever the estuary's waters become too clear.

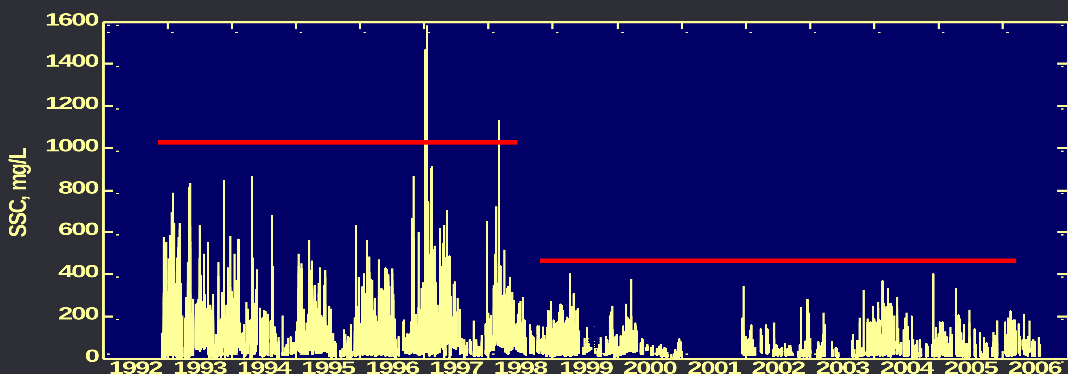
Just as they have in the past, human actions can change the amount of sediment available to the estuary in the future. Watershed management influences the timing, amount and type of sediment delivered by rivers and streams to the estuary, increasing or decreasing supply depending on the approach taken. Useful management approaches may include: stream flow management, retrofitting or removing dams, restoring naturalistic connections between watersheds and the estuary's tidal wetlands, reducing sand mining, and re-use of sediment dredged from the estuary and excavated from terrestrial areas.

Innovative thinking about sediment sources, transport and delivery to the Estuary is now an urgent priority. Restoring natural processes of sediment movement in the watershed and creating artificial methods to deliver sediment (such as reusing channel dredge spoils) are both valuable approaches for restoring this fundamental physical driver. Like fresh water, sediment is a precious resource that is essential for keeping the Estuary healthy.

[PHOTO Caption]
Marshes that harbor endangered Ridgway's rails also buffer the shorelines behind them from rising seas, extreme storms, king tides, and wave erosion. Sediment is an essential ingredient to keep these useful ecosystems sustainable over the long run. Photo: Rick Lewis

Sediment Supply Reduction

Hydraulic Mining in Yuba County in 1870
http://www.sierracollege.edu/ejournals/jsnhb/v2n1/monitors.html



Schoellhamer 2011

State of the Estuary Report 2015

Summary

ECOLOGICAL PROCESSES – Flood Events Indicators Summary

Prepared by Christina Swanson
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September 2015

State of the San Francisco Estuary 2015

ECOLOGICAL PROCESSES – Flood Events Indicators
Summary

Prepared by Christina Swanson
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September 2015

What are the indicators?
The State of the Estuary Report uses two indicators to measure and evaluate the frequency, magnitude and duration of ecologically important flood events. The Yolo Floodplain Flows indicator measures seasonal inflows into the Delta from the Yolo Bypass, the large, partially managed floodplain immediately upstream of the Estuary in the lower Sacramento River basin. The Flood Inflows indicator measures flood events in terms of high volume freshwater inflows to the Bay.

Table 1.

Attribute	**Indicators**	**Benchmarks**
Ecological Processes (Flood Events)	Two indicators measure the frequency, magnitude and duration of: 1) floodplain inundation and flood flows into the Delta (Yolo Floodplain Flows indicator); and 2) high volume flows from the Delta into the Bay (Flood Inflows indicator).	Benchmarks (or reference conditions) are based on: 1) unimpaired flow and flood data records; 2) biological information on floodplain habitat, productivity dynamics, and utilization for spawning, rearing, and outmigration of juvenile salmonids; and 3) current regulatory standards for minimum Bay inflows (i.e., State Water Resources Control Board, 2006 Water Quality Control Plan).

Why are flood events important?
Following winter rainstorms and during the height of the spring snowmelt in the Sacramento-San Joaquin watershed, the estuary's tributary rivers may flood, spilling over their banks to create ecologically important floodplain habitat and sending high volumes of fresh water into the estuary. These seasonal high flows drive multiple ecological processes including: primary and secondary production in inundated floodplains and the upper estuary; downstream transport or organisms, sediment, and nutrients to the Bay; creation of spawning and rearing habitat for a numerous fish species; and mixing of Bay waters and creation of productive brackish, or "low-salinity," habitat in the Bay's upstream Suisun and San Pablo regions. High flows also improve habitat conditions in riverine migration corridors for both adult fish moving upstream as well as young fish moving downstream. All

of these provide conditions favorable for many native fish, invertebrate and other wildlife species. 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.

Several factors have had and are having substantial impacts on the frequency, magnitude and duration of high flow, or flood, events into the estuary. 1) Flows in most of the Bay's largest tributary rivers have been greatly altered by dams, many of which built for the purpose of reducing downstream flooding and to store the mountain runoff for later use and export to other regions in the state. This has deprived the estuary and its tributary rivers of regular seasonal flooding, an important physical and ecological process 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. Dams also physically block the flow of sediment, which starves riverine and estuarine wetlands and marshes of the materials they need to sustain (and restore) themselves. 2) 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 (as well as nutrients, primary production and plankton), even during of relatively high flows (see Freshwater Inflow Index). This reduces the amount of fresh water that flows into the estuary and can decrease inflow to levels below important thresholds for floodplain inundation, habitat creation and sediment transport. 3) The lower reaches of the estuary's largest tributary rivers, the Sacramento and San Joaquin Rivers, are confined by man-made levees that prevent or restrict inundation of adjacent floodplains during high flow events, essentially disconnecting the estuary's tributary rivers from their floodplains.

What are the benchmarks? How were they selected?

The benchmarks (or reference conditions) for the two indicators are based on: 1) unimpaired flow and flood data records; 2) biological information on floodplain habitat, productivity dynamics, and utilization for spawning, rearing and migration; and 3) current regulatory standards for minimum Bay inflows (i.e., State Water Resources Control Board, 2006 Water Quality Control Plan).

What are the status and trends of the indicators and Index?

The two flood events indicators show that the frequency, magnitude and duration of floodplain inundation and high volume inflows to the estuary are all too low to drive or support important ecological processes in the lower watershed and estuary. Inundation of the Yolo Bypass is (and has been for decades) too rare, too little and too short to promote primary and secondary productivity, support floodplain spawning, rearing and migration of native fishes, and export sediment, nutrients and organisms to the Delta and estuary. High volume inflow events to the Bay have declined significantly since the 1940s. For the last decade (or two decades), the condition of flood-related ecological process has been "poor" in almost all years.

Table 2.

Indicator	**CCMP Goals** Fully met if goal achieved in >67% of years since 1990 Partially met if goal achieved in 33-67% of years Not met if goal achieved in <33% of years	**Trend (long term; 1930-2014)**	**Trend since 1990**	**Current condition** (average for last 10 years)
Yolo Floodplain Flows	Not met; goals achieved in 8% of years	Stable (in poor condition)	Mixed	Poor Frequency, magnitude and duration too low to support ecological processes
Flood Inflows	Not met; goals achieved in 12% of years	Decline	Mixed	Poor Frequency and duration of high volume inflows cut by 60-75%

What does it mean? Why do we care?

Floodplain inundation and high volume, flood flows into the estuary are key physical and ecological drivers, stimulating and supporting primary and secondary productivity (creating food for fish and wildlife); transporting sediment (essential for marsh restoration and maintenance, including in the face of sea level rise), nutrients and organisms downstream; and creating spawning, rearing and migratory habitat for fish and wildlife. Man-made reductions in the ecological processes (i.e., from dams and water management operations) measured by these two indicators correspond to declines in food and habitat availability, reduced growth, survival and reproductive success for a number of species, and population declines for a number of fish and wildlife species. In addition to changes in water management operations to selectively restore periodic high volume flood flows to the watershed's and estuary's hydrograph, there are opportunities to manage the Yolo Bypass to create inundated floodplain habitat at lower Sacramento River flows. There is broad agreement that floods and floodplain habitat are important for native fish and wildlife, and that ecosystem restoration and management actions that restore these functions and habitat would likely be effective, but few specific restoration actions have been implemented to date.

State of the Estuary Report 2015

Technical Appendix

ECOLOGICAL PROCESSES – Flood Events Indicators
Technical Appendix

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June 2015

State of the San Francisco Estuary 2015

ECOLOGICAL PROCESSES – Flood Events Indicators Technical Appendix

Prepared by Christina Swanson
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June 2015

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 volumes of fresh water into the estuary. These seasonal high flows drive multiple ecological processes including: primary and secondary production in inundated floodplains and the upper estuary; downstream transport or organisms, sediment, and nutrients to the Bay; creation of spawning and rearing habitat for a numerous fish species; and mixing of Bay waters and creation of productive brackish, or "low-salinity," habitat in the Bay's upstream Suisun and San Pablo regions (Jassby et al. 1995; Sommer et al. 2001; Kimmerer 2002, 2004; Schemel et al. 2004; Feyrer et al. 2006a, b; del Rosario et al. 2013). All of these provide conditions favorable for many native fish, invertebrate and other wildlife species. 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, they also improve habitat conditions in riverine migration corridors for both adult fish moving upstream as well as young fish moving downstream.

In the Estuary's Sacramento-San Joaquin watershed, several factors have had and are having substantial impacts on the frequency, magnitude and duration of high flow, or flood, events into the estuary. First, flows in most of the Bay's largest tributary rivers have been greatly altered by dams, many of which built for the purpose of reducing downstream flooding and to store the mountain runoff for later use and export to other regions in the state. 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, 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 (as well as nutrients, primary production and plankton), even during relatively high flow (see Freshwater

Inflow Index). This reduces the amount of fresh water that flows into the estuary and can decrease inflow to levels below important thresholds for floodplain inundation, habitat creation and sediment transport. And finally, the lower reaches of the estuary's largest tributary rivers, the Sacramento and San Joaquin Rivers, are confined by man-made levees that prevent or restrict inundation of adjacent floodplains during high flow events. Thus, even under high flow conditions, adjacent floodplains that would have been inundated if there were no levees are not. In essence, many of the estuary's tributary rivers have been disconnected from their floodplains, reducing or eliminating creation of ecologically important floodplain habitat.

The State of the Estuary Report uses two indicators to measure and evaluate the frequency (or "how often?"), magnitude ("how much?") and duration ("how long?") of ecologically important flood events. The Yolo Floodplain Flows indicator measures seasonal inflows into the Delta (the upstream region of the San Francisco Estuary) from the Yolo Bypass, the large, partially managed floodplain immediately upstream of the Estuary in the lower Sacramento River basin. The Flood Inflows indicator measures flood events in terms of high volume freshwater inflows to the Bay from the Delta and the Sacramento-San Joaquin watershed.

II. Data Source

Each of the indicators was calculated for each year using daily freshwater inflow data from the California Department of Water Resources (CDWR) DAYFLOW model (Delta inflow from the Yolo Bypass, QYOLO, for the Yolo Floodplain Flows; Delta outflow, QOUT, for Flood Inflows to the Bay; and Sacramento River flow at Freeport, QSAC, for calibration and development of reference conditions for the Yolo Floodplain Flows indicator). 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 1930-2014, although data for Yolo Bypass flows are only available for 1940-2014.[2] Additional information on unimpaired Sacramento River flows and Delta outflow (or Bay inflow), used to inform development of reference conditions and interpret indicator results, was from CDWR's California Central Valley Unimpaired Flow dataset.[3]

III. 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.

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

[2] Dayflow data for Yolo Bypass discharges, as compared to other potentially applicable data on Sacramento River flow or stage, Yolo Bypass inflows or inundation levels, was selected for calculation of this indicator based on the long record, completeness and quality of the data, as well as its easy accessibility.

[3] This 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.

However, examination of unimpaired flow and flood data records as well as biological information on floodplain habitat, productivity dynamics, and utilization for spawning, rearing and juvenile salmonid outmigration provide useful information for establishing ecologically relevant threshold levels and reference conditions for flood frequency, magnitude and duration.

For each indicator and its frequency, magnitude and duration component metrics, a primary reference condition, the quantitative value against which the measured value was compared, was established. Measured values that were higher than the primary reference condition were interpreted to mean that aspect of flood flow conditions met the CCMP goals and corresponded to "good" ecological conditions. Specific information on the primary reference condition and additional intermediate reference conditions is provided below for each indicator.

Effects of Water Year Type on Flood Flows and the Indicators: 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. Even in the current system, in which flows are highly altered by dams and water diversion, high volume flood flows are larger and occur for more frequent and longer durations in wet years compared to drier years. However, for evaluation of these two indicators, water year type was not considered. Instead the indicators measure actual flow conditions for each year, and those measured levels are compared to a single reference condition that does not vary with water year type. Therefore, measured values for frequency, magnitude and duration of flood flows and the evaluation results relative to ecological condition and ecological services provided by flood flows (i.e., “good” v “poor”) are lower in dry years (and multi-year droughts) than in wetter years. (In contrast, the Peak Flows indicator of the Freshwater Inflow Index measures changes in the number of days of flood flows compared to unimpaired flow conditions that have been normalized to account for difference in water year type.)

IV. Indicators

A. Yolo Floodplain Flows indicator

1. Rationale

The Yolo Bypass is a designated floodway located west of the Sacramento River and north of the Delta (Figure 1). The bypass conveys flood flows from the Sacramento Valley, including the Sacramento River, Feather River, American River, Sutter Bypass, and westside streams, directly into the northern Delta at Cache Slough. Inundation of the Yolo Bypass is largely controlled by the Fremont Weir (completed in 1924), located on the Sacramento River: during high flow events, the Sacramento River overtops

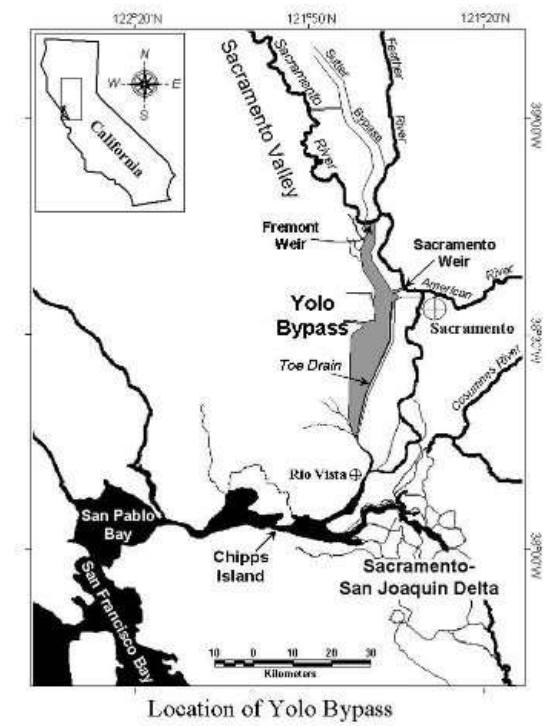

Figure 1. Location of the Yolo Bypass. Source: http://www.americanrivers.org/initiative/floods/projects/yolo-bypass-and-the-fremont-weir

the weir and water flows into the Bypass, inundating up to 60,000 acres of shallow floodplain habitat.

In the Sacramento-San Joaquin watershed, floodplain habitat is most ecologically valuable during the later winter and spring, the period when high flows would typically occur (see Freshwater Inflow Index, Figure 2). In addition to its high primary and secondary productivity, many species use floodplain habitat for spawning, rearing and migration (Sommer et al. 2001; Schemel et al. 2004; Feyrer et al. 2006a, b; del Rosario et al. 2013).[4] Proposals for managed restoration of seasonal floodplain habitat by modifying the Fremont weir to allow more frequent flooding of the Yolo Bypass are prominent elements of Bay-Delta ecosystem restoration planning efforts and species protection plans but none have been implemented yet.

2. Methods and Calculations

The Yolo Floodplain Flows indicator uses three component metrics to assess the frequency, magnitude and duration of occurrence of flood flows from the Yolo Bypass into the San Francisco Estuary during late winter and spring of each year.

Frequency was measured as:

> # of years in the past decade (i.e., ending with the measurement year) with Yolo Bypass flows >10,000 cubic feet per second (cfs) for >45 days during February-June period.[5]

Magnitude was measured as:

> average Yolo Bypass flow (cfs) for the 45 days of highest flows during the February-June period.

Duration was measured as:

> total # days during the February-June period with Yolo Bypass flows >10,000 cfs.[4]

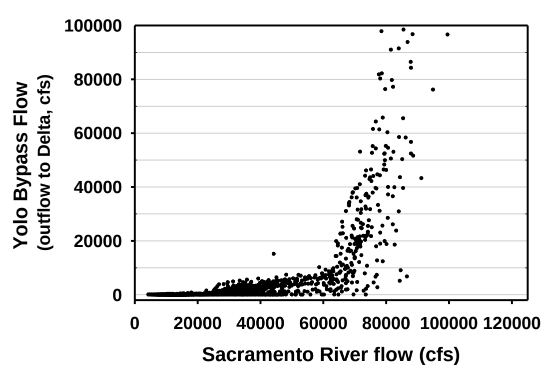

Figure 2. The relationship between Sacramento River flows and Yolo Bypass inflows to the San Francisco Estuary. Data source: California Department of Water Resources, Dayflow.

The late winter-spring period was used based on biological studies that demonstrate the ecological importance of floodplain habitat during this period (Sommer et al. 2001; Schemel et al. 2004; Feyrer et al 2006b; del Rosario et al. 2013). The Yolo Bypass flow level of >10,000 cfs was established based on examination of the relationship between Sacramento River flows and Yolo Bypass flows, which indicated that this level of Yolo Bypass flows, which corresponds to Sacramento River flows of approximately 60,000

[4] The references cited here are only some of the extensive published research on the Yolo Bypass. A comprehensive list and web links to access these and other articles is available at: http://www.water.ca.gov/aes/yolo/yolo_pubs.cfm.
[5] Neither the 45-day period used as part of the reference conditions or nor the count of numbers of days with Yolo Bypass flows >10,000 cfs used in metric calculations required that these days be consecutive.

cfs, is a threshold at which Yolo Bypass flows increased markedly with relatively small increases in Sacramento River flow (Figure 2). The time period of 45 days was based on the time needed for reproduction of splittail, a native floodplain spawner, including access the floodplain, spawning, egg incubation and larval rearing and migration downstream to the Delta (Sommer et al. 1997; Feyrer et al. 2006b). It is likely that, following an initial inundation event and Yolo Bypass flows >10,000, the Yolo Bypass remains inundated for some days after outflows from the floodplain fall below the 10,000 cfs threshold and reference condition used of the indicator metrics; therefore flood events that meet the (non-consecutive) 45 day reference condition threshold may in fact inundate the Yolo Bypass for more than 45 days.

For each year, the Yolo Floodplain Flows 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 Reference Conditions section below.

3. Reference Conditions

The primary reference conditions for the component metrics of the Yolo Floodplain Flows indicator were established as Yolo Bypass flow magnitude of >10,000 cfs for at least 45 days during the February through June period in at least 3 out of 10 years. The bases for the 10,000 cfs and 45 days primary benchmarks are described above. The primary reference condition for frequency was based on an ecological objective to provide spawning habitat for splittail and outmigration and rearing habitat for young salmonids with a return period, 3 out of 10 years, that was relevant to the species’ population dynamics.[6] Yolo Bypass flows that met or exceeded these benchmarks were considered to reflect “good” conditions and meet the CCMP goals. Additional information on Yolo Bypass flows under actual flow conditions (Figure 2), unimpaired Sacramento River flows, and primary and secondary productivity dynamics on the floodplain (e.g., Schemel et al. 2004) was used to develop the other intermediate reference condition levels. Table 1 below shows the quantitative reference conditions that were used to evaluate the results of the component metrics for the Yolo Floodplain Flows indicator.

Table 1. Quantitative reference conditions and associated interpretations for results for each of the three component metrics of the Yolo Floodplain Flows indicator. The primary reference condition, which corresponds to “good” conditions, is in bold italics.

Yolo Floodplain Flows				
Quantitative Reference Conditions			Evaluation and Interpretation	Score
Frequency	Magnitude	Duration		
≥5 years out of 10	>20,000 cfs	>60 days	“Excellent,” similar to unimpaired conditions	4
≥3 years out of 10	***>10,000 cfs***	***>45 days***	***“Good,” meets CCMP goals***	***3***
≥2 years out of 10	>5,000 cfs	>15 days	“Fair”	2
≥1 years out of 10	>2,000 cfs	>5 days	“Poor”	1
0 years out of 10	≤2,000 cfs	≤5 days	“Very Poor,” chronic absence of floodplain habitat	0

[6] Splittail live for 5 to 7 years and can spawn in multiple years (Sommer et al. 1997). Chinook salmon typically return to spawn as 2- to 4-year old fish; therefore creation of floodplain migration habitat in 3 of 10 years would provide benefit to approximately one third of the salmon population (more information available at: http://www.nmfs.noaa.gov/pr/species/fish/chinook-salmon.html.

4. Results

Results of the Yolo Floodplain Flows indicator are shown in Figures 3 and 4.

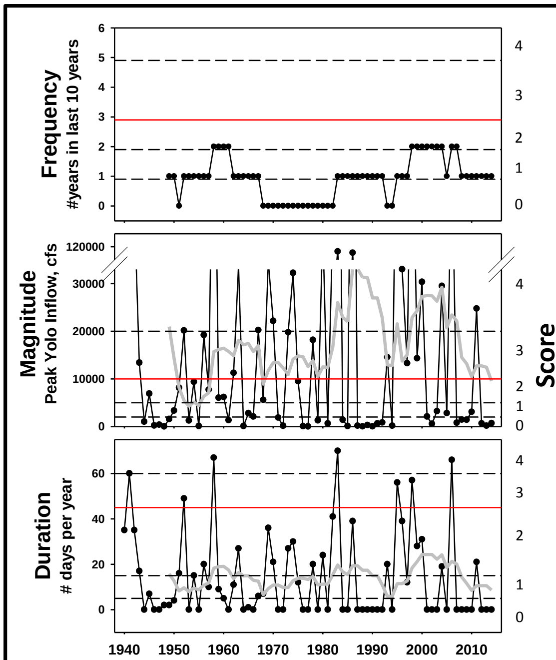

Figure 3. Results of the frequency (top panel), magnitude (middle panel) and duration (bottom panel) component metrics of the Yolo Floodplain Flows indicator. Score is shown on the right Y axis. Each point shows the result for that year and, for the magnitude and duration metrics, the heavy solid grey line shows the 10-year running average. The horizontal red red and dashed lines show the reference conditions for each metric and the numeric score is shown on the right Y axis.

The frequency of creation of inundated floodplain habitat in the Yolo Bypass is low (Figure 3, top panel).

During the past 75 years, the Yolo Bypass has flooded and discharged flows greater than 10,000 cfs for 45 days during the late winter and spring in an average of only one year out of 10 years (10% of years; range 0-20% of years). For a 15 year period from 1968 to 1982, the Yolo Bypass never flooded to the primary reference conditions levels. Based on the relationship between Sacramento River flows and Yolo Bypass flows (Figure 2), this is much less frequent than the Yolo Bypass would have flooded under unimpaired conditions (and with the current Fremont Weir configuration), when it would have flooded with at least 10,000 cfs of flow for at least one month in 54% of years and for at least two months in 26% of years. The last time the Yolo Bypass flooded with >10,000 cfs for at least 45 days was eight years ago, in 2006. Based on frequency of occurrence, floodplain flow and habitat conditions have been consistently poor or very poor.

The magnitude of flood flows from the Yolo Bypass is variable and has not changed over time (Figure 3, middle panel).

Floodplain inundation, as measured by the magnitude of flood flows from the Yolo Bypass is highly variable and, over the 75-year data record, has not changed significantly (regression, $p>0.5$). Since 1940, average flood flows from the Yolo Bypass have been greater than10,000 cfs in 39% of years. The highest flows from the Yolo Bypass occurred in 1983 and 1986, when floodplain discharge to the Delta exceeded 10,000 cfs for several months. The last time average Yolo Bypass flood flows were greater than 10,000 cfs was in 2011. In 2014, a critically dry year, the average of the highest 45 days of late winter-spring flows from the Yolo Bypass was less than 700 cfs.

The duration of flood flows from the Yolo Bypass is low in most years (Figure 3, bottom panel).

Flood flows in excess of 10,000 cfs have occurred for more than 45 days in only 7 of the past 75 years (9% of years). In 34 of 75 years (45% of years) there were no days with Yolo Bypass flood flows greater than 10,000 cfs. The duration Yolo Bypass flood flows is lower than would have occurred under unimpaired conditions: based on unimpaired Sacramento River flows, the Yolo

Bypass would flood with monthly average flows greater than 10,000 cfs for at least one month in most years and at least two months a quarter of years. Flood flow duration is highly variable and has not changed over time (regression, p>0.5). The last time flood flows exceeded 10,000 cfs for 45 days was in 2006. In 2014, Yolo Bypass flows never exceeded 10,000 cfs during the late winter or spring seasons.

Results of the Flood Events indicator, which combines the results of the frequency, magnitude and duration metrics, are shown in Figure 4.

Floodplain flows on the Yolo Bypass are too rare, too low and too short to support ecological processes.
Although Yolo Bypass flows exceed the 10,000 cfs reference condition threshold in more than a third of years, the duration of the those flows is too short to stimulate and support ecological processes and produce ecologically valuable floodplain habitat, as they are defined by the reference conditions established for this indicator. As a result, the frequency of occurrence of "good" floodplain conditions is too low to support important ecological processes in the upstream reaches of the San Francisco Estuary and provide environmental benefits on a relevant timeframe to the population dynamics of floodplain-dependent species. Based on the indicator, the ecological and habitat conditions provided by Yolo floods flows have been "poor" or "very poor" in 70% of years.

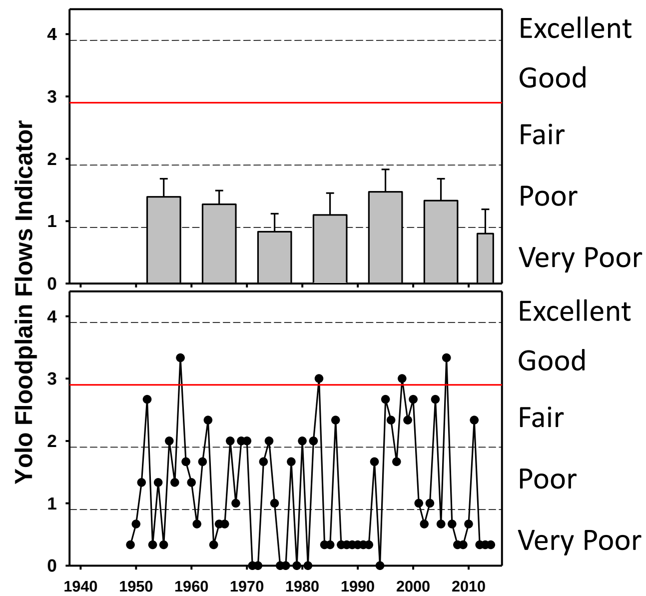

Figure 4. Results for the Yolo Floodplain Flows indicator, which combines the results of the frequency, magnitude and duration component metrics (Figure 3) for 1949 to 2014. The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red and dashed lines show the reference conditions and the indicator evaluation categories are at right.

Based on the Yolo Floodplain Flows indicator, CCMP goals to restore healthy estuarine habitat and function have not been met.
For the past 75 years, the frequency, magnitude and duration of inundation the Yolo Bypass and creation of floodplain habitat immediately upstream of the estuary, have been insufficient to provide ecologically important conditions for primary and secondary productivity, and spawning, downstream migration and rearing of estuarine and anadromous fishes. Since the early 1990s, when the CCMP was implemented, flood conditions have been "good" in only 2 years (8% of years) and have been "very poor" in 13 years (52% of years).

B. Flood Inflows indicator

1. Rationale

High volume, flood inflows of fresh water to the San Francisco Bay occur following winter rainstorms and during the spring snowmelt. Flood inflows transport sediment and nutrients to the Bay, 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, flood flow events are also a form of "natural disturbance" (Kimmerer 2002, 2004; Moyle et al., 2010).

2. Methods and Calculations

The Flood Events indicator uses three component metrics to assess the frequency, magnitude and duration of occurrence of high inflow, or flood events, in the San Francisco Estuary each year.

Frequency was measured as:

# of years in the past decade (i.e., ending with the measurement year) with Bay inflows >50,000 cubic feet per second (cfs)[7] for more than 90 days during the year.

Magnitude was measured as:

average inflow (cfs) during the 90 days of highest inflow in the year.

Duration was measured as:

days during the 90 days of highest inflow that inflow>50,000 cfs.

High volume, flood flow was defined as the 5-day running average of actual daily freshwater Bay inflow>50,000 cfs. Selection of this threshold value was based on two 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; and 2) flows of this magnitude shift the location of low salinity habitat downstream to 50-60 km[8] into Suisun and upper San Pablo Bays (depending on antecedent conditions), driving primary and secondary productivity and providing favorable conditions for many estuarine invertebrate and fish species (Jassby et al. 1995; Kimmerer 2002, 2004).

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 Reference Conditions section below.

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[7] Freshwater inflow levels were measured as the 5-day running average of "Delta outflow."

[8] 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.

3. Reference Conditions

The primary reference conditions for the component metrics of the Flood Inflows indicator were established as Bay inflow (or Delta outflow) magnitude of >50,000 cfs for at least 90 days during the water year in at least 4 out of 10 years. The basis for the 50,000 cfs benchmark is described above. The primary reference conditions for frequency and duration were based on examination of unimpaired Bay inflows (or Delta outflows) that showed that an average of 5 out of 10 years (51% of years) had four or more months with average flows >50,000 cfs and an additional 13% of years had three months of flows of this magnitude. Bay inflows that that met or exceeded these benchmarks were considered to reflect "good" conditions and meet the CCMP goals. Additional information on unimpaired Bay inflows and current regulatory standards for seasonal Bay inflows was used to develop the other intermediate reference condition levels. Table 2 below shows the quantitative reference conditions that were used to evaluate the results of the component metrics for the Flood Inflows indicator.

Table 2. Quantitative reference conditions and associated interpretations for results for each of the three component metrics of the Flood Inflows indicator. The primary reference condition, which corresponds to "good" conditions, is in bold italics.

Flood Inflows				
Quantitative Reference Conditions			**Evaluation and Interpretation**	**Score**
Frequency	**Magnitude**	**Duration**		
≥6 years out of 10	>100,000 cfs	>120 days	"Excellent," similar to unimpaired conditions	4
4 or 5 years out of 10	***>50,000 cfs***	***>90 days***	***"Good," meets CCMP goals***	***3***
2 or 3 years out of 10	>30,000 cfs	>45 days	"Fair," similar to current regulatory standards	2
1 year out of 10	>10,000 cfs	>10 days	"Poor," below current regulatory standards	1
0 years out of 10	≤10,000 cfs	≤10 days	"Very Poor," Bay inflows "flatlined"	0

V. Results

Results of the Flood Inflows indicator are shown in Figures 5 and 6.

The frequency of occurrence of flood events has declined (Figure 5, top panel).
Frequency of occurrence of high inflow flood events in the San Francisco Bay 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 2014, the estuary experienced only one year (2006) with a flood event that met the primary reference conditions.

Flood magnitude has not changed (Figure 5, middle panel).

Flood magnitude, as measured by average inflows during the 90 days with highest inflows per year, is highly variable and, over the 85-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 68,408 cfs during the last two decades and not significantly different (Mann-Whitney Rank Sum test, p=0.39). High inflows during the most recent decade (2005-2014) are somewhat lower, 51,416 cfs on average, but not significantly different than pre-dam levels (t-test, p=0.16).

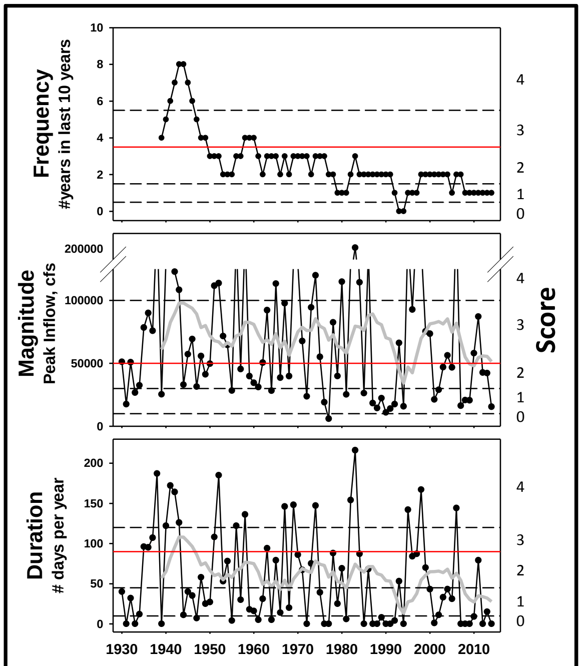

Figure 5. Results of the frequency (top panel), magnitude (middle panel) and duration (bottom panel) component metrics of the Flood Inflows indicator. Score is shown on the right Y axis. Each point shows the result for that year and, for the magnitude and duration metrics, the heavy solid grey line shows the 10-year running average. The horizontal red and dashed lines show the reference conditions for each metric and the numeric score is shown on the right Y axis.

The duration of flood events has declined (Figure 5, bottom panel).

The number of days per year with inflows above the 50,000 cfs flood threshold is also highly variable. Prior to construction of the major dams in the estuary's watershed (the pre-dam period, 1930-1943), high inflows occurred for an average of 82 days per year, significantly more often than during the last decade (2005-2014) when there was an average of just 28 days per year (t-test, p<0.05). Regression analysis also suggests this decline, although due to the variability of data, the decline is not statistically significant (regression, p=0.075). In 2014, a critically dry year, there were zero days with inflows >50,000 cfs.

Results of the Flood Inflows indicator, which combines the results of the frequency, magnitude and duration metrics, are shown in Figure 6.

High inflow flood conditions have declined.

Results of the indicator reveal a steady and significant decline in high inflow, flood event conditions in the Bay (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 85-year data record, 2011, also a wet year, and 2004. Declining flood event conditions were driven by the decline in flood duration, which has fallen by more than 60% and the resultant decline in the frequency of flood events that met the primary reference condition criteria, which has fallen more than 75%.

Based on the Flood Inflows indicator, CCMP goals to restore healthy estuarine habitat and function have not been met. The indicator shows that, for the past five decades, flood inflow conditions, an important physical and ecological process in the Bay, have been mostly "fair" or "poor." Since the early 1990s, when the CCMP was implemented, flood conditions have been "good" in only three years (12% of years) and have been "poor" in 68% of years.

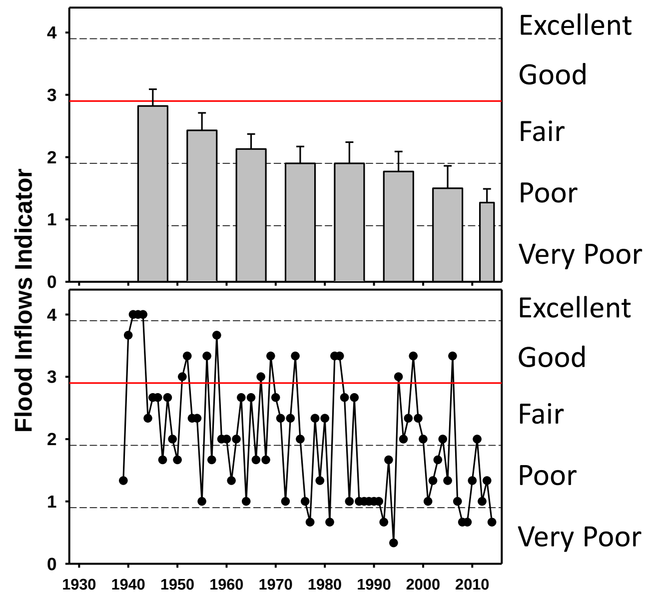

Figure 6. Results for the Flood Inflows indicator, which combines the results of the frequency, magnitude and duration component metrics (Figure 5) for 1939 to 2014. The top panel shows results as decadal averages±1 SEM (and for five years for 2010-2014) and the bottom panel shows results for each year. The horizontal red and dashed lines show the reference conditions and the indicator evaluation categories are at right

VI. References

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State of the Estuary Report 2015

Sidebar Source Material

PROCESSES – Blooms

Prepared by
David Senn, San Francisco Estuary Institute

Nutrients Sidebar

Too Much of a Good Thing?

Nitrogen (N) and phosphorus (P) occur naturally in estuaries, and serve as essential nutrients that promote growth of algae (or "phytoplankton") and other aquatic plants, and thereby support estuarine food webs. However, human activities have dramatically increased N and P loads to estuaries worldwide, in many cases exceeding the amount these ecosystems can handle, leading to severe impacts to habitat, fisheries, and recreation.

The Bay-Delta receives high nutrient loads from over 40 wastewater treatment plants, agricultural runoff, and, to a lesser extent, stormwater runoff. Nutrient concentrations in many areas of the Bay-Delta greatly exceed those in other US estuaries where condition has been impaired by nutrient pollution. However, the Bay-Delta has exhibited resistance to some of the classic symptoms of excessive nutrient concentrations, such as high phytoplankton abundance and low dissolved oxygen, which have plagued other nutrient-enriched estuaries. High turbidity and strong tidal mixing in the estuary tend to limit light levels and algae growth, causing a low proportion of available nutrients to be converted into algae biomass. Large populations of filter-feeding clams have further limited phytoplankton accumulation by grazing the algae from the water column.

However, observations over the past 10 years suggest that the Bay-Delta may no longer be as resistant to its high nutrient loads. These observations include:

- a 2-3 fold increase in summer-fall algal biomass in South Bay since 1999;
- frequent detections in the Bay of algal species that have been shown in other nutrient-rich estuaries to form harmful blooms, and blooms in the Delta of the harmful phytoplankton *Microcystis spp.* that produces the toxin microcystin,
- frequent detection throughout the Bay of two algal toxins, microcystin and domoic acid, and elevated levels of microcystin in the Delta during *Microcystis* blooms,
- an unprecedented red tide bloom (*Akashiwo sanguinea*) in Central Bay (Fall 2004);
- studies hypothesizing that the chemical forms of nitrogen (i.e., NH_4^+ vs. NO_3^-) can decrease phytoplankton productivity or alter their community composition;
- dramatic increases in the density and areal coverage of aquatic macrophytes in the Delta; and
- low dissolved oxygen in some Bay sloughs and creeks, and Delta ship channels.

How should we interpret these observations? In some cases, the observations are concerning because they indicate marked changes in biological response, e.g., increased South Bay algal biomass and Delta macrophyte biomass. In other cases, like the detection of algal toxins, these are new additions to list of parameters that scientists and managers are using to assess condition in the Bay-Delta – so it is impossible to determine whether these toxins appeared only recently or have always been present. Nonetheless, the detection of these toxins, even at low levels, is noteworthy and signals the need for further investigation.

The Bay-Delta is also a large and complex system, and there is no single answer for "good" condition. The Bay-Delta is comprised of multiple subembayments and habitats that receive different nutrient loads. Factors that can strongly influence biological response to nutrients -- flow (i.e., residence time), mixing, suspended sediment concentrations (i.e., light levels), temperature, grazing – vary spatially and seasonally, and have also changed over time.

The complexity of mechanisms controlling the biological response of the Estuary to nutrient loading highlights the importance of continued monitoring, research, and synthesis of science.

Management Response

To address growing concerns about adverse nutrient impacts in the Estuary, both the San Francisco Bay Regional Water Quality Control Board (SFBRWQCB) and the Central Valley Regional Water Quality Control Board (CVRWQCB) have launched efforts to better understand potential nutrient impacts and identify appropriate management actions. The SFBRWQCB worked collaboratively with stakeholders to develop the San Francisco Bay Nutrient Management Strategy in 2012, which lays out an overall approach for building the scientific understanding to support well-informed nutrient management decisions. The CVRWQCB began a similar effort for the Delta in 2014. Scientific investigations and monitoring are underway, or planned, in both regions to

- identify appropriate nutrient-related indicators,
- monitor to determine when and where adverse impacts occur,
- quantify nutrient loads to the estuary,
- model nutrient transport and fate within the Estuary,
- determine the protective nutrient levels, and
- ultimately, identify effective management actions.

Establishing Nutrient-related Indicators in the Bay-Delta

Work is underway to identify appropriate indicators of nutrient-related impacts and the methods needed to accurately measure and interpret them. Because complex physical and biological factors influence an estuary's response to nutrients, nutrient concentrations alone do not tell the whole story. Instead, parameters that track the response of the Estuary to nutrients are considered to be more meaningful indicators. Measurement and interpretation of these indicators can be challenging for the following reasons:

- Spatial variability: Some of the potential indicators are familiar – for example chlorophyll and dissolved oxygen. However, even for these traditional indicators, interpretation can be complicated. For example, the amount of chlorophyll that is associated with a high risk of adverse impact may be different in the open Bay than in shallow slough channels, or even differ between subembayments. In addition, there may be site-specific differences between dissolved oxygen requirements, because some habitats naturally experience wider variations in dissolved oxygen than others (e.g., wetlands) and the local fish and benthic organisms are adapted to this variability.

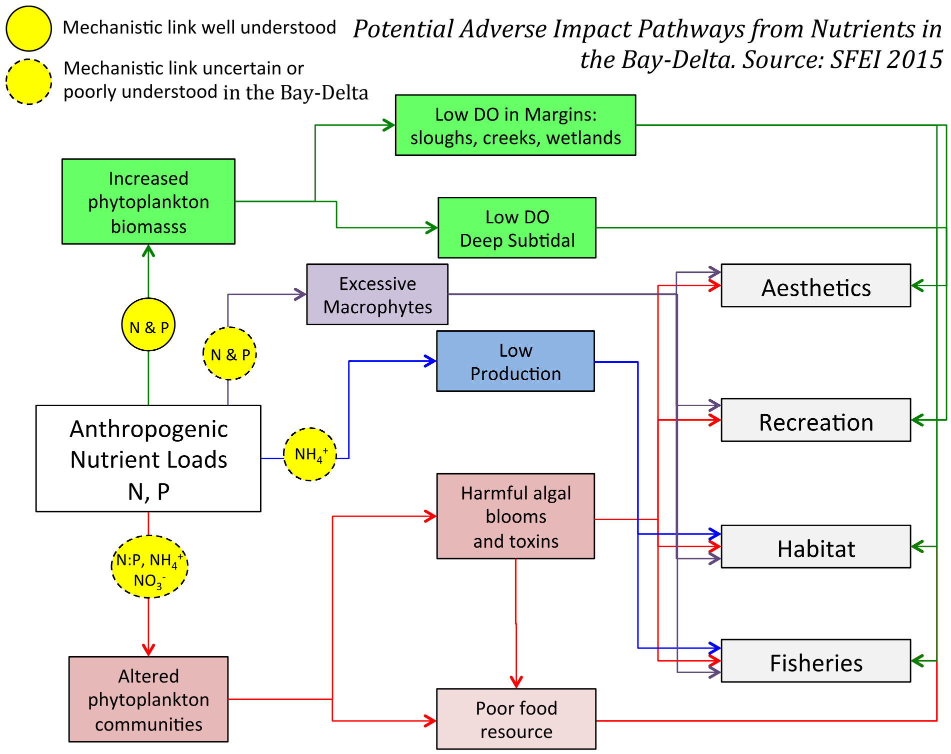
- Our evolving understanding of nutrient impacts: Some proposed adverse impact pathways for the Bay-Delta are still under investigation and have different indicators than the "classic eutrophication" indicators (chlorophyll, dissolved oxygen). Harmful algal species and associated algal toxins are examples. Identifying accurate methods to measure and interpret new types of indicators will require research studies and modeling (left).

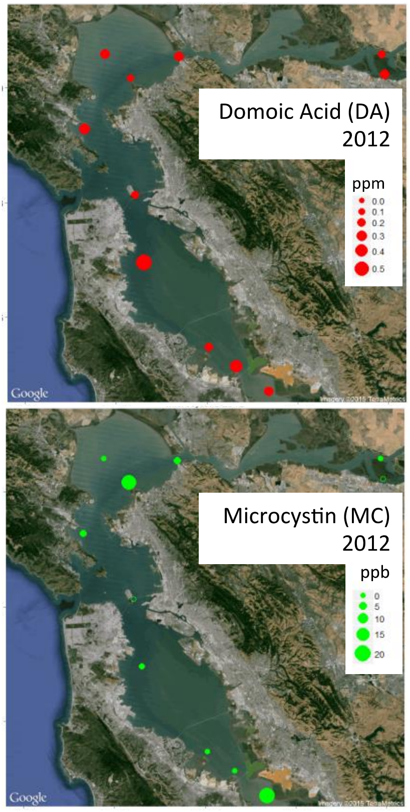
Future Outlook

In the next few years, research for the Bay Nutrient Management Strategy will focus on developing a Science Plan, conducting detailed field studies investigating nutrient cycling and ecosystem response, and developing a nutrient water quality model. For the Delta, the CVRWQCB will convene scientific workgroups regarding nutrients, macrophytes, cyanobacteria, and modeling in 2015 and plans to complete a Nutrient Research Plan by the end of the same year. Nutrient concentrations will likely change substantially in the Delta and Suisun Bay within 10 years due to treatment upgrades at the Sacramento region's wastewater facility. This ecosystem-scale 'experiment' provides a unique opportunity to study the northern Estuary's response to major changes in nutrient inputs. A well-designed pre- and post-upgrade science and monitoring program will be needed to document and accurately interpret any changes in ecosystem response.

Supporting Graphs and Figures

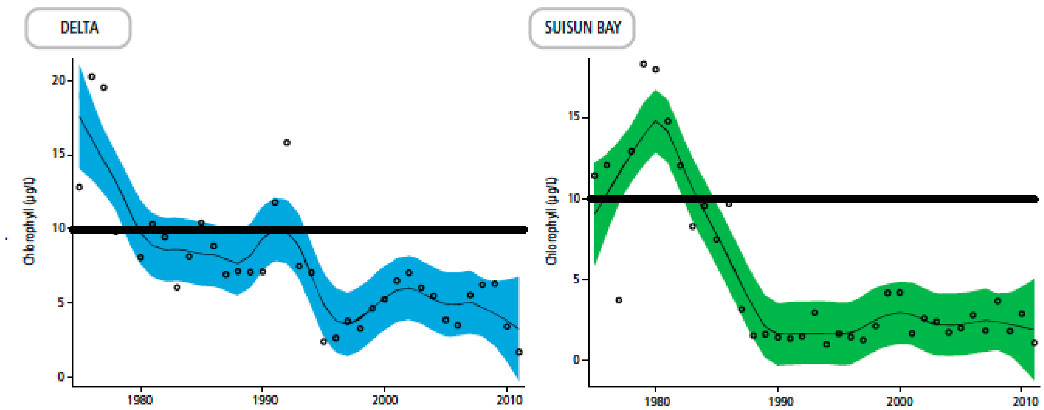
Potential Adverse Impact Pathways from Nutrients in the Bay-Delta. Source: SFEI 2015





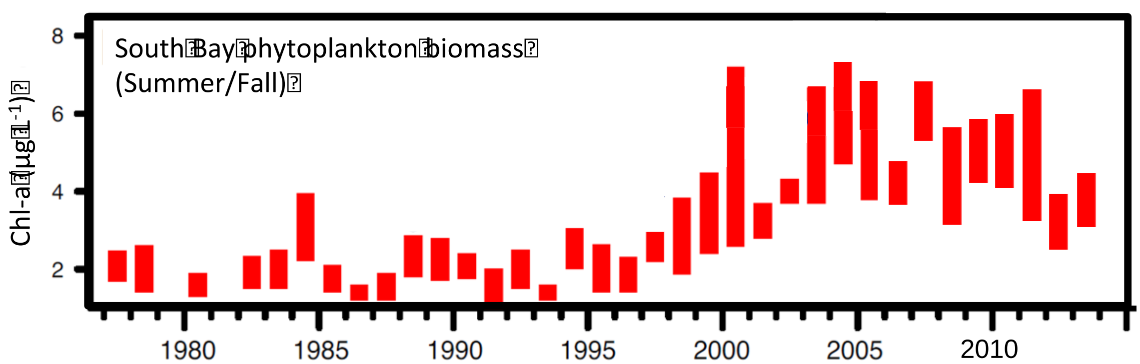
All units wet weight

Domoic acid and microcystin measured in mussels deployed as part of 2012 Mussel Watch monitoring. Circle size indicates measured concentration; empty circles indicate nondetect. Domoic acid was detected in all samples, and microcystin was detected in all but one sample. Although ubiquitous, domoic acid concentrations << 20 ppm regulatory thresholds for shellfish consumption. Source: M Peacock (UCSC), pers. comm.



Footnotes: Delta- and Suisun Bay-wide averages. The trend line displays a Loess fit and the shaded area represents the 90% confidence limits for the trend line. Data from the IEP Environmental Monitoring Program. Monitoring stations included in the Delta-wide averages: C3, C3A, C7, C9, C10, C10A, D2, D4, D10, D11, D12, D14A, D15, D16, D19, D22, D24, D26, D28A, MD7, MD7A, MD10, MD10A, P2, P8, P10, P10A, P12, P12A. Suisun Bay: D2, D7, D8, D9, NZS42, NZ032, S42.

Data source: Tiffany Brown, California Department of Water Resources.



Interquartile range of surface chl-a concentration, a measure of algal biomass, during summer and fall at station between the Bay Bridge and Dumbarton Bridge. Source: Cloern et al. 2007, SFEI 2015

Primary N and P forms: N and P are present in multiple inorganic and organic forms in estuarine waters, and biologically-mediated reactions convert between forms, including uptake by algae and other plants

NO_3^-	nitrate	o-PO_4	Dissolved orthophosphate
NH_4^+	ammonium	❖-PO_4	Particle-complexed PO_4
DIN	dissolved inorganic N NO_3^- + NH_4^+ + minor forms		
PON, DON	Dissolved and particulate organic N	POP, DOP	Dissolved and particulate organic P

State of the Estuary Report 2015

Sidebar Source Material

PROCESSES – Blooms

Submerged and Floating Aquatic Vegetation Cover in the Sacramento-San Joaquin Delta

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Center for Spatial Technologies and Remote Sensing, University of California, Davis

Submerged and Floating Aquatic Vegetation Cover in the Sacramento-San Joaquin Delta

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Center for Spatial Technologies and Remote Sensing, University of California, Davis.

Invasive aquatic plants have far-reaching impacts on the Delta ecosystem and are now widespread. We know that aquatic plants change shoreline habitat by slowing water velocities and increasing water clarity, which act as positive feedbacks furthering the establishment and spread of invasive plants. This dense mat of submerged vegetation can offer predatory fishes places to hide and hunt. Meanwhile, native species like Delta smelt who like to stay in open water are more vulnerable to attack because of less turbid water. Such effects can propagate up or down the food chain affecting the entire ecosystem. Invasive aquatic plants also impede boat travel and are difficult and expensive to control. We measured the distribution and acreage of invasive and native aquatic plant species using remote sensing imagery from 2004 to 2008, and again in 2014. This sidebar provides first estimates of the changes in acreage of submerged and floating aquatic species in 2014 compared to the period from 2004 to 2008.

Between 2004 and 2008, submerged aquatic vegetation (SAV) cover, most of which was invasive Brazilian waterweed, reduced from almost 8000 acres to 50% of its 2004 extent (4300 acres). In 2014, the SAV cover has been measured as 6070 acres (Figure 1).

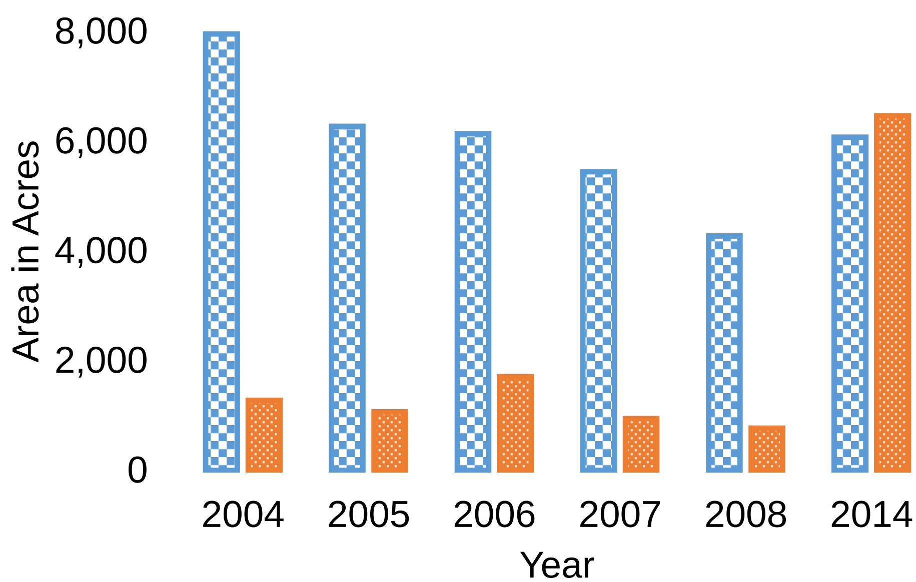

Figure 1: Submerged (blue) and floating aquatic vegetation (orange) cover in the Delta from 2004 to 2014.

From 2004 through 2008, floating species cover varied between 800 to 1700 acres. However, in 2014, floating species cover increased many-fold and now covers 6460 acres. The three dominant floating plants in the Delta from 2004 to 2008 were two invasives, water hyacinth and water primrose, and one native species, pennywort. Pennywort cover was comparable to invasive water hyacinth cover. Water primrose had the least cover. By 2014 however, cover was mainly composed of just the two invasive species, water hyacinth (69%) and water primrose (31%) while pennywort cover was greatly reduced.

The total invaded area in the Delta (SAV + FAV) has increased from the previous recorded maximum of 9,000 acres (in 2004) to more than 12,500 acres. Both SAV and FAV have especially flourished in flooded islands in the Delta, colonizing new areas. These are old subsided islands that flooded when a levee breached and are now like shallow lakes surrounded by remnant levees that protect them to some extent from tidal forces. Below are two examples of flooded islands in the Delta illustrating these changes in distribution.

<u>Rhode</u> <u>Island</u>: In Rhode Island (flooded in 1938), the aquatic plant community shift from SAV to FAV is obvious. Compared with 2008, SAV and FAV cover has increased many-fold. Figure 2 shows their changing distribution for each year that imagery is available while Figure 3 shows the water hyacinth and water primrose distribution in November 2014.

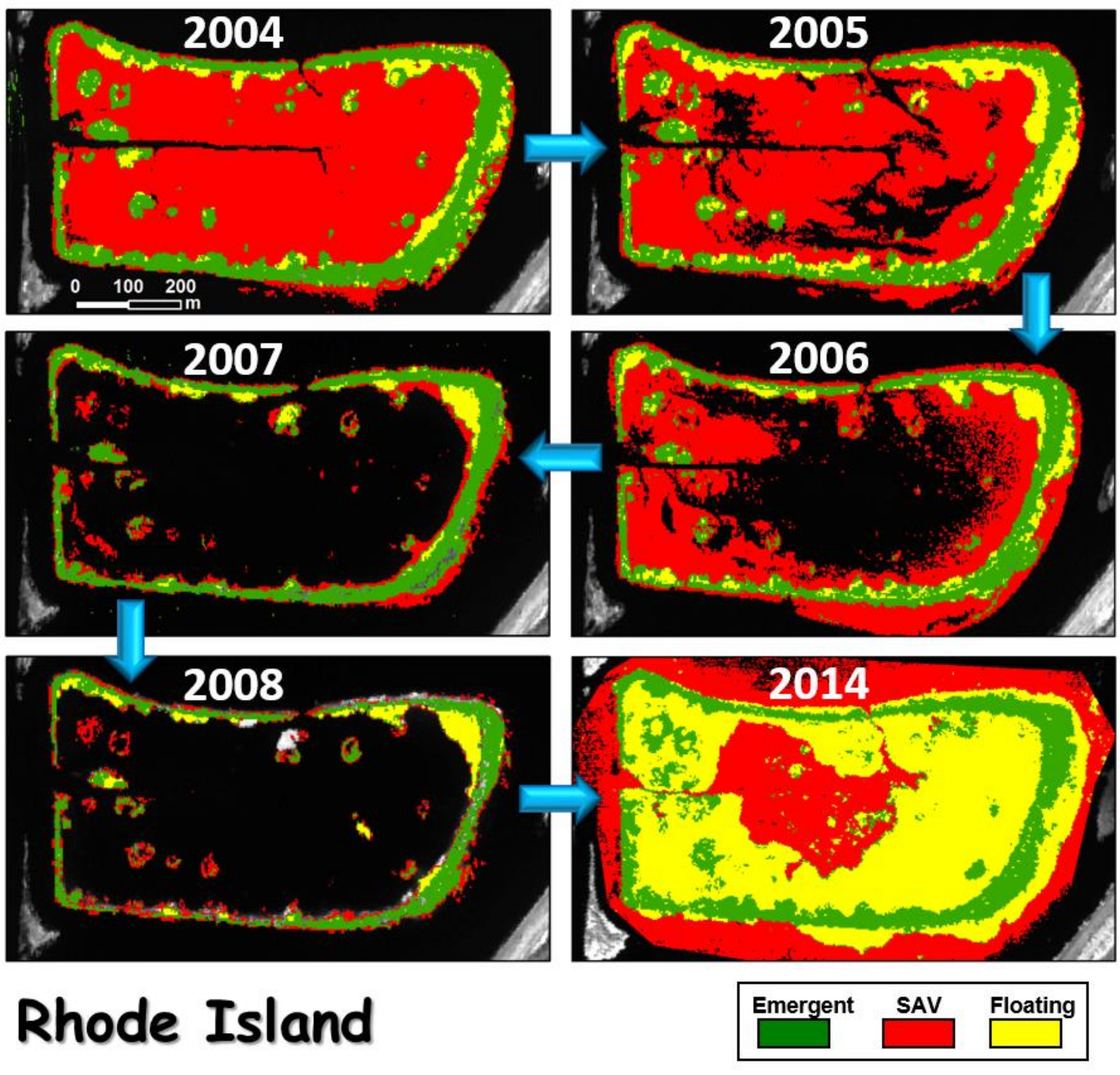

Figure 2: Classification maps of emergent, submerged & floating vegetation in Rhode Island from 2004 to 2014.

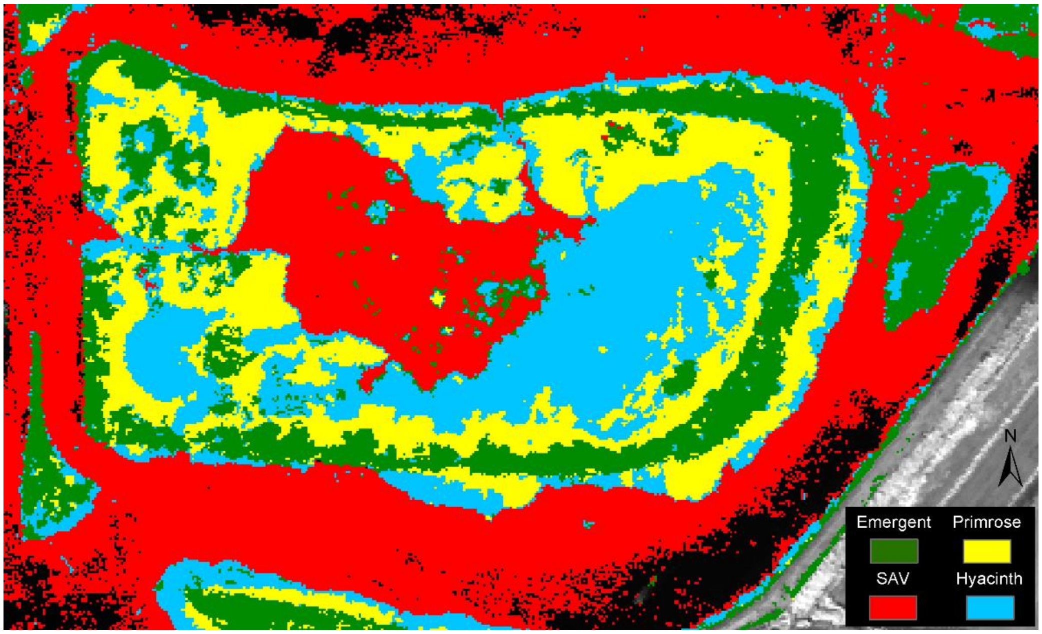

Figure 3: Water hyacinth and water primrose distribution in Rhode Island in November 2014.

Liberty Island: In Liberty Island (flooded in 1998), both SAV and FAV cover has increased over the past 10 years (Figure 4) but the cover of emergent vegetation has also increased as the wetland has expanded outward in a concentric pattern (Figure 4). In previous years, coarse substrate underlying this large, shallow island and wave action due to wind has prevented the establishment of SAV in Liberty Island. But as the emergent and floating species increase in the island, they are providing shelter from these forces to SAV species. Hence, in the past 6 years, between 2008 and 2014, we see a large increase in SAV cover but all of this spread is restricted to the emergent wetlands in the north of the island as expected. The floating vegetation in these wetlands is exclusively water primrose. However, there are a few water hyacinth mats to the south of the island.

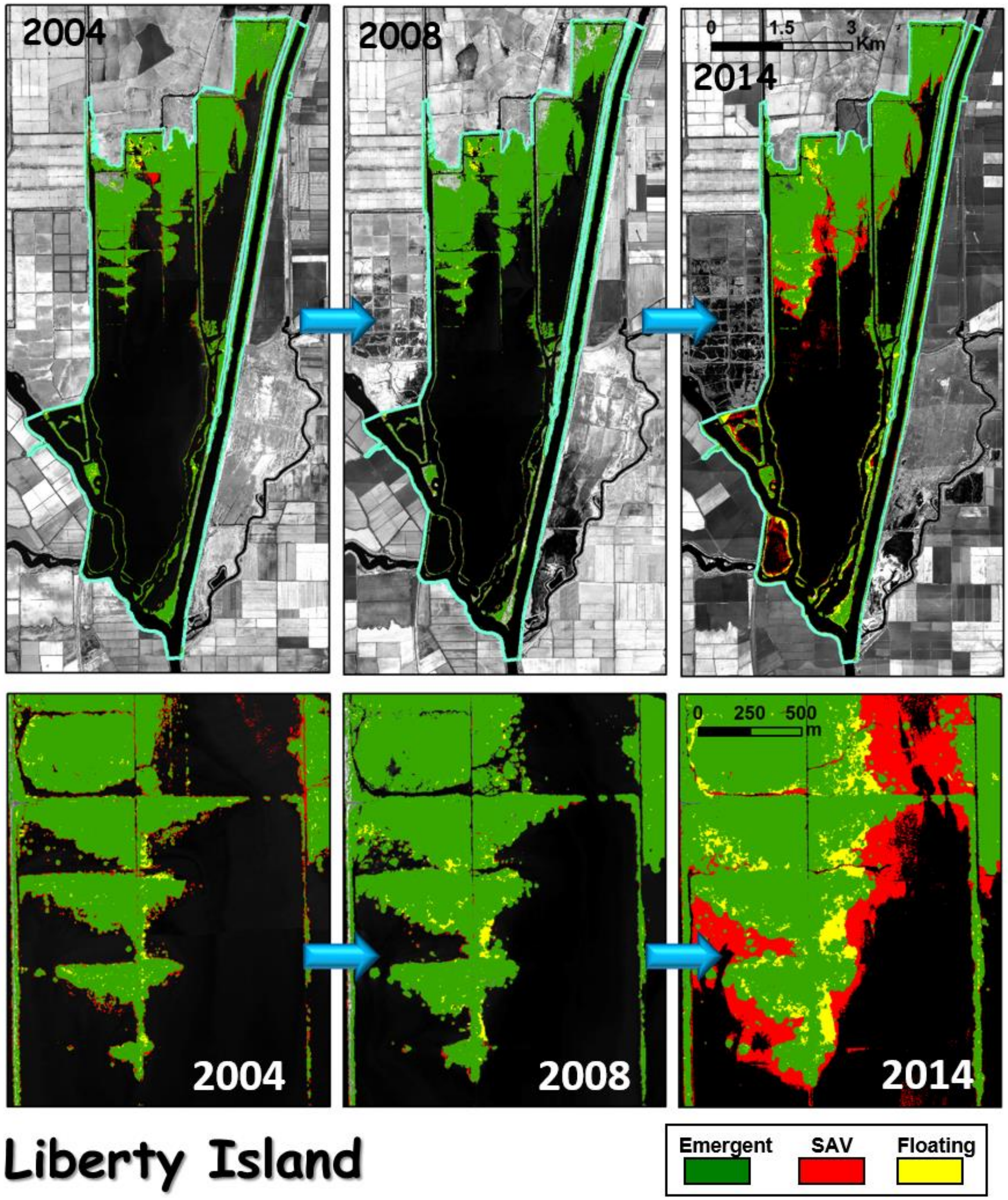

Figure 4: Classification maps of emergent, submerged & floating vegetation in Liberty Island from 2004 to 2014.

There are many possible reasons for the observed changes. In the past few years, permitting problems have delayed spraying of water hyacinth which might have led to its spread. The spraying of water primrose, on the other hand, is not mandated by the government hence there are no control measures to contain the spread of this plant. The prolonged drought has likely reduced water levels and increased shallow habitat with slow moving water ideal for the establishment of SAV and FAV. Coincidently, mild winters and lack of large storms and floodwaters have also favored the establishment and spread of these species. More studies are necessary to tease apart the mechanisms behind the changing distribution of invasive species that are threatening the Delta ecosystem.

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Technical Appendix Combined for WILDLIFE: Upper Estuary Fish and PROCESSES: Fish as Food

(for full appendix, please see wildlife section above)

Fish Assemblage Health Indicators for the Upper San Francisco Bay Estuary, including Suisun Bay, Suisun Marsh, and Delta

Technical Appendix

Alison Weber-Stover and Jonathan Rosenfield, Ph.D.

The Bay Institute

State of the Estuary Report 2015

Summary

PROCESSES – Zooplankton as Food

Prepared by
April Hennessy, California Department of Fish and Wildlife

State of the Estuary Report 2015- Zooplankton Indicator

1. Brief description of indicator and benchmark

Table 1.1

Attribute	Indicator	Benchmarks
Zooplankton	• Mysid and calanoid copepod biomass in Delta and Suisun regions	• Benchmark was the historical average 1974-1986 biomass for each region. At or above benchmark was considered "Good", below benchmark to 25% of benchmark was "fair", and below 25% of benchmark was "poor". Current status determined by 2010-2014 average biomass.

2. Indicator status and trend measurements

Table 1.2

Attribute	Status	Trend	Details
Mysid biomass- Suisun & Delta	Poor	Declined from historic, stable since 2000	Mysid biomass has shown a significant decline since the historical benchmark period and the current status is "poor", due in part to competition with the non-native invasive clam *Potamorcorbula amurensis.*
Calanoid copepod biomass- Suisun	Fair	Declined from historic, stable since 2000	Calanoid copepod biomass has shown a significant decline since the historical benchmark period and the current status is "fair", due in part to competition with and predation from the non-native invasive clam *Potamocorbula amurensis.*
Calanoid copepod biomass- Delta	Good	Increased from historic, stable since 2000	Calanoid copepod biomass has shown a significant increase since the historical benchmark period and the current status is "good".
Zooplankton	Mixed	Declined from historic, stable since 2000	Mysid biomass declined in both the Suisun and Delta regions and the current status is "poor", due in part to competition with the non-native invasive clam *Potamorcorbula amurensis.* In Suisun, calanoid copepod biomass declined and the current status is "fair", due in part to competition with and predation from the non-native invasive clam *Potamocorbula amurensis.* In the Delta, calanoid copepod biomass has significantly increased and the current status is "good".

3. Brief write-up of scientific interpretation

Provide 2-3 sentences to answer the question: What is this indicator?

Provide 2−3 sentences to answer the question: Why is it important?

Zooplankton are small aquatic invertebrates that provide an important trophic link between primary producers and fish. Most larval and juvenile fish eat zooplankton, and some small fish such as Delta Smelt and Longfin Smelt rely on zooplankton for food throughout their lives. To assess trends in fish food resources in the upper San Francisco Estuary (SFE), the Interagency Ecological Program's Zooplankton Study has provided annual zooplankton abundance estimates since 1972. Calanoid copepods and mysids are crustaceans that were chosen for the zooplankton indicator because they are important food items for Delta Smelt and Longfin Smelt, two listed fish species in the upper SFE.

Zooplankton samples were collected by the California Department of Water Resources during their monthly water quality monitoring cruise at 16 to 22 stations using plankton nets. The samples were preserved and returned to the California Department of Fish and Wildlife's Laboratory in Stockton, CA for processing. Average annual biomass, which is a measure of available carbon, of calanoid copepods and mysids was calculated using March through November data from 14 stations (6 in Suisun region and 8 in Delta region, see map) that have been consistently sampled since 1974.

Provide 2−3 sentences to answer the questions: What is the benchmark? How was it selected?

Provide 2−3 sentences to answer the question: What is the status and trend for this indicator?

Since the late 1980s, zooplankton biomass has decreased in most areas of the upper SFE, particularly in the low salinity zone. This decrease has been attributed in large part to the introduction of *Potamocorbula amurensis* in 1986, an invasive clam found in the low salinity zone (see benthic indicator for more about this clam). Competition with *P. amurensis* for phytoplankton, a shared food resource, as well as clam consumption of copepod nauplii (babies) has reduced zooplankton. The historical reference period 1974-1986 was selected as the benchmark, as it is the earliest data available before the disturbance caused by *P. amurensis.* At or above the 1974-1986 average biomass was considered "good", below to 25% of this benchmark was considered "fair", and below 25% of this benchmark was considered "poor". The current status was considered the average 2010-2014 biomass. The trend was determined by a linear slope of the annual data, and the significance of this slope was determined by a Mann-Kendall test ($p<0.005$ was considered significant).

Mysid biomass has declined in both the Suisun and Delta regions of the upper SFE since monitoring began. Since 2001, mysid biomass in both regions has been "poor". The 2010-2014 average biomass was 1.6 milligrams of carbon per cubic meter of water sampled for the Suisun region and 0.4 milligrams of carbon per cubic meter of water sampled for the Delta region, placing the current status of both regions as "poor". There was a significant downward trend in annual mysid biomass for both regions from 1974-2014 ($p<0.001$). However, since 2000 there was no significant trend, indicating that although mysid biomass is lower than it was historically, it does not appear to be declining further.

Calanoid copepod biomass has declined in the Suisun region of the upper SFE since monitoring began. Since 1988, calanoid copepod biomass in the Suisun region has fluctuated between "fair" and "poor", with small peaks occurring during higher flow years such as 2006 and 2011. The current status of calanoid copepods in the Suisun region is "fair". Like the mysids, there was a significant downward trend from 1974-2014 in the Suisun region ($p<0.001$). However, from 2000 through 2014 there was no significant trend, indicating that although calanoid copepod biomass is lower than it was historically, it does not appear to be declining further in the Suisun region.

Calanoid copepod biomass has increased in the Delta region of the upper SFE since monitoring began. In the Delta region calanoid copepod biomass has fluctuated between "good" and "fair" for the entire monitoring period, however biomass during the historical reference period used to establish the benchmark was low. The current status of calanoid copepods in the Delta region is "good". There was a significant upward trend in biomass from 1974-2014 in the Delta region ($p=0.0047$); however from 2000-2014, there was no trend.

Provide 4–6 sentences to answer the questions: What does it mean? Why do we care?

Zooplankton plays a key role in the food web by providing the means for energy to move up the food chain from phytoplankton to fish. The food web of the upper SFE has been highly altered by the introduction of non-native invasive species, particularly the clam *P. amurensis*. The resultant zooplankton decline in the low salinity zone has been implicated as one of the many causes of the pelagic organism decline (POD) which described the dramatic decline of several pelagic fish species. Recovery of listed fish species such as the Delta Smelt and Longfin Smelt relies in part on food availability.

State of the Estuary Report 2015

Technical Appendix

PROCESSES – Zooplankton as Food

Prepared by
April Hennessy, California Department of Fish and Wildlife

5. Technical appendix- Zooplankton Indicator

- **Background and Rationale**
  - **Discuss how the indicator relates to the ecological health of the estuary.**
  - **Include historical information about the indicator and any current programs to evaluate it.**
  - **Explain why this indicator and this calculation approach were chosen**.

Zooplankton is an important component of the pelagic food web, providing a key trophic link between fish and phytoplankton. Most larval and juvenile fish in the upper San Francisco Estuary (SFE) feed on zooplankton, and some smaller fish like Delta Smelt and Longfin Smelt feed on zooplankton throughout their lives. Summer to fall survival of Delta Smelt has been positively linked with zooplankton biomass (Kimmerer 2008).

Monitoring of zooplankton in the upper SFE is conducted by the California Department of Fish and Wildlife's Zooplankton Study as part of the Interagency Ecological Program for the San Francisco Estuary (IEP). Since the late 1980s, zooplankton has decreased in most areas of the upper SFE, particularly in the low salinity zone. This decrease has been attributed in large part to the introduction of *Potamocorbula amurensis* in 1986, an invasive clam found in the low salinity zone (see benthic indicator for more about this clam). Competition with *P. amurensis* for phytoplankton, a shared food resource, as well as predation on copepod nauplii (babies) by *P. amurensis* has reduced zooplankton (Kimmerer and Orsi 1996, Orsi and Mecum 1996, Kimmerer et al. 1994). The decline is particularly evident in Suisun Bay, a region heavily impacted by *P. amurensis.*

Zooplankton biomass, as calculated for this indicator, is an estimate of the relative amount of carbon available in calanoid copepods and mysids for comparison between years, and is not meant to estimate the total carbon available from the entire zooplankton population. Calanoid copepods and mysids were chosen as the representative zooplankton taxa for the indicator because these are important food items for Delta Smelt and Longfin Smelt, 2 listed fish species in the upper SFE (Chigbu and Sibley 1998, Nobriga 2002, Hobbs et al. 2006, Slater and Baxter 2014).

- **Benchmark**
  - **Describe the benchmark and why it was chosen.**

The average biomass during the historical reference period 1974-1986 was chosen as the benchmark, because there is no established standard threshold for "healthy" zooplankton biomass in the upper SFE. Therefore, the earliest available data prior to the disturbance caused by the introduction and spread of *Potamocorbula amurensis* around 1987 was used as the benchmark. The benchmark was established as the 1974-1986 average and was considered "good", less than the benchmark down to 25% of the benchmark was considered "fair", and less than 25% of the benchmark was considered "poor". The current status was reported as the average 2010-2014 biomass. The trend was determined by a linear slope of the annual data and the significance of this slope was determined by a Mann-Kendall test where $p<0.005$ was considered significant.

- **Discuss any limitations of the benchmark and how it might be improved in the future.**

The historical reference period (1974-1986) used to set the benchmark for "good" zooplankton biomass is biologically relevant, since most zooplankton declined after the introduction of *P. amurensis*. However, many human-induced changes occurred in the upper SFE before this period, therefore it probably does not truly reflect a pristine state of zooplankton biomass. The benchmark was based on the average biomass in each region during the historical reference period, therefore in areas like the Delta where the historical biomass was lower the benchmark was lower. Zooplankton data is highly variable, both in time and space, so one way to improve the benchmark in the future may be to take a station by station approach and look at how many stations in each region or season reached a specific threshold rather than taking the average of all stations in the region from March through November.

The indicator could be improved in the future by limiting analysis to the most utilized copepod species. Some fish like Delta Smelt tend to utilize certain calanoid copepod species more than others and also utilize some species of cyclopoid copepods (Slater and Baxter 2014).

- **Data Sources**
  - **Describe the data used and where they came from.**

The data used for this indicator was from the IEP's Zooplankton Study, a long-term monitoring survey which has been monitoring zooplankton in the upper SFE since 1972. Zooplankton samples were collected by the California Department of Water Resources during their monthly water quality monitoring cruise at approximately 16 to 22 stations using plankton nets. The number of stations sampled varied depending on the location of the floating entrapment zone stations (where sampling location is determined by a bottom specific conductivity of 2 and 6 mS/cm, approximately 1 and 3 ‰), and specific conductance (3 stations in San Pablo Bay and Carquinez Strait were only sampled when surface specific conductance was less than 20 mS/cm). The study area extends from eastern San Pablo Bay through the Delta (see complete station map at www.dfg.ca.gov/delta/data/zooplankton/stations.asp). Only a subset of selected stations in the Delta and Suisun regions were used for this indicator. Data is available through a password protected ftp site; access information can be obtained by contacting April Hennessy of the California Department of Fish and Wildlife (April.Hennessy@wildlife.ca.gov).

At each station, plankton nets were towed obliquely for 10 minutes, to get samples that were representative of zooplankton in the entire water column from the bottom to the surface. Two conical nets arranged next to each other on a sled were used to target zooplankton of different sizes. A meso-zooplankton net was used to target adult and juvenile calanoid copepods (160 micron mesh) and a macro-zooplankton net was used to target mysids (500 micron mesh). A General Oceanics flowmeter was mounted in the mouth of each net to measure the volume of water sampled during the tow. Pump samples were also collected to target smaller zooplankton like rotifers and smaller cyclopoid copepods, however this data was not used to develop the zooplankton indicator presented here. Samples were preserved in 10% formalin and returned to the California Department of Fish & Wildlife's Laboratory in Stockton, CA for processing. Organisms in samples were identified and enumerated using a dissecting microscope.

Meso-zooplankton samples were rinsed and then diluted in a beaker of water to an approximate organism concentration of 200-400 organisms per milliliter of water (called dilution volume). Subsamples were taken 1 milliliter at a time with an auto pipette, and placed on a Sedgewick-Rafter slide for identification and enumeration. Between 5 and 20 slides were examined for each sample, this number varied depending on the dilution volume with a target of 6% of the sample examined. For example if the sample volume was 100 ml, then 6 slides were examined. Catch-per-unit-effort (CPUE) was then calculated for each sample as:

CPUE = ((C/S)L)/V

Where:
CPUE = the number of a taxon per cubic meter of water filtered
C = the cumulative number of a taxon counted for the sample
L = the reconstituted sample volume (dilution volume) in milliliters
S = the number of Sedgewick-Rafter cells examined (1 ml ea)
V = the volume of water filtered through the net (m^3) (where volume filtered is estimated by: VolFiltered = (end meter – start meter) * calibration factor * mouth area)

Biomass-per-unit-effort (or BPUE, also referred to simply as "biomass" for this indicator) was then calculated by multiplying CPUE by a carbon weight for each taxon in micrograms, using some literature based values and some provided by Dr. Wim Kimmerer of Romberg-Tiburon Center for Environmental Studies (Kimmerer 2006, Uye et al. 1983, Culver et al. 1985, Hoof and Bollens 2004). BPUE was then converted from micrograms to milligrams per cubic meter of water sampled, as reported here, by dividing by 1000.

Macro-zooplankton samples were rinsed with water to remove excess formalin, and then placed in a sorting tray for processing. Samples that appeared to have more than 400 mysids were subsampled by placing a quadrant splitter into the tray. If the first quadrant appeared to have more than 400 mysids, then this quadrant was split again into 4 more quadrants. This process was repeated until the subsample appeared to have no more than 400 mysids in it. Each quadrant was processed until a minimum of 400 mysids were processed and the number of quadrants processed determined the subsample. The first 100 non-gravid females (those not carrying eggs), males, and juveniles, as well as the first 30 gravid females (those carrying eggs) of each species were measured from the tip of the eye to the base of the telson. Measurements were rounded up to the nearest millimeter and recorded. All remaining mysids of each species in the subsample were counted and the total was recorded as the plus count. The total number of each mysid species was determined by the equation:

$N = C/S$
N = total number of each mysid species in the sample
C = total of mysids in the sub-sample (number measured + plus count)
S = sub-sample or fraction of the sample examined.

Biomass was calculated from length-frequency using length-weight regression equations for *Neomysis mercedis* and *Hyperacanthomysis longirostris* (J. Orsi CDFW, unpublished). The length weight equation for *N. mercedis* was used for all other species besides *H. longirostris.* Weight was then summed by species for each sample date and station, and biomass estimated using a carbon to dry weight ratio of 40% (Uye 1982).

- **Methods**
    - **Describe the calculation methods.**

The sum of total calanoid copepods BPUE and total mysid BPUE was calculated for each sample date and station. Annual averages for each region were then calculated using March-November data for all stations in each region. The benchmark was calculated as the average of the 1974-1986 annual averages, and the "fair-poor" scoring break as 25% of the benchmark. The current status was calculated as the average of the 2000-2014 annual averages. The trend was calculated by a linear slope of the annual averages for each region, and the significance of the slope tested using a Mann-Kendall test where $p<0.005$ was considered significant.

- **Include a description of the assumptions and uncertainties.**

Zooplankton are sampled monthly at fixed stations, therefore data are limited temporally and spatially. It is assumed that BPUE at these stations is representative of zooplankton BPUE throughout the upper SFE, however zooplankton is highly variable temporally and spatially.

- **Peer Review**

    - **Describe how the indicator was vetted with other experts in the community as per the SOTER Peer Input Guidelines.**

Several venues were used for peer review of the zooplankton indicator. Consultation with fellow State of the Estuary contributors Elizabeth Wells (California Department of Water Resources) and Hildegarde Spautz (California Department of Fish and Wildlife), as well as Dr. Wim Kimmerer from the Romberg Tiburon Center for Environmental Studies, and Kathryn Hieb (DFW) led to several revisions of the indicator. Drafts of the indicator ideas, calculations, and results were presented at State of the Estuary meetings as well as at several California Estuary Monitoring Workgroup meetings, and were discussed in meetings of the Living Resources subgroup of the California Estuary Monitoring Workgroup. Further discussion of indicator benchmarks and scoring was conducted with Letitia Grenier and Amy Richey (both of the San Francisco Estuary Institute), as well as with Elizabeth Wells and Hildegarde Spautz.

- **Results**

    - **In addition to summarizing the status and trend, this is a place to provide greater detail on the results that may not be possible in the limited space of the main report.**

Mysids have declined in both the Suisun and Delta regions of the upper SFE since monitoring began. The historical reference period from 1974-1986 established the benchmarks of 17.3 milligrams of carbon per cubic meter of water (a measure of biomass) sampled for the Suisun region and 7.7 milligrams of carbon per cubic meter of water sampled for the Delta region. After a slight upturn in 2000, biomass in both regions has been below the “fair” threshold of 4.3 milligrams of carbon per cubic meter of water sampled for the Suisun region and 1.9 milligrams of carbon per cubic meter of water sampled for the Delta region. The 2010-2014 average biomass was 1.6 milligrams of carbon per cubic meter of water sampled for the Suisun region and 0.4 milligrams of carbon per cubic meter of water sampled for the Delta region, placing the current status of both regions as “poor”. The annual data had a significant downward trend for both regions from 1974-2014 ($p<0.001$). However, from 2000 through 2014 there was no significant trend, indicating that although mysid biomass is lower than it was historically, it does not appear to be declining further.

Calanoid copepods have declined in the Suisun region of the upper SFE since monitoring began. The 1974-1986 average of 15.3 milligrams of carbon per cubic meter of water sampled was the historical reference period used to establish the benchmark for the Suisun region. At or above this benchmark was considered “good”, below this benchmark down to 25% of this benchmark was considered “fair”, and less than 25% was considered “poor”. In the Suisun region calanoid copepod biomass fluctuated between “good” and “fair”, until 1988 when it fell below the threshold of 3.8 milligrams of carbon per cubic meter of water sampled to “poor”. Since 1988, calanoid copepod biomass in the Suisun region fluctuated between “fair” and “poor”, with small peaks occurring during higher flow years like 2006 and 2011. The current status of calanoid copepods in the Suisun region is “fair” with the 2010-2014 average of 4.7 milligrams of carbon per cubic meter of water sampled. Similar to the mysids, there was a significant downward trend from 1974-2014 in the Suisun region ($p<0.001$). However, in recent years from 2000 through 2014 there was no significant trend, indicating that although calanoid copepod biomass is lower than it was historically, it does not appear to be declining further in the Suisun region.

Calanoid copepod biomass has increased in the Delta region of the upper SFE since monitoring began. The 1974-1986 average of 5.6 milligrams of carbon per cubic meter of water sampled was the historical reference period used to establish the benchmark for the Delta region. At or above this benchmark was considered “good”, below this benchmark down to 25% of this benchmark was considered “fair”, and less than 25% was considered “poor”. In the Delta region calanoid copepod biomass has fluctuated between “good” and “fair” for the entire monitoring period, however biomass during the historical reference period used to establish the benchmark was low. The current status of calanoid copepods in the Delta region is “good” with the 2010-2014 average of 7.9 milligrams of Carbon per cubic meter of water sampled. From 1974 through 2014 there was a significant upward trend in calanoid copepod biomass in the Delta region ($p=0.0047$), however from 2000-2014 there was no trend.

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State of the Estuary Report 2015

Summary

PROCESSES – Feeding Chicks, Brandt's Cormorant

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State of the Estuary 2015: Processes
Brandt's Cormorant Reproductive Success Indicator

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1. Brief description of indicator and benchmark; background

The indicator is the number of fledged young produced per breeding pair at the breeding colony of Brandt's Cormorants (*Phalacrocorax* penicillatus) on Alcatraz Island, in San Francisco Bay. The indicator provides a measure of the health of the aquatic foodweb in the San Francisco Bay estuary. Brandt's Cormorants are entirely piscivorous and, thus, the reproductive success of breeding pairs reflects availability of food for the young in the open water of San Francisco Bay. The indicator has been studied on Alcatraz Island since 1995, as part of the monitoring conducted by the National Park Service. A comparable time series has been collected at the Brandt's Cormorant colony on the Farallon Islands by Point Blue Conservation Science since 1972.

The specific calculation used for assessment purposes is the mean reproductive success for the most recent 3 years. We compare that value to the reproductive success required to maintain a stable population, which is the benchmark criterion for Good. If the indicator value meets or exceeds the latter, the indicator is scored Good. The criterion for Poor was 60% that of Good, i.e., 40% reduction. If the indicator is at or below the scoring criterion for Poor the criterion is scored Poor.

2. Indicator status and trend measurements

Results for this indicator for 1995-2014 are displayed in Figure 1. The indicator was relatively stable between 1995 and 2007. In 2008, it exhibited a slight drop, followed by a strong decline in 2009. From 2009 to 2012, inclusive, the indicator was either scored Poor (2009, 2010, 2012) or Fair (2011). However, in the two most recent years (2013, 2014), the indicator has shown a sharp rebound. As a result, there is no significant ($P > 0.05$) linear trend. In addition, the mean value for the most recent three years is 1.67. This is above the benchmark value of 1.50 and is therefore scored as Good.

3. Brief write-up of scientific interpretation

What is this indicator?

The indicator is the number of fledged young produced per breeding pair at the breeding colony on Alcatraz Island, in San Francisco Bay. This colony has been studied since 1995 (Robinson et al. 2014) using standardized focal site surveys, comparable to studies conducted on the Farallon Islands (Boekelheide et al. 1990, Nur and Sydeman 1999a).

Why is it important?

There are two essential reasons for tracking and evaluating the reproductive success of the Brandt's Cormorant in San Francisco Bay. Above all, this metric provides a reliable index of prey availability for foraging seabirds in the bay, and thus provides an indicator of functioning of the aquatic foodweb in the bay. Brandt's Cormorant are piscivorous (Ainley and Boekelheide 1990), and, moreover, are apex marine predators. That the ability of parent birds to adequately feed their chicks is a good measure of prey availability has been well established through numerous studies, including long-term studies on the Farallon Islands nearby (Nur and Sydeman 1999a). Secondly, success at rearing chicks is a necessary requirement for healthy, self-sustaining populations (Nur and Sydeman 1999b).

What is the benchmark? How was it selected?

The benchmark for Good is the level of reproductive success that produces a stable population (given what is known regarding all other relevant demographic parameters). On the basis of calculations in Nur et al. (1994) (see also Nur & Sydeman 1999a, this value is 1.50 chicks fledged per breeding pair. The criterion for Poor is 60% that of Good. Thus, a reproductive success below 0.90 chicks per pair is the criterion for Poor. Reproductive success below 0.90 chicks for an extended period of time would have marked population consequences. We note that three of the recent years (2009, 2010, 2012) were below the Poor benchmark value.

Status and Trends

The most recent three-year mean is 1.67 young fledged per pair, which is scored Good. In fact, the most two recent years (2013, 2014) were 2.3 and 2.1 young fledged, respectively, which is in the top half of all results for the 20-year time series. In contrast, 2012 was an especially low value. Thus, after a four-year period of moderate to low reproductive success (2009-2012), Brandt's Cormorant success appears to have fully rebounded.

Over the period 1995-2008 there was no significant trend in the indicator. However, adding 2009-2012 to the time series resulted in a significant, linear declining trend. But with the addition of 2013 and 2014, there is currently no significant linear trend over the entire 20-year period, 1995-2014 ($P > 0.05$, for linear regression of chicks fledged in relation to year).

Significance/Interpretation

Starting in 2008, Brandt's Cormorants displayed a declining trend in reproductive success, which, in 2009, 2010, and 2012, reached extremely low levels. Similarly low reproductive success was observed on the Farallon Islands during these years, specifically 2008-2012, inclusive (Warzybok et al. 2014). Such low success indicated especially low prey availability in those years. In 2013 and 2014, however, reproductive success for Brandt's Cormorant on Alcatraz Island was very high, demonstrating a complete reversal of the earlier decline. Thus, a well-functioning foodweb, supporting forage fish and their predators, is indicated for the two most recent years. In general, principal prey species for Brandt's Cormorants are Northern anchovy, rockfish (several species) and flatfish such as the Pacific sanddab (Nur and Sydeman 1999a). While anchovy have been rare or absent in recent years, in 2013 and 2014 rockfish and sand dabs have been well-represented (Point Blue unpublished).

Two consecutive years of high reproductive success is encouraging, but evaluation of reproductive success in 2015 will be required to confirm whether the situation continues to be favorable.

4. Related Figures.

Indicator results are displayed in Figure 1.

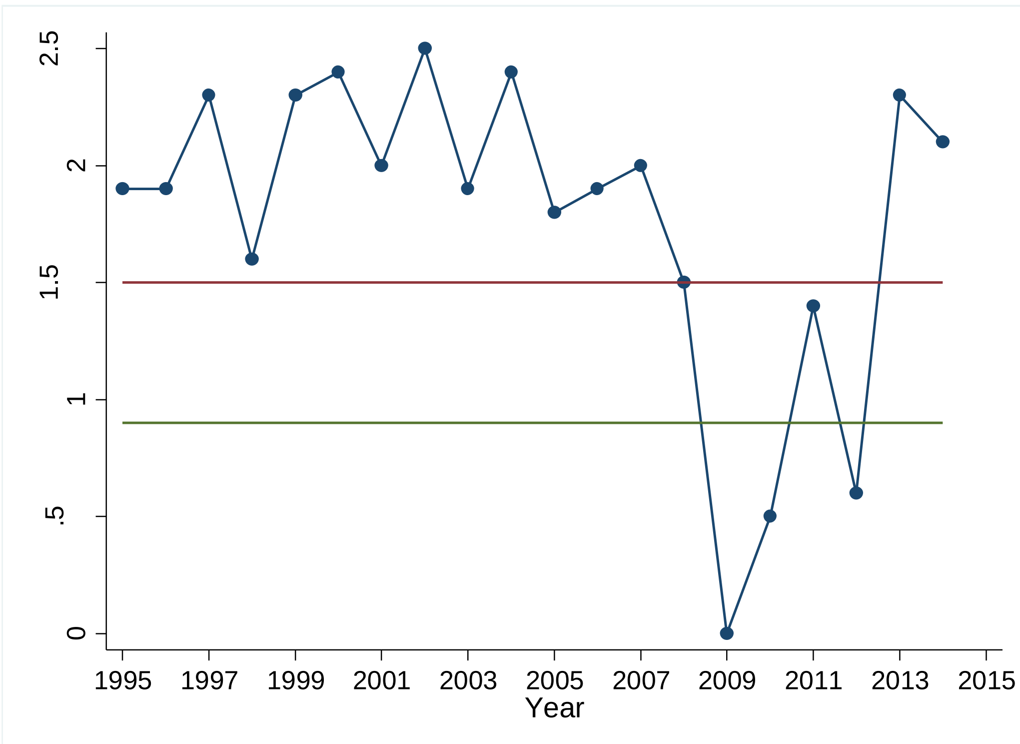

Figure 1. Brandt's Cormorant Reproductive Success Indicator. Mean reproductive success is shown for each year. The Benchmark Value for Good is 1.50 fledged young per pair (see text). The scoring criterion for Poor is 60% of the benchmark, i.e., 0.90 fledged young per pair.

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5. Technical Appendix.

Background and Rationale

The reproductive success of seabirds and other waterbirds is a well-established indicator of ecosystem health. In particular, reproductive success of seabirds has been shown to be a good indicator of food availability for vertebrate predators (Parsons et al. 2008). Use of this indicator for Brandt's Cormorants on Alcatraz Island, work which was initiated in 1995, provides an especially informative indicator because we also have a long time series for the same species on the nearby Farallon Islands (Boekelheide et al. 1990, Nur and Sydeman 1999a, Warzybok et al. 2014). The same methods have been used in both colonies, facilitating comparison among the two time series.

From 1995 until 2007, reproductive success on Alcatraz Island remained high and relatively stable from year to year. In 2008, the lowest value as of then was observed: 1.50 chicks reared per pair. However in 2009 no chicks at all were reared in the colony (Robinson et al. 2014). In 2010 and 2012 reproductive success was very low (less than 0.75 chicks reared per pair) and even in 2011, reproductive success was lower than in any year observed between 1995 and 2008. However, in 2013 and 2014 reproductive success returned to the levels observed in 1995-2007.

Benchmark

Choice of benchmark: As noted above, the number of chicks reared to fledging is a well-established indicator of food availability for marine predators. At the same time, reproductive success is a key and necessary parameter to maintain healthy wildlife populations (Nur and Sydeman 1999b). Due to previous, intensive studies of Brandt's Cormorants on the nearby Farallon Islands (Boekelheide et al. 1990 and Nur and Sydeman 1999a), we are able to estimate the reproductive success per pair needed to produce a stable population, given our knowledge regarding survival rates of juveniles and adults and age of first breeding (Nur et al. 1994). That value, 1.50 chicks reared per pair, provides the benchmark value used here. Furthermore, we use the average value for the three most recent years for comparison. Use of a single year's value would not be as informative due to year to year variation in this indicator.

Data sources and Methods

The data collected and calculations made for this indicator were carried out by several investigators working on Alcatraz Island under the auspices of the National Park Service, most recently Robinson et al. (2014). That reference provides information on data collection and calculation of annual reproductive success. The time series on Brandt's Cormorant reproductive success on Alcatraz Island was initiated in 1995 and has continued through the present. Reproductive success is estimated at multiple sub-colonies on the island; in 2014, data were gathered from six sub-colonies. Sample size in each year has usually exceeded 150 pairs for the island. In 2014 the sample size was 167 breeding pairs. In a few years in the time series sample size was less than 100 (Robinson et al. 2014).

For analysis of the indicator we calculated the mean reproductive success over the past 3 years (i.e., during the current period) and compared that to the benchmark value (see above). We did not calculate a standard error around the three-year average using the original field data since we did not have access to those data, but only the summary statistics. However, we do note that the three-year average is based on a sample of 541 breeding pair-years, and so confidence in that average is high.

Uncertainties and assumptions: Reproductive success among breeding pairs is well estimated on Alcatraz Island: a large sample of breeding pairs are monitored, spanning six or more sub-colonies in each year. However, not all mature Brandt's Cormorant attempt to breed in each year. Thus, poor food availability can lead to skipping of breeding (Nur and Sydeman 1999a) and this phenomenon cannot be captured with the current metric.

The choice of benchmark value depends on assumptions regarding survival and age of first breeding. The conclusion that two out of the last three years exceeded the benchmark value is a robust one and does not depend on the exact benchmark value used, since in 2013 and 2014 annual reproductive success was 2.1 or greater, substantially greater than the calculated benchmark value of 1.5. The greater uncertainty concerns the reproductive success achieved in 2015 and in future years. Will the favorable conditions observed in 2013 and 2014, similar to observations from 1995 to 2007, continue?

Peer Review and Acknowledgments
Thanks to Meredith Elliott for providing review of the indicator. Thanks are also due to Victoria Seher and Bill Merkle for providing assistance in completion of this indicator for the 2015 State of the Estuary Report. Methods used in analysis have been published in Nur and Sydeman (1999a).

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Summary

PROCESSES – Heron and Egret Brood Size

Prepared by John P. Kelly, Cypress Grove Research Center,
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State of the San Francisco Estuary 2015
Processes – Heron and Egret Brood-size Indicator

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What is the indicator?

As top wetland predators that operate over large areas of the San Francisco Estuary, herons and egrets depend on extensive tidal marshes, seasonal wetlands, and associated freshwater systems. The State of the Estuary Report uses prefledging brood size among successful heron and egret nests to assess ecological conditions across broad wetland landscapes. The section on "Feeding Chicks," in the "Processes" chapter of the 2015 State of the Estuary Report, summarizes the Heron and Egret Brood-size Indicator, described more fully here. This indicator uses the number of young produced in successful nests to index conditions that affect the availability of food, the productivity of estuarine food webs, and the quality of wetland feeding areas.

The "Wildlife" chapter of the 2015 State of the Estuary Report includes two additional heron and egret indicators, for nest density and nest survival. The Heron and Egret Nest Density Indicator provides an index of regional heron and egret population sizes. The Heron and Egret Nest Survival Indicator is based on the survival of nesting attempts through the breeding cycle and is used to assess the dynamics of nest-predator populations, human disturbance, and changes in human land use that can affect the size and distribution heron and egrets nesting colonies

Attribute	Indicator	Benchmark
Food web	Great Egret/Great Blue Heron Brood Size (number of young produced per successful nest)	The benchmark is the number of young produced per nest from 1991-2000, across Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all three areas combined.

How are the current Brood-size Indicator conditions measured?

The Heron and Egret Brood-size Indicator was evaluated using methods and analysis described in Kelly *et al*. (1993, 2007) and Kelly and Condeso (2014). The Brood Size Indicator is calculated as the mean prefledging brood size, between species. The Brood Size Indicator is also calculated separately for Great Egrets and Great Blue Herons (see Technical Appendix).

The indicator provides insight into change over time in brood size prior to fledging among nests that successfully fledge one or more young. Brood-size measurements were conducted when Great Blue Heron nestlings were known to be 5-8 weeks old and Great Egrets are known to be 5-7 weeks old (Pratt 1970, Pratt and Winkler 1985); during these periods, nestlings are too young to hop away from their nests and old enough to have survived the period when most brood reduction occurs (Pratt 1970; Pratt and Winkler 1985). The Brood Size Indicator is calculated as the mean prefledging brood size (number of young produced in successful nests), between species, based on observations at 40-50 colony sites within foraging range (10 km) of the historic tidal wetland boundary (ca.1770–1820; San Francisco Estuary Institute 1999). Brood size is sampled in approximate proportion to colony size and averaged annually (1991-2014) among nests within and across the three major subregions of northern San Francisco Bay (Central San Francisco Bay, San Pablo Bay, and Suisun Bay).

Trends in the indicator values were measured as proportional annual change, converted to percent change, over the 24 years, 1991-2014, or before/after years with minimum/maximum values, and by comparisons of indicator values between recent years (2009-2014) and a ten-year baseline (1991-2000), weighted equally among years. Patterns of change over time were modeled as quadratic trends with increasing or decreasing slopes, *if and only if* the quadratic term in the model was significant (P<0.05); otherwise changes over time were estimated as linear trends.

What is the benchmark for the Brood-size Indicator and how was it selected?

The benchmark for the Heron and Egret Brood-size Indicator is the mean annual, prefledging brood size (number of young produced in successful nests, as described above) during the first ten years of regional monitoring, 1991-2000. This period was chosen to be consistent with the benchmark selected for the Heron and Egret Nest Density, which was selected because the densities of nesting herons and egrets were relatively stable during this period, compared to subsequent years.

What is the status and trend of Brood-size Indicator in each area?

Heron and Egret Brood Size (Figure 1, Table 1) exhibited a shallow annual decline across northern San Francisco Bay, to 2.02 young per successful nest in 2008, 4.1% below a the baseline average of 2.03 young per nest, then increased slightly in recent years (quadratic trend: $F_{2,3239}$=11.43, P<0.001).

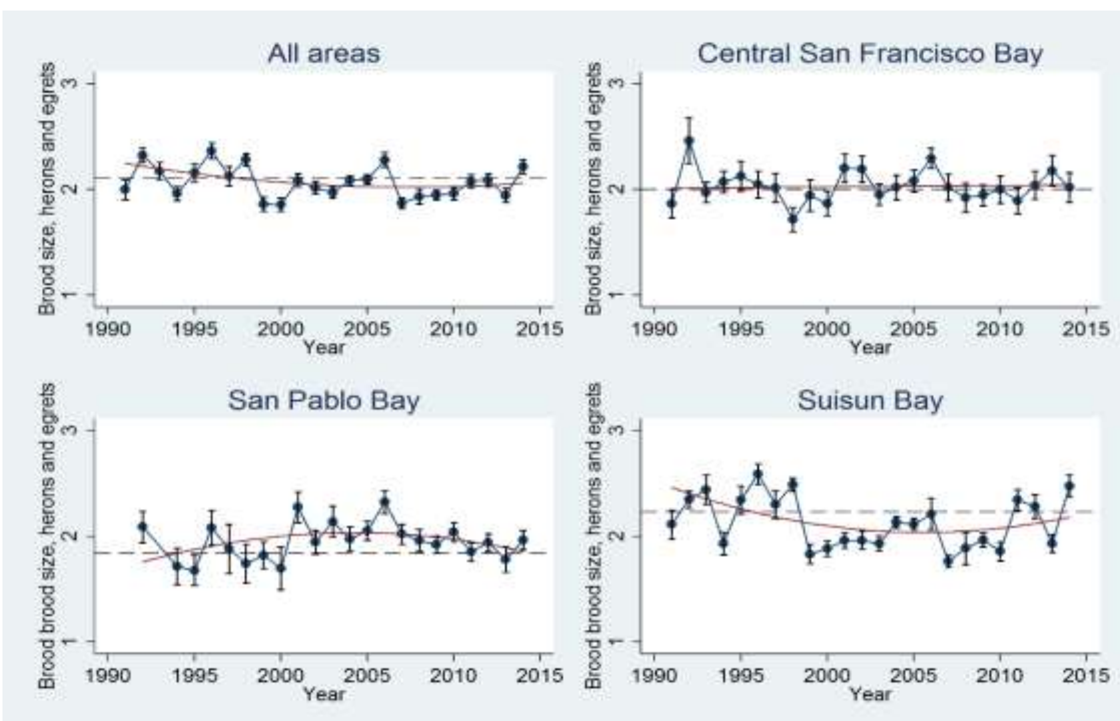

Figure 1. Annual heron/egret brood size and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

Productivity during 2009-2014 was slightly lower than baseline levels in 1991-2000 ($F_{1,14}$=11.8, P<0.001; Table 1).

In San Pablo Bay, Heron and Egret Brood Size increased to a maximum of 2.00 young per successful nest in 2004, then declined by 1.3% per year ($\log_e$ trend over the entire period: -0.013±0.005, P=0.07). Brood sizes were 5.2% higher, on average, in 2009-2014 than during the baseline period, although the difference was not significant ($F_{1,14}$=1.8, P=0.17; Table 1).

In Suisun Bay, Heron and Egret Brood Size declined to a minimum of 2.0 young per successful nest in 2006 ($F_{2,1729}$=20.78, P<0.001), then increased through 2014 ($F_{2,1729}$=20.78, P<0.001). During recent years (2009-2014), brood size averaged 7.5% lower than the baseline period ($F_{1,14}$= 12.3, P<0.001; Table 1).

Table 1. Heron and Egret Brood Size Indicator (species combined) results, including the mean and standard error (SE) of annual brood size, weighted equally among years, during the current period, 2001-2014, and the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the *F*-value and significance (*P*) of the change.

Area	Current (2009-2014)	SE	Baseline (1991-2000)	SE	Percent change	$F_{1,14}$	*P*
All areas combined	2.0	0.03	2.1	0.02	**-5.4**	**11.8**	**0.001**
Central San Francisco Bay	2.0	0.06	2.0	0.04	2.0	0.3	0.56
San Pablo Bay	1.9	0.04	1.8	0.06	5.2	1.8	0.17
Suisun Bay	2.1	0.04	2.3	0.03	**-7.5**	**12.3**	**<0.001**

In general, what do the results mean and why are they important?

Heron and Egret Brood size is relatively stable across the region but suggests a very gradual, long-term decline in wetland productivity. Within San Pablo Bay, an apparent decline in productivity of successful heron and egret nests since 2005 is consistent with the leveling off of nest densities there in recent years, suggesting a reduction in the quality of wetland feeding areas or, alternatively, the presence of foraging competition.

How does heron and egret brood size relate to the ecological health of the estuary?

The Brood size Indicator is sensitive changes in the extent and quality of foraging habitat, or the supply or availability of prey needed to provision nestlings, 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, Frederick 2002, Kushlan and Hancock 2005). The two target species reflect productivity responses related to the use of different feeding habitats: 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.

What is the historical use of this indicator and current programs for evaluation?

Audubon Canyon Ranch (ACR) 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. ACR and continues to sustain this effort on an ongoing basis, and to produce regular reports based on this information (e.g., Kelly et al. 1993, 2005, 2006, 2007, 2008, 2014, Kelly and Rothenbach 2012).

How suitable are the reference conditions and targets for monitoring wetland condition?

The Heron and Egret Brood-size Indicator provides a particularly suitable target for monitoring wetland conditions at landscape scales (Kelly et al. 2008). The productivity of successful nests was relatively stable across the northern San Francisco Bay area during 1991-2000.

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Technical Appendix

Great Blue Heron Brood Size

Great Blue Heron brood size declined in the Suisun Bay until 2005, falling to 12.9% below the baseline, then increased gradually to near baseline levels in 2014 ($F_{2,992}$=15.1, P<0.001). During 2009-2014, mean brood sizes in Suisun Bay averaged 2.00±0.12 young per successful nest, which was 4.4% below the baseline level ($F_{1,14}$=5.9, P<0.001; Table 2).

In Central San Francisco Bay, Great Blue Heron brood size declined since 2000, but by less than one percent annually ($F_{1,522}$ =3.47, P=0.06). In 2009-2014, mean Great Blue Heron broods averaged 2.02 young per successful nest, which was 5.6% lower, than the 1991-2000 baseline ($F_{1,14}$=3.8, P=0.053; Table 2).

Great Blue Heron brood size declined in the Suisun Bay until 2005, falling to 12.9% below the baseline, then increased gradually to near baseline levels in 2014 ($F_{2,992}$=15.1, P<0.001). During 2009-2014, mean brood sizes in Suisun Bay averaged 2.00±0.12 young per successful nest, which was 4.4% below the baseline level ($F_{1,14}$=5.9, P<0.001; Table 2).

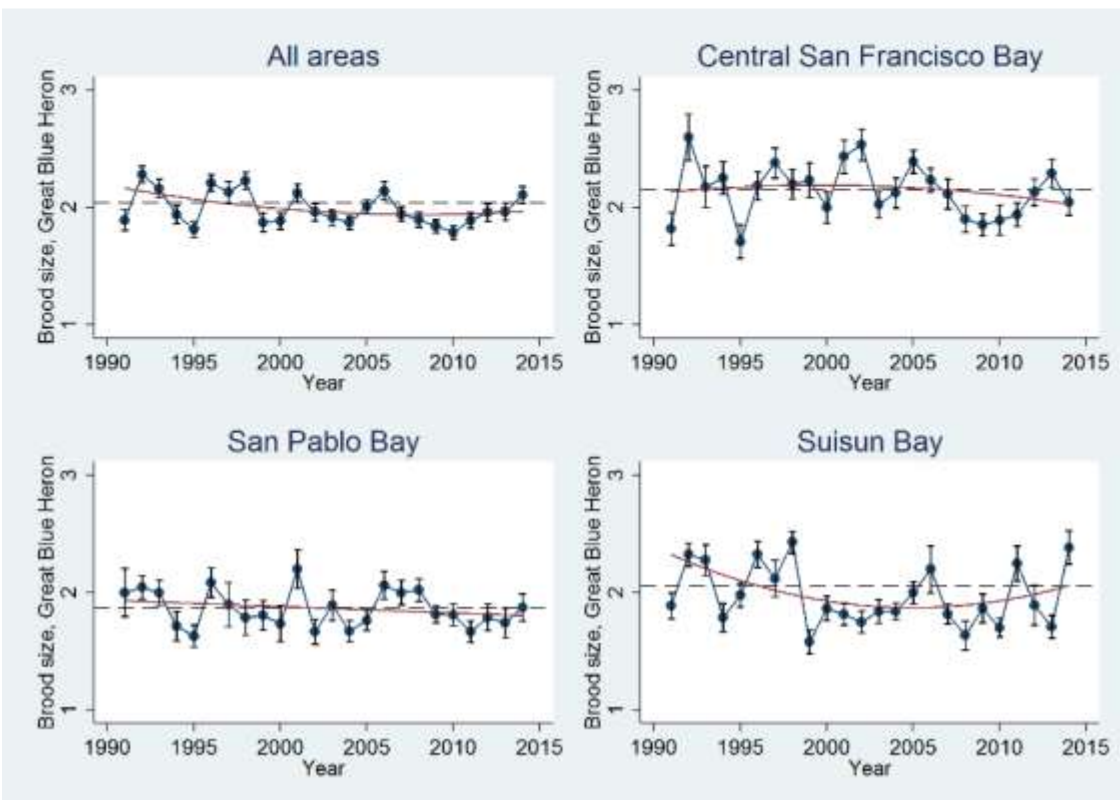

Figure 2. Annual Great Blue Heron brood size and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

***Great Egret Brood Size*.**

Great Egret Brood Size (Figure 3, Table 3) declined in the Central San Francisco Bay until 2006, falling to 5.3% below the baseline, then increased gradually to near baseline levels in 2014 ($F_{2,3907}$=21.0, P<0.001). Mean annual brood sizes in 2009-2014 did not differ significantly from baseline levels ($F_{1,14}$=0.43, P=0.52).

In Central San Francisco Bay, Great Egret broods sizes showed significant quadratic trend (Figure 3; $F_{2,686}$=10.02, P<0.001), suggesting a decline to 1.8 young per nest at the end of the baseline period in 2000, followed by increasing productivity, but with no difference between recent (2001-2014) and baseline (1991-2000) years ($F_{1,14}$=1.9, P=0.17; Table 3).

Table 2. Great Blue Heron Brood Size Indicator results, including the mean and standard error (SE) of annual brood size, weighted equally among years, during the "current" period of recent years, 2009-2014, and the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the *F*-value and significance (*P*) of the change.

Area	Current (2009-2014)	SE	Baseline (1991-2000)	SE	Percent change	$F_{1,\,14}$	*P*
All areas combined	1.9	0.03	2.1	0.02	**-6.9**	**15.2**	**<0.001**
Central San Francisco Bay	2.0	0.05	2.1	0.04	**-4.6**	2.3	0.13
San Pablo Bay	1.8	0.04	1.9	0.04	**-5.4**	3.0	0.08
Suisun Bay	2.0	0.05	2.1	0.04	**-7.8**	**6.8**	**0.01**

Baseline brood-sizes among Great Egrets nesting in San Pablo Bay during the 1991-2000 reference period were relatively low (although samples were smaller and less precise than in later years), but increased to 2.4 young per nest in 2004 (19.4% above baseline), then declined to lower levels in recent years ($F_{2,698}$=10.8, P<0.001). Productivity among successful nests was relatively stable in 2009-2014 and did not differ significantly from the baseline period ($F_{1,11}$=0.03, P=0.86; Table 3).

In Suisun Bay during the 1991-2000 baseline period, annual mean productivity in successful Great Egret nests was relatively high, averaging 2.36±0.100 young per nest. However, reduced reproductive output was apparent through 2006, when productivity leveled and began to increase gradually at an annual rate of 1.4% ($F_{2,2517}$=36.7, P<0.001). This increasing trend led to a mean brood size in 2014 that exceeded the estimated baseline mean, but overall productivity remained below baseline levels during 2009-2014 ($F_{1,14}$=10.33, P<0.001; Table 3).

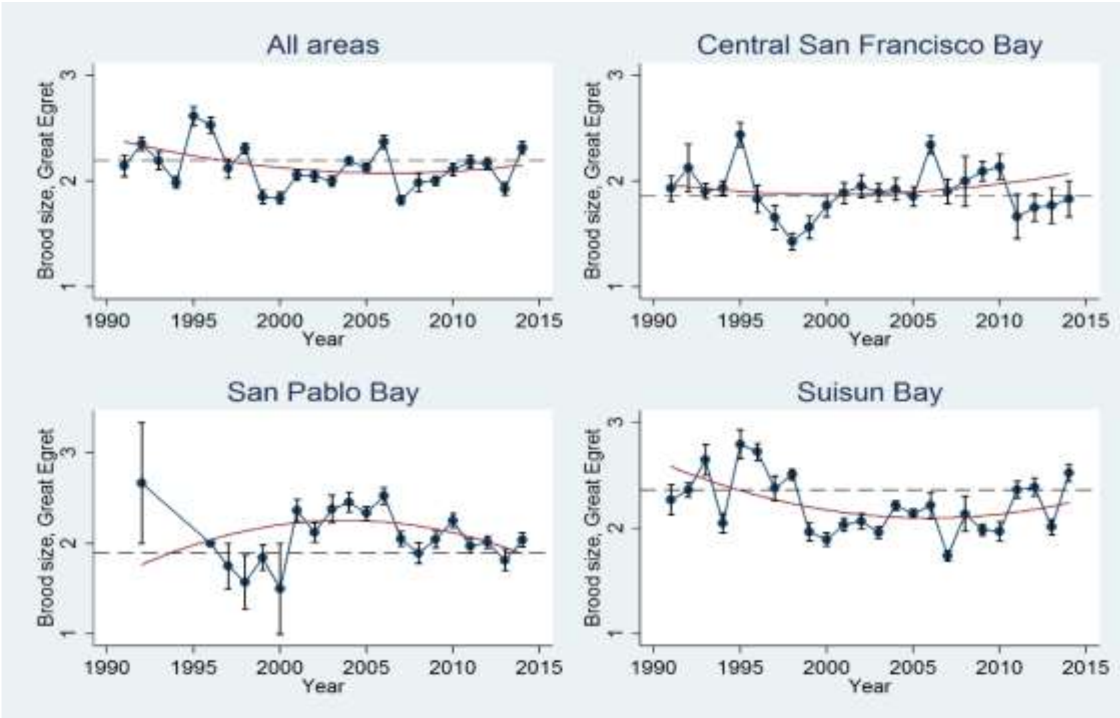

Figure 3. Annual Great Egret brood size and trends in Central San Francisco Bay, San Pablo Bay, and Suisun Bay, and all areas combined, 1991-2014. Error bars represent standard errors; red lines indicate the linear or quadratic trends, 1991-2014; dashed lines indicate the mean values (benchmarks) for the reference period 1991-2000.

Table 3. Great Egret Brood Size Indicator results, including the mean and standard error (SE) of annual brood size, weighted equally among years, during the "current" period of recent years, 2009-2014, and the baseline period, 1991-2000, the mean percent change between current and baseline periods, and the *t*-value and significance (*P*) of the change.

Area	Current (2009-2014)	SE	Baseline (1991-2000)	SE	Percent change	$F_{1,\,14}$	*P*
All areas combined	2.1	0.02	2.2	0.02	**-5.2**	**13.0**	**<0.001**
Central San Francisco Bay	2.0	0.06	1.9	0.03	5.2	1.9	0.17
San Pablo Bay	2.0	0.03	1.9	0.10	9.8	2.9	0.09
Suisun Bay	2.2	0.03	2.4	0.03	**-8.6**	**24.5**	**<0.001**

What is the source of these data?

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.

What assumptions and uncertainties are involved?

Heron and Egret Brood Size Indicator 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. It assumes that brood reduction has declined and that nestlings are 5-8 weeks of age closely reflect the number of young fledged from successful nests (Pratt 1970, Pratt and Winkler 1985). Uncertainties are related primarily to unobserved nestlings concealed by vegetation and (2) estimation of nesting ages and (3) timing of the brood-size reduction portion of the nesting cycle. However the conspicuousness of heron and egret nests facilitates the successful use of this indicator.

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State of the Estuary Report 2015

Technical Appendix

PEOPLE – Conserving Water

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TECHNICAL APPENDIX: URBAN WATER USE

CONTEXT

The San Francisco Bay Area uses about 1 million acre-feet per year of water, 90% of which supports urban activities in homes, businesses, institutions, and industries. Most of this urban or municipal water[1] – about 75%- is imported, primarily from the Delta and Central Valley watersheds with smaller amounts from the Russian River and Tomales Bay. Less than 10% is supplied from local Bay-draining (non-Delta) watersheds, such as the Napa River and Alameda, Coyote, Los Gatos and San Mateo Creeks. The remaining 15% is supplied from groundwater, which is a locally significant supply source to urban users in the Santa Clara and Livermore Valleys, and in Fremont and the North Bay; some of that groundwater is derived from the recharge of imported surface water.

Using less water (conservation) and using water more efficiently by reducing the amount of water needed for any activity while still accomplishing the goals of that activity (e.g. toilet flushing, irrigation) has many actual and potential benefits for the Bay Area including:

- Reduces the demand on already-over-drawn supply sources, leaving more water to maintain the habitats, living resources, and ecological processes of the Bay and its watersheds
- Reduces the financial and energy costs of treating and transporting water
- Reduces the need to develop new supplies;
- Reduces pollutant loads from irrigating lawns, gardens and crops;
- Reduces the vulnerability of supplies to disruption by earthquakes, droughts, floods, rising sea level, and regulatory requirements to protect endangered species.
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With four straight years of low runoff from the Delta watersheds and the record low snowpack in 2015, state and local agencies have made water conservation and water efficiency both a priority and a mandate for urban residents and water suppliers.

INDICATOR

This indicator assesses the region's water use and the efficiency of that use over time by measuring total urban water use and just the residential portion with two metrics: the annual potable volume in acre-feet; and the per-person (per capita) use in gallons per capita per day (gpcd).[2] The period of assessment is 1986-2014, a period long enough to evaluate how the Bay Area urban use is affected over time by population growth, climate, plumbing codes, conservation measures and economic conditions. 1986 is just prior to the 1987-92 drought, the longest duration drought experienced by the Bay Area. Major plumbing code changes were also instituted in the early 1990's. From

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[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] Measures of potable or drinkable water do not include recycled water.

2007 to 2009 the Bay Area experienced a 3-year dry period and economic downturn and since 2012 the region is in the midst of another prolonged drought.

This indicator measures the consumption of the water used inside and outside of the residences, businesses, and industries in the Bay Area. It does not measure the total water footprint, which is the volume of water that is required to produce all the goods and services that are consumed and which is many times greater than the direct consumption.[3]

Residential use, which includes both single family and multi-family residences, consists of indoor uses (waste elimination, washing clothes and dishes, bathing, drinking) and outdoor uses (irrigation and cleaning). 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. Residential use 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.

Residential per capita use is sometimes used to compare water use within and across watershed boundaries or among water agencies.[4] Total per capita 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 individuals use less accurate. The total municipal per-capita use for the Bay Area, however, 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

All of the Bay Area municipal water suppliers measure the water use of their customers in order to bill them based upon the volume of use. The retail water suppliers separate the customers into different sectors or types of use, often distinguished by the size and type of water meter. Residential water use is normally 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. The water suppliers generally report the water use on a monthly or bi-monthly basis in gallons or cubic feet or occasionally acre-feet. For this indicator the volume of annual water use is compiled in acre-feet per year. An acre-foot is equal to 325,851 gallons.

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[3] 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)

[4] This assumes that the agencies are defining the single family and multi-family residential customer class similarly, which is not always true. E.g some agencies separate mobile home parks and dedicated landscaping meters at multi-family complexes.

This indicator also requires population data in order to calculate the per-capita use. Water suppliers also report their population, which is usually derived from census data although sometimes the population is estimated based upon the number of customer accounts.

Annual water use data for the entire 1986-2014 period is available from water suppliers that serve about 93% of the 6.65 million people that reside in the municipalities in the local Bay-draining watersheds. Total municipal and residential water use and population data for the 1986-2014 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). 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. Municipalities and areas not included because data back to 1986 was not available include Novato, Petaluma, Sonoma Valley, Napa Valley communities not including City of Napa, Vallejo, American Canyon, Benicia, Fairfield, and Suisun City; the combined population of these areas in 2014 is about 450,000.

Data for the 1986-2014 period was obtained directly from the water suppliers, from reports that the suppliers produce, and the state agencies and associations to which they report their data. The specific sources include:

1. Some or all of the 1986-2014 period directly from the following suppliers: EBMUD, SFPUC, MMWD, BAWSCA, Napa, CCWD, Zone 7, SCVWD
2. 1986-2004 data compilations for the Bay Area Water Agencies Coalition (BAWAC- a coalition of the major Bay Area water agencies); some of this data was superseded by data obtained directly from suppliers)
3. Pre-2013 from Department of Water Resources Public Water System Survey (PWSS).[5]
4. 2013 and 2014 from State Water Resources Control Board Drinking Water Program database
5. California Urban Water Conservation Council (CUWCC) database
6. Urban Water Management Plans (UWMP) for selected suppliers

The water use data reported by retail suppliers to the PWSS, the Drinking Water Program and the CUWCC is not always consistent with the data for the same year contained in agency reports including the BAWAC report and their Urban Water Management Plans. These inconsistencies were brought to the attention of the suppliers who provided the water use directly to me.

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

[5] The PWSS are available up through 2012. Beginning in 2013, DWR no longer requested suppliers to submit a PWSS and monthly water use data is reported by suppliers to the Drinking Water Program database housed at the State Water Resources Control Board.

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 gallons per capita per day (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).[6]

BENCHMARKS, 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 Table 3 from which to measure changes in total water use, population, and per-capita use.

Benchmarks used to evaluate progress on the water use efficiency metrics are based on state legislation. The Water Conservation Act of 2009, Senate Bill x7-7 (2009 Act) established a goal of reducing urban per-capita water use from a baseline usage 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. A water supplier 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. The Method 3 target is ninety-five percent of the applicable hydrologic region target derived from the State's 20x2020 Water Conservation Plan.[7] The benchmark for the total per-capita metric, based upon 95% of the region's target of 131 gpcd, is currently 125 gpcd for 2020 and 137 gpcd for 2015.[8] These benchmarks are shown in Figure 2 to assess progress for the region, although they are not meant to be used to determine 2009 Act compliance.

A second benchmark to evaluate the total volume of use derives from the 2015 emergency drought regulations to reduce urban use statewide by 25% through February 2016 as result of the 4th consecutive year of low precipitation and runoff. The State Water Board translated the statewide reduction goal into specific reduction targets from the 2013 water use for each Bay Area urban water supplier separately. The required reductions range from 8% for San Francisco to 36% for Hillsborough. Population-weighting the individual supplier targets derives a Bay Area-wide reduction target of 18%. Applying this 18% reduction to the 2013 Bay Area urban water use of

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[6] It is possible that some of the visitors using the water in the municipalities are using residential water (e.g. bed and breakfasts, other short-term rentals) 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.

[7] The 20 by 2020 Water Conservation Plan follows from the 2008 governors executive order requiring state agencies to develop a plan to reduce statewide per capita urban water use by 20 percent by the year 2020.

[8] The 131 gpcd regional target is reported in the 2010 UWMP for SFPUC. According to Peter Brostrom, DWR water use efficiency section chief, The SBx7-7 target for the San Francisco Bay hydrological region is not a fixed number but that for purposes of this assessment the 131 gpcd can be used (pers com, Sept 10, 2015)

946,000 results in a reduction target of about 775,000 ac-ft for a 12-month period. This value is not a compliance target but is useful as a benchmark for water use in 2015. [9]

RESULTS

Figures 1 and 2 and Table 3 document the fluctuation and eventual overall decline in total water use and per-capita use in the San Francisco Bay region in the 1986-2014 period. Total urban water use in the Bay Area is 24 percent or 266 thousand acre-feet (TAF) less in 2014 than it was 1986, a remarkable achievement given that the population has increased by 26 percent; water use so far in 2015 indicates that the decline will be around 30%. These impressive reductions are the result of requirements and incentives for more efficient water-using devices and landscapes, combined with the recent requirements for mandatory conservation because of the continuing drought. Periodic droughts over the last three decades along with greater public outreach have increased the consumer's awareness of water use, which helps increase water use efficiency. Prior to the more widespread of imposition of mandatory conservation in 2014 and 2015, total use in 2013 had declined 14% from 1986. Residential use did not decline as much percentage-wise as the total use - only 16% or 93 TAF in the 1986-2014 period- reflecting the residential growth in the region that has been greater in the hotter, lower-density inland areas. Commercial and industrial water use has also declined proportionally more due to shrinkage of water-intensive manufacturing and industry and economic incentives for the use recycled water and increased water use efficiency.

The per-capita use metrics use also demonstrates the significant increases in water use efficiency. Since 1986, the total per-capita use has declined by 40%, down to 119 gpcd, an even greater percentage reduction than the volumetric reduction because of the population increase. The per-capita residential use declined 33% to 72 gpcd during that same period.

Bay Area water agencies have collectively exceeded the legislative requirements for a 10% reduction in the per-capita use required by 2015. Furthermore the 2014 overall use exceeds the 20% reduction in per-capita use required by 2020. All of these trends have been impacted by the further reductions required by the State Board as of 2015 due to the increasing severity of the drought. Data from June, July and August of 2015 indicate that the Bay Area has reduced its outdoor water use significantly from the corresponding period in 2013. If the trend continues the region overall will achieve State Board goals although individual suppliers may not be fully compliant with their targets. However If drought restrictions are lifted, these gains could slip as evidenced that the 2013 per-capita use was just at the 2015 benchmark value of 137 gpcd.

The change in total and residential 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

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[9] The emergency regulations proscribe compliance for the 9-month period from June 2015 to February 2016.

between the smaller lots in the older cities and larger lots in the newer suburbs; SFPUC and CCWD represent the two extremes with a greater than two-fold difference in the total and per-capita water use. Variations in water use are 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. For example 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 30 years. 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 with the greatest reductions in the areas with higher outdoor water use.

THREATS & CHALLENGES

Responding to persisting drought will require still more efficiency, and the Bay Area faces the additional challenge of accommodating population growth. Every new person, family, or business presents increasing demand for new supply at a time when the region remains more vulnerable than ever to the warming climate. The Bay Area is still highly dependent on imports from watersheds reliant on shrinking natural snow storage. The warming climate will also increase outdoor water use, which currently represents about 40% of the total urban use in the region and offers the greatest potential for additional water savings. Efficiency improvements need to go beyond traditional conservation measures that reduce potable water use, however. Improvements must also encompass greater use of locally derived non-potable sources such as recycled wastewater and the on-site reuse of gray water, rainwater, and stormwater. The ongoing drought is also stimulating behavioral changes in how we use water. If demand stays at these reduced levels due to continued conservation 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.

Whether collective action will lead to permanent reductions in urban water use and an increase in freshwater flows to the Bay and through rivers and streams, flows vital to fish and ecosystem health, remains to be seen. Current policy and upstream water management do not provide the Estuary the extra freshwater inflow that greater water use efficiency and reliance on locally sustainable sources could provide.

Table 1: Water Agencies in the San Francisco Bay Region

Agency	Type	County / region served	2014 Population	Primary sources of water
Alameda County Water District (*ACWD*)	Retail	South Alameda	340,000	SWP, SFPUC, and ground water
Bay Area Water Supply and Conservation Agencies (*BAWSCA*)[10]	Association	San Mateo, north Santa Clara, south Alameda	1,745,116 (874,415)[11]	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	471,422	CVP, and direct diversion from the Delta
East Bay Municipal Utility District (*EBMUD*)	Retail	North Alameda, north and central Contra Costa	1,379,000	Mokelumne River and local surface water
Marin Municipal Water District (*MMWD*)	Retail	South and central Marin	187,500	Lagunitas Creek, and Russian River surface water
San Francisco Public Utilities District (*SFPUC*)	Retail and Wholesale	San Francisco	848,903	Tuolumne River and local runoff in Alameda and San Mateo County
Santa Clara Valley Water District (*SCVWD*)	Wholesale	Santa Clara	1,868,558[12]	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	238,373	SWP, local surface and ground water
City of Napa	Retail	Napa	86,051	SWP, local surface water

[10] BAWSCA does not deliver water but is an association of the 26 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.

[11] BAWSCA includes ACWD and agencies that are part of SCVWD. The bracketed number represents the population *excluding* those entities.

[12] SCVWD population includes South County

Table 2 - Total and Residential Water Use for the San Francisco Bay Region

	Year	Population Served	Total Use		Residential Use	
			Acre-feet	GPCD	Acre-feet	GPCD
	1985					
	1986	4,926,783	1,095,075	198	589,835	107
Drought Period	1987	4,979,501	1,115,781	200	589,065	106
	1988	5,037,887	1,054,355	187	544,857	97
	1989	5,104,278	947,070	166	514,297	90
	1990	5,161,134	981,503	170	514,416	89
	1991	5,194,112	859,548	148	450,112	77
	1992	5,248,028	876,048	149	482,453	82
	1993	5,319,206	908,995	153	514,013	86
	1994	5,363,939	957,448	159	531,947	89
	1995	5,394,104	961,710	159	542,424	90
	1996	5,450,714	1,016,822	167	572,912	94
	1997	5,522,039	1,066,884	172	600,685	97
	1998	5,598,163	1,009,597	161	563,015	90
	1999	5,668,259	1,060,497	167	596,470	94
	2000	5,750,656	1,090,438	169	612,620	95
	2001	5,817,604	1,093,009	168	621,477	95
	2002	5,849,746	1,089,017	166	619,335	95
	2003	5,869,093	1,059,250	161	634,344	96
	2004	5,922,332	1,082,049	163	641,958	97
	2005	5,950,543	1,031,193	155	612,521	92
	2006	5,997,222	1,030,924	153	616,989	92
Dry Period and Recission	2007	6,062,945	1,060,596	156	631,236	93
	2008	5,940,947	986,819	148	587,079	88
	2009	5,978,758	906,759	135	534,628	80
	2010	5,987,069	864,667	129	514,350	77
	2011	6,043,490	875,742	129	509,676	75
Drought	2012	6,105,603	897,884	131	543,926	80
	2013	6,166,108	945,989	137	557,358	81
	2014	6,199,597	828,660	119	496,964	72
Percent Change (%)		**26%**	**-24%**	**-40%**	**-16%**	**-33%**

Table 3: Total and Residential Water Use in 2014 for Individual Agencies in the San Francisco Bay Area

Agency	Population change since 1986	2014 Water Use			Change in water use 1986-2014		Per capita water use		Change in *per capita* water use 1986-2014	
		Total (AF[13])	Residential (AF)	Resid. % of total[14]	Total % change	Residential % change	Total (GPCD)	Resid. (GPCD)	Total % change	Residential % change
Alameda County Water District (*ACWD*)	+42%	40,647	26,168	64%	-10%	-13%	107	69	-37%	-39%
Bay Area Water Supply and Conservation Agencies (*BAWSCA*)[15]	+25%	222,896 1	137,732	62%	-22%	-14%	114	70	-37%	-31%
Contra Costa Water District (*CCWD*)	+ 53%	104,500	55,734	53%	-21%	+16%	198	106	-49%	-24%
East Bay Municipal Utility District (*EBMUD*)	+21%	188,820	112,438	60%	-21%	-17%	122	73	-35%	-32%
Marin Municipal Water District (*MMWD*)	+12%	24,521	16,934	69%	-25%	-27%	117	81	-33%	-27%
San Francisco Public Utilities District (*SFPUC*)	+13%	73,696	42,672	58%	-35%	-24%	79	46	-42%	-33%
Santa Clara Valley Water District (*SCVWD*)[16]	+31%	318,000	191,340	60%	-13%	0%	153	92	-34%	-24%
Zone 7 Alameda County (*Zone 7)*	+112%	36,148	23,066	60%	+34%	+28%	137	87	-36%	-39%
City of Napa	+31%	13,217	7,977	61%	+3%	-1%	137	83	-22%	-24%

[13] Units: AF = acre-feet (325,831 US Gal., or 1233.48 m^3); GPCD = gallons per capita per day
[14] Residential water use as % of total water use not including recycled water
[15] BAWSCA values includes ACWD and agencies that are part of SCVWD.
[16] SCVWD values are for 2013; residential use estimated based upon ratio of residential use to total use given in 2010 UWMP

FIGURE 1

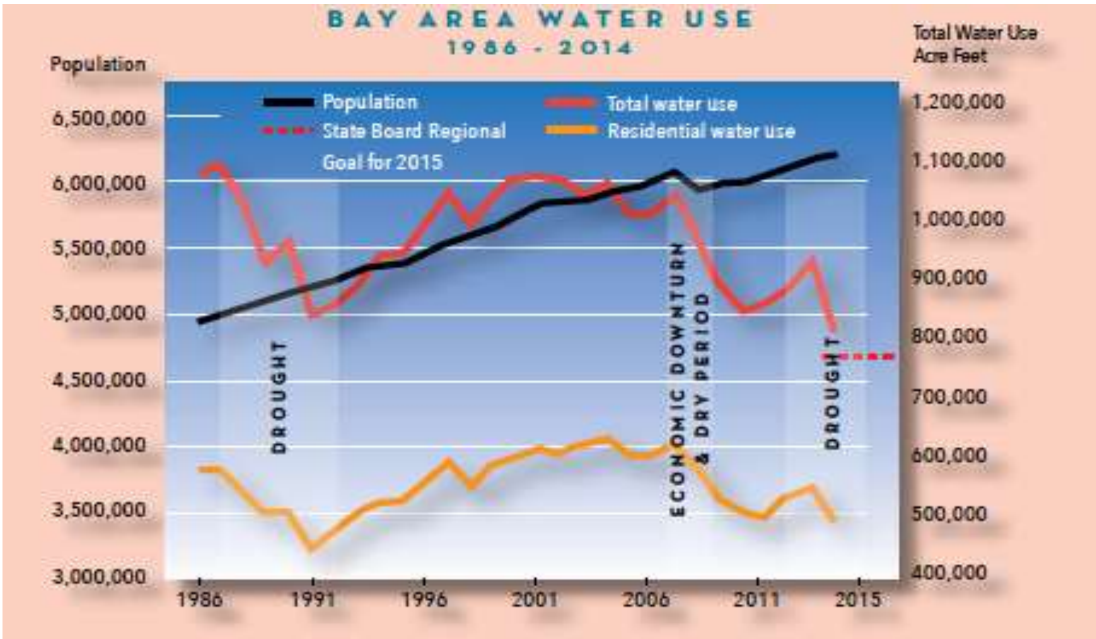
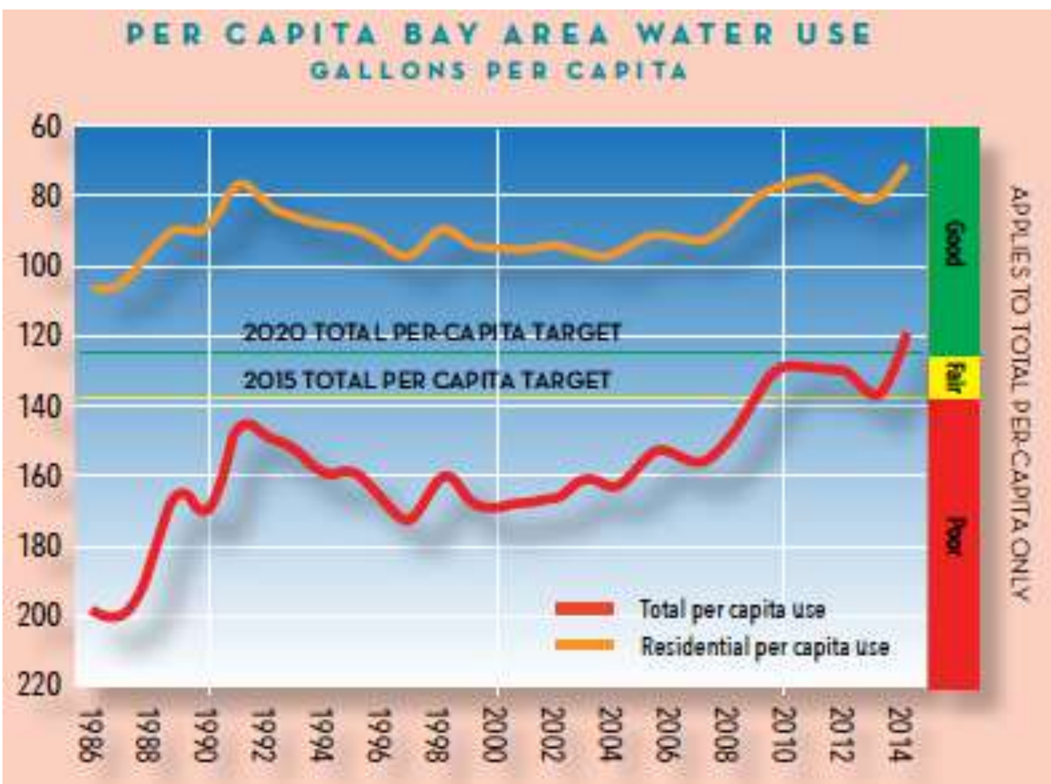


FIGURE 2



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Technical Appendix

People – Recycling Water

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TECHNICAL APPENDIX: RECYCLED WATER USE

CONTEXT

Most of the surface and ground water consumed by urban users in the Bay Area is treated to drinking water standards, used once, treated again to remove pollutants, and discharged from 34 publicly owned treatment works (POTW) into the Bay and its tidal sloughs and streams (see Figure 1).[1] Much of this consumption, including the 40% used for landscaping, does not necessarily require drinking water for its use. Until recently, treating the wastewater to recyclable standards was expensive compared to treating and distributing freshwater diverted from local and imported sources.[2] Reusing gray water from showers, bathroom sinks and laundry on-site was prohibited until recently. As a result of these limitations, a relatively small amount of intentional water recycling has occurred over the last 50 years in the Bay Area. Efforts over the last three decades have increased and made recycled water a more important part of the Bay Area's water portfolio. On-site reuse of graywater, rainwater, and stormwater is also increasing but still a very small percentage of the total recycled water use.

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 flow to wetlands or create new ones. 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 and appropriately, 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

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[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 season when runoff is higher.

[2] Recycled water must meet the standards in Title 22 of the California Code of Regulations and in accordance with a Regional Water Quality Control Board (RWQCB) permit, such as National Pollutant Discharge Elimination System (NPDES), waste discharge requirements (WDR), or water recycling requirements (WRR)

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[3] Energy consumption for recycled water depends on the distance and elevational difference of the end user and treatment plant.

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. It can also reduce the amount of nitrogen and phosphorous-rich wastewater discharged into the Bay, which locally can accelerate algal growth. In sum water recycling can meet multiple resource management and protection objectives.[4]

INDICATOR

Recycled water is quantified for four years (2001, 2005, 2010, and 2014) with two metrics[5]: 1) the total amount of water recycled, treated, and distributed from wastewater treatment plants to provide for a beneficial use. Beneficial uses, as defined by state water quality law, include domestic supply, agriculture, aquaculture, recreation, navigation, water quality, and fish and wildlife preservation; and 2) the surface and groundwater supply usable for drinking water, which the recycled water offsets (potable offset).

The recycled water that offsets potable supply is quantified into four categories based upon its end use: landscape irrigation, commercial, industrial, and agriculture .[6] The recycled water that is used in a way that does not offset potable water but still provides a beneficial use is quantified in two categories. The largest category is for creating and enhancing wetland habitat around the Bay.[7] The second category is recycled water applied by treatment plants to non-irrigated surroundings to grow grass or forage crops. In both of these cases, the discharged water is getting additional treatment, expanding the region's available water portfolio, and providing a beneficial use.

DATA SOURCES

Multiple data sources must be used to quantify annual recycled water use because the reporting requirements and definitions for the different categories recycled water use have not been standardized. Data inconsistencies arise because of 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. The California Department of Water Resources (DWR), through its 2015 guidelines for preparing Urban Water Management Plans (UWMP's) and coordination with recycled water surveys by the State Water Resources Control Board (State Board) and the Bay Area Clean Water

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[4] Recycling is not without its critics who note its high capital costs, increased concentration of total dissolved solids, and growth-inducing aspects.

[5] Consistent annual data is not readily available.

[6] Recycled water used for crop irrigation (e.g. vineyards) and cultural practices (dairies) which replaces untreated groundwater or surface water that is potentially potable is quantified as a potable offset.

[7] A small amount of recycled water is used for wildlife habitat in ponds in parks and other public spaces.

Agencies (BACWA,) is attempting to standardize the reporting of recycled water use (DWR 2015).

The following sources were used in the data compilation:

1. The San Francisco Bay Regional Water Quality Control Board (Regional Board) 96-011 recycled water annual reports required by Order 96-011 General Water Reuse Requirements for Municipal Wastewater and Water Agencies. Many but not all POTW's and water agencies that provide recycled water file 96-011 reports or report their use through their NPDES permit reports. The Regional Board now receives most of the 96-011 reports in digital form. Much of the 2014 data compilation started with these reports although some were missing data or reported the quantity of recycled water inconsistently with other reports. When available 96-011 reports for the earlier years (2005 and 2010) were also used.
2. Some of the 2010 and 2014 data was obtained directly from the wastewater treatment plant operator (e.g. Napa Sanitation District, Las Gallinas Sanitary District) or from the distributor or consumer of the water, either a water agency (e.g. EBMUD) or the direct consumer (e.g. turf farm). The quantities reported in the 96-011 reports and the amounts the water suppliers reported were not always consistent.[8]
3. The 2010 UWMP's for the water supply agencies distributing recycled water were consulted, especially for projects that did not have 96-011 reports for 2005 or 2010 although that data was not always consistent with the other sources.
4. 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).

BENCHMARKS, TARGETS AND GOALS

There are no standardized benchmarks, goals or targets for assessing progress of recycled water use in the Bay region.[9] The following are the approaches used in this assessment:

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[8] The discrepancies may arise when the amount of recycled water actually used and reported in the 96-011 reports can be less than the amount treated to the recyclable standards but not used. Or sometimes the amount demanded by the entity using the recycled water (e.g Chevron refinery) exceeds the amount that the POTW can produce who then must use potable water to supplement the recycled water to meet the demand.

[9] Statewide goals for recycled water have been established by the State Water Board but they are not based upon any regional assessments; rather they are based upon existing uses in 2003 and projections for use in 2020 and 2030 (Toni Pezzetti, pers com)

1. Comparing recycled water used to the total amount of water treated or "produced" at wastewater treatment plants, usually expressed as a percentage. Regional Water Quality Control Board records indicate that the amount treated at San Francisco Region POTW's was about 194 billion gallons or about 595,000 ac-ft. [10] Some entities establish goals to maximize the use of wastewater. E.g. the City of San Jose established a goal in 2007 to beneficially reuse 100% of its wastewater by 2022 (https://www.sanjoseca.gov/index.aspx?NID=2951)
2. Comparing recycled water used for potable offset to the potable water demand. For example, Santa Clara Valley Water District has a goal of expanding recycled water use so that it supplies at least 10% of countywide water demands by 2025 (SCVWD 2014). The 2014 potable water demand for the San Francisco Bay region is 828,660 ac-ft .
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 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 compiled, recognizing that many of the assumptions made in 1999 about demand and funding availability have not been realized. The projections made for 2015 in the 2010 UWMP's were compiled by DWR (Toni Pezzitti, May 29 e-mail) and by BACWA in their 2011 survey. The two compilations were similar, with a projection of 70 TAF of recycled use in 2015.[11]

[10] E-mail from Vince Christian, Region 2 staff engineer, on June 18, 2015

[11] BACWA's 2011 survey included about 10.4 TAF of internal use at POTW's which DWR does not include. When the internal use is removed, DWR and BACWA numbers are nearly the same.

RESULTS

Results of this assessment are displayed in Table 1 for 2001, 2005, 2010, 2014 and Figure 2, which graphically displays the results for 2001 and 2014. For 2014 Table 1 displays the amount of recycled water for each of the major categories by each "recycler" or producer of recycled water. The recyclers are grouped by region (East Bay, South Bay, Peninsula, and North Bay).

Total use steadily grew from 2001 to 2014 by 23 thousand acre-feet (TAF), an 80% increase, to 52 TAF, which represents about 9% of the wastewater produced at POTW's. The amount that offset potential potable water grew more - 26 TAF or a 158% increase- up to 42 TAF, which represents about 5% of the urban demand in 2014. The biggest increase since 2001 in offsetting potable use was by the Chevron refinery and the new and expanded use by power plants for process and cooling water. Offsetting landscape and agricultural irrigation demand grew over 10 TAF since 2001, nearly doubling, with over 1500 sites receiving recycled water for irrigation in 2014. The use of recycled water to create and enhance wetlands and further clean the water prior to its discharge into the Estuary increased modestly with the addition in 2014 of the Napa-Sonoma salt pond complex. The recycled water use in 2014 appears to have declined from 2013 in some areas as the mandatory conservation requirements reduced demand for landscape irrigation and reduced water treated at the POTW's.

Recycled water use is a small but an increasingly important part of the Bay Area's water portfolio. The extended drought and new infusion of government grants will accelerate the use of it. The projections of recycled water use are expected to increase when they are updated again later in 2015 as the continuing drought has amplified the value and reliability of recycled water.

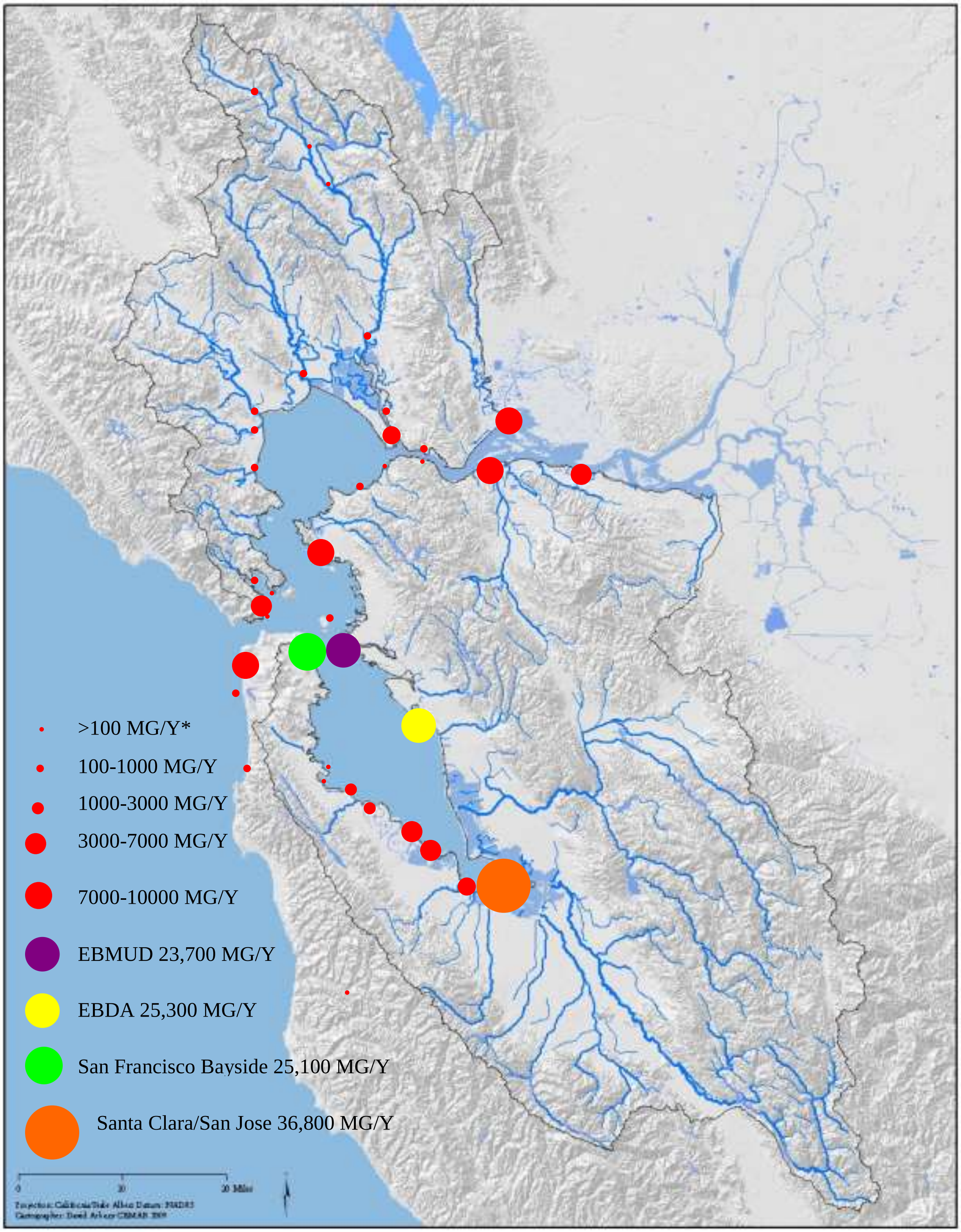
The region, however, has not been able to achieve targets and projections for its use and lags behind other urbanized regions of the State in its use. The 2014 recycled water use is about 74% of the projections made in 2010 for 2015, and 34% of the ambitious but outdated targets established in 1999 for 2010 use in the 5-county region (excluding the North Bay) by the Bay Area Regional Water Recycling Program. The shortfall in developing recycled water up to now is due to project costs and funding limitations, market demand, and customer/public acceptance.

THREATS AND CHALLENGES

Despite the Bay Area's extreme dependence on imported water, its relatively high reliability and low cost up to now has inhibited the use of recycled water. The current awareness of our regions vulnerability to drought, warming climate and natural disasters has significantly heightened the interest in the use of recycled water. A portion of the wastewater stream may never be economically feasible to develop for traditional irrigation, industrial, and commercial uses of recycled water from POTW'S because of the current mismatch between wastewater discharge locations and recycled water demand locations. On-site treatment and reuse for indoor and outdoor non-potable uses is becoming more feasible particularly in new developments. Residential users are also increasing their on-site reuse to meet their outdoor water demand as new building codes have facilitated its acceptance.

The greatest potential to significantly increase recycled water use in the near term is for groundwater recharge into the aquifers of the Santa Clara Valley, southern and eastern Alameda County and the North Bay for indirect potable reuse. The direct potable reuse of wastewater by putting it into reservoirs and distribution pipelines is in the more distant future, even though direct potable reuse already occurs in other parts of the world and water-short areas in the United States. 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.

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

TABLE 1: RECYCLED WATER USE IN THE SAN FRANCISCO BAY REGION IN 2014, 2010, 2005, AND 2001

All values in acre-feet for 2014 except where noted

Region	County	Distributor / Customer	Source Water Operator	Total	Potable Offset	Landscape Irrigation	Industrial	Commercial	Agriculture	Agriculture non-offset	Wetlands & Wildlife
East Bay	Alameda	EBMUD	EBMUD Main WWTP	173	173	173	0	1	0	0	0
East Bay	Alameda	EBMUD	San Leandro WPCP	321	321	321	0	0	0	0	0
East Bay	Alameda	San Leandro	San Leandro WPCP	301	301	301					
East Bay	Alameda	Hayward	Oro Loma/Castro Valley	242	242	242	0	0	0	0	0
East Bay	Alameda	Livermore	City of Livermore	1,078	1,078	997	77	1	3	0	0
East Bay	Alameda	Hayward Marsh	Union Sanitary District	3,810	0	0	0	0	0	0	3,810
East Bay	Alameda	Russell City Energy Center	Hayward WPCP	1,747	1,747		1,747				
East Bay	Contra Costa	CCWD	Central Contra Costa SD	437	437	436		1	0	0	0
East Bay	Contra Costa	CCWD	Delta Diablo Sanitation District	7,824	7,824	526	7,297	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	2,572	2,572	2,413	158	0	0	0	0
East Bay	Contra Costa	EBMUD- R.A.R. E.	West County Wastewater District	2,289	2,289	0	2,289	0	0	0	0
East Bay	Contra Costa	EBMUD- Chevron	West County Wastewater District	3,771	3,771	0	3,771	0	0	0	0
East Bay	Contra Costa	EBMUD	from Dublin San Ramon	715	715	715	0	0	0	0	0
Peninsula	San Francisco	SFPUC	Southeast Treatment Plant	2	2		2				
Peninsula	San Mateo	Redwood City	South Bayside System Authority	714	714	702	13	0	0	0	0
Peninsula	San Mateo	SFPUC/Daly City	Daly City	792	792	792	0	0	0	0	0
Peninsula	San Mateo	Pacifica/North Coast CWD	North Coast County WD	21	21	21					
Peninsula	San Mateo	SF Airport	San Francisco Int'l Airport	26	26		26				
South Bay	Santa Clara	San Jose etc.	San Jose/Santa Clara WPCP	10,884	10,884	7,073	3,811	0	0	0	0
South Bay	Santa Clara	Sunnyvale	Sunnyvale WPCP	205	205	195	8	2	0	0	0
South Bay	Santa Clara	Palo Alto	Palo Alto Regional WQCP	1,858	881	516	363	2	0	0	941
North Bay	Marin	MMWD	Las Gallinas SD	588	588	588	0	0	0	0	0
North Bay	Marin	North Marin WD	Novato SD	1,770	259	259	0	0	0	951	560
North Bay	Marin	North Marin WD	Las Gallinas SD	159	159	159					
North Bay	Marin	Mill Valley	Sewerage Agency South Marin	30	21	21					9
North Bay	Napa	American Canyon	American Canyon	141	141	115	0	0	26	0	0
North Bay	Napa	Napa and Ag	Napa Sanitation District	1,777	1,336	1,305	17		14	441	0
North Bay	Napa	Calistoga	Calistoga	399	341	336	5	0	0	58	0
North Bay	Napa	Yountville and Ag	Yountville	367	367	34	0	1	331	0	0
North Bay	Solano	Turf farm	Fairfield Suisun Sewer District	1,171	1,171	0	0	28	1,143	0	0
North Bay	Sonoma	Petaluma	Petaluma WR Facility	1,574	666	635	25	0	6	909	0
North Bay	Sonoma	Agriculture	Sonoma Valley County SD	2,431	1,803	0	0	0	1,803	0	629
East Bay				27,521	21,471	6,125	15,340	2	3	0	6,050
Peninsula				1,555	1,555	1,514	41	0	0	0	0
South Bay				12,947	11,970	7,784	4,183	4	0	0	977
North Bay				10,408	6,852	3,453	46	29	3,323	0	1,198
2014 total	9 counties			52,432	41,848	18,876	19,610	35	3,327	2,359	8,225
2010 total	9 counties			46,108	35,608	14,945	17,055	512	3,096	2,958	7,542
2010 minus North Bay	5 counties			37,105	30,124	12,719	16,924	478	3	0	6,982
2005 total	9 counties			35,756	25,187	12,057	12,922	6	173	3,685	6,886
2001 total	9 counties			29,094	16,219	9,392	4,865	32	1,930	5,559	7,317
% change from 2010 -2014				14%	18%	26%	15%	-93%	7%	-20%	9%
% change from 2001-2014				80%	158%	101%	303%	11%	72%	-58%	12%

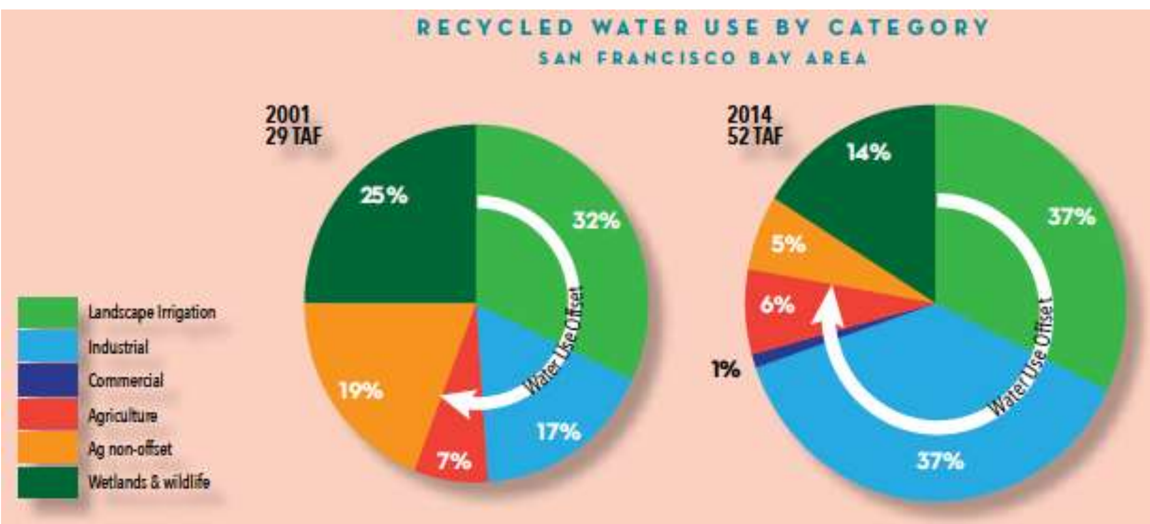
TABLE 2. RECYCLED WATER USE BENCHMARKS, TARGETS AND PROJECTIONS

Year	Total recycled use	Potable offset	Potable offset as a % of total	1999 BARWRP target	2011 BACWA projections	Total wastewater produced	Total recycled as % of wastewater produced	Potable demand in Bay region	Potable offset as % of demand
2001	29,094	16,219	56%						
2005	35,756	25,187	70%						
2010	46,108	35,608	77%	125,000		613,000	7.5%	864,667	4.1%
2014	52,432	41,848	80%			595,000	8.8%	828,660	5.1%
2015					69,806				
2020					87,324				

Notes

1. Values in acre-feet unless otherwise noted
2. BACWA projections without internal use
3. BARWRP target for 5 county region - does not include North Bay counties

FIGURE 2



Citations

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